Impacts of Geographic Variation on Aluminum Lightweighted Plug-in Hybrid Electric Vehicle Greenhouse Gas Emissions

by

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List of Abbreviations

LW% - Percentage Lighweighted

AEP – American Electric Power

BPA – Bonneville Power Administration

BREC – Big Rivers Electric Corporation

CERC – Clean Energy Research Center

CCPUD – Chelan County Public Utility District

CD – Charge Depleting

CS – Charge Sustaining

eGRID – Emissions and Generation Resource Integrated Database

EPA – Environmental Protection Agency

EF – Emission Factor

EFRI – Emerging Frontiers in Research and Innovation

GREET – Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation

GHG – Greenhouse Gas

HWFET – Highway Fuel Economy Driving Schedule

IAI – International Aluminium Institute

ICE – Internal Combustion Engine

NPCC – Northeast Power Coordinating Council

NWPP – Northwest Power Pool

MA – Mid Atlantic

MW - Midwest

NERC – North American Electric Reliability Corporation

Nested Avg. Prod. – Recommended Average Production

NHTSA – National Highway Traffic Safety Administration

NYPA – New York Power Authority

NYUP – Northeast Power Coordinating Council, Upstate New York

PCA – Power Control Area

PHEV – Plug-in Hybrid Electric Vehicle

PHEV-40 – Plug-in Hybrid Electric Vehicle, 40 miles all-electric range

PNW – Pacific Northwest

Prod. – Production

RESIN – Resilient and Sustainable Infrastructures

RFC – Reliability First Corporation

RFCW – Reliability First Corporation, West

SE - Southeast

SERC – SERC Reliability Corporation

SRMW – SERC, Mississippi Valley

SRTV – SERC, Tennessee Valley

SRVC – SERC, Virginia/Carolina

Tot. – Total

UDDS – Urban Dynamometer Driving Schedule

U.P. – Use Phase

VMT – Vehicle Miles Travelled

WECC – Western Electricity Coordinating Council

Introduction

Over the past two decades, the environmental impact of steel-bodied, internal combustion engine powered automobiles has been well quantified (Keoleian et al. 1997; Weiss et al. 2000). This configuration has dominated vehicle construction and propulsion for the last 100 years and relies on construction materials and fuels that are largely insensitive to the location of vehicle production or use. Therefore, the environmental impact of these vehicles has typically been characterized without regard to vehicle location beyond the continental level. However, increasing fuel prices, environmental concerns, and fuel efficiency regulations are influencing the adoption of new vehicle construction and propulsion technologies that are more sensitive to the location of production and use (U.S. EPA 2012a).

Two new vehicle technologies of interest, plug-in hybrid electric powertrains and lightweight automotive aluminum, are especially location sensitive. Aluminum is an extremely electricity-intensive metal to produce, requiring an average 56.4 MJ per kg of aluminum ingot (compared to steel at 27.5 MJ per kg) (World Steel Organization 2011; IAI 2007). Plug-in hybrid electric vehicles, using large batteries that store grid-supplied electricity for propulsion, consume 100,000's of MJ of electricity over their lifetime[1]. This reliance on electrical energy links the environmental impact of both technologies to electricity production emissions, which are in turn highly location dependent (U.S. EPA 2012c). These technologies are also expected to be applied to the same vehicle platform because expensive vehicle plug-in hybrid electric vehicle batteries can be downsized as vehicle mass is reduced (Bull 2011; NRC 2010; Markel 2007).

This thesis focuses on the regional variations in greenhouse gas emissions associated with aluminum lightweighted plug-in hybrid electric vehicles. Work was divided into two portions: a

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[1] Estimated lifetime electricity use of a 2013 Chevrolet Volt based on U.S. EPA combined, electric-only energy consumption of 0.35 kWh per mile, a utility factor of 63.5%, and lifetime vehicle miles travelled of 160,000 miles.

high resolution study of the regional variations in U.S. primary aluminum production greenhouse gas emissions and a case study in the net savings in greenhouse gas emissions from the application of U.S. primary aluminum to a plug-in hybrid electric vehicle. The characterization of U.S. primary aluminum production, presented in the first chapter of this thesis, also examines how electricity emissions are assigned to electricity consumers. No standards currently exist for the assignment of electricity emissions to electricity consumers and the dynamic and interconnected nature of the U.S. electrical grid creates a high level of uncertainty as to the origins of consumed electricity (Marriott and Matthews 2005; Weber et al. 2010). The second chapter of this thesis explores the impact of aluminum production region and vehicle charge location on the lifetime greenhouse gas emissions of a plug-in hybrid electric vehicle. This exploration takes place through the case study of a plug-in hybrid electric vehicle with 40 miles of all-electric range which is equipped with a hood constructed of either conventional steel, high strength steel, or aluminum. Lifetime greenhouse gas emissions for plug-in hybrid electric vehicles utilizing aluminum and high strength steel hoods are characterized in comparison to the baseline, conventional steel hood-equipped vehicle. This case study also accounts for the impacts of vehicle charge location by exploring the variation in carbon intensity between ten different U.S. vehicle charge locations.

By understanding the impact of regional variations in material production and vehicle use, we wish to inform decision makers of potential hotspots within their vehicle design and material supply chain strategies. This information can help direct attention and improvements to the most impactful parts of the vehicle’s lifecycle and ensure that strategies designed to lower the lifetime greenhouse gas emissions of personal transport have the desired effect.

A manuscript based on the first chapter of this thesis been submitted to the Journal of Industrial Ecology for publication. The second chapter is being prepared for submission.

Chapter 1: Regional Variations in U.S. Primary Aluminum Production Greenhouse Gas Emissions

Summary

This chapter explored the effects of production location and electricity emissions allocation on primary aluminum production greenhouse gas (GHG) emissions in the U.S. The generating resources that supply electricity to aluminum producers are critical in determining the environmental impact of primary aluminum production. Yet, no clear protocol exists to assign generating resources to electricity consumers in a temporally and spatially variable, interconnected electrical distribution system. In an effort to better understand and represent the connection between electricity generation resources and consumers, we explored the variation in assigned emissions associated with four existing electricity allocation protocols. We also proposed two new protocols that utilize inter-regional trade information and localized emission factors to combine generating pools that are sub-sets or super-sets of one another. This new nested approach increases the likelihood of capturing important inter-regional electricity trading and the appropriate assignment of generator emissions to consumers of local and regional electricity. When applying the new and existing protocols to the U.S. primary aluminum industry, our analysis found greenhouse gas emission factors that were dramatically different than those reported in previous literature. We calculated production-weighted average emission factors of 19.0 and 19.9 kilograms CO2-equivalent per kilogram of primary aluminum ingot produced when using our two nested electricity allocation protocols. Previous studies reported values of 10.5 and 11.0, at least 42% lower than those found by our work.

Introduction

Primary aluminum production is electricity intensive and highly dispersed (Choate and Green 2003; Bray, Wallace, and Miller 2011). When spread across the large geographic area and heterogeneous electrical distribution network present in the United States (U.S.), these regional differences are amplified (U.S. EPA 2012c). Therefore, modeling that accounts for these differences is required to fully assess the environmental impacts of production.

When assessing the environmental impacts of energy-intensive industries like primary aluminum production, life cycle analysts are often faced with the difficult task of quantifying electricity emissions. An interconnected, regionally variable, and temporally dynamic electrical grid makes the physical tracking of electrons from electricity generators to consumers virtually impossible (Marriott and Matthews 2005; Weber et al. 2010). To estimate emissions with limited knowledge about the electricity's origin, analysts often use average emission factors (EF) at various levels of aggregation (WRI/WBCSD 2007; U.S. EPA 2008). However, depending on which average EF is used, two identical processes can be assessed as having dramatically different emissions from consuming the same quantity of electricity. Average EFs also smooth over differences in emissions between regions.

We propose a new methodology that creates a local EF for electricity based on the weighted average of local and regional electricity generation resources. We use the trade between local and regional generators to determine the amount of electricity that comes from each generating pool and assign a percentage of each pool's emissions to the final quantity of consumed electricity. This approach is based on real grid boundaries and dynamics and can be applied to any region of the U.S.

The exploration and creation of new electricity allocation protocols allows us to better profile the U.S. primary aluminum industry. In addition, we demonstrate how current estimates of the environmental impact of U.S. primary aluminum are problematic, and provide an updated estimate.

Previous studies and frameworks offer guidance on allocating electricity emissions. Soimakallio et al. suggest using EFs based on purchase agreements and real-time consumption data for single process attributional life cycle assessments (Soimakallio, Kiviluoma, and Saikku 2011). Weber et al. suggest reporting a range of EFs including those calculated for the U.S. Environmental Protection Agency's (EPA) Emissions and Generation Resource Integrated Database (eGRID) sub-regions, grid operator region, and Interconnect region (Weber et al. 2010). Weber also concludes that the use of political boundaries should be avoided. Koch and Harnisch, who studied the subject of primary aluminum production in Europe, offered no recommended boundary due to Europe's complex electricity supply structure (Koch and Harnisch 2002). The U.S. has a similarly complex structure of ownership and trading.

The World Resource Institute and World Business Council for Sustainable Development (WRI/WBCSD) offer an EF hierarchy for businesses that consume electricity. WRI/WBCSD recommend site-specific EFs as being the most accurate but acknowledge that this may not apply to many electricity consumers. If site-specific EFs are unavailable, they suggest using power pool EFs. These power pools could take the form of regions or sub-regions within a nation.WRI cautioned against the use of state boundaries and, if power pool EFs are not available, suggest using national average EFs (WRI/WBCSD 2007). These boundary recommendations account for some of the complexities of an interconnected grid but do not acknowledge trading.

The U.S. EPA also offers guidance on estimating emissions from purchased electricity when quantifying greenhouse gas (GHG) emissions from U.S. industries. They recommend the use of national level EFs for industries that are spread evenly throughout the country and a disaggregated method that assigns electricity consumption to North American Electricity Reliability Corporation (NERC) regions for more concentrated industries. The EPA used the second method when allocating emissions of purchased electricity to the alumina and aluminum industry (U.S. EPA 2008).

Although no explicit advice on allocation protocols is given, the technique used by the International Aluminium Institute (IAI) to attribute consumed electricity fuel source to primary aluminum production facilities is especially important. PE Americas use this as the foundation for their grid estimation techniques and those EFs are also used in the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation Model (GREET) (PE Americas 2010; Wang, Burnham, and Wu 2012). The IAI instructs facility owners to report the percentage of electricity generation fuel source used, information that presumably comes from the utility that the facility contracts with. This information is aggregated and provided to the public at the continental level. It is unclear how facility owners account for electricity generation fuel sources or how the IAI corroborates the reported information (IAI 2012).

Previous assessments of the U.S. primary aluminum industry have been conducted but only profiled environmental impact as a national average (Choate and Green 2003; PE Americas 2010; Wang, Burnham, and Wu 2012). Each study utilized slightly different methodologies and datasets, which created a range of final EFs. We discovered two major trends when analyzing these studies: first, the inconsistent inclusion of upstream electricity generation emissions; second, extreme variations in the assumed electricity fuel mix of U.S. primary aluminum production. These two factors can drastically alter the final EF attributed to primary aluminum production.

Studies of primary aluminum production that account for regional variations were only completed for the global (comparisons between continents) and European primary aluminum production industries (Koch and Harnisch 2002; McMillan and Keoleian 2009). These and other assessments all highlight the importance of off-site electricity production emissions allocation

(PE Americas 2010; Koch and Harnisch 2002; Briem et al. 2000; Huglen and Kvande 1994; Liu, Bangs, and Müller 2011; Norgate and Rankin 2001; IAI 2007; Liu and Müller 2012).

To quantify the consequences of electricity allocation in primary aluminum production, we first explored the impacts of assigning different regional collections of electricity generators to primary aluminum production facilities. We refer to these different methods of generator assignment as “electricity allocation protocols”. We also developed and applied a new method for assigning electricity generators to primary aluminum producers and analyzed the effects of different protocols on environmental impacts.

Regional differences in primary aluminum production also arise due to differences in production technology (Koch and Harnisch 2002; Briem et al. 2000; IAI 2007; Moors 2006). Thus, we examine the rate of energy consumption and process emissions for each production facility in our model. Our examination of primary aluminum production was exclusively from cradle to exit gate and excludes any considerations for end-of-life processes.

Electricity Allocation Boundaries

The U.S. electrical grid is commonly divided into smaller regions using one of five different aggregation methods defined by the U.S. EPA’s eGRID database: power control areas (PCA), state boundaries, eGRID sub-regions, NERC regions, and U.S. average (from lowest to highest degree of aggregation), shown in Figure 1. “Degree of aggregation” refers to the number of generators used to define the electricity of a region. Typically, an area of lower aggregation (for example, an eGrid sub-region) will be nested within an area of higher aggregation (a NERC region). The assignment of a facility’s electricity consumption to the generators located in the regional boundary created by one of the five aggregation methods is called the electricity allocation protocol. Therefore, each facility can have five electricity allocation protocols when using average EFs from eGRID’s database: PCA, state, eGRID sub-region, NERC region, and

U.S. average. For each electricity allocation protocol, the GHG emissions from each group of generators are assigned to a facility's electricity consumption. Later, we describe our methodology for blending across allocation protocols to capture the electricity trading inherent in the market.

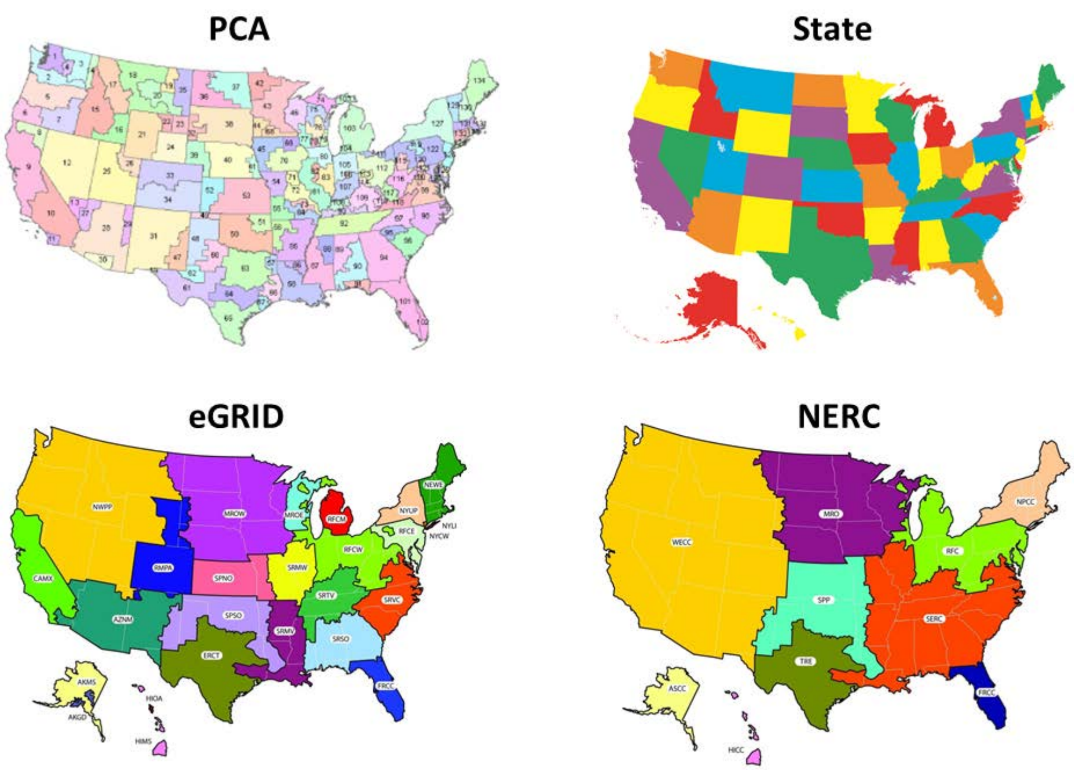


Figure 1. Regional boundaries associated with each electricity allocation protocol. Clockwise from top left: PCA, state borders, NERC regions, and eGRID sub-regions (Sattler et al. 2012; USGS 2013; U.S. EPA 2012a).

PCAs, defined as "a portion of an integrated power control grid for which a single dispatcher has operational control of all electric generators", are the least aggregated delineations of the U.S. electric grid available in eGRID (TranSystems | E.H. Pechan 2012). The generation plants are grouped into PCAs by the utility entity that owns the regional transmission and distribution infrastructure. State-level emissions were calculated as the production-weighted average of emissions from all generating resources located within a state. The EPA created eGRID sub-regions, often equivalent to the Integrated Planning Model sub-regions, using the guidance of PCA and NERC region boundaries and affiliations. The two largest regional boundaries are the NERC regions and the U.S. national boundary. NERC regions divide the U.S. into ten regions. The U.S. national EF is composed of the production-weighted average emissions from all U.S. generating resources active during 2009 (U.S. EPA 2012c).

No single boundary properly accounts for all complexities of a dynamic and interconnected electricity generation, transmission, and distribution system. The generators that create the electricity consumed by industrial processes can vary with time of day, time of year, geographic location, and electricity markets. The use of any of the five boundaries will not fully characterize the electricity consumed inside that region. Table 1 provides a brief overview of the advantages and disadvantages of each allocation boundary.

Table 1. Advantages and disadvantages of the five existing electricity allocation protocols.

Electricity Allocation Protocol	Advantages	Disadvantages
PCA / Contract Mix	-Captures emissions from the nearest generators -Aligns with purchase agreements between consumers and generators - Incentivizes the use of low carbon electricity	-Doesn't account for inter-PCA trade -Boundaries overlap, depend on transmission ownership, and are not geographic in nature -No guarantee that electricity purchased from a PCA is the actual electricity consumed
State	-Accounts for trading between all PCAs located within the state -Boundaries are geographic, no overlap	-Doesn't account for inter-state trade -Does not reward those who purchase cleaner electricity or locate near cleaner electricity producers -Boundary not related to grid operation
eGRID Sub-region	-Accounts for inter-state trade -Boundaries are geographic, no overlap	-Doesn't account for trading that occurs across eGrid sub-regions -Does not reward those who purchase cleaner electricity or locate near cleaner electricity producers -Created for use in policy-making model, unclear connection to actual grid layout
NERC Region	-Accounts for trading that occurs between eGRID sub-regions -Boundaries are geographic, no overlap -Boundaries based on grid composition	-Reduces regional differences -Does not reward those who purchase cleaner electricity or locate near cleaner electricity producers -May incorrectly allocate emissions from far away plants to electricity consumers -Doesn't account for inter-NERC trade
U.S. Average	-Accounts for all domestic trade	-Doesn't account for international trade -Eliminates regional differences in electricity mix -Does not reward those who purchase cleaner electricity or locate near cleaner electricity producers

Nested Average Electricity Allocation Protocols

The selection and use of any of the five electricity allocation protocols above result in an EF unlikely to reflect inter-regional trade. Depending on the magnitude of this trade, electricity could be associated with the incorrect pool of generators. We propose merging two allocation protocols, bridging the gap between protocols that account well for local generation but ignore trade and protocols that account well for inter-regional trade but dilute the contributions of local resources.

To capture local generation, we use the EF of any on-site or PCA generators. To capture the average emissions of the pool of generators from which the PCA imports we include the NERC region EF. Both the on-site / PCA and NERC region EFs are then weighted by the percentage of local versus imported electricity used in the PCA over a one year period. The geographical relationship between local and regional generators is shown below in Figure 2.

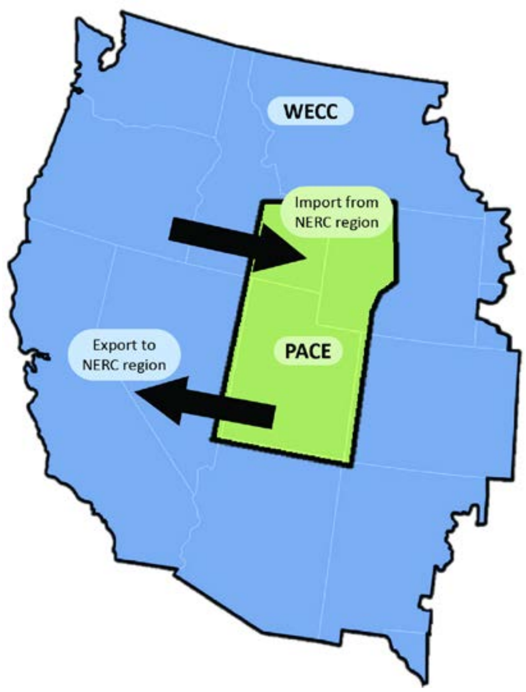


Figure 2. Illustration of trade between a small pool of generators (the PacifiCorp East (PACE) power control area) located within a larger region of generators (the Western Electricity Coordinating Council (WECC) NERC region) (U.S. EPA 2012a).

The electricity imported into a PCA may generally be used to either supply local customers or transmit to a neighboring PCA. Electricity is imported for local customers when local generators are insufficient or uneconomical to operate. Electricity is passed through a PCA and onto another if the neighboring regions do not have a technical or economically viable transmission pathway. To properly characterize each scenario, we provide two separate protocols. The first protocol, called the *import/export nested average*, assigns the EF of local generators to all electricity generated within and exported from the PCA. All imported electricity is assigned the EF from the surrounding, adjusted NERC region. An adjusted NERC region EF is created by subtracting all generators from the PCA located within the NERC region, thereby accounting for only emissions coming from generators within the NERC external to the PCA. This avoids double counting. The import/export nested average is presented below in equation (1).

$$EF_{I/E} = \frac{Gen_{PCA} \times EF_{PCA} - Exp_{PCA} \times EF_{PCA} + Imp_{PCA} \times EF_{Adj\ NERC}}{Gen_{PCA} - Exp_{PCA} + Imp_{PCA}} \tag{1}$$

where $EF_{I/E}$ is the import/export nested average EF for the PCA in kg CO_2-eq per kWh, Gen_{PCA} is the amount of electricity generated in the PCA in kWh per year, EF_{PCA} is the EF of PCA in kg CO_2-eq per kWh, Exp_{PCA} is the amount of electricity exported from the PCA in kWh per year, Imp_{PCA} is the amount of electricity imported to the PCA in kWh per year, and $EF_{Adj\ NERC}$ is the EF of the adjusted NERC region. This protocol assumes that all exported electricity is consumed outside the boundary of the PCA and does not count towards emissions within the PCA. This protocol also assumes that all imported electricity is consumed within the PCA and accepts these emissions created by outside generators.

The second protocol examines electricity trading on a net basis. We call this scenario a *net nested average* electricity allocation protocol. If the PCA is a net importer of electricity, the EFs of the local PCA and surrounding adjusted NERC region are averaged based on the amount of electricity imported. If the PCA is a net exporter, all electricity is assumed to come from the local PCA as net export would only take place if a surplus of electricity was generated. Even though large quantities of electricity may be imported, if an equal or greater amount of electricity is exported, we assume that no imported electricity was locally consumed. The proposed protocol is further illustrated in equation (2).

$$EF_{Net} = EF_{PCA,i} \times W_{PCA} + EF_{Adj\ NERC} \times (1 - W_{PCA}) \tag{2}$$

where EF_{Net} is the net nested average EF of the PCA in kg CO_2-eq per kg aluminum ingot, EF_{PCA} is the EF of the PCA in which the production facility resides in kg CO_2-eq per kWh, W_{PCA} is the percent of local electricity provided to the facility during the preceding year, and $EF_{Adj\ NERC}$ is the adjusted EF of the NERC region in which the production facility resides in kg CO_2-eq per kWh.

U.S. Primary Aluminum Production

Methods

To explore the impact of using each electricity allocation protocol on the calculated emissions of U.S. primary aluminum production, we calculated cradle-to-gate GHG EFs on a per-kg-aluminum-ingot basis for nine U.S. smelters operational in 2010. Cradle-to-gate primary aluminum production is broadly composed of six stages: bauxite mining, alumina production, anode production, electrolysis, ingot casting, and transportation and ancillary activities. We used smelter-specific data to characterize electrolysis and industry average data for all other stages. Electricity consumption EFs were composed of 2009 at-plant emissions as reported in the U.S. EPA's eGRID database and upstream fuel emissions as reported by GREET2 2012 (Wang, Burnham, and Wu 2012; U.S. EPA 2012c).

We calculated annual smelter electricity consumption by scaling peak annual energy consumption (peak power consumed over 365 days of continuous operation) by a capacity utilization ratio composed of annual smelter aluminum output divided by annual smelter nameplate capacity. This is characterized by equation (3):

$$Q_{Annual,i} = \dot{Q}_{Peak,i} \times 8{,}760 \times \frac{P_{Annual,i}}{P_{Capacity,i}} \qquad (3)$$

where $Q_{Annual,i}$ is the annual energy consumption in megawatt hours (MWh) per year of aluminum smelter i, $\dot{Q}_{Peak,i}$ is the peak power consumption of aluminum smelter i in megawatts (MW), 8,760 is the number of hours in a year, $P_{Annual,i}$ is the annual aluminum production of aluminum smelter i in tonnes, and $P_{Capacity,i}$ is the annual nameplate capacity of aluminum smelter i in tonnes.

We determined energy consumption on a kilowatt-hour (kWh) per kg aluminum ingot basis by dividing annual smelter energy consumption, $Q_{Annual,i}$, by annual aluminum ingot output, $P_{Annual,i}$, as seen in equation (4).

$$E_{Electrolysis,i} = \frac{Q_{Annual,i} \times 1{,}000}{P_{Annual,i}} \qquad (4)$$

where $E_{Electrolysis,i}$ is the energy consumption of smelter i in kWh on a per kg aluminum ingot basis and 1,000 is the number of kWh in a MWh. We assume that the energy consumption of an aluminum smelter scales linearly with aluminum output.

We determined smelter peak power consumption, $\dot{Q}_{Peak,i}$, through a variety of data sources including smelter websites, power control area websites, court and regulatory documents, and news stories (table provided in the Appendix). The United States Geological Survey (USGS) reported 2010 annual smelter nameplate capacity, $P_{Capacity,i}$, and Harbor Aluminum Intelligence reported annual smelter output, $P_{Annual,i}$ (Bray, Wallace, and Miller 2011; Harbor Aluminum Intelligence Unit 2011).

We used equation (5) to calculate primary aluminum GHG EFs for each primary aluminum smelter in the U.S. for location-specific electricity allocation protocols:

$$EF_{i,j} = E_{Electrolysis,i} \times \epsilon_{Elecricity,j} + E_{Process,i} + B_{Avg} + A_{Avg} + T_{Avg} \tag{5}$$

where $EF_{i,j}$ is the EF of primary aluminum production from smelter i with an electricity consumption EF from regional area j in kg CO_2-eq per kg aluminum ingot, $E_{Electrolysis,i}$ is the energy consumption (primarily from electrolysis) of smelter i in kWh per kg aluminum ingot, $\epsilon_{Elecricity,j}$ is the full fuel-cycle electricity consumption EF from regional electrical grid area j in kg CO_2-eq per kWh, $S_{Process,i}$ is the process emissions of smelter i in kg CO_2-eq per kg aluminum ingot, and B_{Avg}, A_{Avg}, and T_{Avg} are the emissions from bauxite mining, alumina refining, and transportation and ancillary activities in kg CO_2-eq per kg aluminum ingot, respectively.

Regional electricity consumption EFs, seen in equation (5) as $\epsilon_{Electricity,j}$, were provided for the geographic area in which each aluminum smelter was located for each of the seven electricity allocation protocols. These full fuel-cycle EFs included upstream emissions from fuel resource recovery, transportation, and processing and the emissions released during fuel combustion. We used equation (6) to create specific upstream EFs for each electricity allocation protocol by using the regional fuel mix:

$$\epsilon_{Elecricity,j} = \left[\left(\sum_{k=1}^{11} FS_k \times EF_{Upstream,k}\right) + EF_{Combustion,j}\right] \times (1 + TD) \tag{6}$$

where k is the fuel type with 1 through 11 corresponding to coal, oil, natural gas, nuclear, hydro, biomass, wind, solar, geothermal, other fossil fuels, and unknown/purchased generation respectively, FS_k is the percentage of each fuel type used in the regional grouping provided by eGRID 2012, $EF_{Upstream,k}$ is the upstream EF in kg CO_2-eq per kWh of the fuel type provided by GREET1 2012, $EF_{Combustion,j}$ is the production-weighted average at-plant combustion emissions in kg CO_2-eq per kWh for regional area j provided by eGRID 2012, and TD is the regional transmission and distribution line loss factor in percent provided by eGRID 2012 (U.S. EPA 2012c; Wang 2012).

The U.S. EPA's Greenhouse Gas Reporting Program's database reported process emissions for each primary aluminum smelter (U.S. EPA 2012d). This included perfluoromethane (PFC-14), perfluoroethane (PFC-116), and CO_2 emissions. We converted all emissions to CO_2-eq values using a 100-year time horizon (Solomon et al. 2007). While it is certain that the process EFs captured process emissions from electrolysis, we assumed process emissions from anode production and ingot casting were also captured. We made this assumption due to the tendency for most primary aluminum smelters to have these operations co-located with their smelting facilities (U.S. EPA 1996). Therefore, we assumed that the emissions from all three activities were accounted for in the smelter's process emissions.

We used industry-average data from the IAI's 2005 lifecycle assessment of primary aluminum production to quantify emissions associated with bauxite mining, alumina refining, and transportation and ancillary activity. According to the IAI, GHG emissions from these stages accounted for only 21% of lifecycle GHG emissions and, due to low levels of electricity utilization, are assumed to have more stable (and location independent) emissions factors (IAI 2007). The characteristics of each aluminum smelter are provided in the Appendix.

Primary aluminum producer websites, court and regulatory documents, and news stories often mention which entity an aluminum producer purchases electricity from. For all but two aluminum producers, Warrick and Massena West, this corresponded to a PCA recognized within eGRID's database. For the Massena West facility, our references indicated that power was purchased from the New York Power Authority (NYPA), a state power organization not listed in eGRID as a PCA (New York Power Authority 2012). The Warrick smelter has a coal-fired power plant located on site to meet its electricity needs (Alcoa 2008). As no standard PCA EF was available, we created our own local EFs for these facilities. We did the same when profiling the generating resources of American Electric Power – Ohio (AEP). While eGRID recognized AEP as a PCA, the reported EF was calculated based only on AEP's generating resources located in the southern U.S. and did not represent AEP – Ohio's emissions (AEP-Ohio 2013).

In this paper, we refer to all on-site, utility contracted or PCA contracted electricity as coming from a PCA. This places all of the closest electricity generating resources into the same category for comparison.

We found the amount of generated, imported, and exported electricity for each PCA from U.S. Federal Energy Regulatory Commission (FERC) regulatory filings and PCA-published annual reports. We assumed that on average, electricity imported into a PCA originated from generators within the surrounding NERC region.

Results and Discussion

Figure 3 presents cradle-to-gate GHG EFs for the nine U.S. primary aluminum smelters operational in 2010. Total cradle-to-gate GHG emissions from the production of aluminum ingot at each smelter are reported using electricity allocation protocols that assign EFs from the smelter's applicable PCA, state, eGRID sub-region, NERC region, U.S. average, and a nested average electricity generating resources to smelter electricity consumption.

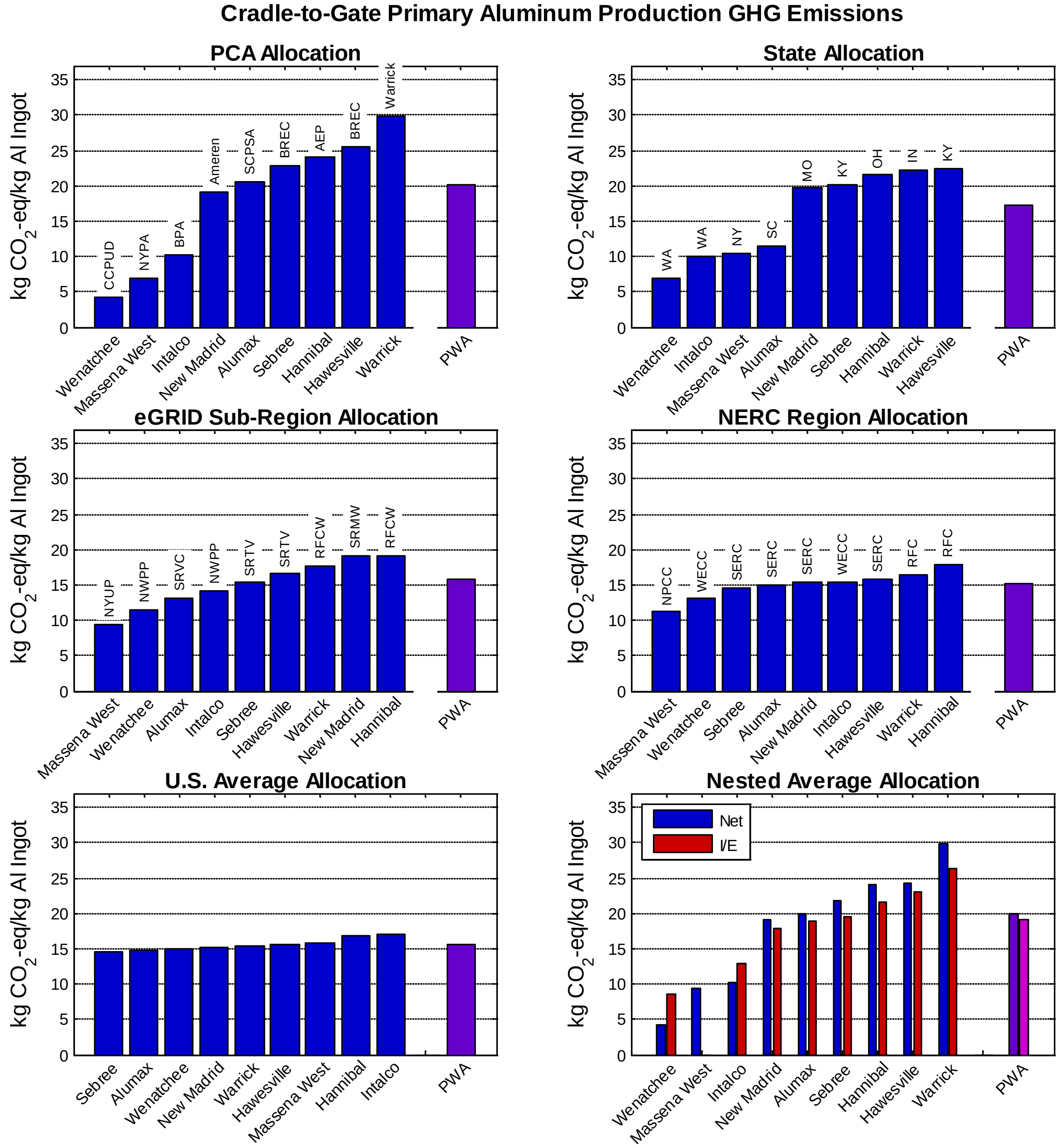


Figure 3. Cradle-to-gate primary aluminum production GHG emissions for the nine aluminum smelters operational in the U.S. in 2010 (blue and red) and national production weighted average (PWA) GHG emissions (purple). Protocol regions are displayed above each smelter's EF. No import/export nested average EF was calculated for Massena West because no import/export information was publically available for NYPA.

Smelter emissions were affected by location (and therefore the composition of regional generating resources) and the electricity allocation protocol applied. These influencing factors created a large range of EFs for each smelter and between smelters. An approximately 700% difference between the least and most carbon intensive production locations, 4.3 to 30.0 kg CO_2-

eq per kg aluminum ingot, occurred when electricity consumption was allocated to the most local generators (PCA). When allocation protocols expanded the geographic boundary and assigned electricity consumption to a larger number of generators, EFs began to converge towards approximately 15 kg CO_2-eq per kg aluminum ingot. The production-weighted average of all GHGs emitted during primary aluminum production in the U.S. in 2010 ranged from 15.7 to 19.8 kg CO_2-eq per kg aluminum ingot, depending on the electricity allocation protocol used. The results present a range of EFs not seen in previous characterizations of U.S. primary aluminum production and a higher national production weighted average EF.

We find that the nested average electricity allocation protocols allows us to best represent trading and the relative contributions of the regions close to and immediately surrounding each aluminum smelter. The results are almost identical to those calculated when all electricity consumption was allocated to PCAs. This can be attributed to low levels of imported electricity for most PCAs in which aluminum production facilities are located. This is logical as most aluminum production facilities were originally located near areas of abundant and inexpensive electricity. The net nested average protocol produces an EF closer to that calculated by the PCA allocation protocol because imported electricity is assumed to pass through the region instead of be consumed by the aluminum smelter. The local generating resources therefore contribute more of the consumed electricity. The import/export nested average protocol produces EFs that trends towards those seen when the national average EF was applied. The import/export nested protocol assumes all imported electricity is consumed locally. As imported electricity comes from NERC regions that tend to have EFs closer to the U.S. average than smaller, more homogeneous PCAs, this pulls the EFs of each smelter toward the national average.

When we calculated emissions using the U.S. average electricity allocation protocol, process emissions dominate in determining carbon-intensity. Intalco, a smelter located in a region with large amounts hydroelectric generation, went from being one of the least carbon-intensive smelters to the most due to higher than average process emissions. These process emissions were attributed to above average perfluorocarbon (PFC) emissions making Intalco's process emissions 5.5 kg CO_2-eq per kg aluminum ingot (U.S. EPA 2012d). These process emissions are almost two times higher than the industry average.

The use of a single electricity EF also helped isolate the impact of smelter production energy intensity. If all smelters consume electricity with the same EF, then smelters that consume less electricity per kg of aluminum ingot produced have lower electricity consumption emissions. Plant energy intensity, measured in kWh of electricity required to produce one kg of aluminum ingot through electrolysis, ranged from 14.4 kWh per kg aluminum ingot for Intalco to 17.9 kWh per kg aluminum ingot for Hannibal. Interestingly, Intalco's above-average smelter efficiency was offset by high levels of PFC emissions. These trends are shown in Figure 4.

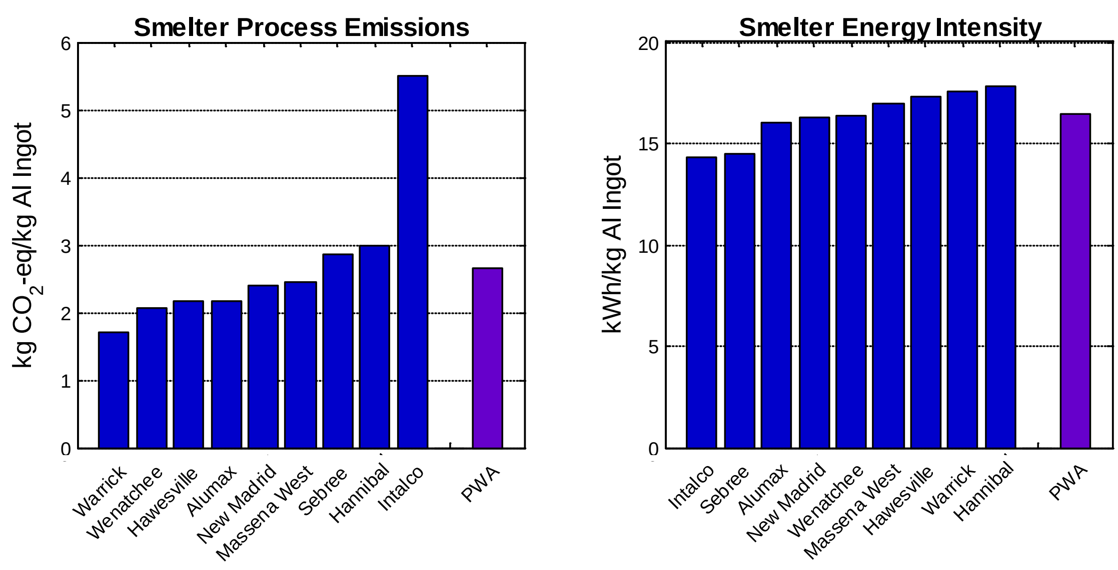


Figure 4. Primary aluminum production smelter process emissions (left) and energy intensity (right). Process greenhouse gas emissions include CO2, PFC-14, and PFC-116. Production weighted averages (PWA) are represented by the purple bars.

In anticipation of the application of this work to more broad material sourcing studies, we also broke down primary aluminum production GHG emissions by material production area, shown in Figure 5.

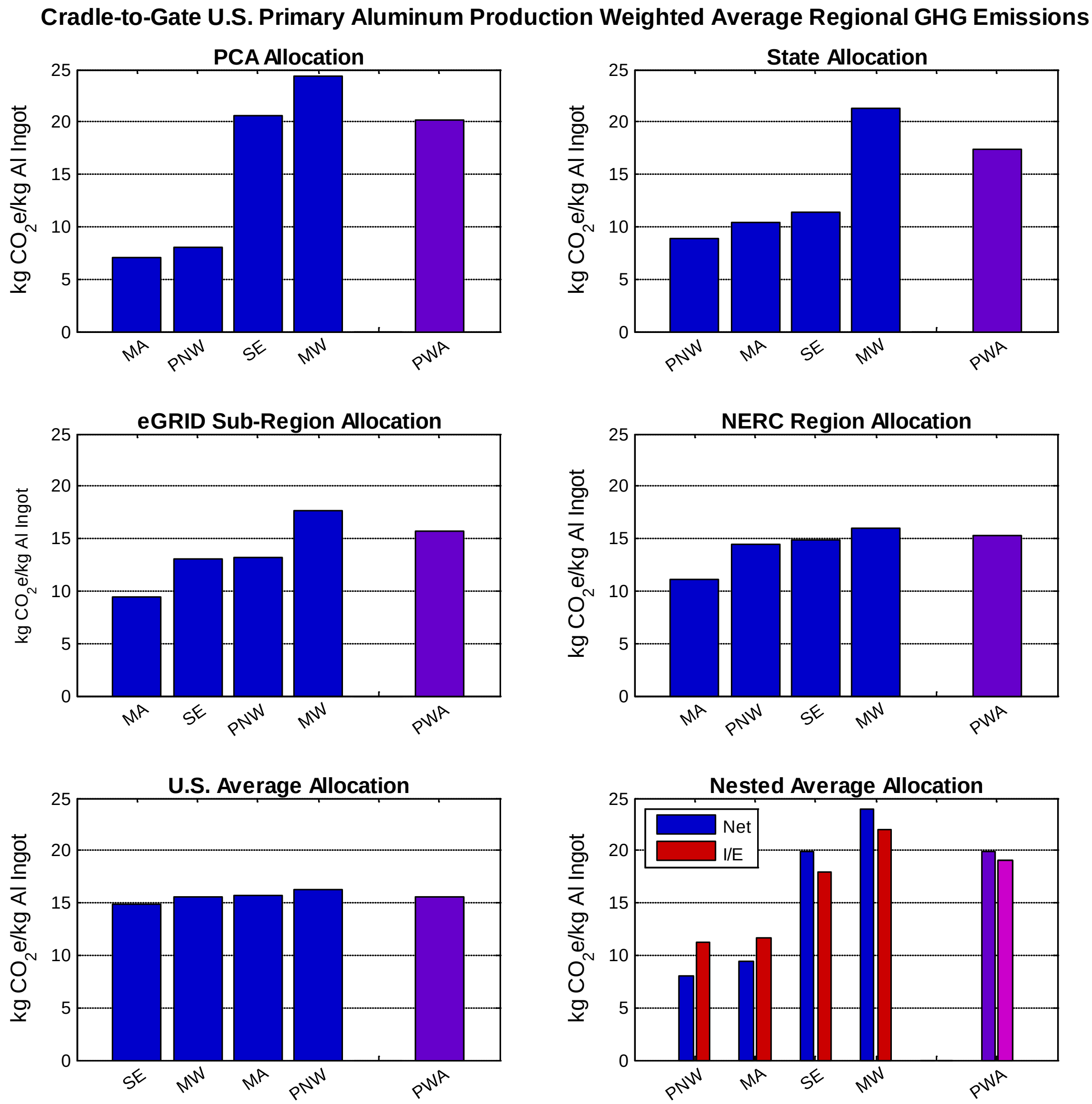


Figure 5. Cradle-to-gate primary aluminum production regional production weighted average GHG emissions for the Pacific Northwest (PNW), Midwest (MW), Mid Atlantic (MA), and Southeast (SE) (all blue or red) and national production weighted average (PWA) (all purple). In the nested average allocation chart, blue bars represent the net nested average protocol (net) and the red bars represent the import/export nested average protocol (I/E).

Sensitivity Analysis

We assessed the sensitivity of our model to changes in electricity generation fuel type, smelter energy intensity, and smelter capacity utilization. The end of life allocation protocols used to assess primary aluminum production can also have a significant impact on overall EF calculations (Johnson, McMillan, and Keoleian 2013; Nicholson et al. 2009). In our model, we used the recycled content method, assigning all the burdens of primary production to our calculated EF and not counting any credits for material recycling. We chose this with the intent that our calculated primary EF be used as a building block for more complex analyses of the aluminum product life cycle. As the focus of our study was electricity allocation in primary aluminum material production, we leave the exploration of additional end-of-life allocation protocols to future research.

We calculated the EF for the entire production weighted U.S. primary aluminum industry as if all smelters were to consume electricity generated from a single fuel type, seen in Figure 6. We sourced the electricity consumption EFs of these single-fuel-source scenarios from GREET1 and accounted for the full fuel-cycle emissions burdens (Wang 2012). Production weighted EFs ranged from 23.3 kg CO_2-eq/kg Al for an all-coal fueled electricity grid, 15.1 kg CO_2-eq/kg Al for natural gas, to 4.9 kg CO_2-eq/kg Al for hydro and renewables. This captures the full range of production weighted EFs seen in our study (15.4 to 19.8 kg CO_2-eq per kg Al ingot). The 479% difference in GHG EFs seen between primary aluminum production based on coal-generated electricity versus hydro and renewable-generated electricity highlights the large influence of electricity generation fuel source on the final GHG EFs of primary aluminum production. The sensitivity of overall GHG emissions to electricity fuel source suggests that aluminum carbon intensity is strongly tied to grid carbon intensity.

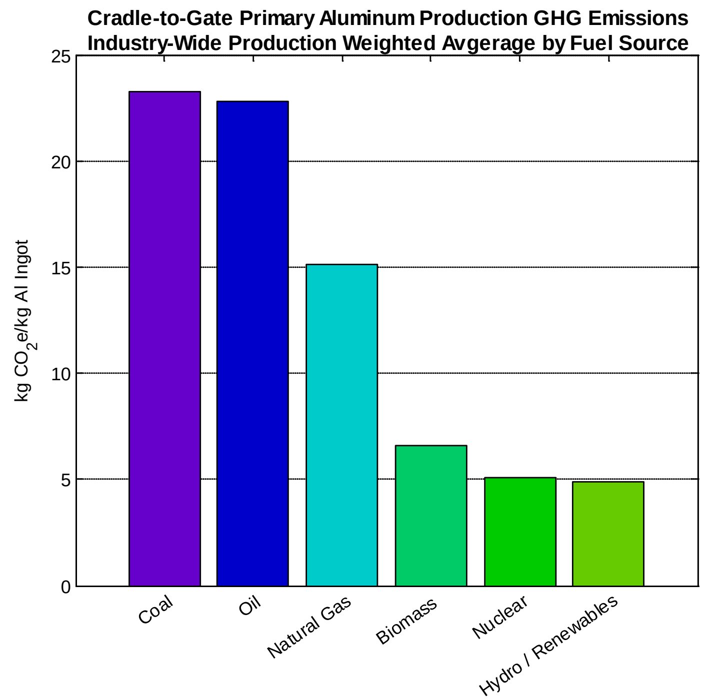


Figure 6. Cradle-to-gate primary aluminum production weighted average GHG emissions by electricity generation fuel source**.**

Replacing smelter specific energy intensity with the global average energy intensity, shown in Figure 7, resulted in EFs that ranged between 11.4% lower for Warrick and 4.1% higher for Sebree. Minimum and maximum global average smelter energy intensity values encapsulate all smelter-specific EFs calculated in this study. While variation existed between smelter-specific and global average energy intensity, the overall trends discussed in the results section remained; EFs were most divergent when electricity emissions were allocated to the smallest group of generating resources and converged as allocation boundaries expanded.

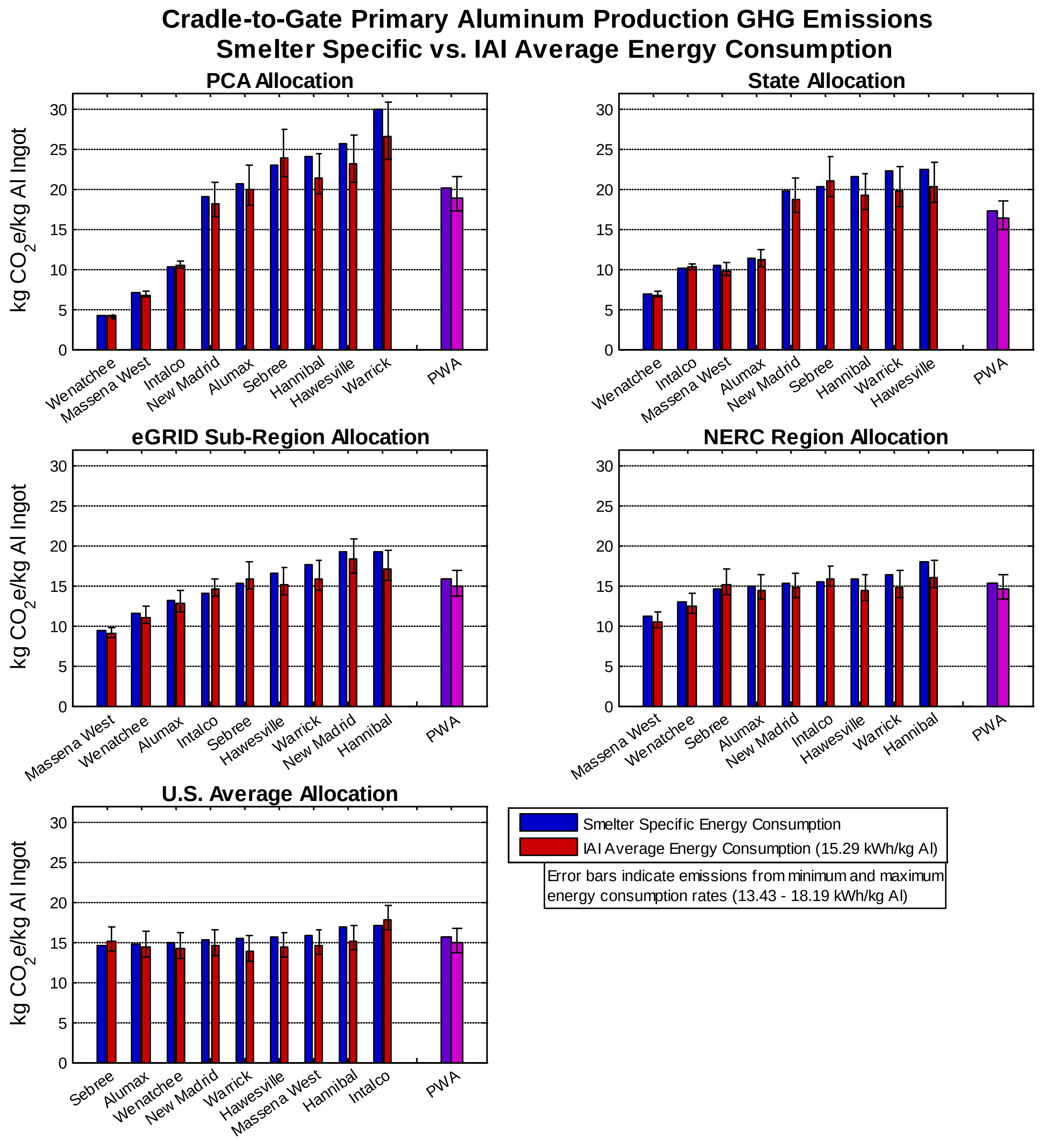


Figure 7. Cradle-to-gate primary aluminum production GHG emissions for smelter specific (blue) and 2005 International Aluminium Institute (IAI) global average (red) energy consumption. U.S. national production weighted average (PWA) emission factors are represented by dark purple and light purple bars.

This sensitivity analysis also illustrates the potential impacts of increasing plant efficiency. For example, if Warrick were to decrease smelter energy intensity from 17.6 kWh per kg aluminum ingot to 13.4 kWh per kg aluminum (its current smelter efficiency and the average lowest potential smelter efficiency respectively), its emissions would decrease from 30.0 to 23.8

kg CO2-eq per kg aluminum ingot, a 20.5% decrease. For comparison, if Warrick instead switched the fuel source of the electricity generating resources it relied on from coal to natural gas or hydroelectricity, its emissions would decrease to 14.9 or 3.9 kg CO2-eq per kg aluminum ingot, decreases of 50.4% and 86.7%, respectively. For aluminum smelters seeking to lower their carbon footprint, these results suggest that the focus should be on the fuel source of electricity generating resources.

Future demand for aluminum products is likely to increase as a number of industries look to take advantage of aluminum’s lightweight and highly conductive material properties (Nappi 2013). 2010 U.S. primary aluminum production capacity utilization rates ranged from 54 to 99% with a national average of 79%. An additional 462,000 tonnes of aluminum could be produced at the U.S. production maximum (Bray, Wallace, and Miller 2011; Harbor Aluminum Intelligence Unit 2011). This scenario includes Alcoa’s Massena East smelter re-starting (it is currently off-line for renovation) and bringing 125,000 tonnes of primary aluminum back into production.

Figure 8 shows the results of bringing all U.S. primary aluminum production facilities up to 100% of nameplate capacity. For all but one electricity allocation protocol, the production weighted average EF decreased. For our net and import/export nested average protocols, the EFs fell from 19.9 and 19.0 to 18.4 and 18.1 kg CO_2-eq per kg aluminum ingot respectively. We attribute this to the large increases in production by aluminum smelters located in in the Pacific Northwest and Mid-Atlantic where most electricity is generated by hydroelectric dams. When using the PCA electricity allocation protocol, we assumed that increased production would be met by current electricity production resources and also did not alter the ratio of imported or exported electricity for the nested average protocols.

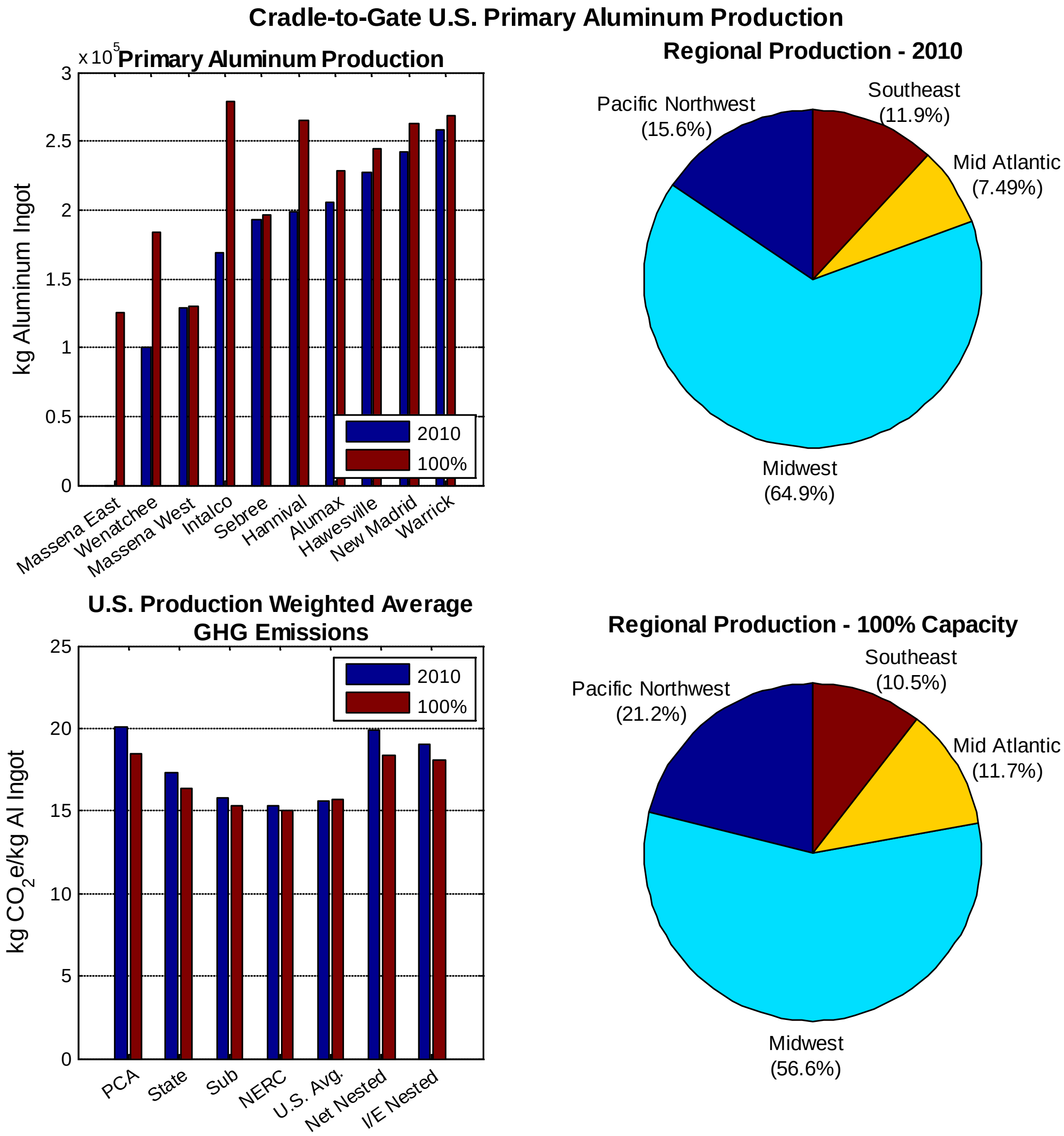


Figure 8. U.S. primary aluminum production in 2010 and at 100% capacity (top left), 2010 U.S. primary aluminum production by region (top right), U.S. primary aluminum production by region at 100% capacity (bottom right), and U.S. primary aluminum production weighted average EFs for each electricity allocation protocol for 2010 and 100% production rates (bottom left) (Harbor Aluminum Intelligence Unit 2011; Bray, Wallace, and Miller 2011).

Conclusion

Nested average electricity allocation protocols were developed to account for locally generated and imported electricity using a weighting scheme tied to actual electricity trade. These protocols can be used to model any of the 119 U.S. PCAs and offer a standardized method of profiling electricity consumption emissions. Our example of this model relies on PCAs and

NERC regions because of their nested geographic relationship but could be applied to any two nested grid boundaries in either the U.S. or globally.

We created both the net and import/export nested average models to provide the means to assess different end uses of imported electricity: consumption or export. Information on the local generating resources within the PCA will offer guidance as to which nested average protocol should be applied. In scenarios where the amount of imported electricity far exceeds the amount exported, we recommend using the import/export method. Imported electricity is likely being used to supplement local generating resources that serve customers within the PCA. We also recommend using the import/export method for the inverse scenario, where much more electricity is being exported than imported.

However, in scenarios where the amount of imported and exported electricity are nearly even, either net or import/export may be appropriate. If local generating resources are highly seasonal, electricity may be imported to cover the temporary loss of base load generation resources. For these circumstances, it would be more appropriate to use the import/export nested average. If a PCA instead relied on consistent local generation and owned extensive transmission infrastructure, imported electricity would likely pass through the region without being consumed and the net nested average would be more appropriate.

While nested average electricity allocation protocols account well for emissions from local and regional generating resources, the current version is unable to account for electricity generated outside of and potentially imported into the larger regional boundary. NERC regions may import electricity from other NERC regions and from generators located across national boundaries. Future versions of this model could account for this trade with the addition of another layer and its trade-weighted EF. Nested average protocols are also limited by the amount of trade information known about a region and how imported electricity is utilized within a region.

The U.S. average aluminum production EF reported in our study differ significantly from current models and literature, especially when using the nested allocation protocols. U.S. GHG EFs ranged from 10.1 to 11.1 kg CO_2-eq per kg aluminum ingot in current models and literature while national production-weighted average values from this study ranged between 15.7 to 19.8 kg CO2-eq per kg aluminum ingot produced in 2010 (PE Americas 2010; Choate and Green 2003).

To help understand differences between the EFs calculated by our model and those in the literature, we examined GREET2 2012. We selected GREET2 because the boundaries used to calculate EFs aligned well between our model and the GREET2 model. GREET2 utilizes a vastly different estimation of electricity fuel sources than that used here. GREET's default electricity fuel mix comes from that assumed by PE Americas', with electricity coming from 69.2% hydroelectric, 29.7% coal, 0.3% natural gas, and 0.6% nuclear sources. To justify this electricity fuel source mix, PE Americas assumed that the 2007 International Aluminium Institute (IAI) electricity fuel source mix for North America was also representative of the fuel source mix used by U.S.-only primary aluminum producers (PE Americas 2010). This assumption is likely unrealistic when the locations of U.S. hydroelectric resources are considered in relation to U.S. primary aluminum production facilities. While 64.9% of aluminum was produced in the Midwest in 2010, only 21.2% of the nation's hydroelectricity came from the three NERC regions with Midwestern territory in 2009 (U.S. EPA 2012c; Harbor Aluminum Intelligence Unit 2011). We chose to replace the default fuel mix in GREET2 with the fuel mix assumed by Choate and Green and Marriot and Matthews (Choate and Green 2003; Marriott and Matthews 2005). The results of using these alternative grid electricity mixes more closely track the results of our model. Using the Choate and Green electricity fuel mix, an EF of 16.0 kg CO2-eq/kg Al was calculated. An EF of 16.2 kg CO_2-eq/kg Al was calculated using the fuel mix proposed by Marriot and Matthews.

With the introduction of nested average electricity allocation protocols, we offer a new methodology that better reflects the composition, boundaries, and regional trade of the U.S. electrical grid. When characterizing the U.S. primary aluminum industry, the impact of these new allocation protocols is dramatic, leading to calculated EFs nearly twice that of previous studies.

Chapter 2: Impact of Material Production and Battery Charge Location on PHEV-40 Lifetime Greenhouse Gas Savings: Aluminum Hood Case Study

Summary

This chapter details the GHG emissions impacts of lightweighting a plug-in hybrid electric vehicle's hood with aluminum sourced from the four different U.S. production regions detailed in Chapter 1. We assess high strength steel (HSS) in addition as it is commonly considered by automakers when lightweighting vehicle components. We also account for charging the vehicle's battery from ten different U.S. vehicle charge locations. We focused on production region and charge location to highlight the impact of a heterogeneous U.S. electrical supply system in which GHG emissions can vary greatly depending on location. We also accounted for the impact of which methods are used to assign consumed electricity to generating resources. For electricity-intensive activities like primary aluminum production and plug-in hybrid electric vehicle operation, these variations should be accounted for when assessing environmental performance. We found that production location strongly influenced potential lifetime GHG emission savings of PHEVs equipped with aluminum over PHEVs equipped with conventional steel hoods. Depending on aluminum production location, lifetime GHG emissions for aluminum lightweighted PHEVs ranged from 98.8 below to 65.3 kg CO_2-eq above the baseline, steel hood-equipped PHEV.

Introduction

The U.S. Environmental Protection Agency (EPA) and the Department of Transportation's National Highway Traffic Safety Administration (NHTSA) recently finalized greenhouse gas (GHG) emission standards for 2017-2025 model years of light-duty vehicles. These standards place a GHG emission cap on passenger cars of 143 grams per mile by 2025. This is equivalent to a fuel efficiency of 54.5 miles per gallon (mpg) (U.S. EPA 2012a).

To understand how vehicle manufacturers would most likely meet these new standards, the EPA commissioned multiple studies and reports from automotive consulting firms outlining GHG reduction potentials, costs, and adoption rates for a variety of automotive technologies. These studies found that plug-in hybrid electric drivetrains and lightweight vehicle construction materials, among other vehicle technologies, would likely be used to reduce vehicle GHG emissions. Plug-in hybrid electric vehicles (PHEVs) are ideal platforms for lightweight material use because lighter vehicles can use smaller, less expensive propulsion motors and batteries (U.S. EPA 2012a; Markel 2007; Bull 2011). This is especially important as the batteries used to propel electrified vehicles are extremely expensive (NRC 2010).

Aluminum is a popular light weight vehicle construction material featuring high strength and low density that can be used to construct components traditionally made out of steel to lower vehicle mass. While aluminum's mass saving properties enable lightweight vehicles to consume less fuel during operation, it is also an extremely energy and GHG intensive material to produce. For comparison, industry trade groups estimate that steel takes 27.5 MJ/kg steel to produce with GHG emissions of 2.5 kg CO2-eq per kg while aluminum requires 56.4 MJ/kg aluminum ingot and produces 10.5 kg CO2-eq per kg (World Steel Organization 2011; IAI 2007). It is therefore necessary to analyze whether or not the GHG emissions reductions due vehicle lightweighting are negated by the GHG emissions created during the production of energy intensive lightweight construction materials.

To explore this tradeoff, we created a scenario in which a production-like PHEV with 40 miles of all-electric range (PHEV-40) was lightweighted by replacing the vehicle's conventional steel hood with a lighter aluminum or high strength steel hood. We then used life cycle assessment methods to determine GHG emissions from the vehicle's material production, manufacturing, use, and end of life phases (International Organization for Standardization 2006).

Background and Methodology

To evaluate the impacts of decreasing PHEV-40 mass via aluminum lightweighting, we utilized the U.S. primary aluminum material production emission factors calculated in Chapter One to profile regional aluminum material production burdens, calculated the ratio at which aluminum and high strength steel could be substituted for conventional steel, and modeled the energy consumption and emissions of a production-like PHEV-40 with differing curb weights. For this case study, we focused on substituting the material of a single component, the vehicle's hood.

Model Structure

To inform the structure of our model, we reviewed previous life cycle GHG analyses of conventional, internal combustion engine (ICE) powered aluminum intensive vehicles (Stodolsky et al. 1995; Das 2000; Field, Kirchain, and Clark 2000; Gaines and Cuenca 2000; Ungureanu, Das, and Jawahir 2007; Cáceres 2009; Bertram, Buxmann, and Furrer 2009; H.-J. Kim et al. 2010; Stasinopoulos et al. 2011). All but the most recent studies analyzed American family-sized sedans produced during the 1990's with all-steel body-in-whites (an industry term for the vehicle's inner body structure to which the doors, hood, truck, and windows are secured), mass reduction scenarios between 10-30%, and linear fuel efficiency to mass relations. Advanced powertrains were not explored.

Many studies presented the trade-off between material production and use-phase emissions as a "payback period", the point in the vehicle's lifetime at which the emissions penalty from increased production emissions was overcome by reductions in use phase emissions. Others provided the net amount of GHG emissions emitted above or below a baseline scenario based on lifetime vehicle miles travelled. We adopt the latter methodology and compare material production GHG burdens to use-phase GHG savings for a baseline PHEV-40 with a conventional steel hood to a PHEV-40 with a lightweight aluminum or high-strength steel hood. All other material production, manufacturing, and end-of-life emissions from vehicle component construction other than the hood were ignored as we focused on only this single component. Secondary mass savings were not considered as decreases in total vehicle mass were less than one percent. However, in scenarios where larger reductions in vehicle mass are possible, secondary mass savings should be considered (Alonso et al. 2012). This framework is represented mathematically in equation *(7)* and is used to underpin our study.

$$\Delta GHG = \left(E_{MP,Total} + E_{UP,Total}\right)_{Basline} - \left(E_{MP,Total} + E_{UP,Total}\right)_{Light\,Weight} \qquad (7)$$

where ΔGHG is the difference in lifetime GHG emissions between the PHEV-40 with the baseline configuration of a conventional steel hood and the PHEV-40 with a hood of either aluminum or high strength steel in kg CO_2-eq, $E_{MP,Total}$ is the total material production and manufacturing emissions in kg CO_2-eq for the vehicle hood, and $E_{UP,Total}$ is the total use phase vehicle emissions in kg CO_2-eq.

Material Production

We quantified U.S. primary aluminum ingot production GHG emissions at the smelter level in in Chapter One. When using a national, production weighted average emission factor and assuming consumed electricity emissions came from the Nested Average electricity allocation protocol, we calculated an emission factor of 19.04 kg CO_2-eq per kg aluminum ingot for U.S. primary aluminum production. We also quantified primary aluminum production emissions factor differences based on how electricity was allocated to generation resources, and the variations between four production regions. These regions included the Pacific Northwest, the Midwest, the Mid-Atlantic, and the Southeast.

U.S. secondary aluminum, and primary and secondary steel material production GHG emission factors are from GREET2 2012 (Wang, Burnham, and Wu 2012, 2). We used national average emission factors for secondary aluminum and primary and secondary steel because their energy requirements are much lower and suggest a lesser impact from variation. We assumed high strength steel had the same material production emissions as conventional steel based on discussions with industry experts.

Material Substitution

To determine the amount of aluminum required to replace a steel component while still providing equal functionality (strength, resistance to dents etc.), we used both the material index method and data from previous studies (Ashby 2011; Mcguire 2003; Wohlecker et al. 2006; Bertram, Buxmann, and Furrer 2009; Lotus Engineering Inc. 2010; European Aluminium Association 2011; Singh 2012). We calculated the material index for common automotive aluminum and steel alloys to explore the range of potential material substitution ratios available. We assumed that the material loading would resemble that of a light, stiff panel. From these calculated material indices, we established material substitution ratios by calculating the ratio of their material indices. For automotive grade aluminum and conventional steel, we calculated average material indices of 0.747 and 1.52. This yielded a material substitution ratio of 2.04, meaning that 2.04 kilograms (kg) of steel could be replaced with 1 kg of aluminum. Material properties and indices for all materials considered are available in the Appendix.

We acknowledge that material indices do not account for the intricacies of highly engineered automotive components that must meet demanding durability targets and safety regulations. To further account for these considerations, we looked to real-world examples from literature where vehicle hoods of differing materials were actually engineered and produced. Previous studies, detailed in the Appendix, indicated that between 1.52 and 2.69 kg of steel could be replaced with 1 kg of aluminum, between 1.13 and 1.30 kg of conventional steel could be replaced with 1 kg of high strength steel, and between 1.46 and 2.03 kg of high strength steel could be replaced with 1 kg aluminum. We used the average substitution ratios for each category and applied them to the actual weight of a production PHEV-40's aluminum hood. Based on a known aluminum hood weight of 7.67 kg, we calculated a high strength steel hood weight of 13.38 kg, and a conventional steel hood weight of 15.90 kg (Suburban Chevrolet of Ann Arbor 2012).

Manufacturing

We utilized GREET2 2012 scrap rates and GHG emission factors for rolling and stamping primary and secondary aluminum and steel into vehicle hoods (Wang, Burnham, and Wu 2012, 2). Our assumed manufacturing processes for primary and secondary aluminum and steel vehicle hoods are shown below in Table 2.

Table 2. Manufacturing processes modeled for primary and secondary steel and aluminum vehicle hoods.

Primary Aluminum	**Secondary Aluminum**	**Primary Steel**	**Secondary Steel**
Aluminum Prod.	*Scrap Collection*	*Steel Prod.*	*Scrap Collection*
Hot Rolling	*Casting*	*Hot Rolling*	*Casting*
Cold Rolling	*Hot Rolling*	*Skin Mill*	*Hot Rolling*
Stamping	*Cold Rolling*	*Cold Rolling*	*Skin Mill*
	Stamping	*Galvanizing*	*Cold Rolling*
		Stamping	*Galvanizing*
			Stamping

Based on the manufacturing processes outlined above, we calculated total material production and manufacturing emissions with equation *(8)*,

$$E_{MP,Total} = m\left[W_P \sum \left(EF_{p,i} \times SF_{p,i}\right)_{Primary} + (1 - W_P) \sum \left(EF_{p,i} \times SF_{p,i}\right)_{Secondary}\right] \quad (8)$$

where $E_{MP,Total}$ is the total emissions for material production and manufacturing process in kg CO_2-eq, m is the mass of the hood, W_P is the percentage of primary material in the vehicle hood, $EF_{p,i}$ is the emission factor for production process p at stage i in kg CO_2-eq per kg stage i aluminum , and $SF_{p,i}$ is the scrap factor for production process p at stage i in kg stage i aluminum per kg final stage aluminum. Production processes, p, represent the collection steps required to make primary and secondary aluminum and steel hoods. Stages, i, represent each manufacturing step contained within a production process (i.e. hot rolling or stamping).

Vehicle Use – Vehicle Characterization

We surveyed five production or near-production PHEVs to inform our simulated PHEV-40's powertrain architecture, component size, and vehicle characteristics. In our assessment of the Toyota Prius Plug-in, Ford C-MAX Energi, Ford Fusion Energi, Honda Accord Plug-in and Chevrolet Volt, we identified commonalities including the use of a split series / parallel hybrid powertrain architecture and Atkinson cycle engines (with the exception of the Chevrolet Volt). Vehicle mode control and all-electric propulsion range differed (Uehara et al. 2012; Burress et al. 2011; Rask et al. 2010; Toyota 2012; Rahman et al. 2011; Chevrolet 2012; Edmonds; Ford Motor Company 2012c; Ford Motor Company 2012b; Ford Motor Company 2012a; Honda Motor Company 2012). The final characterization of our production-like PHEV-40 borrowed heavily from the specifications of the Chevrolet Volt and is presented in the Appendix. Finally, we found the utility factor (UF), or the ratio of total electric-driven miles to total driven miles, to be 0.635 from the Society of Automotive Engineer's J1711 PHEV fuel economy test standard (SAE 2010).

Vehicle Use- Simulation

We modeled our production-like PHEV-40 using Argonne National Laboratory's Autonomie vehicle simulation software package (Argonne National Laboratory and LMS 2013).

We created three different vehicle configurations, each reflecting the total curb weight of a PHEV with a conventional steel, high strength steel, or aluminum hood. We then simulated each vehicle configuration over the EPA's Urban Dynamometer Driving Schedule (UDDS) and Highway Fuel Economy Driving Schedule (HWFET) in both charge depleting (CD) and charge sustaining (CS) vehicle operating modes (U.S. EPA 2012b; Elgowainy et al. 2009; Shiau et al. 2009; Freyermuth, Fallas, and Rousseau 2007). CD mode assumes that the vehicle is operated in an electric-only propulsion mode and all energy is supplied from the vehicle's battery directly to the electric propulsion motor. CS mode assumes that the vehicle's electric battery is depleted and cannot be used for electric-only propulsion. In this mode, the vehicle performs as a conventional hybrid electric vehicle, using the internal combustion engine for primary propulsion with assistance from the electric motor only during acceleration from energy stored during regenerative braking. This creates 12 vehicle simulation scenarios, summarized below in Table 3.

Table 3. Vehicle simulation scenarios based on hood type, operating mode, and driving type.

Hood Type	**Operating Mode and Driving Type**	
Aluminum	CD – UDDS	CS – UDDS
	CD – HWFET	CS – HWFET
High Strength Steel	CD – UDDS	CS – UDDS
	CD – HWFET	CS – HWFET
Conventional Steel	CD – UDDS	CS – UDDS
	CD – HWFET	CS – HWFET

We assumed all energy consumed while the vehicle was in CD mode was electrical energy stored in the vehicle's battery. All energy consumed during CS mode was assumed to come from the gasoline engine. We acknowledge that this is a simplification from more advanced blended mode operation that many PHEVs may utilize. However, advanced control strategies are proprietary to vehicle manufacturers and difficult to discern without advanced testing (N. Kim, Rousseau, and Rask 2012). While the use of different control strategies will influence overall PHEV efficiency, the development of a production-like blended control algorithm was beyond the scope of this study.

Vehicle Use – Energy Consumption and Emissions

The Autonomie simulations provided energy consumption data for each of the 12 driving scenarios described above. Using the U.S. EPA's assumed 55% to 45% city to highway driving ratio, we used equation *(9)* to calculate the average, combined energy consumption for CS and CD driving modes,

$$C_{i,j,Comb} = \frac{1}{\frac{0.55}{C_{i,j,UDDS}} + \frac{0.45}{C_{i,j,HWFET}}} \qquad (9)$$

where i is the vehicle hood material type (aluminum, high strength steel, or conventional steel), j is the vehicle operating mode (either CD or CS), $C_{i,j,Comb}$ is the combined energy consumption in Wh per mile (CD) or gallons of gasoline per mile (CS), $C_{i,j,UDDS}$ is the energy consumed during urban driving in Wh per mile (CD) or gallons of gasoline per mile (CS), and $C_{i,j,HWFET}$ is the energy consumed during highway driving in Wh per mile (CD) or gallons per mile (CS) (U.S. EPA 2013). We then used the calculated combined energy consumption values from CD and CS operating modes, along with the UF, to calculate vehicle emissions. This is detailed in equation *(10)*,

$$E_{UP,Total} = \left(UF \times C_{CD,Comb} \times EF_k + (1 - UF) \times C_{CS,Comb} \times EF_{Gasoline}\right) \times VMT_{Total} \qquad (10)$$

where $E_{UP,Total}$ is the total use phase GHG emissions of the vehicle in kg CO_2-eq, UF is the utility factor, $C_{CD,Comb}$ is the combined energy consumption of the vehicle in charge depleting mode in Wh per mile, EF_k is the full fuel-cycle emission factor of the consumed electricity in kg

CO_2-eq per Wh from North American Electric Reliability Corporation (NERC) region k, $C_{CS,Comb}$ is the combined energy consumption of the vehicle in charge sustaining mode in gallons of gasoline per mile, $EF_{Gasoline}$ is the full fuel-cycle emission factor of gasoline combustion in kg CO_2-eq per gallon of gasoline, and VMT_{Total} is the total vehicle miles traveled over the vehicle's lifetime in miles. Emission factors were assigned to gasoline combustion and electricity generation based on GREET2 2012 and the U.S. EPA's Emission and Generating Resource Integrated Database (eGRID) (Wang, Burnham, and Wu 2012, 2; U.S. EPA 2012c).

End of Life

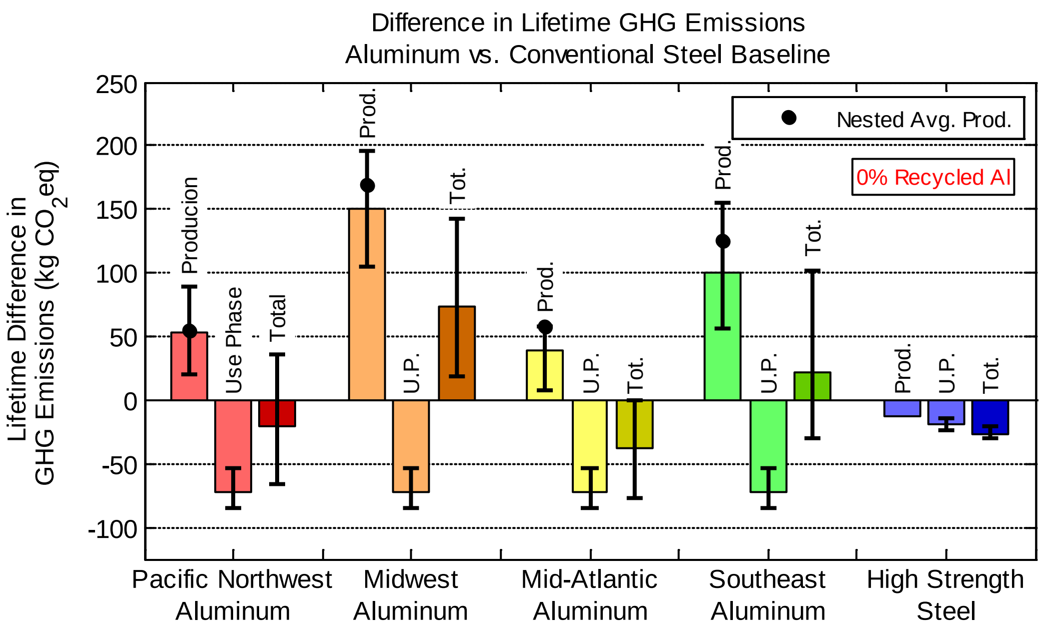
We followed the recycled content method when allocating primary and secondary material production burdens to the vehicle hoods (Johnson, McMillan, and Keoleian 2013). We assumed all other end-of-life considerations were identical between vehicles and are therefore ignored. Additional end of life allocation methods are explored later in the Sensitivity Analysis.

Sources of Variation

Our model also accounted for changes in electricity allocation protocol for primary aluminum production, changes in the amount of recycled aluminum content in aluminum vehicle hoods, and differing locations of vehicle battery charging during the vehicles use phase. In Chapter 1, we found changes in electricity allocation protocol to have a strong influence on the emission factors calculated for different primary aluminum production locations. As these protocols have yet to be standardized, we included the full range of potential production emission factors based on changes in allocation protocol. Variations in the amount of recycled content in and vehicle charge location were also included to represent the full range of potential lifetime GHG emissions based on potential construction methods and vehicle production and use locations.

Results and Discussion

Figure 9 presents differences in lifetime GHG emissions between the baseline PHEV-40 equipped with a conventional steel hood and the lightweighted PHEV-40s with aluminum or high strength steel hoods. Aluminum lightweighted PHEV-40s are further broken down by production location to examine the impacts of regional material sourcing. Error bars represent the uncertainty associated with material production electricity emissions allocation and vehicle charge location, topics explored in detail in Chapter One. We also calculated lifetime GHG emissions for 0, 15, and 30% recycled content. The black dots represent the nested average production values, which we found to best reflect grid boundaries and inter-regional trade.



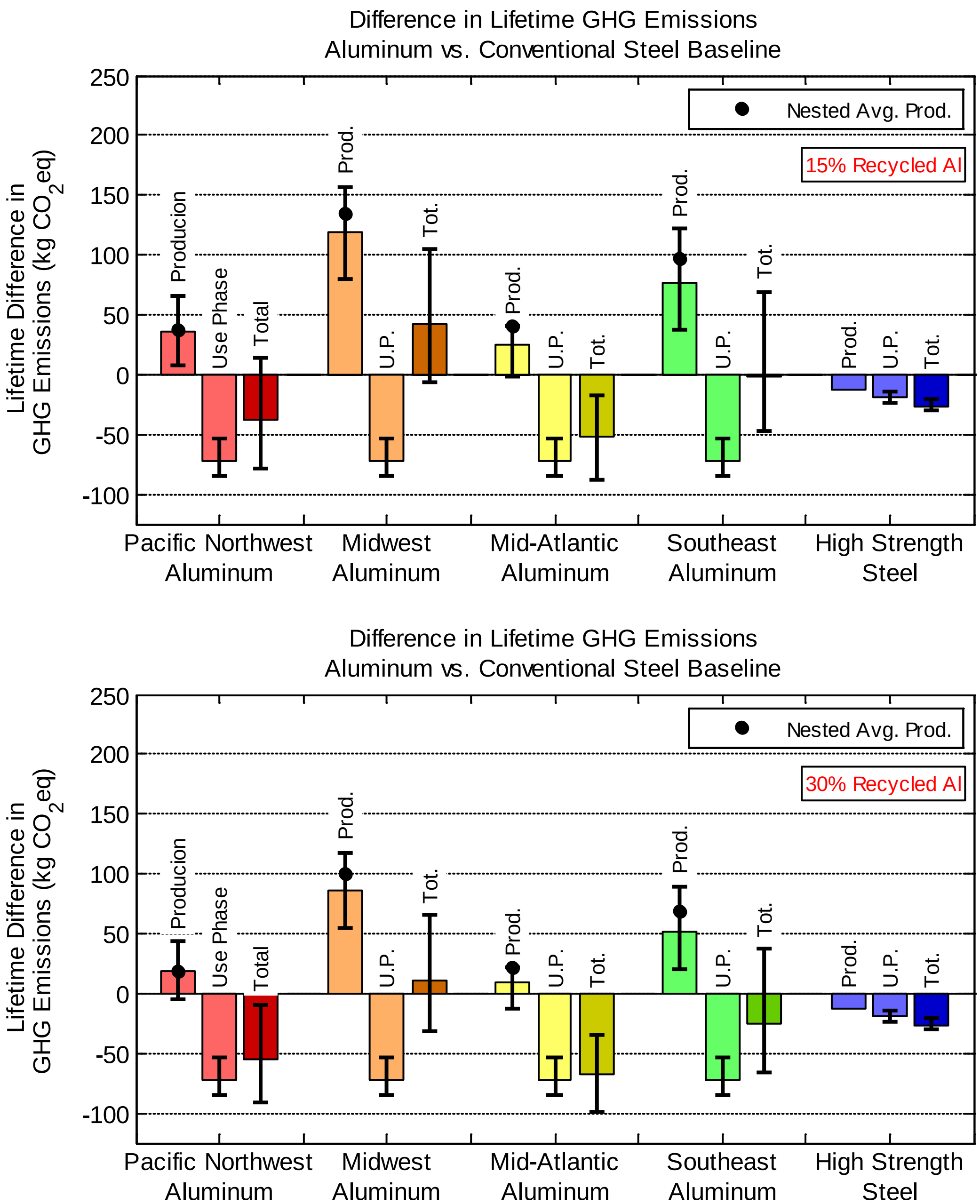


Figure 9. Differences in lifetime GHG emissions between the baseline PHEV-40 with a conventional steel hood and a lightweighted PHEV-40 with hoods made from aluminum (red, orange, yellow, or green bars) or high strength steel (blue bars). Material production, use-phase, and combined emissions are shown for aluminum manufacturing in the Pacific Northwest, Midwest, Mid-Atlantic, and Southeast production regions. Production (prod.) error bars display the spread associated with consumed electricity emissions allocation and error bars shown for the use phase (U.P.) represent the spread in carbon intensity of vehicle charge locations. Error bars in the combined phase show the combined spread of production and use phase emissions. Black dots represent nested average production emissions values (Nested Avg. Prod.) and assign all consumed electricity emissions using the nested average electricity allocation protocol developed in Chapter One.

Aluminum production location strongly influenced lifetime GHG emissions savings over the conventional steel. Aluminum produced in the Mid-Atlantic had the lowest material production burdens and we calculated lifetime GHG emissions of between 0.02 kg CO_2-eq higher and 77.6 kg CO_2-eq lower than the lifetime GHG emissions of baseline vehicle when only virgin, Mid-Atlantic sourced aluminum was used. Aluminum produced in the Midwest did not show any lifetime GHG emissions savings when the vehicle hood was composed of 100% primary aluminum, creating between 19.3 and 142.9 kg CO_2-eq more than the baseline vehicle. All aluminum lightweighted PHEV-40s out-perform the conventional and high strength steel equipped PHEVs during vehicle use but, when using 100% primary aluminum, only provided lifetime GHG emissions savings under the least carbon intensive production and vehicle charging scenarios.

When the recycled aluminum content was increased to 15%, Pacific Northwest and Mid-Atlantic aluminum sourced aluminum offered lifetime GHG reductions in almost all production scenarios and charge locations. Midwestern and Southeastern aluminum potentially offered savings but, under many circumstances, led to increased GHG emissions. When 30% recycled aluminum content was used, all Pacific Northwest and Mid-Atlantic aluminum sourcing scenarios offered lifetime GHG emissions savings over the baseline vehicle. Midwestern and Southeastern sourced aluminum offered more scenarios where GHG savings were possible but still exhibited increased emissions for many production scenarios and charge locations. The maximum GHG savings achieved in this scenario was 98.8 kg CO_2-eq using 70% Mid-Atlantic sourced primary aluminum and 30% recycled aluminum. This compares with the findings of Bertram et al who calculated a ~200 kg CO_2-eq savings when replacing the steel hood of conventionally powered large family sedan with an aluminum hood (Bertram, Buxmann, and Furrer 2009). Bertram's study used a different vehicle powertrain and class but the order of magnitudes are similar, which is to be expected despite the size and powertrain differences.

The large variation displayed during production indicates the heavy influence of electricity allocation protocol selection on primary aluminum production. These protocols, methodologies used to assign electricity generation emissions to electricity consumers, can alter primary aluminum production emission factors dramatically. When we explored this relationship extensively in the previous chapter, we found differences of 100 – 300% when any of the four commonly used electricity allocation protocols were applied to primary aluminum smelters. When use phase savings are small, the choice in electricity allocation protocol can mean the difference between realizing lifetime GHG savings or increasing lifetime GHG burdens compared to the baseline PHEV.

To eliminate the large spread in aluminum hood production emissions, we constructed a scenario in which the electricity allocation protocol and amount of recycled aluminum were held constant. We selected the "nested-average" electricity allocation protocol, a trade-weighted average of the electricity generated within the power control area (PCA) that each aluminum smelter contracts with and the surrounding NERC region in which the PCA is nested inside of. Recycled aluminum content was held constant at 30% based on discussions with automotive industry experts. Aluminum material production values calculated during this scenario were indicated in Figure 9 as recommended production values but are reproduced in Figure 10 to emphasize the much smaller variation in combined lifetime GHG savings.

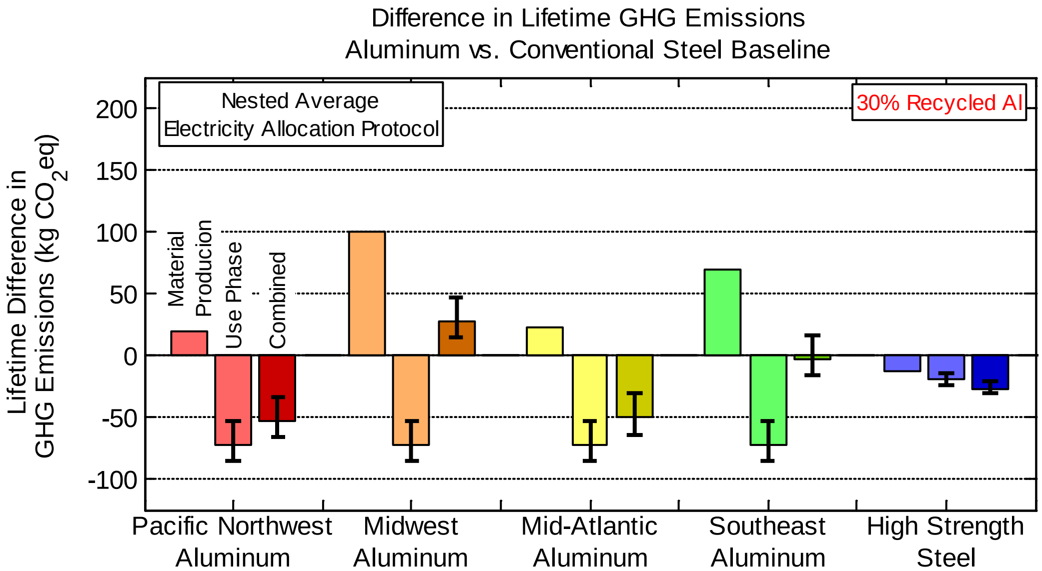


Figure 10. Differences in lifetime GHG emissions between the baseline PHEV-40 with a conventional steel hood and a lightweighted PHEV-40 with hoods made from aluminum and high strength steel. Variations in material production are eliminated by allocating all electricity to a trade-weighted average of electricity generators in the power control area supplying electricity to the smelter and the surrounding North American Electricity Reliability Corporation grid regions.

When aluminum is sourced from the Pacific Northwest and Mid-Atlantic, we found reductions in lifetime GHG emissions of 33.7 to 66.5 and 31.0 to 63.8 kg CO_2-eq, respectively. Sourcing aluminum from the Midwest resulted in increases in lifetime GHG emissions between 14.0 and 46.7 kg CO_2-eq, the maximum range for this scenario. Aluminum sourced from carbon intensive aluminum production locations like the Midwest and Southeast pose a challenge as production emissions must be offset by larger use-phase savings than would be needed for less carbon intensive production locations.

We identified several important trends to help extend our results, where a small component was lightweighted on a single vehicle, to a larger fleet of PHEVs. First, 76.8% of U.S. primary aluminum was produced in the carbon-intensive Midwest and Southeast regions in 2010 (Harbor Aluminum Intelligence Unit 2011). This suggests that industry average material production burdens are more similar to those seen in the Midwest and Southeast. This stands in contrast the location of most electrified vehicle sales, where top Designated Market Areas include San Francisco, San Diego, and Los Angeles, California, and Seattle/Tacoma, Washington, cities with below average grid emissions (Libby 2012; U.S. EPA 2012c). Therefore, decision makers should view the average, fleet-wide impact of aluminum lightweighting as coming from more carbon intensive production locations but less carbon intensive vehicle charge locations. However, future electricity generation emissions are likely to decline as future increases in U.S. electricity generation are projected to come from natural gas and renewables, further decreasing the impact of both production and vehicle use (U.S. EIA 2013).

Sensitivity Analysis

We assessed the sensitivity of our model to changes in lifetime vehicle miles traveled (VMT), percentage of vehicle mass reduced (LW%), and recycling allocation methods. Changes in lifetime VMT and LW% were analyzed using the nested average electricity allocation protocol and with the recycled aluminum content held constant at 30%. Increases in lifetime vehicle miles traveled, seen in Figure 11, decreased lifetime GHG emissions of the lightweighted PHEVs when compared to the baseline, conventional steel-equipped PHEV. Pacific Northwest and Mid-Atlantic sourced aluminum resulted in lifetime GHG emissions savings in all scenarios. Southeastern sourced aluminum began to see lifetime GHG savings when VMT was higher than 151,670 while Midwestern sourced aluminum required over 219,365 miles of vehicle travel before use phase savings outweighed production burdens. For every 10,000 mile increase in lifetime travel, vehicles equipped with aluminum hoods saw lifetime GHG emissions decrease by 4.54 kg CO_2-eq when compared to vehicles outfitted with conventional steel hoods. High strength steel equipped vehicles saw lifetime GHG savings of 1.24 kg CO2-eq compared to conventional steel vehicles for every 10,000 mile lifetime travel increase.

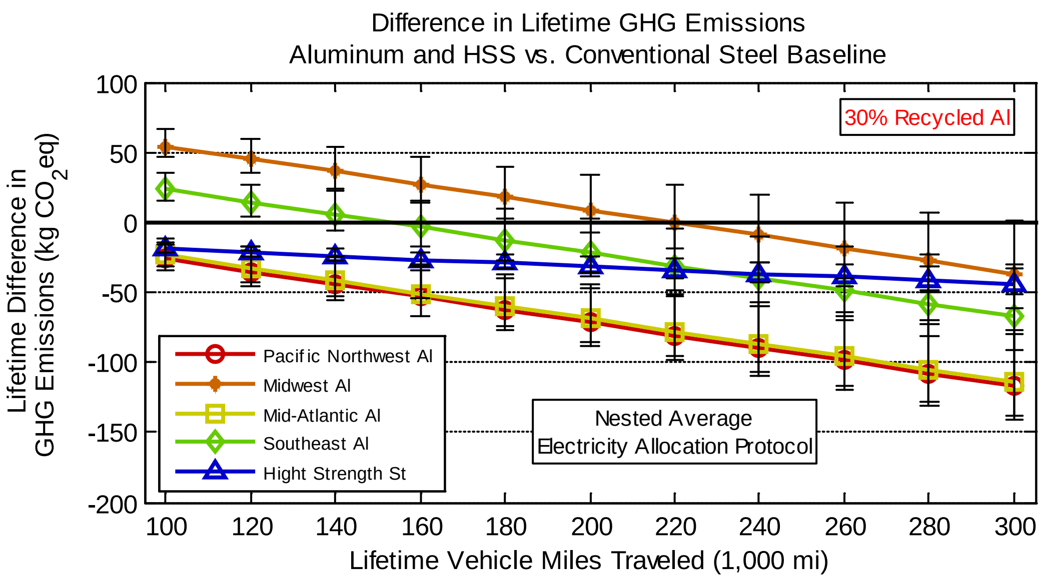


Figure 11. Difference in lifetime GHG emissions between the baseline PHEV-40 equipped with a conventional steel hood and a lightweighted PHEV-40 equipped with an aluminum or high strength steel hood as a function of lifetime VMT. Aluminum material production locations include the Pacific Northwest, Midwest, Mid-Atlantic, and Southeast. Error bars represent the spread in vehicle charge location carbon intensity.

When we increased the amount of aluminum or high strength steel incorporated in the vehicle design as a percentage of curb weight, seen in Figure 12, lifetime GHG emissions for all lightweighted vehicles fell with respect to those of their conventional steel counterparts with one exception. When Midwestern sourced aluminum use was increased, use phase GHG emissions savings failed to outweigh production burdens. Aluminum lightweighted PHEV-40 average lifetime GHG emissions actually increased by 210.6 kg CO_2-eq above the baseline vehicle for every 5% decrease in vehicle mass brought about by Midwestern sourced aluminum. Every 5% decrease in vehicle mass caused by incorporating Mid-Atlantic and Pacific Northwest aluminum resulted in average savings of 605.7 and 634.6 kg CO_2-eq above the baseline vehicle, respectively. Southeastern aluminum use saved an average of 112.2 kg CO_2-eq for every 5% decrease in mass. High strength steel displayed savings of 1066.1 kg CO2-eq for every 5% decrease in vehicle mass. While this compares favorably with aluminum, a 5% decrease in vehicle mass corresponds with replacing 548.4 kg of conventional steel with 462.64 kg of high strength steel, a scenario that would require extensive high strength steel incorporation throughout the vehicle. A 5% mass reduction done using aluminum would only require replacing 165.9 kg of conventional steel with 80.1 kg of aluminum. Large differences in total mass replacement between scenarios make their comparison difficult. Additionally, no secondary mass savings were considered in our analysis but are likely to occur in situations where vehicle mass reductions are large (Alonso et al. 2012). This would have the effect of increasing use-phase emissions savings and in turn, decreasing the slope of the lines presented in Figure 12.

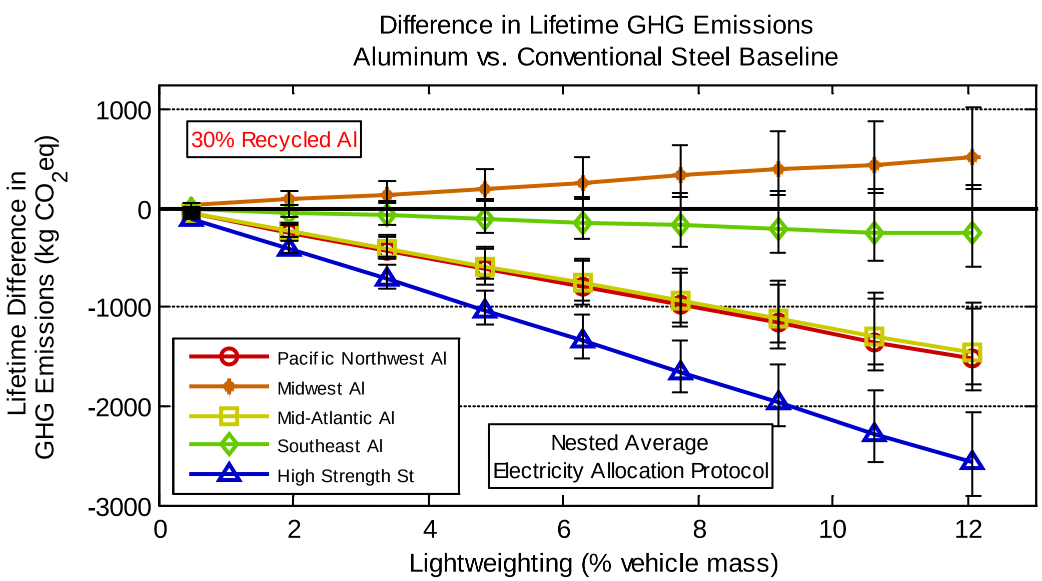


Figure 12. Differences in lifetime GHG emissions between a conventional steel PHEV-40 and aluminum and high strength steel lightweighted PHEV-40s as a function of percent curb weight reduction. Aluminum material production locations include the Pacific Northwest, Midwest, Mid-Atlantic, and Southeast. Error bars represent the spread in vehicle charge location carbon intensity.

Two different end of life recycling strategies were explored: the recycled content method, and the end of life recycling method (Johnson, McMillan, and Keoleian 2013). The recycled content method was used throughout this report and assigned production emissions based on the amount of primary and secondary aluminum and steel content that were incorporated into the hood at the time of production. We assumed 70% primary and 30% secondary aluminum and 73.6% primary and 26.4% secondary steel use based on literature and conversations with automotive industry experts (Wang, Burnham, and Wu 2012). The end of life recycling method assumes 100% primary material was used during production but offers a credit for all materials recovered and recycled. We assumed 86.5% of aluminum and 89.3% of steel was recovered and recycled during hood production, use, and end-of-life (Wang, Burnham, and Wu 2012; World Steel Organization 2012; The Aluminum Association 2009; Pomykala et al. 2007). Results for production weighted average U.S. primary aluminum can be seen in Figure 13. We again allocated all electricity emissions using the nested average method previously detailed in Chapter One.

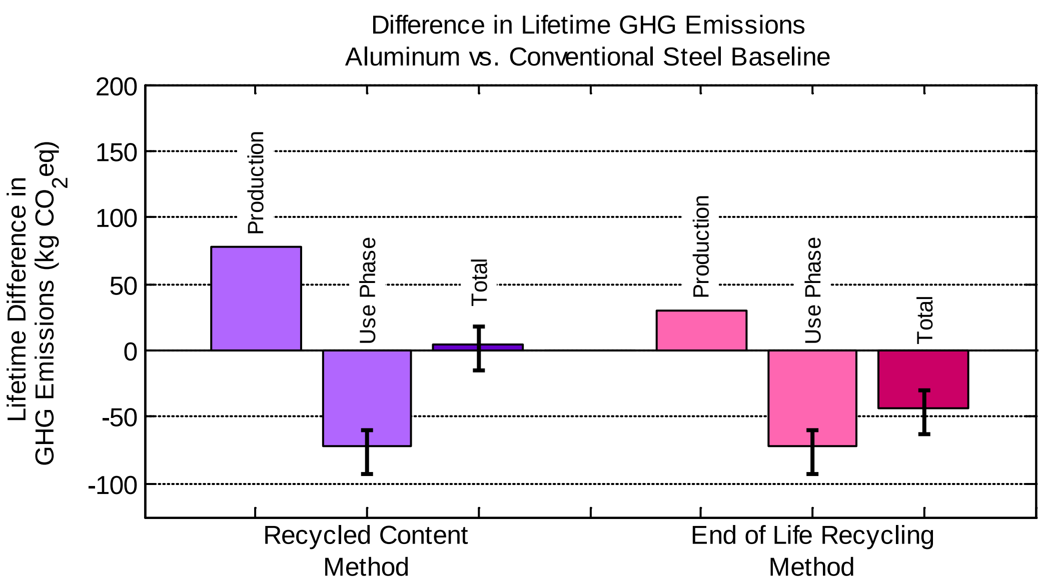


Figure 13. Differences in lifetime GHG emissions between a conventional steel PHEV-40 and an aluminum lightweighted PHEV-40 for recycled content and end of life recycling allocation methods. A 70% primary, 30% secondary aluminum blend was assumed for the recycled content method. A 86.5% and 89.3% material recovery rate were assumed for the end of life recycling method.

Using the recycled content method, lifetime GHG emissions from an aluminum lightweighted PHEV were between 24.69 above and 8.12 kg CO_2-eq below baseline PHEV lifetime GHG emissions depending on vehicle charge location. The end of life recycling method decreased production emissions from 77.54 to 29.55 kg CO_2-eq above the baseline, conventional steel production scenario. This led to total lifetime GHG savings of between 23.30 and 56.11 kg CO_2-eq when compared to the baseline. While both end of life scenarios could potentially be appropriate, we used the recycled content approach for this study because of the uncertainty associated with wrought aluminum recycling processes and infrastructure.

Conclusion

Our results indicate that aluminum production carbon intensity has the potential to negate the lifetime GHG emissions savings of an aluminum lightweighted PHEV. Midwestern aluminum, representing 64.9% of all 2010 U.S. primary aluminum production, is so carbon intensive that lifetime GHG emissions savings are not realized under VMT scenarios of less than 219,000 miles. This was found even as the percentage of lightweight material use was increased, meaning that use phase GHG emissions savings never outpaced production burdens under baseline VMT scenarios. Aluminum production carbon intensity was also strongly influenced by electricity allocation protocols and end of life considerations. With the knowledge that production location, electricity emissions allocation, and end of life considerations all have the potential to turn a net savings in GHG emissions into a net increase, we encourage decision makers to fully map and understand the lifecycle of their lightweight components. Awareness of the hot spots we have identified can help influence decisions on where aluminum is procured and ultimately lead to the net savings in GHG emissions expected when vehicle components are lightweighted.

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Appendix

Table A 1. Smelter peak power consumption and contract utility information sources.

Plant	Peak Power Consumption	Contract Utility	Source
Warrick	540 MW	On-site	(Alcoa 2003; Todd 2008)
Intalco	457 MW	Bonneville Power Administration	(Beyers, O'Corroll, and Sorensen 2006)
Goose Creek	418 MW	Santee Cooper	(Pulley 2006)
Wenatchee	343.89 MW*	Chelan County Public Utility District	(Beyers, O'Corroll, and Sorensen 2006; Courtney 2011)
Messena West	252.35 MW**	New York Power Authority	(Caufield 2007)
Hawesville	482 MW	Big Rivers Electric Corporation	(Pidherny and Espino 2011)
New Madrid	489.18 MW***	Ameren Missouri	(Ailor 2011; Tomich 2011)
Hannibal	540 MW	American Electric Power – Ohio	(Pfeifer 2011)
Sebree	320 MW	Big Rivers Electric Corporation	(Matyi 2012; SebreeWorks)

**Peak was reported in source as 428 MW for a capacity of 229,000 mt/year. Scaled to match current USGS reported nameplate capacity of 184,000 mt/year for 2010.*

***Reported as combined consumption between Messena East and Messena West facilities of 495 MW. Scaled by USGS reported nameplate production at each plant.*

****Peak was reported as 465 MW in 2003 when capacity was reported as 250,000 mt/year by USGS. Scaled for 2010 production level of 263,000 mt/year.*

Table A 2. 2010 U.S. primary aluminum smelter characteristics (Bray, Wallace, and Miller 2011; Harbor Aluminum Intelligence Unit 2011; U.S. EPA 2008).

Smelter	Location	2010 Nameplate Capacity (mt Al.)	2010 Actual Output (mt Al.)	2010 Process Emissions (mt CO_2 eq.)	Peak Power Consumption (MW)	Power Control Area	State	NERC Sub-Region	NERC Region
Warrick	Evansville, IN	309,000	258,000	441,073	540	On-site	IN	RFCW	RFC
Intalco	Ferndale, WA	279,000	169,000	930,960	457	BPA	WA	NWPP	WECC
Alumax	Mount Holly, SC	229,000	205,000	446,760	418	SCPSA	SC	SRVC	SERC
Wenatchee	Wenatchee, WA	184,000	100,000	206,989	343	CCPUD	WA	NWPP	WECC
Massena West	Massena, NY	130,000	129,000	316,344	252	NYPA	NY	NYUP	NPCC
Hawesville	Hawesville, KY	244,000	227,000	491,291	482	BREC	KY	SRTV	SERC
New Madrid	New Madrid, MO	263,000	242,000	579,879	489	Ameren	MO	SRMW	SERC
Hannibal	Hannibal, OH	265,000	199,000	596,315	540	AEP-Ohio	OH	RFCW	RFC
Sebree	Sebree, KY	196,000	193,000	553,416	325	BREC	KY	SRTV	SERC

Table A 3. Power control area and on-site import and export characteristics. Unless otherwise noted, all values from U.S. Federal Energy Regulatory Commission Form 714 or 1 for 2010.

PCA / On-Site Resource	Annual Net Generation (MWh)	Annual Imports (MWh)	Annual Exports (MWh)	Net Transaction (MWh)	Generated to Exported Ratio	PCA Weighting Factor (W_{pca})
Warrick (Alcoa Generating Corporation)	4,524,811	1,205,966	1,1760,043	-29,923	-0.01	0.99
Bonneville Power Administration	102,381,882	22,965,037	73,469,606	50,504,569	0.49	1.00
Santee Cooper (SCPSA)	25,404,460	13,410,034	10,325,622	-3,084,412	-0.12	0.88
Chelan County Public Utility District	7,635,736	790,782	6,762,292	55,971,510	0.78	1.00
New York Power Authority*	24,400,000	-	-	-12,400,000	-0.51	0.49
Big Rivers Electric Corporation	11,340,083	5,209,224	3,784,150	-1,425,074	-0.13	0.87
Ameren Missouri (Union Electric)	48,046,798	2,089,483	9,795,367	7,705,884	0.16	1.00
American Electric Power – Ohio**	61,289,647	8,848,869	11,482,579	2,633,710	0.04	1.00

*New York Power Authority's annual generation and imported electricity data taken from their annual report (NYPA 2011).

*American Electric Power – Ohio imported and exported electricity data taken from Ohio Power, Wheeling Power, and Columbus Southern Power Company FERC Form 1 filings.

Table A 4. Material substitution material indices for popular aluminum and steel alloys used in vehicle closures.

Material Type	Tensile Strength (MPa)	Yield Strength (MPa)	Modulus of Elasticity (GPa)	Density (g/cm^3)	Material Index ($\frac{E^{1/3}}{\rho}$)	Source
Cold Rolled Carbon Steel (DS Type B)	317	193	200	7.87	**0.743**	(MatWeb 2012a)
Bake Hardened Steel (BH 210)	320	210	200	7.87	**0.743**	(MatWeb 2012b)
Electrogalvanized Dual Phase Steel (DI-Form T500)	300	547	210	7.87	**0.755**	(ArcelorMittal 2009; MatWeb 2012c)
				Average	**0.747**	
2000 Series Aluminum (2036-T4)	340	195	70.3	2.75	**1.50**	(ASM International 2012)
5000 Series Aluminum (5182-0)	276	138	69.6	22.65	**1.55**	(ASM International 2012)
6000 Series Aluminum (6009-T4)	234	131	69	2.71	**1.51**	(ASM International 2012)
				Average	**1.52**	
			Average	**Ratio**	**2.04**	

Table A 5. Material substitution ratios calculated from previous studies. Conventional steel (CSt), high strength steel (HSS) and aluminum (Al) hood weights and substitution ratios are compiled from six different sources.

Application	CSt Hood (kg)	HSS Hood (kg)	Al Hood (kg)	CSt : Al Ratio	CSt : HSS Ratio	HSS : Al Ratio	Source
Luxury Sedan	26	-	17	1.52: 1	-	-	(Mcguire 2003)
Light Truck	30.6	23.5	16	1.91 : 1	1.30 : 1	1.46 : 1	(Mcguire 2003)
SUV (outer hood only)	7.1	-	3.3	2.17 : 1	-	-	(Mcguire 2003)
Small Sedan	12.6	-	8.1	1.56 : 1	-	-	(Wohlecker et al. 2006)
Large Family Car	-	17.5	10.1	-	-	1.73 : 1	(Bertram, Buxmann, and Furrer 2009)
Crossover (low)	16.8	-	6.3	2.69 : 1	-	-	(Lotus Engineering Inc. 2010)
Crossover (high)	16.8	-	7.2	2.33 : 1	-	-	(Lotus Engineering Inc. 2010)
Small Sedan	14	12.4	6.1	2.30 : 1	1.13 : 1	2.03 : 1	(European Aluminium Association 2011)
Midsize Sedan	14.7	-	7.0	2.1 : 1	-	-	(Singh 2012)
			Minimum Ratio	1.52 : 1	1.13 : 1	1.46 : 1	
			Maximum Ratio	2.69 : 1	1.30 : 1	2.03 : 1	
			Average Ratio	**2.07 : 1**	**1.22 : 1**	**1.75 : 1**	

Table A 6. Production-like PHEV-40 vehicle characteristics.

Architecture	Split Series/Parallel
Engine Size, Type (kW)	62, conventional
Motor/Generator 1 Size (kW)	110
Motor/Generator 2 Size (kW)	55
Battery Type	Lithium Ion
Battery Capacity (kWh)	16.53
Battery Maximum Power (kW)	115
Battery Weight (kg)	190
Assume Accessory Load (kW)	0.8
All electric range (mi, UDDS/HWFET)	40.32 / 39.31
Utility Factor	0.635
Curb Weight – Al hood (kg)	1715
Curb Weight – HSS Hood (kg)	1720.8
Curb weight- CS Hood (kg)	1723.2
Passenger and cargo mass (kg)	136
Coefficient of Drag (C_d)	0.28
Frontal Area (m^2)	2.42
Tire Radius (m)	0.334
Tire Rolling Resistance	0.0088
Lifetime Vehicle Mileage (mi)	160,000
Urban / Highway Split	55% / 45%