Comparative Environmental Life Cycle Assessment of Conventional and Electric Vehicles

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Supporting information is available on the *JIE* Web site

Summary

Electric vehicles (EVs) coupled with low-carbon electricity sources offer the potential for reducing greenhouse gas emissions and exposure to tailpipe emissions from personal transportation. In considering these benefits, it is important to address concerns of problem-shifting. In addition, while many studies have focused on the use phase in comparing transportation options, vehicle production is also significant when comparing conventional and EVs. We develop and provide a transparent life cycle inventory of conventional and electric vehicles and apply our inventory to assess conventional and EVs over a range of impact categories. We find that EVs powered by the present European electricity mix offer a 10% to 24% decrease in global warming potential (GWP) relative to conventional diesel or gasoline vehicles assuming lifetimes of 150,000 km. However, EVs exhibit the potential for significant increases in human toxicity, freshwater eco-toxicity, freshwater eutrophication, and metal depletion impacts, largely emanating from the vehicle supply chain. Results are sensitive to assumptions regarding electricity source, use phase energy consumption, vehicle lifetime, and battery replacement schedules. Because production impacts are more significant for EVs than conventional vehicles, assuming a vehicle lifetime of 200,000 km exaggerates the GWP benefits of EVs to 27% to 29% relative to gasoline vehicles or 17% to 20% relative to diesel. An assumption of 100,000 km decreases the benefit of EVs to 9% to 14% with respect to gasoline vehicles and results in impacts indistinguishable from those of a diesel vehicle. Improving the environmental profile of EVs requires engagement around reducing vehicle production supply chain impacts and promoting clean electricity sources in decision making regarding electricity infrastructure.

Introduction

Our global society is dependent on road transport, and development trends project substantial growth in road transport over the coming decades. According to a study commissioned by the World Business Council for Sustainable Development (2004), light-duty vehicle[1] ownership could increase from roughly 700 million to 2 billion over the period 2000–2050. Globally, light-duty vehicles account for approximately 10% of global energy use and greenhouse gas (GHG) emissions (Solomon et al. 2007). These patterns forecast a dramatic increase in gasoline and diesel demands, with associated energy security concerns as well as implications for climate change and urban air quality.

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Among available transport alternatives, electric vehicles (EVs) have reemerged as a strong candidate. The European Union (EU) and the United States, among others, have provided incentives, plans, and strategies, at different levels of ambition, for the introduction of EVs (European Commission 2010; Greater London Authority 2009; IEA 2009; U.S. Department of Energy 2011). One of the more ambitious targets is proposed by a consortium of the International Energy Agency (IEA) (2009) and eight countries (China, France, Germany, Japan, South Africa, Spain, Sweden, and the United States), which aims to reach a combined total of 20 million full and plug-in hybrid EVs by 2020. Meanwhile, battery-powered EVs are becoming an important component of automotive manufacturers' strategies. Both Mercedes and Ford have clear ambitions in this area (Daimler AG 2010b; Ford Motor Company 2011). The first generation of mass-produced EVs has just entered the market (e.g., the Mitsubishi i-MiEV, Nissan Leaf, Renault Kangoo, GM Volt, and Ford Electric Focus).

EVs offer advantages in terms of powertrain efficiency, maintenance requirements, and zero tailpipe emissions, the last of which contributes to reducing urban air pollution relative to conventional internal combustion engine vehicles (ICEVs) (Wang and Santini 1993). This has led to a general perception of EVs as an environmentally benign technology. The reality is more complex, requiring a more complete account of impacts throughout the vehicle's life cycle. Consistent comparisons between emerging technologies such as EVs and their conventional counterparts are necessary to support policy development, sound research, and investment decisions.

In an earlier stage of this research, we reviewed life cycle assessment (LCA) studies of EVs (Hawkins et al. 2012). For conventional ICEVs, although the use phase accounts for the majority of global warming potential (GWP) impact, vehicle production is not insignificant, contributing on the order of 10% to the life cycle GWP. When considering a suite of environmental impacts of ICEVs, the need for a full LCA including manufacturing is well documented (Daimler AG 2005, 2007a, 2007b, 2008a, 2008b). Accounting for production impacts is even more important when comparing technologies with significantly different powertrains such as ICEVs and EVs. In particular, the production of electronic equipment requires a variety of materials, which poses a challenge for recycling and raises concerns about toxicity (Johnson et al. 2007).

A few studies consider battery and/or EV production explicitly, at varied levels of detail and transparency. Samaras and Meisterling (2008) focus on energy and GWP, providing an inventory based primarily on energy consumption within life cycle stages. Burnham and colleagues (2006) provide a stylized representation of vehicle production, relying on material content to estimate GWP criteria, air pollution, and energy use to give a basis for comparing EVs with other technologies within the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model. Van den Bossche and colleagues (2006) and Matheys and colleagues (2008) perform a more complete assessment of traction batteries within the EU-sponsored Sustainable Batteries (SUBAT) project. Their results are generally presented as EcoIndicator points and are based on confidential inventories. Daimler AG (2009) presents results from a comparative study of a hybrid and a conventional version of the same car from a full LCA perspective. This is likely the most complete life cycle inventory (LCI) of an EV; however, it is for a hybrid rather than a full-battery EV. Zackrisson and colleagues (2010) provide a well-documented inventory for comparison of two prospective production processes for lithium iron phosphate ($LiFePO_4$) next-generation batteries. Notter and colleagues (2010) present one of the most transparent LCA studies of an EV based on a lithium manganese oxide ($LiMn_2O_4$) battery. Their inventory focuses on battery production and places these results in the context of the EV life cycle. Majeau-Bettez and colleagues (2011) provide another transparent inventory for production of nickel metal hydride (NiMH), lithium nickel cobalt manganese (LiNCM), and $LiFePO_4$ batteries designed to be adapted into a more complete study of the full EV life cycle.

The primary objective of this LCA is to provide an appropriate comparison of an EV and an ICEV over their entire life cycle. A second objective is to provide a transparent inventory that can be used for assessing other vehicle and fuel options. Results are presented for a suite of ten relevant environmental impact categories, including GWP, toxicity impacts, and metal depletion. To address uncertainty and the difficulty of predicting aspects of technological development, results of a sensitivity analysis with respect to key parameters are presented.

In order to understand the composition of a small ICEV and an EV, we found it necessary to create our inventory with more detail than can be readily obtained from present public inventories. This study thus contributes a transparent comparison of an ICEV and an EV to the publicly available literature. The material content of vehicle components and the processes used to produce them are estimated based on secondary data and well-reasoned assumptions. With respect to prior EV LCAs, our study offers significantly more resolution regarding the manufacture of vehicle components, full transparency, consideration of a range of battery technologies, and includes a broader array of environmental impacts. In this way, it provides a basis upon which the next generation of LCA studies of generic vehicles can be built and a context within which proprietary LCA studies can be placed.

Method and System Details

General Considerations, Goal, Scope, and Data

LCA involves compiling an inventory of the environmentally relevant flows associated with all processes involved in the production, use, and end of life of a product and translating this inventory into impacts of interest (Curran 1996; Guinée et al. 2002). The goal of this study is to provide a scoping-level comparative LCA of a conventional ICEV and a first-generation battery EV representative of a typical small European car, including all relevant processes and a cross section of relevant impacts.

An appropriate comparison of an EV and a conventional ICEV requires that the system boundary be set to include all relevant differences between the two alternatives. Our scope includes vehicle production, use, and end of life together with all relevant supply chains. To ensure the comparability of the EVs and ICEVs, we established a common generic vehicle glider (vehicle without a powertrain; see glider components in Table 1) and customized powertrains for gasoline, diesel, and EVs. The assumption of a common glider platform for multiple drivetrains seems reasonable considering industry signals regarding forthcoming generations of vehicles (Daimler AG 2010a). In the use phase we tracked electricity and fuel consumption, together with their full supply chains. Use phase energy requirements are based on the performance of the Mercedes A-series ICEV and the Nissan Leaf EV, vehicles of comparable size, mass, and power. Performing the analysis in this way guaranteed the comparability of our case vehicles during the production, use, and disposal phases of their lives, thereby isolating the core differences. For the end of life, we model treatment and disposal of the vehicle and batteries.

The functional unit is 1 kilometer (km)[2] driven under European average conditions. Our LCA is attributional and process based. The foreground LCI was compiled using secondary data sources. We put a premium on transparency and thereby sacrificed the additional detail associated with confidential, manufacturer-specific data. Detailed industry inventories and reports regarding materials, masses, and processes were used whenever these were publicly available, but we avoid the use of rolled-up LCIs. Different modeling assumptions of vehicle composition, efficiency, lifetime, and fuel use are assessed in a sensitivity analysis. Ecoinvent v2.2 (Ecoinvent Centre 2010) was used as a background dataset, and impact assessment was performed using the ReCiPe characterization method for midpoint indicators, from the hierarchical perspective (Goedkoop et al. 2009). Sensitivity analysis was performed to test the effect of modeling assumptions regarding vehicle composition, efficiency, lifetime, and fuel use.

Our inventory was compiled as a technical requirement and a stressor intensity matrix. The requirement matrix was built in a triangularized hierarchical manner, following Nakamura and colleagues (2008). Material and processing requirements were tracked in matrices for each vehicle component with columns representing subcomponents and rows representing production requirements based on original source data. A second matrix was then developed for each component to associate production requirements based on original source data to the closest matching Ecoinvent v2.2 processes (Ecoinvent Centre 2010). It was always possible to find a good match or an appropriate proxy such that we are confident that our results offer a decent scoping-level life cycle representation of material and process requirements. Further details on system definitions, component matrices, correspondence matrices, variables, and calculations are provided in supporting information S2 available on the Journal's Web site.

Table 1 Vehicle components

Category	*Component*	*ICEV*	*EV, $LiFePO_4$*	*EV, LiNCM*	*Data sources*
Glider	Body and doors	X	X	X	a–d
	Brakes	X	X	X	a, e–g
	Chassis	X	X	X	a, h
	Final assembly	X	X	X	h
	Interior and exterior	X	X	X	a, i
	Tires and wheels	X	X	X	a, h–k
ICEV	Engine	X			a, h, b, c
	Fluids	X			a, b, i, j
	Other powertrain	X			a, i, l
	Transmission	X			d, h, m
	PbA batteries	X			a, i, o, p
EV	Motor, control, and inverter		X	X	g, n
	Fluids		X	X	a, b, i, j
	Differential		X	X	g, h
	$LiFePO_4$ battery		X		q
	LiNCM battery			X	q

Note: ICEV = internal combustion engine vehicle; EV = electric vehicle; LiNCM = lithium nickel cobalt manganese; $LiFePO_4$ = lithium iron phosphate; PbA = lead acid.
a = Burnham et al. (2006); b = Sullivan et al. (1998); c = USAMP (1999); d = Daimler AG (2008a); e = Tami (1991); f = Garg et al. (2000); g = Röder (2001); h = Schweimer and Levin (2000); i = IDIS 2 Consortium (2009); j = Nemry et al. (2008); k = NCDNR (2010); l = Lloyd et al. (2005); m = Volkswagen AG (2008a, 2008b); n = ABB (2010a, 2010b, 2010c, 2010d, 2010e); o = Rantik (1999); p = Delucchi (2003); q = Majeau-Bettez et al. (2011).

Vehicle Production

We first established the inventory of a generic vehicle glider, which was devoid of any component specific to ICEVs or EVs. We then added the ICEV and EV powertrains. In the case of the EV, two battery types were investigated (i.e., $LiFePO_4$ and LiNCM). Table 1 provides a list of the different vehicles' components, which are comprised of roughly 140 subcomponents. The detailed inventories and vehicle properties are provided in supporting information S2 on the Web.

The GREET 2.7 vehicle cycle model (Burnham et al. 2006) served as a starting point for modeling the glider and ICEV powertrain. It was rescaled and adapted to match the characteristics of the Mercedes A-Class (Daimler AG 2008a), further subdivided to gain additional component-level detail, and then supplemented by data from detailed industry inventories and reports. Notably, the engine composition is based on the Volkswagen A4 (Schweimer and Levin 2000). The EV powertrain configurations were modeled roughly after that of the Nissan Leaf EV (Nissan 2010b). Battery inventories were adapted in full resolution from Majeau-Bettez and colleagues (2011). Battery masses of 214 and 273 kilograms (kg) were selected for LiNCM and $LiFePO_4$, respectively, so as to have equal charge capacities of 24 kilowatt-hours (kWh).[3]

Use Phase

All use phase energy requirements were based on industry performance tests with the New European Driving Cycle, following the UNECE 101 regulation (UNECE 2005). These tests combine four elementary urban driving cycles and one extra-urban driving cycle, with regenerative charging and energy losses during overnight charging included for EVs. Use phase energy requirements were assumed to be 0.623 megajoules/kilometer (MJ/km)[4] for the EV, 68.5 milliliter/kilometer (mL/km)[5] for the gasoline ICEV, and 53.5 mL/km for the diesel ICEV, based on the Nissan Leaf (Nissan 2010a), the Mercedes A-170, and an average of the Mercedes CDI A-160 and A-180 results (Daimler AG 2008a). These vehicles were selected because of their comparable sizes, masses, and performance characteristics (0 to 100 kilometer/hour [km/h] acceleration between 11.5 and 13.5 seconds).

To ensure the comparability of these test results, we checked that the energy at wheel was similar for the different vehicles, taking into account typical battery, engine, and transmission losses (see sheet 24 of supporting information S2 on the Web) (Åhman 2001; Karden et al. 2007; Larminie and Lowry 2003; Matheys et al. 2008; Tanaka et al. 2001; Van den Bossche et al. 2006) and found our use phase energy requirements imply energy delivered to the wheel to be 0.42 MJ/km for the ICEVs and 0.48 MJ/km for the EV. The slight increase in energy use is consistent with simulation results considering battery and structural mass differences between an ICEV and an EV (Shiau et al. 2009).

Use phase gasoline, diesel, and electricity inputs to vehicles are representative of average European conditions and import mixes. Results for natural gas and coal electricity use by the LiNCM EV are also provided, and additional electricity sources are represented in the sensitivity analysis. Brake wear is estimated based on work by Garg and colleagues (2000) and tire wear is based on work by Röder (2001). Maintenance and parts replacement is estimated based on available reports and our own assumptions documented in the supporting information on the Web.

End of Life

Vehicle and battery lifetimes are assumed to be 150,000 km driven, which is well aligned with typical lifetime assumptions used by the automotive industry (Daimler AG 2008a; Volkswagen AG 2008b; Ford Motor Company 2007), although lifetimes found in the literature range between 150,000 and 300,000 km (Hawkins et al. 2012). Results for alternative lifetimes are presented in the sensitivity analysis section. End-of-life vehicle treatment is based on Ecoinvent v2.2 (Burnham et al. 2006). Battery treatment consists of dismantling and a cryogenic shattering process. The impacts associated with material recovery and disposal processes are allocated to the vehicle life cycle.

Results

Overview

Figure 1 compares six transportation technologies in terms of ten life cycle environmental impact categories. Detailed numerical results are presented in section I of supporting information S1 on the Web. The cases represent an LiNCM or $LiFePO_4$ EV powered by European average electricity (Euro), an LiNCM EV powered by either natural gas (NG) or coal (C) electricity, and an ICEV powered by either gasoline (G) or diesel (D). Impacts are broken down in terms of life cycle stages and normalized to the greatest impact. Differences between the impacts of the two EV options arise solely from differences in the production of the batteries.

For all scenarios, human toxicity potential (HTP), mineral depletion potential (MDP), and freshwater eco-toxicity potential (FETP) are caused primarily by the supply chains involved in the production of the vehicles. On the other hand, the use phase dominates for GWP, terrestrial eco-toxicity potential (TETP), and fossil depletion potential (FDP). End-of-life treatment adds only a marginal contribution across all impact categories. The EV production phase is more environmentally intensive than that of ICEVs for all impact categories with the exception of terrestrial acidification potential (TAP). The supply chains involved in the production of electric powertrains and traction batteries add significantly to the environmental impacts of vehicle production. For some environmental impact categories, lower emissions during the use phase compensate for the additional burden caused during the production phase of EVs, depending on the electricity mix. However, this is not always the case.

Global Warming Potential

For all scenarios analyzed, the use phase is responsible for the majority of the GWP impact, either directly through fuel combustion or indirectly during electricity production. When powered by average European electricity, EVs are found to reduce GWP by 20% to 24% compared to gasoline ICEVs and by 10% to 14% relative to diesel ICEVs under the base case assumption of a 150,000 km vehicle lifetime. When powered by electricity from natural gas, we estimate LiNCM EVs offer a reduction in GHG emissions of 12% compared to gasoline ICEVs, and break even with diesel ICEVs. EVs powered by coal electricity are expected to cause an increase in GWP of 17% to 27% compared with diesel and gasoline ICEVs.

In contrast with ICEVs, almost half of an EV's life cycle GWP is associated with its production. We estimate the GWP from EV production to be 87 to 95 grams carbon dioxide equivalent per kilometer (g CO_2-eq/km), which is roughly

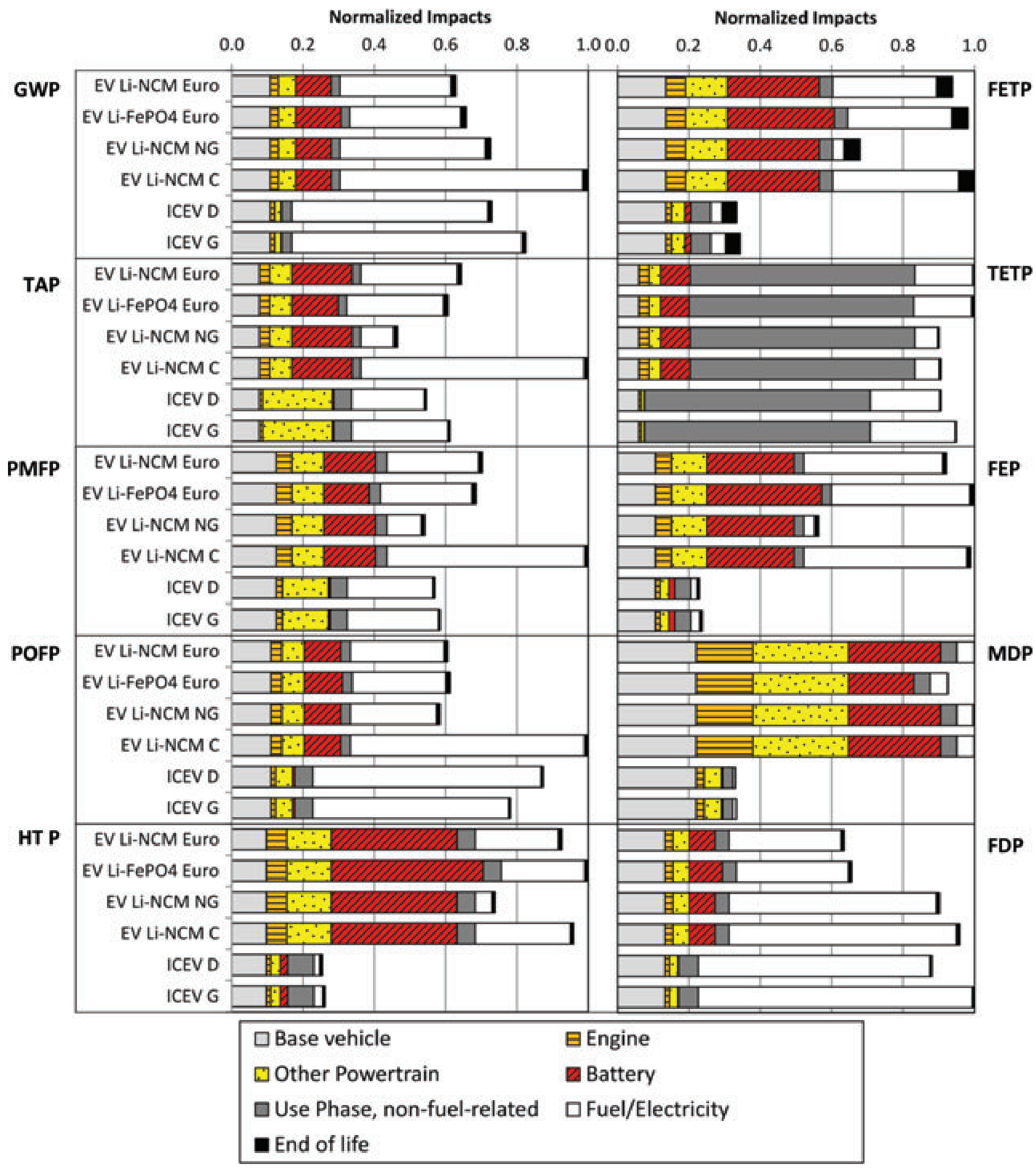

Figure 1 Normalized impacts of vehicle production. Results for each impact category have been normalized to the largest total impact. Global warming (GWP), terrestrial acidification (TAP), particulate matter formation (PMFP), photochemical oxidation formation (POFP), human toxicity (HTP), freshwater eco-toxicity (FETP), terrestrial eco-toxicity (TETP), freshwater eutrophication (FEP), mineral resource depletion (MDP), fossil resource depletion (FDP), internal combustion engine vehicle (ICEV), electric vehicle (EV), lithium iron phosphate ($LiFePO_4$), lithium nickel cobalt manganese (LiNCM), coal (C), natural gas (NG), European electricity mix (Euro).

twice the 43 g CO_2-eq/km associated with ICEV production. Battery production contributes 35% to 41% of the EV production phase GWP, whereas the electric engine contributes 7% to 8%. Other powertrain components, notably inverters and the passive battery cooling system with their high aluminum content, contribute 16% to 18% of the embodied GWP of EVs.

Under the assumption of identical life expectancies, LiNCM EVs cause slightly less GWP impact than $LiFePO_4$ EVs due to the greater energy density of their batteries. With the European electricity mix, the LiNCM and $LiFePO_4$ vehicles present life cycle GWP intensities of 197 and 206 g CO_2-eq/km, respectively.

Because production impacts are more significant for EVs than conventional vehicles, assuming a vehicle lifetime of 200,000 km exaggerates the GWP benefits of EVs to 27% to 29% relative to gasoline vehicles or 17% to 20% relative to diesel because production-related impacts are distributed across the longer lifetime. An assumption of 100,000 km decreases the benefit of EVs to 9% to 14% with respect to gasoline vehicles and results in impacts indistinguishable from those of a diesel vehicle. Although not discussed in detail due to space constraints, the sensitivity to lifetime assumption follows a similar pattern for other impact categories as well, with impacts associated with vehicle production being effected more significantly than those more closely associated with the use phase.

Other Potential Impacts

The TAP impacts caused by the production phase of the EVs and ICEVs are similar, but their underlying causes differ. With structural path analysis (Defourny and Thorbecke 1984; Treloar 1997; Peters and Hertwich 2006), the acidification impact of EV production can be traced back to the nickel, copper, and, to a lesser extent, aluminum requirements of the battery and the motor (see section IV of supporting information S1 on the Web). On the other hand, more than 70% of the production phase TAP of the ICEV is caused by the production of platinum-group metals for the exhaust catalyst. It should be noted that there is significant variability between the LCIs of primary platinum-group metals (Classen et al. 2009). The acidifying emissions reported for Russian and South African production processes differ by more than an order of magnitude. Our study uses a European consumption mix of these two sources and secondary platinum-group metals.

As more than 70% of the life cycle TAP is caused by sulfur dioxide (SO_2) emissions, the sulfur intensity of the use phase fuel largely determines the relative performances of the different transportation technologies in terms of TAP. Because of its share of hard coal and lignite combustion, the use of average European electricity for EV transportation does not lead to significant improvements relative to ICEVs. Significant benefits may only be expected for EVs using electricity sources with sulfur intensities comparable to or lower than that of natural gas.

Particulate matter formation potential (PMFP) follows a trend similar to that of TAP. Structural path analysis identifies the same metal supply chains—nickel, copper, and aluminum—as the dominant sources of emissions from the production phase, and SO_2 emissions are the leading cause of PMFP for all life cycle transportation scenarios (35% to 46% of impact). EVs using natural gas electricity perform best with regard to PMFP due to the relative purity of natural gas and the completeness of its combustion. The use of average European or coal-based electricity leads to a potential increase in PMFP relative to ICEVs, though this impact is spatially and to some extent temporally distanced from the use phase.

The photochemical oxidation formation potential (POFP), or smog formation potential, is one of the environmental impact categories for which EVs perform best, with European and natural gas electricity mixes allowing for reductions of 22% to 33% relative to ICEVs. For all scenarios, releases of nitrogen oxides are the predominant cause of impact. These are mostly caused by combustion activities, but also from blasting in mining activities.

Human toxicity potential (HTP) stands out as a potentially significant category for problem-shifting associated with a shift from ICEVs to EVs. We estimate that HTP increases for EVs relative to ICEVs both in the production and the use phase. The different EV options have 180% to 290% greater HTP impacts compared to the ICEV alternatives. The additional production phase toxicity impacts of EVs stem mostly from additional copper requirements and, in the case of NCM EVs, nickel requirements. Toxic emissions from the production chain of these metals mostly occur in the disposal of the sulfidic mine tailings, which accounts for roughly 75% of the HTP from the production phase. The rest of the impact is caused predominantly by the disposal of spoils from lignite and coal mining, which are important sources of energy throughout the life cycle of the EV.

Freshwater ecotoxicity potential (FETP) and eutrophication potential (FEP) impacts demonstrate patterns similar to HTP. In fact, these three impact categories are dominated by the same processes (i.e., disposal of sulfidic tailings and spoils from coal and lignite mining). For all three impacts, the use of electricity from natural gas yields substantial benefits relative to the other electricity mixes.

Terrestrial ecotoxicity potential (TETP) is dominated by the use phase emissions of zinc from tire wear (approximately 40%), and copper and titanium from brake wear (25%). Given the uncertainty of the characterization of this impact (Huijbregts et al. 2000; Lenzen 2006), there is no clear difference among the vehicle options considered.

Metal depletion potential (MDP) is a commonly cited concern with EVs (e.g., Gaines and Nelson 2009, 2010), due to their reliance on metals of differing scarcities. This analysis suggests that the MDP of EVs is roughly three times that of ICEVs. However, as this investigation was not specifically focused on MDP, results are more uncertain than for other impact categories. Depending on the component and the metal, our inventory either relies on primary sources or on average consumption mixes of primary and secondary sources (see sheets 6-19 of supporting information S2 on the Web). It should be noted that the ReCiPe method does not include MDP characterization factors for lithium.

Fossil depletion potential (FDP) may be decreased by 25% to 36% with electric transportation relying on average European electricity. EVs with natural gas or coal electricity, however, do not lead to significant reductions.

Sensitivity Analysis and Uncertainty

EVs have only recently entered mass production, their ongoing development is still very much open-ended, and technologies and production processes are evolving rapidly. Therefore it is difficult to fix specific values for some of the parameters influencing the impacts of EVs. In addition, parameters such as the consumption and the carbon intensity of use phase energy are influenced by driving patterns and local conditions that vary between users and regions. We performed a sensitivity analysis to understand the robustness of our results against changes in key parameters. Figure 2 provides the variations in LiNCM and $LiFePO_4$ EVs with European electricity and ICEVs with gasoline and diesel. The base case for each technology is represented with a vertical bar, and the horizontal bars describe

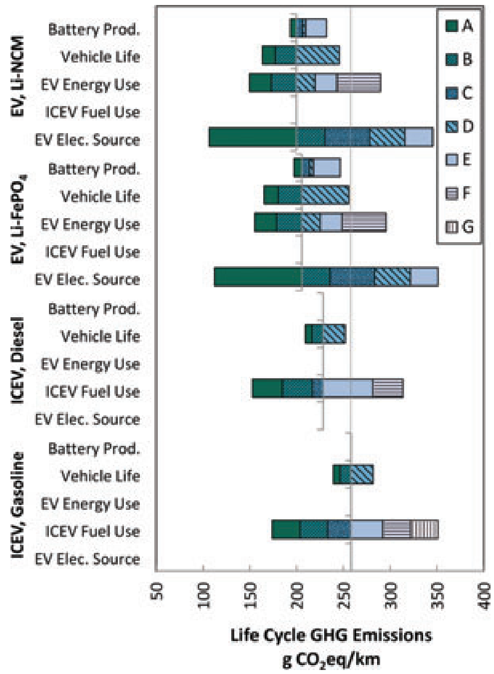

Figure 2 Sensitivity of total life cycle greenhouse gas emissions to key parameters. Vertical lines represent the base case for each technology, whereas the bars demonstrate the variation associated with the following changes in the parameters listed at left: battery prod (mass of battery required, normalized to base case): 0.8 (A), 1.0 (B), 1.2 (C), 1.3 (D), 2 (E); vehicle lifetime (km): 250,000 (A), 200,000 (B), 150,000 (C), 100,000 (D); EV energy use (MJ/km): 0.3 (A), 0.45 (B), 0.6 (C), 0.75 (D), 0.9 (E), 1.2 (F); ICEV diesel use (L/km): 0.03 (A), 0.04 (B), 0.05 (C), 0.06 (D), 0.07 (E), 0.08 (F); ICEV gasoline use (L/km): 0.04 (A), 0.05 (B), 0.06 (C), 0.07 (D), 0.08 (E), 0.09 (F), 0.1 (G); EV use phase electricity source: wind (A), natural gas (B), oil (C), coal (D), lignite (E). ICEV = internal combustion engine vehicle; EV = electric vehicle; $LiFePO_4$ = lithium iron phosphate; LiNCM = lithium nickel cobalt manganese; GHG = greenhouse gas; g CO_2-eq/km = grams carbon dioxide equivalent per kilometer; km = kilometer; MJ/km = megajoules per kilometer; L/km = liters per kilometer.

the deviation in GWP impact associated with changes in the parameters. The sensitivity analysis of other impact categories is presented in section III of supporting information S1 on the Web.

Figure 2 demonstrates that changing our assumptions regarding battery mass, vehicle lifetime, vehicle efficiency, and electricity mix can all potentially alter our base case ranking for GWP. Therefore care must be taken in interpreting and drawing conclusions based on our results or across other studies of the environmental impacts of EVs and ICEVs.

Across most impact categories, the environmental intensity of the use phase electricity is the single most influential variable in the EV life cycle (see figure 2 for GWP and section III of supporting information S1 on the Web for others). Our sensitivity analysis predicts net benefits in terms of TAP, PMFP, and POFP for EVs using electricity from natural gas, relative to ICEVs. A key issue raised by these results is our ability to effectively compare electricity generation technologies across a diverse range of impacts. For cleaner, renewable, and less carbon-intensive energy sources, such as wind energy, these benefits are intensified and accompanied by gains in terms of GWP and FDP. Wind power electricity would allow electric transportation with life cycle carbon footprints as low as 106 g CO_2-eq/km. On the other hand, the use of electricity from lignite combustion leads to a life cycle GWP of 352 g CO_2-eq/km, significantly worse than the comparable ICEV performance. The use of electricity from natural gas combustion seems to constitute the break-even point for EVs relative to diesel ICEVs in terms of GWP. However, human and water toxicities along with metal depletion potentials are always greater for electric transportation independent of the electricity source.

Variations in fuel and electricity efficiencies have a significant effect on GWP and on other, predominantly use-phase, impacts. While our base case efficiency of 0.623 MJ/km is derived from the Nissan Leaf's New European Driving Cycle test results, previous studies have estimated use phase efficiencies between 0.4 MJ/km (Elgowainy et al. 2009; Shiau et al. 2009, 2010) (for PHEVs in charge-depleting mode) and 0.8 MJ/km (Graham and Little 2001; Huo et al. 2010; Parks et al. 2007). With an efficiency of 0.9 MJ/km, the studied EVs would have a GWP footprint between that of the base case diesel and gasoline ICEVs. Conversely, a fuel consumption between 40 and 50 mL/km would allow the ICEV to break even with the base case EVs in terms of GWP.

As the industry matures, the design of EVs will probably converge toward "typical" battery sizes and capacities, based on engineering constraints and consumer demand. Depending on the desired driving range (Shiau et al. 2009) and probable improvements in battery energy densities (Armand and Tarascon 2008; Shukla and Kumar 2008), different battery masses may be envisioned for future EVs. At present, however, the typical size of future EV batteries is still uncertain, and this, in turn, influences the level of certainty with which the battery lifetime can be estimated, since lifetime is largely determined by the number and characteristics of the charge–discharge cycles (Majeau-Bettez et al. 2011; Matheys et al. 2008). Our base case assumes masses consistent with battery capacities of 24 kWh and a 150,000 km lifetime equal to that of the vehicle. Deviations from this scenario are accounted for by considering a parameter by which the battery demand per vehicle lifetime is multiplied, which can be thought of as accounting for variation in battery mass, failure rate, or lifetime. Within the range considered (0.8 to 2.0 times battery mass), this source of sensitivity was not as significant as the use phase considerations; however, it is not insignificant. Uncertainties relevant to differences between LiNCM and $LiFePO_4$ battery types are such that it is not possible to distinguish their relative production impacts in our study. Varying battery requirements by ±20% is enough to alter their relative ranking.

Due to the greater emissions intensity of its production phase, changing the vehicle lifetime has a greater effect on the GWP per kilometer for EVs than it does for ICEVs. Increasing the lifetime of EVs from 150,000 km to 250,000 km potentially decreases the GWP by as much as 40 g CO_2-eq/km, down to roughly 165 g CO_2-eq/km, whereas the same lifetime increase for ICEVs only decreases the GWP per kilometer by 19 g CO_2-eq/km. Selecting an appropriate lifetime assumption for EVs is challenging, as many uncertainties arise related to battery degradation and failure rates, cost of operation and retirement decisions, and the driving patterns associated with EV use.

In addition to modeling uncertainties, some level of uncertainty is associated with the use of generic Ecoinvent processes (Ecoinvent Centre 2010). The linked unit process structure and thorough documentation allow for identifying sources of background uncertainty. According to the authors of the LCI on sulfidic tailings disposal—a dominant cause of human and freshwater eco-toxicity impacts in our system—"the uncertainty in [the tailings disposal] dataset is high," due largely to the inherent stochasticity of the tailings composition (Classen et al. 2009). However, they point to the use of conservative modeling assumptions "to avoid overestimations." Furthermore, the great uncertainties associated with the characterization of toxicity issues are a well-known challenge for environmental assessment (Goedkoop et al. 2009). Of greater importance for the ICEV catalyst than for EVs, the broad range of acidifying emissions intensities across the different producers of platinum-group metals has already been alluded to. Classen and colleagues (2009), whose data was based on an LCA study of autocatalysts, point to "satisfactory" data quality for mining and metallurgy, but greater uncertainties for beneficiation and disposal of tailings.

Our estimates of potential environmental impacts from end-of-life treatment are small (figure 1), and allocating them differently would not have significantly altered our results. With regard to materials in the vehicle production phase, some assumptions were necessarily made as to the mix of primary and secondary sources used by the industry. It should be noted, however, that the production mixes for many key materials in this study are still largely dominated by primary sources (>75%). This is notably the case for the use of platinum-group metals in the catalyst industry (Saurat and Bringezu 2008), refined copper (IISD 2010), and refined nickel (Reck et al. 2008).

Discussion

Benchmarking and Limits of Scope

Our best estimate for the GWP impact of EV production (87 to 95 g CO_2-eq/km) is almost twice the impact potential reported by previous studies (Baptista et al. 2010; Burnham et al. 2006; Notter et al. 2010; Samaras and Meisterling 2008), due in part to higher battery-related impacts and the inclusion of electronic components not previously considered. For ICEV production, our cradle-to-gate GWP intensity (5 kg CO_2-eq/kg of car) is within the range of intensities (4 to –6.5 kg CO_2-eq/kg of car) determined from various literature sources (Daimler AG 2005, 2007a, 2007b, 2008a, 2008b; Samaras and Meisterling 2008; Volkswagen AG 2008b, 2008c). A few studies also provide emissions results related to acidification, human toxicity, and resource use, among others, related to various vehicle types (Daimler AG 2009; Duvall 2005; Mccleese and Lapuma 2002; Parks et al. 2007; Wang et al. 1997). While the results of the present study are reasonably well aligned with these previous results, this inventory offers a significant improvement in transparency.

Although this study incorporates the cradle-to-gate portion of the battery inventories by Majeau-Bettez and colleagues (2011), we treat the use phase in a different manner. This study assumes that the batteries have a lifetime equivalent to that of the vehicle, regardless of their chemistry or their charge capacity, whereas Majeau-Bettez and colleagues expressed lifetimes as the number of expected charge–discharge cycles. In this study, equal lifetime assumptions significantly increase the per kilometer impacts of the $LiFePO_4$ and shift its ranking relative to LiNCM.

The cradle-to-gate battery production impacts estimated by Majeau-Bettez and colleagues (2011) (22 kg CO_2-eq/kg) are substantially higher than the estimates by Notter and colleagues (2010) (6 kg CO_2-eq/kg) or Samaras and Meisterling (2008) (9.6 kg CO_2-eq/kg). These differences, which mostly stem from differing assumptions concerning manufacturing energy requirements and system boundaries, are indicative of the need for better public primary inventory data from the battery industry.

In undertaking this study, we have striven to provide as much detail as possible while maintaining a transparent inventory. This allows impacts from manufacturing for the different vehicle configurations to be distinguished. Improved resolution also permitted better quality control of our model than is possible with GREET 2.7 (Burnham et al. 2006). Although our model may be less detailed than what automotive manufacturers produce for internal use, to the best of our knowledge it represents the best combination of detail, transparency, and completeness in a publicly available vehicle inventory.

This study provides an attributional perspective that can be more easily adopted in and compared to future vehicle LCA studies. While acknowledging their relevance, we leave consequential and scenario-based considerations for future work. There are a number of considerations that should be worked through in relation to specific decisions regarding adoptions of vehicle technologies, such as (1) the additional stress that a large fleet of EVs would place upon electricity production and distribution infrastructures (Farrell et al. 2007); (2) the potential impacts of large-scale EV adoption on the quality of metal ores, extraction costs, and impacts; and (3) structural changes in society or the rebound effects (Hertwich 2005) that may result from a large-scale adoption of EVs.

General Findings and Policy Implications

Our results demonstrate the importance of including vehicle manufacturing impacts when considering electric transportation policies. The GWP from EV production is about twice that of conventional vehicles. Our results suggest a potentially greater gap between the two technologies for other impact categories, such as HTP and MDP. Environmental evaluations relying solely on fuel and powertrain efficiencies miss key differences associated with the production of different vehicle types and could lead to misguided comparisons across technologies. While the EU has made efforts to include life cycle approaches for benchmarking various biofuels and determining appropriate support mechanisms (European Commission 2009), EVs warrant yet another layer of complexity. Assessments excluding the impacts from vehicle production are likely to lead to biased conclusions and suboptimal results. The environmental performance of EVs is critically dependent on the combination of the vehicle and electricity production impacts as well as key factors such as energy use and battery and vehicle lifetimes. For example, performing the calculation assuming a lifetime of 200,000 km for the ICEV and assuming a battery replacement within the lifetime of the EV would result in lower GWP impact for the diesel ICEV with respect to the EV charged with European average electricity.

Although EVs are an important technological breakthrough with substantial potential environmental benefits, these cannot be harnessed everywhere and in every condition. Our results clearly indicate that it is counterproductive to promote EVs in areas where electricity is primarily produced from lignite, coal, or even heavy oil combustion. At best, with such electricity mixes, local pollution reductions may be achieved. Thus EVs are a means of moving emissions away from the road rather than reducing them globally. Only limited benefits are achieved by EVs using electricity from natural gas. In the absence of foreseeable improvements to electricity mixes, a more significant reduction in GWP could potentially be achieved by increasing fuel efficiency or shifting from gasoline to diesel ICEVs without significant problem-shifting (with the exception of smog).

Conversely, the combination of EVs with clean energy sources would potentially allow for drastic reductions of many transportation environmental impacts, especially in terms of climate change, air quality, and preservation of fossil fuels. The many potential advantages of EVs should therefore serve as a motivation for cleaning up regional electricity mixes, but their promotion should not precede commitment to grid improvement. Consideration of alternative vehicle technologies should be undertaken from the perspective of benefits across time. While EVs may only offer minor benefits or even setbacks under an initial grid, their development and market penetration should be evaluated together with realistic scenarios for grid development in the long term.

Our results point to some probable problem shifts, irrespective of the electricity mix. EVs appear to cause a higher potential for human toxicity, freshwater eco-toxicity, freshwater eutrophication, and metal depletion impacts. Uncertainties and risk assessment play an important role in this trade-off, however. As previously discussed, these impacts have significant uncertainties associated with both release inventories and characterization factors. The promotion of EVs by policy instruments may boil down to achieving clear reductions for emissions that have well-understood impact potentials, such as GWP and FDP, at the expense of uncertain increases in emissions that potentially cause poorly understood impacts, such as FETP. In view of this trade-off, a promotion of EVs should be accompanied by stricter life cycle management and life cycle auditing. Considering how the potential problem shifts mostly arise from material requirements of EV production, effective recycling programs and improved EV lifetimes would constitute an appropriate first response. A thorough material flow strategy is warranted, including the evaluation of secondary sources, alternative materials, and component recyclability.

The shift in emissions that EVs are poised to bring about—an elimination of tailpipe emissions at the expense of increased emissions in the vehicle and electricity production chains—brings new opportunities and risks for policy makers and stakeholders. On the one hand, EVs would aggregate emissions at a few point sources (power plants, mines, etc.) instead of millions of mobile sources, making it conceptually easier to control and optimize societies' transportation systems (McKinsey & Company 2009). On the other hand, the indirect nature of these emissions—which are embodied in internationally traded commodities such as copper, nickel, and electricity—challenges us as a society. It poses the question of how serious are we about life cycle thinking, and how much control and oversight we, customers, and policy makers believe should be exerted across production chains.

Conclusion

We provide a new level of transparency and detail to the ongoing public discussion on the life cycle merits of EVs relative ICEVs. The production, use, and end of life of these two technologies were inventoried in a manner ensuring an appropriate comparison. The production phase of EVs proved substantially more environmentally intensive. Nonetheless, substantial overall improvements in regard to GWP, TAP, and other impacts may be achieved by EVs powered with appropriate energy sources relative to comparable ICEVs. However, it is counterproductive to promote EVs in regions where electricity is produced from oil, coal, and lignite combustion. The electrification of transportation should be accompanied by a sharpened policy focus with regard to life cycle management, and thus counter potential setbacks in terms of water pollution and toxicity. EVs are poised to link the personal transportation sector together with the electricity, the electronic, and the metal industry sectors in an unprecedented way. Therefore the developments of these sectors must be jointly and consistently addressed in order for EVs to contribute positively to pollution mitigation efforts.

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Notes

1. Light-duty vehicles include cars, SUVs, mini vans and personal-use light trucks. There is no single definition for light-duty vehicles covering these vehicles in different countries.
2. One kilometer (km, SI) ≈ 0.621 miles (mi).
3. One kilowatt-hour (kWh) ≈ 3.6×10^6 joules (J, SI) ≈ 3.412×10^3 British thermal units (BTU).
4. One megajoule (MJ) = 10^6 joules (J, SI) ≈ 239 kilocalories (kcal) ≈ 948 British thermal units (BTU).
5. One milliliter per kilometer (mL/km) ≈ 0.425×10^{-3} gallons per mile (gal./mi).

References

ABB. 2010a. *Environmental product declaration: AC 800M controller*. Helsinki, Finland: ABB Industry Oy/Machines Group.

ABB. 2010b. *Environmental product declaration: AC low voltage cast iron motor, type M3BP 315*. Helsinki, Finland: ABB Industry Oy/Machines Group.

ABB. 2010c. *Environmental product declaration: AC machine type HXR 355, 250 kW power*. Helsinki, Finland: ABB Industry Oy/Machines Group.

ABB. 2010d. *Environmental product declaration: ACS 600 frequency converter, 250 kW power*. Helsinki, Finland: ABB Industry Oy/Machines Group.

ABB. 2010e. *Environmental product declaration: DriveIT low voltage AC drive, ACS 800 frequency converter, 250 kW power*. Helsinki, Finland: ABB Industry Oy/Machines Group.

Åhman, M. 2001. Primary energy efficiency of alternative powertrains in vehicles. *Energy* 26(11): 973–989.

Armand, M. and J. M. Tarascon. 2008. Building better batteries. *Nature* 451(7179): 652–657.

Baptista, P., M. Tomás, and C. Silva. 2010. Plug-in hybrid fuel cell vehicles market penetration scenarios. *International Journal of Hydrogen Energy* 35(18): 10024–10030.

Burnham, A., M. Wang, and Y. Wu. 2006. *Development and applications of GREET 2.7—The transportation vehicle-cycle model. ANL/ESD/06*. Argonne, IL, USA: Argonne National Laboratory, University of Chicago, U.S. Department of Energy.

Classen, M., H.-Jörg Althaus, S. Blaser, W. Scharnhorst, M. Tuchschmid, N. Jungbluth, and M. Faist Emmenegger. 2009. *Life cycle inventories of metals*. Final report Ecoinvent data v2.1 no, 10. Dübendorf, Switzerland: Swiss Center for Life Cycle Inventories.

Curran, M. A. 1996. *Environmental life-cycle assessment*. New York: McGraw-Hill Professional.

Daimler AG. 2005. *Environmental certificate S-class. Mercedes-Benz*. Stuttgart, Germany: Daimler AG.

Daimler AG. 2007a. *Environmental certificate C-class. Mercedes-Benz*. Stuttgart, Germany: Daimler AG.

Daimler AG. 2007b. *Environmental certificate C-class estate. Mercedes-Benz*. Stuttgart, Germany: Daimler AG.

Daimler AG. 2008a. *Environmental certificate A-class. Mercedes-Benz*. Stuttgart, Germany: Daimler AG.

Daimler AG. 2008b. *Environmental certificate B-class. Mercedes-Benz*. Stuttgart, Germany: Daimler AG.

Daimler AG. 2009. *Life cycle – Environmental certificate for the S-400 hybrid. Mercedes-Benz*. Stuttgart, Germany: Daimler AG.

Daimler AG. 2010b. *Sustainable mobility strategy – The road to emission-free mobility*. Stuttgart, Germany: Daimler AG.

Daimler AG. 2010a. *Electrification of the drivetrain*. Stuttgart, Germany: Daimler AG.

Defourny, J. and E. Thorbecke. 1984. Structural path analysis and multiplier decomposition within a social accounting matrix framework. *Economic Journal* 94(373): 111–136.

Delucchi, M. A. 2003. *A lifecycle emissions model (LEM): Lifecycle emissions from transportation fuels, motor vehicles, transportation modes, electricity use, heating and cooking fuels, and materials—Documentation of methods and data*. UCD-ITS-RR-03-17-MAIN REPORT. Davis, CA, USA: Institute of Transportation Studies, University of California, Davis.

Duvall, M. S. 2005. Battery evaluation for plug-in hybrid electric vehicles. Paper presented at the IEEE Vehicle Power and Propulsion Conference, 7–9 September, Chicago, IL, USA.

Ecoinvent Centre. 2010. *Ecoinvent data and reports v2.2*. Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories.

Elgowainy, A., A. Burnham, M. Wang, J. Molburg, and A. Rousseau. 2009. *Well-to-wheels energy use and greenhouse gas emissions analysis of plug-in hybrid electric vehicles*. ANL/ESD/09-2. Argonne, IL, USA: Argonne National Laboratory, University of Chicago, U.S. Department of Energy.

European Commission. 2009. Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing directives 2001/ 77/EC and 2003/30/EC. Brussels, Belgium: European Commission.

European Commission. 2010. European strategy on clean and energy efficient vehicles. http://ec.europa.eu/enterprise/sectors/automotive/competitiveness-cars21/energy-efficient/index_en.htm. Accessed 17 December 2011.

Farrell, A. E., D. Lemoine, and D. M. Kammen. 2007. Effects of plug-in hybrid electric vehicles in California energy markets. Paper presented at the Transport Research Board 86th Annual Meeting, 21–25 January, Washington, DC, USA.

Ford Motor Company. 2007. *Product sustainability index*. Ford Motor Company. Köln, Germany: Ford of Europe.

Ford Motor Company. 2011. Ford electric vehicle plan. http:// corporate.ford.com/news-center/news/press-releases/press-releases- detail/pr-ford-electric-vehicle-plan-offers-33793. Accessed 17 December 2011.

Gaines, L. and P. Nelson. 2009. *Lithium-ion batteries: Possible materials issues*. Argonne, IL, USA: Argonne National Laboratory.

Gaines, L. and P. Nelson. 2010. *Lithium-ion batteries: Examining material demand and recycling issues*. Argonne, IL, USA: Argonne National Laboratory.

Garg, B. D., S. H. Cadle, P. A. Mulawa, P. J. Groblicki, C. Laroo, and G. A. Parr. 2000. Brake wear particulate matter emissions. *Environmental Science & Technology* 34(21): 4463–4469.

Goedkoop, M., R. Heijungs, M. A. J. Huijbregts, A. D. Schryver, J. Struijs, and R. V. Zelm. 2009. *ReCiPe 2008*. Netherlands: Dutch Ministry of the Environment, The Hague.

Graham, R. and A. D. Little. 2001. *Comparing the benefits and impacts of hybrid electric vehicle options*. Palo Alto, CA, USA: Electric Power Research Institute (EPRI).

Greater London Authority. 2009. *Turning London electric – London's electric vehicle infrastructure strategy*. London, UK: Greater London Authority.

Guinée, J. B., M. Gorrée, R. Heijungs, G. Huppes, R. Kleijn, A. de Koning, L. van Oers, A. Wegener Sleeswijk, S. Suh, H. A. Udo de Haes, H. de Bruijn, R. van Duin, and M. A. J. Huijbregts. 2002. *Handbook on life cycle assessment – Operational guide to the ISO standards*. Dordrecht, the Netherlands: Kluwer Academic.

Hawkins, T. R., O. M. Gausen, and A. H. Strømman. 2012. Environmental impacts of hybrid and electric vehicles – a review. *International Journal of Life Cycle Assessment* 17(8): 997–1014.

Hertwich, E. G. 2005. Consumption and the rebound effect: An industrial ecology perspective. *Journal of Industrial Ecology* 9(1–2): 85–98.

Huijbregts, M. A. J., U. Thissen, and T. Jager. 2000. Priority assessment of toxic substances in life cycle assessment. Part II: Assessing parameter uncertainty and human variability in the calculation of toxicity potentials. *Chemosphere* 41(4): 575–588.

Huo, H., Q. Zhang, M. Q. Wang, D. G. Streets, and K. He. 2010. Environmental implication of electric vehicles in China. *Environmental Science & Technology* 44(13): 4856–61.

IDIS 2 Consortium. 2009. *International Dismantling Information System (IDIS) v4.30*. Saarbruecken, Germany: IDIS Management/tec4U GmbH.

IEA (International Energy Agency). 2009. *Technology roadmaps – Electric and plug-in hybrid electric vehicles*. *International Energy Agency*. Paris, France: Organisation for Economic Cooperation and Development.

IISD (International Institute for Sustainable Development). 2010. *Sustainable development and the global copper supply chain: International research team report*. Winnipeg, Manitoba, Canada: IISD.

Johnson, J., E. M. Harper, R. J. Lifset, and T. E. Graedel. 2007. Dining at the periodic table: Metals concentrations as they relate to recycling. *Environmental Science & Technology* 41(5): 1759–1765.

Karden, E., S. Ploumen, B. Fricke, T. Miller, and K. Snyder. 2007. Energy storage devices for future hybrid electric vehicles. *Journal of Power Sources* 168(1): 2–11.

Larminie, J. and J. Lowry. 2003. *Electric vehicle technology explained*. Chichester, UK: John Wiley & Sons.

Lenzen, M. 2006. Uncertainty in impact and externality assessments – Implications for decision-making. *International Journal of Life Cycle Assessment* 11(3): 189–199.

Lloyd, S. M., L. B. Lave, and H. S. Matthews. 2005. Life cycle benefits of using nanotechnology to stabilize platinum-group metal particles in automotive catalysts. *Environmental Science & Technology* 39(5): 1384–1392.

Majeau-Bettez, G., T. R. Hawkins, and A. H. Strømman. 2011. Life cycle environmental assessment of lithium-ion and nickel metal hydride batteries for plug-in hybrid and battery electric vehicles. *Environmental Science & Technology* 45(10): 4548–4554.

Matheys, J., J. V. Mierlo, J. M. Timmermans, and P. Van den Bossche. 2008. Life-cycle assessment of batteries in the context of the EU directive on end-of-life vehicles. *International Journal of Vehicle Design* 46(2): 189–203.

McKinsey & Company. 2009. *Paths to a low-carbon economy – Version 2 of the global greenhouse gas abatement cost curve*. Chicago, IL, USA: McKinsey & Company.

Mccleese, D. L. and P. T. Lapuma. 2002. Using Monte Carlo simulation in life cycle assessment for electric and internal combustion vehicles. *International Journal of Life Cycle Assessment* 7(4): 230–236.

NCDNR (North Carolina Department of Natural Resources). 2010. *Anatomy of a tire*. Raleigh, NC, USA: North Carolina Department of Natural Resources. http://infohouse.p2ric.org/ref/11/10504/html/intro/tire.htm. Accessed 16 February 2011.

Nakamura, S., K. Nakajima, Y. Kondo, and T. Nagasaka. 2008. The waste input-output approach to materials flow analysis. *Journal of Industrial Ecology* 11(4): 50–63.

Nemry, F., G. Leduc, I. Mongelli, and A. Uihlein. 2008. *Environmental impact of passenger cars*. Seville, Spain: Joint Research Centre, Institute for Prospective Technological Studies.

Nissan. 2010a. *Nissan Leaf*. Nissan Europe – France www.nissan.fr/etc/medialib/nissaneu/_FR_fr/_Brochures/_Electric_Vehicles/103857.Par.62053.File.pdf. Accessed 20 February 2011.

Nissan. 2010b. *Nissan Leaf*. Nissan North America. www.nissanusa.com/ev/media/pdf/specs/FeaturesAndSpecs.pdf. Accessed 20 February 2011.

Notter, D. A., M. Gauch, R. Widmer, P. Wäger, A. Stamp, R. Zah, and H. J. Althaus. 2010. Contribution of Li-ion batteries to the environmental impact of electric vehicles. *Environmental Science & Technology* 44(17): 6550–6556.

Parks, K., P. Denholm, and T. Markel. 2007. *Costs and emissions associated with plug-in hybrid electric vehicle charging in the Xcel Energy Colorado service territory*. NREL/TP-640-41410. Golden, CO, USA: National Renewable Energy Laboratory.

Peters, G. P. and E. G. Hertwich. 2006. Structural analysis of international trade: Environmental impacts of Norway. *Economic Systems Research* 18(2): 155–181.

Rantik, M. 1999. *Life cycle assessment of five batteries for electric vehicles under different charging regimes*. Stockholm, Sweden: Chalmers University of Technology.

Reck, B. K., D. B. Müller, K. Rostkowski, and T. E. Graedel. 2008. Anthropogenic nickel cycle: Insights into use, trade, and recycling. *Environmental Science & Technology* 42(9): 3394–400.

Röder, A. 2001. Integration of life-cycle assessment and energy planning models for the evaluation of car powertrains and fuels. Ph.D. dissertation, ETH-14291, Swiss Federal Institute of Technology, Zurich, Switzerland.

Samaras, C. and K. Meisterling. 2008. Life cycle assessment of greenhouse gas emissions from plug-in hybrid vehicles: Implications for policy. *Environmental Science & Technology* 42(9): 3170–3176.

Saurat, M. and S. Bringezu. 2008. Platinum group metal flows of Europe, part 1. *Journal of Industrial Ecology* 12: 754–767.

Schweimer, G. W. and M. Levin. 2000. *Life cycle inventory for the Golf A4*. Wolfsburg, Germany: Volkswagen AG.

Shiau, C.-S. N., N. Kaushal, C. T. Hendrickson, S. B. Peterson, J. F. Whitacre, and J. J. Michalek. 2010. Optimal plug-in hybrid electric vehicle design and allocation for minimum life cycle cost, petroleum consumption, and greenhouse gas emissions. *Journal of Mechanical Design* 132(9): 1–11.

Shiau, C.-S. N., C. Samaras, R. Hauffe, and J. J. Michalek. 2009. Impact of battery weight and charging patterns on the economic and environmental benefits of plug-in hybrid vehicles. *Energy Policy* 37(7): 2653–2663.

Shukla, A. K. and T. P. Kumar. 2008. Materials for next-generation lithium batteries. *Current Science* 93(3): 314–331.

Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller, eds. 2007. *Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, 2007*. Cambridge, UK: Cambridge University Press.

Sullivan, J. L., R. L. Williams, S. Lester, E. Cobas-Flores, S. Chubbs, S. Hentges, and S. Pomper. 1998. Life cycle inventory of a generic US family sedan: Overview of results USCAR AMP project. SAE Technical Paper Series no. 982160. Warrendale, PA, USA: SAE International.

Tami, R. M. 1991. *Material safety data sheet: Brake lining, non-asbestos friction material – 2530-01-298-3259*. Ridgeway, PA, USA: Motion Control Industries Division, Carlisle Corporation.

Tanaka, T., K. Ohta, and N. Arai. 2001. Year 2000 R&D status of large-scale lithium ion secondary batteries in the national project of Japan. *Journal of Power Sources* 97(1–2): 2–6.

Treloar, G. J. 1997. Extracting embodied energy paths from input-output tables: Towards an input-output-based hybrid energy analysis method. *Economic Systems Research* 9(4): 375–391.

UNECE (United Nations Economic Commission for Europe). 2005. Regulation no. 101 revision 2. Geneva, Switzerland: UNECE.

U.S. Department of Energy. 2011. *One million electric vehicles by 2015*. Washington, DC, USA: U.S. Department of Energy.

USAMP (U.S. Automobile Materials Partnership). 1999. *Life cycle inventory analysis of a generic vehicle. Final report*. Ann Arbor, MI, USA: US Automobile Materials Partnership, Ecobalance, National Pollution Prevention Center.

Van den Bossche, P., F. Vergels, J. Van Mierlo, J. Matheys, and W. Van Autenboer. 2006. SUBAT: An assessment of sustainable battery technology. *Journal of Power Sources* 162(2): 913–919.

Volkswagen AG. 2008a. *The DSG dual-clutch gearbox: Environmental commendation – Background report*. Wolfsburg, Germany: Volkswagen AG.

Volkswagen AG. 2008b. *The Golf – Environmental commendation – Background report*. Wolfsburg, Germany: Volkswagen AG.

Volkswagen AG. 2008c. *The Passat – Environmental commendation – Background report*. Wolfsburg, Germany: Volkswagen AG.

World Business Council for Sustainable Development (WBCSD). 2004. *Mobility 2030: Meeting the challenges to sustainability*. Geneva, Switzerland: WBCSD.

Wang, M. Q., S. Plotkin, D. J. Santini, J. He, L. Gaines, and P. Patterson. 1997. *Total energy-cycle energy and emissions impacts of hybrid electric vehicles*. Argonne, IL, USA: Argonne National Laboratory, University of Chicago, U.S. Department of Energy.

Wang, Q. and D. L. Santini. 1993. *Magnitude and value of electric vehicle emissions reductions for six driving cycles in four U.S. cities with varying air quality problems*. Argonne, IL, USA: Argonne National Laboratory.

Zackrisson, M., L. Avellan, and J. Orlenius. 2010. Life cycle assessment of lithium-ion batteries for plug-in hybrid electric vehicles – Critical issues. *Journal of Cleaner Production* 18(15): 1519–1529.

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Supporting Information

Additional supporting information may be found in the online version of this article.

Supporting Information S1. This supporting information contains the numerical results on which our article's figures are based. Section I presents life cycle impact assessment (LCIA) potentials for the different vehicles, presented as total impacts, normalized, or broken down across the different life cycle stages or components. Section II presents the material content of the different components in a condensed form. Section III presents the sensitivity analysis for all impact categories. Section IV presents the top paths responsible for the different impact categories, as identified by our structural path analysis.

Supporting Information S2. This supporting information contains our detailed inventory and system description. For every component group, the detailed inventory of each component/activity is presented in a matrix form, along with the links between this inventory and the Ecoinvent background. The matrix-based approach to LCA is briefly presented. Finally, we examine the comparability of the conventional vehicle and the electric vehicle in their use phase.

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