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Sensitivity Analysis in the Life-Cycle Assessment of Electric vs. Combustion Engine Cars under Approximate Real-World Conditions

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Abstract: This study compares the environmental impacts of petrol, diesel, natural gas, and electric vehicles using a process-based attributional life cycle assessment (LCA) and the ReCiPe characterization method that captures 18 impact categories and the single score endpoints. Unlike common practice, we derive the cradle-to-grave inventories from an originally combustion engine VW Caddy that was disassembled and electrified in our laboratory, and its energy consumption was measured on the road. Ecoivent 2.2 and 3.0 emission inventories were contrasted exhibiting basically insignificant impact deviations. Ecoinvent 3.0 emission inventory for the diesel car was additionally updated with recent real-world close emission values and revealed strong increases over four midpoint impact categories, when matched with the standard Ecoinvent 3.0 emission inventory. Producing batteries with photovoltaic electricity instead of Chinese coal-based electricity decreases climate impacts of battery production by 69%. Break-even mileages for the electric VW Caddy to pass the combustion engine models under various conditions in terms of climate change impact ranged from 17,000 to 310,000 km. Break-even mileages, when contrasting the VW Caddy and a mini car (SMART), which was as well electrified, did not show systematic differences. Also, CO_2-eq emissions in terms of passenger kilometers travelled (54–158 g CO_2-eq/PKT) are fairly similar based on 1 person travelling in the mini car and 1.57 persons in the mid-sized car (VW Caddy). Additionally, under optimized conditions (battery production and use phase utilizing renewable electricity), the two electric cars can compete well in terms of CO_2-eq emissions per passenger kilometer with other traffic modes (diesel bus, coach, trains) over lifetime. Only electric buses were found to have lower life cycle carbon emissions (27–52 g CO_2-eq/PKT) than the two electric passenger cars.

Keywords: BEV (battery electric vehicle); LCA; life cycle assessment; real-world driving; real-world life-cycle inventory; battery production; battery second use; battery size; break-even mileages; vehicle size effect; climate change impact; traffic modes; passenger kilometers travelled; diesel; electric bus

1. Introduction

Transport accounts for 23% of the global energy-related CO_2 emissions. Unlike other sectors, emissions from transport did not decrease but continued to increase annually by 2.5% on average between 2010 and 2015 [1]. In the European Union (EU), road transport in 2012 represented 82% of the total transport-related final energy use, with passenger cars contributing 60% to this share [2]. Electrification is seen as an essential element to decrease CO_2 emissions and resource use of the transport sector [1]. Some institutions project "zero emissions" when full electrification is achieved [3]. Zero carbon emissions, however, can only be achieved during the use phase of a vehicle and only if understood as the absence of direct emissions from a combustion engine. Considering that any electricity source, even a renewable one, will lead to some life cycle carbon emissions, an electric vehicle can deliver low, but never zero, carbon emissions [4]. However, future transportation modes as well as new vehicles need to be decarbonized as much as possible. The degree of de-carbonization can be best evaluated through standardized life-cycle assessment (LCA) in addition to the more common well-to-wheel analysis (for a comparison, see Moro and Helmers 2017 [5]). Although many LCA studies solely focus on climate change [6], further impact categories must be considered to avoid unintended environmental consequences. This is a lesson learned from the European transport emission policy of the past two decades: the primary focus on saving CO_2 led to EU policy boosting diesel cars to the disadvantage of petrol cars [7,8], which resulted in massive additional NO_x emissions and subsequent health costs [9,10]. In this context, it is important to realize that, so far, during LCA modelling, real-world NO_x emissions are not considered (and integrated in databases); instead, type-approval data from laboratory measurements are considered, which are lower by approximately one order of magnitude (e.g., reference [11]). So far, the insights resulting from the so-called "diesel scandal" [12] have not yet entered the life-cycle modelling of combustion engine vehicles, which may falsify the comparison of electric and combustion engine vehicles to some extent: While the NO_x emissions of petrol cars were reduced continuously according to the legislative emission thresholds following Euro 1 to Euro 6 stages in Europe, the NO_x emissions of diesel cars in 2012 were even higher than in 1993. These striking insights first became public due to two long-term remote-sensing campaigns in Europe collecting hundreds of thousands of measurements (e.g., references [11,13]). Real-world NO_x emissions of diesel cars did not begin to decrease before the year 2015. Corresponding to this, there has been no significant difference until 2015 in the NO_x emissions between Euro 5 and Euro 6 diesel cars [14].

In this context, it appears essential to cover not only the climate change impact, but many more impact categories in LCA of electric vehicles. However, this information is largely missing. From our counting, only a minority of 23 studies out of 85 peer-reviewed LCA studies on electric cars, published between 2010 and 2019, cover on average seven impact categories beyond climate change impact (e.g., the multi-impact studies from references [15,16]). Our own earlier study on electrifying a SMART [17] is the only one covering all 18 ReCiPe impact categories so far.

Although common agreement seems to exist that electric vehicles are the key technology to shift road traffic into a sustainable future [18,19], there is a discussion with regard to optimizing them. The electrification of luxury class or sport utility vehicles (SUVs) has been criticized because of huge batteries needed and corresponding weight increases [20,21]. Ellingsen et al. [22] investigated this size effect, concluding that smaller electric vehicles (EVs), equipped with smaller batteries, are more quickly overtaking combustion engine cars with respect to the carbon footprint (also see [23]). The question arises whether these problems have been adequately addressed in detail by life cycle assessment so far—almost all LCA reports quantifying the impacts of electric vehicles are based on standardized inventories and type approval registration data (reviewed in reference [24]). The vast majority of LCA results published so far is based on virtual (non-existing) vehicles traced back to the inventory of a VW Golf A4 from the year 2000, and still employed 17 years later (e.g., reference [25]). Also the fuel/electricity consumption in the use phase has been standardized (e.g., [26]), which, depending on the carbon footprint of the energy supply, can be the dominating impact throughout the life cycle (e.g., references [4,27]). Within the 85 peer-reviewed LCA studies published between 2010 and 2019, we could not identify another study (next to our own earlier study from 2017) about electrifying a SMART (Helmers et al. [17]) that captures the impacts of a real vehicle. In the same selection, only four LCA studies included a documentation of full material cakes of the vehicles [28–31].

High divergences between type-approval and real-world CO_2-emissions are well known for conventional passenger cars in the EU reaching above 40% in the year 2015 [32,33]. When it comes to electric cars, the type-approval to real-world deviations of measured electricity consumption as quantified in Europe range from +25% in Germany [34] to +34% in Finland under summer conditions, on top of that +31% when switching from summer temperatures to −20 °C in winter [35]. Concluding, the common picture of electric cars' life cycle impacts should be verified or refined based on real existing vehicles. As for example, the influence of an EV's size on its energy consumption and relative environmental impacts has to be analyzed under real-world conditions.

Disappointingly, even IPCC reports modelling the climate change impact of the transportation system only mention infrastructure and production costs but did not quantify or consider them in the most recent report [36]. Moreover, their modelling seems to be based on vehicle type approval CO_2 emissions [36] measured under laboratory conditions and reported by the manufacturers, which still seems to be common practice when it comes to discussing vehicle emissions in the public, however misleading it is [37].

Battery production has been identified as causing the second most important impact in an electric vehicles' lifecycle (e.g., references [4,28,38]), next to the use phase. When the life cycle impacts of the first electric vehicles were modelled, there were doubts whether the battery would survive more than 100,000 to 150,000 km [39]. Today, batteries can offer > 90% of the original capacity even at 200,000 km [40,41]. Use phase mileages between 150,000 and 200,000 km were most often applied in scientific reports [26,42].

In a predecessor project to the present report, the impact of an electric SMART's Li-ion cells produced in China was found to dominate throughout the life cycle across five impact categories, which was traced back to electricity supply dominated by coal fired power plants in China [17]. To conclude, electricity provision alternatives during battery production are expected to establish an essential sensitivity parameter when it comes to the question, if and when—at which mileage—an electric vehicle trumps a combustion vehicle in lifetime impact. We believe such investigations are essential in the present situation in which massive investments into battery cell production are made globally. Additionally, a further opportunity to decrease impacts from battery production evolved. Today, batteries do not have to be recycled after having reached 80% of their initial state-of-health, a value which is defined as the batteries' end-of-life criterion in a car [43]. Actually, the battery can be further moved to a stationary storage for fluctuating renewable electricity [44–46]. The influence of this battery second use case on the BEV's life cycle impact must be examined accordingly.

2. Materials and Methods

2.1. General Purpose

We address the shortcomings described above by compiling our own life-cycle inventory data obtained from a laboratory project in which a SMART Fortwo and a Volkswagen Caddy were electrified between 2011 and 2016. In a unique point of view, we can thus compare combustion engine and electric alternatives based on the same vehicle gliders converted in our laboratory.

The energy consumption of these vehicles was measured on the road, and the material cakes of these cars were documented during dismantling and re-assembly for the most part. Thus, this project seeks to deliver an alternative materials cake for future modelling. Major impact results obtained from the SMART conversion project were already published [17], and the impact assessment of the VW Caddy is reported here and compared with that of the smaller SMART vehicle.

The results contribute new and more accurate life-cycle inventory data, capturing the actual environmental impacts of electric vehicles under real-world operating conditions on the road.

2.2. Modelling Approaches

Goal, Scope, Software, and Databases

The goal of this study is to provide a comparative LCA of an electric car vs. combustion engine counterparts based on conditions which are as close to real-world as possible. The following scope items have been defined: BEV and ICEV versions are 1:1 comparable because they were (dis-) assembled in our workshop and tested under real-world conditions. The foreground and background systems and the boundaries are shown in Figure 1 and are similar to those in the previous study [17]. Our models describing the combustion engine vehicles included 409 input/intermediate/output materials, emissions, wastes, and amounts of energy (electric VW Caddy: 413). Also, 143 process modules (transitions) describe the system of the combustion engine vehicles (electric VW Caddy: 262).

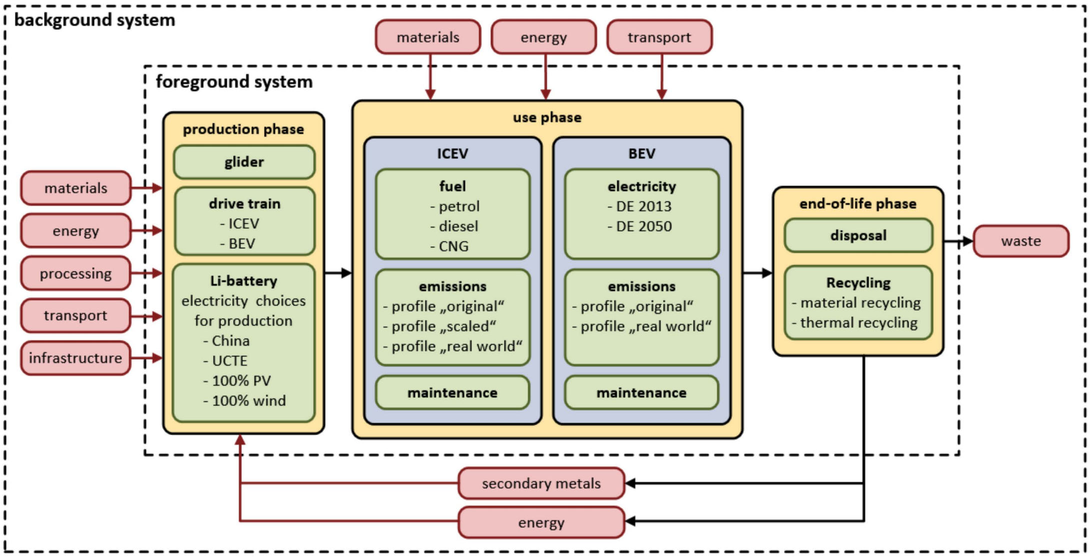


Figure 1. Product system as modelled in this project (modified from Habermacher 2011 [47], which is based on Althaus and Gauch, 2010 [48]). ICEV = integrated combustion engine vehicle, BEV = battery electric vehicle, UCTE electricity = see Table 3, CNG = compressed natural gas, DE 2013/2050 electricity = see Table 1. For metal recovery percentages, see Supplement #1 in Supplementary Materials.

The use phase is defined here as 150,000 km of mileage. For climate change (CC) impact modelling, we considered an extended use phase of 200,000 km in addition.

For quantifying break-even mileages, no limit was put on lifetime mileage. Continuous CC impacts per mileage driven by both the electric SMART and the electric VW Caddy were calculated and compared with the ICE versions until the mileages when the BEV undercuts the CC impact of the respective ICE versions (break-even). At this, the four different battery production scenarios (battery China, UCTE, PV, wind), the two different electricity mixes in the use phase (DE 2013 and DE 2050) as well as the battery second use case (with and without) and, additionally for the Caddy, two different battery sizes (25.9 and 51.8 kWh) were considered.

Altogether, we quantified the following sensitivity parameters and combined them:

- Size of the car (small vs. mid-sized, carbon footprint only);
- Emission profile (laboratory based vs. real-world);
- Fossil fuel choice (diesel, petrol, natural gas);
- Electricity choices during battery production and use phase;
- Battery size and battery second use;
- Mileage (150,000 and 200,000 km).

For the impact assessment, the ReCiPe method (2012 version) was applied, and its results covering all 18 impact categories are reported here. Results have been recalculated in terms of impact equivalents/km as functional unit, which means the life cycle impact was divided by 150,000 km or 200,000 km, respectively, where useful (results available in Tables S13 and S14 in Supplement #2 in Supplementary Materials).

The LCA modelling was performed between 2015 and the end of 2018 with Umberto 5.6 software connected to the Ecoinvent (Ei) database. The relevant Ei modules are listed in the Table S12 in

Supplement #2 in Supplementary Materials. Ei is presently operating version 3.6 (Ecoinvent 2020) [49], while version 2.2 is still available for modelling. Continued modelling based on Ei 2.2 data was essential to maintain comparability with the earlier sister project [17]. Additionally, a majority of current scientific LCA literature on electric vehicles is based on Ei 2.2 (e.g., [22]). However, to enable the connection to the Ei version 3 database, inventories used here are updated with emission numbers specified under Ei 3 as an additional sensitivity (see below); Section 3.1 provides a comparison of impacts from both inventories.

Sensitivity combinations: guide to the principal modelling approaches (VW Caddy)

(A) ICEV (Vehicle #1–6, Table 1)

The three combustion engine choices are based on the petrol version, while the diesel and natural gas vehicle just deviates in the use phase inventory. When it comes to modelling the use phase of the combustion engine vehicles (#1–6, Table 1) they are based on the Ecoinvent modules "operation, passenger car, petrol, EURO 5" and "operation, passenger car, diesel, EURO 5," respectively. Modelling the ICEV use phase is split into three alternatives considering the fuel consumption, combustion engine and abrasion emissions:

(1) "Euro 5 original" indicates that the original Ei emission module is kept, but on input side the fuel consumption and on the output side the corresponding CO_2 emissions were adjusted to the respective numbers measured. This corresponds to the standard (recommended) modelling attempt, but the approximation to the reality is limited—although many emissions vary relative to fuel consumption, they are kept constant here.

(2) "Euro 5 scaled" (Table 1), on the other hand, indicates that all combustion engine emissions are linearly corrected relative to the fuel consumption. This is performed by quantifying a mileage to run the model corresponding to the individual fuel consumption specified vs. the standardized fuel consumption fixed by the module. Both modelling cases due to (1) and (2) represent simplifications—neither are emissions always constant, nor are they fully to scale with respect to the fuel consumption.

(3) The third modelling alternative of the combustion engine vehicle ("Euro 5 real-world") is based on "Euro 5 original" but includes two deviations. On the one hand, a high number of additional chemical species were manually added to the modules "operation, passenger car, petrol, EURO 5" and "operation, passenger car, diesel, EURO 5," respectively. The respective emission numbers are based on the Ei 3 amendments as published by Simons (2013) [50]. On the other hand, and specifically for the diesel car (#6, Table 1), the emissions of a few species were updated to better describing real-world emissions as documented in recent scientific publications (see more detailed explanations in Section 2.3.6).

(B) BEV (vehicle #7–13, Table 1).

Quantifying the abrasion emissions of the electrified VW Caddy is based on two alternatives only. First, the use phase emission modelling is based on the Ei module "operation, passenger car, electric, $LiMn_2O_4$." Generally, we modified this Ei module by updating the weight of the vehicle (which is 1632 kg in the original module) thus scaling all use phase emissions of the electric vehicle, resulting in the emission profile "abrasion original" (Table 1). In a second step multiple additional chemical species taken from Ei 3 (provided in Simons 2013) [50] were added, this way building an emission profile called "abrasion real-world" (Table 1, see more detailed explanations in Section 2.3.6).

Finally, the EV is modelled based on two alternative electricity choices depicting the use phase: (1) Average net electricity from 2013 in Germany, which, regarding its power plant mix and climate change (CC) impact, is close to the European average mix [5]. (2) A realistic German renewable electricity mix of the future, called DE 2050 (Table 1). Additionally, the modelling of the electric vehicles distinguishes four different types of electricity provision during battery production (Table 1). For explanations see Section 2.3.3.

Table 1. Directory of main models quantified (VW Caddy).

<table>
<tr><th rowspan="2">VW Caddy Propulsion (Fuel)</th><th colspan="3">Main Sensitivity Parameters (1–3)</th><th rowspan="2">Vehicle Number #</th></tr>
<tr><th>1: Electricity Choice During Battery Production (See also Table 3)</th><th>2: Emission Profile</th><th>3: Use Phase Electricity Choice</th></tr>
<tr><td rowspan="3">Petrol</td><td rowspan="7"></td><td>Euro 5 original</td><td rowspan="7"></td><td>1</td></tr>
<tr><td>Euro 5 scaled</td><td>2</td></tr>
<tr><td>Euro 5 real-world</td><td>3</td></tr>
<tr><td>Natural gas (CNG)</td><td>Euro 5 real-world</td><td>3a</td></tr>
<tr><td rowspan="3">Diesel</td><td>Euro 5 original</td><td>4</td></tr>
<tr><td>Euro 5 scaled</td><td>5</td></tr>
<tr><td>Euro 5 real-world</td><td>6</td></tr>
<tr><td rowspan="7">Electric</td><td rowspan="4">China</td><td rowspan="2">abrasion original</td><td>DE 2013 *</td><td>7</td></tr>
<tr><td>DE 2050 **</td><td>8</td></tr>
<tr><td rowspan="2">abrasion real-world</td><td>DE 2013 *</td><td>9</td></tr>
<tr><td>DE 2050 **</td><td>10/10a</td></tr>
<tr><td>European average</td><td>abrasion real-world</td><td>DE 2013 *</td><td>11</td></tr>
<tr><td>100% PV</td><td>abrasion real-world</td><td>DE 2013 *</td><td>12</td></tr>
<tr><td>100% wind</td><td>abrasion real-world</td><td>DE 2013 *</td><td>13</td></tr>
</table>

(*) DE 2013 = German grid electricity of 2013, 707.4 g CO_2-eq/kWh (Note: this includes self-consumption of power plants and all losses along the grid); (**) DE 2050 = future renewable grid electricity mix proposed for Germany, 130.6 g CO_2-eq/kWh (for details and justification see Helmers et al. 2017 [17]). For electricity carbon footprints during battery production see Table 3. Model 3a extrapolated to CNG (compressed natural gas) use based on Model 3. Model 10a: battery capacity doubled. All models based on materials substitution as EOL modelling choice.

2.3. Inventory Development

2.3.1. Vehicle Composition, Assembly, and Use

The material cakes were developed from the ground up which starts at material composition data provided by manufacturers. Such data are usually restricted or rudimentary and need to be supplemented by own measurements and literature information. This is the case for the VW Caddy in total (Table 2): Volkswagen published a shortened composition specifying the percentages of nine groups of materials—steel and ferrous metals, light metals, non-ferrous metals, special metals, polymer materials, process polymers, further materials, electronics and electrics, operating materials, and accessories (source B in Table 2). These data were taken as a frame to develop a more detailed material cake (Tables S1–S5 in Supplement #2 in Supplementary Materials), particularly by utilizing detailed composition data provided by Habermacher (2011) [47], who also worked on the VW Golf (a sister model of the VW Caddy investigated here). Propellants were removed because the material balance refers to an empty vehicle. Generally, percentages of materials compiled this way were related to the measured mass of the particular car electrified in the laboratory and to its dismantled components individually weighed. Additionally, the materials composition of the engine, the gearbox, and other parts of the vehicle have been quantified separately and then related to the overall composition data. The combustion engine and the original gearbox of the VW Caddy were not disassembled but were instead sold and re-used. The materials composition of the petrol engine was obtained from a Renault publication because the combustion engine specified was very similar in replacement, weight and power (Table 2 and Supplement #1 in Supplementary Materials).

In the next step, the materials quantified were allocated to Ei modules (see Supplements #1 and #2 in Supplementary Materials). The amount of energy necessary for glider production was taken from Habermacher (2011) [47] and rescaled. Missing data (e.g., process loss of materials, manufacturing processes) were taken from Notter et al. (2010) [51]. Expenditures due to transportation of all parts from the respective factories to the Volkswagen plant in Poland (original fabrication of the Caddy) and, later, to the workshop for electric conversion were considered in the inventory (Table 2).

All VW Caddy ICEV versions are modelled based on the same material cakes of the petrol version, as displayed in Table 2. The material cake of the SMART electrified earlier in our laboratory is first published here and contrasted with that of the VW Caddy (see Supplementary Materials). Electrification procedures and technologies were similar in both cases. The only technical difference between the two vehicles was that the SMART kept the original gearbox (in which one gear was fixed), while a specialized one-speed gearbox was mounted during the electrification of the VW Caddy (Table 2).

Table 2. VW Caddy* production inventory guide (based on a petrol engine vehicle which was subsequently electrified in the author's laboratory). For more detailed inventory data, particularly of the electrified vehicle, see the Supplements #1 and #2 in Supplementary Materials.

Category	Component (Specification)	Location of Production (Distances)	Determination of Weight and Composition	Composition Details (Total Weight)
Glider		200 km away from Poznań, Poland	A, B, C	see Tables S1 and S2 in Supplement #2 in Supplementary Materials
ICEV powertrain (for complete composition see Tables S3–S8 in Supplement #2 in Supplementary Materials)	Motor (1.6L-petrol engine, 75 kW, 148 Nm, manufacturer ID: BGU 196175)	Salzgitter, Germany	A, D, E	(157 kg) 70.2% conventional steel, 17.3% Al, 6.5% plastics, 2.3% stainless steel, 1.1% rubber, 0.5% Cu, 1.5% polyamide, 0.6% polypropylene
	gearbox	Kassel, Germany	A, E	(38.5 kg) 30.9% Al, 69.1% high-tensile steel
	Pb battery (61 Ah)	Hannover, Germany	A, F	(16.7 kg) 68.3% Pb, 14% H_2O, 8% H_2SO_4, 4.1% PP, 2% fiber glass, 1.9% PE, 1% Cu, 0.7% Sb, 0.03% As
	remaining parts (e.g., starter, exhaust system, fuel pump)	400 km average	A	(102.6 kg) see Table S5 in Supplement #2 in Supplementary Materials
BEV powertrain (for complete composition see Supplementary Tables S6–S8)	Motor (FIMEA type N 80, 65 kW, 500 Nm)	Liscate (Italy)	manufacturer data	(128 kg) 34.4% Al, 52.3% high-tensile steel, 10.95% Cu, 2.35% PE
	Gearbox (Novatec AXLE ZG0302 TG050200)	Palazzuolo sul Senio (Italy)	A, E	(27 kg) 30.9% Al, 69.1% high-tensile steel
	Pb battery 26 Ah (RPTechnik, type RPower OGiV 12260)	Rodgau (Germany)	A, F	(8.8 kg) composition equally to ICEV Pb battery
	powerpac (voltage converter, inverter, control device)	Ranica (Italy)	A	(23 kg) composition due to Habermacher (2011) [47]
	charger (TC Charger type TCCH-H192V-36A)	Hangzhou (China)	A, C	(20 kg) composition due to Habermacher (2011) [47]
	BMS (LIGOO type EK-FT-12)	Hefei (China)	A, G	(7.3 kg) 50% Cu, 40% stainless steel, 10% integrated circuits
	remaining parts (e.g. cables, holders, water pump, vacuum pump	400 km average	A	(138.3 kg) see Table S8 in Supplement #2 in Supplementary Materials
Final assembly (ICEV)		Poznań (Poland)	A	detailed weight information provided in the Supplements #1 and #2 in Supplementary Materials
Final assembly (BEV), electric conversion		Birkenfeld (Germany)	A	

(*) VW Caddy Life, Type 2K, first registration 2005, electric conversion 2013–2015 at mileage 45,000 km. Detailed composition data in Tables S1–S10 in Supplement #2 in Supplementary Materials. (A) own measurements. (B) VW (2008) [52]. (C) Habermacher (2011) [47]. (D) Renault (2011) [53]. (E) Notter et al. (2010) [51]. (F) Hawkins et al. 2013 [15] (G) Majeau-Bettez et al. (2011) [54]. PE = Polyethylene, PP = Polypropylene.

2.3.2. Use Phase Energy Consumption

For the VW Caddy, we quantified a consumption of 8.89 L petrol/100 km before the electric conversion and 23.57 kWh/100 km afterward (averaging operation in city, rural, autobahn equally, including 8% of charging losses as measured for the EV). The same route of 94 km length was driven with the vehicle before and after electrification to quantify the energy consumption. We modelled with a consumption of 7.02 L/100 km for the analogous 77 kW-diesel model based on 77 vehicles by www.Spritmonitor.de at the time of database access (2015) [55].

As for the CNG consumption of the natural gas model, this is based on an 80 kW Caddy version produced by Volkswagen AG since many years. The German internet platform Spritmonitor.de (2018) [56] revealed a CNG consumption of 5.99 kg CNG/100 km on average, based on measurements from 456 cars on the streets. The analogous data of the mini car SMART Fortwo are 5.3 L petrol/100 km before, and 13.4 kWh/100 km after electric conversion, respectively [17]. Consumption data taken from internet platforms like Spritmonitor.de (2018) allow to approximate a representative energy consumption, which is much more real-world close than the fuel consumption specified due to type approval of a vehicle (e.g., references [33,57]).

2.3.3. Electricity for Battery Cell Production

A major impact during battery production stems from the high amount of electricity spent in the Li-Ion cell making. We base the electricity consumption during cell production on Majeau-Bettez et al. [54,58]. Interestingly, Majeau-Bettez et al. [58] assume the same electricity demand in the cell production (27 MJ/kg) for both $LiFePO_4$ and Li-ion cells based on a $NiCoMnO_2$-Chemistry (the latter preferred by OEM carmakers).

Whereas electric vehicles can charge electricity from diverse sources and at varying locations, battery cell production takes place in specific factories. Such production plants may locally be provided by 100% renewable electricity, which is why we considered 100% PV and 100% wind electricity, respectively, as modelling choices (Table 3). A battery cell production under provision of 100% PV electricity has been promised by Tesla, Inc. ("gigafactory"; see Tesla, 2018) [59]. A battery cell production under provision of 100% wind electricity is still missing, to the best of our knowledge, but might be a future option in Europe.

Table 3. Electricity provision alternatives assumed during battery cell production.

Label		**China**	**European Average (UCTE [b] 2004, as Utilized by Ecoinvent until 2014)**	**PV**	**Wind**
Fossil Electricity Production at Power Plant (%)	**Coal**	**78.5 [a]**	**26.7 [c]**		
	Fossil (All)	**81.7 [a]**	**51.1 [a]**		
g CO_2-eq/kWh [d]		1180	531	92.5	15.8
Description/ Ecoinvent modules applied		"medium voltage, at grid", China	"medium voltage, at grid, UCTE [b]"	"medium voltage, at grid [DE]" (Ecoinvent 2.2), adapted to 100% renewable each	

(a) Frischknecht et al. 2007 [60]; (b) UCTE = Union for the Coordination of the Transmission of Electricity, now ENTSO-E, see www.entsoe.eu. (c) EU (28) in 2013, taken from taken from Fehrenbach et al. 2016 [61]. (d) quantified using independent LCI (Life Cycle impact) models based on one kWh of each electricity choice. PV = photovoltaic.

The Li-ion cells for the VW Caddy were in fact produced in China with carbon-intensive electricity of 1180 g CO_2-eq/kWh (Table 3). We call this "Chinese electricity" in the following, while acknowledging that there is a strong variability in the carbon footprint of electricity production throughout China [62]. However, most of the Chinese battery production is located in the coastal provinces of Guangdong and Jiangsu [63], whose reported electricity mix and resulting carbon footprint closely resembles the Chinese electricity footprint utilized in this work, even if losses in the grid are considered [62]. As a comparison, we also ran a model that assumed battery production using UCTE electricity (Table 3) from the year 2004, the basic European electricity mix provided under the Ei database in 2014 [17]. This electricity mix has a carbon footprint of 531 g CO_2-eq/kWh, close to the average European electricity production impact recently reported [5] and similar to the European electricity mixes applied for battery production by Majeau-Bettez et al. [54] and Ellingsen et al. [64], allowing the connection of our results to these data published. Until today, the carbon footprint of the European electricity production has remained almost unchanged when compared to UCTE 2004. In 2018, it amounted to 521.74 g CO_2-eq/kWh quantified by LCA (AIB 2019) [65].

Two further energy options editing the Ei module "electricity, medium voltage, at grid [DE]" are considered in additional models, namely 100% PV and 100% wind electricity (see Table 3). A number of European countries base their renewable electricity production mainly on wind energy, and there are regions in central/northern Europe with a surplus in wind energy production [66], such as northern Germany.

2.3.4. Battery Chemistry Alternatives

The battery impact covers the Li-ion cells only (as equally handled by Majeau-Bettez et al. [58] 2011b), while the cell container is allocated to the powertrain in our inventories. $LiFePO_4$-cells of the type SE180AHA from CALB (China) were utilized during electrification of the VW Caddy achieving 25.9 kWh of capacity compared to 14 kWh of capacity installed in the earlier project when electrifying the Smart [17]. As an alternative available at the time of vehicle electrification, Zhejiang GBS (China) produced $LiFeMnPO_4$-cells (type GBS-LFMP200AH). Prior to deciding for one of these cells, their respective battery chemistry impacts were modelled and compared.

The chemical inventories of the Li-Ion cells were generated starting with simplified chemical compositions provided by the battery producers when delivering the cells. CALB, for example, specified mass percentages each for the elements Fe, P, Li, Cu, Al, F, C, Mn, Ca, and Na and for the compounds polyethylene and graphite. Zheijang GBS specified mass percentages for Al, Cu, graphite, $LiFeMnPO_4$, lithiumhexafluoro-phosphate, and polypropylene. Although both compositions provided by the battery makers were certified to specify 100% of composition, essential components were missing as information on the chemical species. We kept the elemental composition as it was specified by the battery companies, but filled in the gaps in battery chemistry based on data by Majeau-Bettez et al. [54] and Yang et al. [67], respectively.

Both battery types contain the same inorganic materials with the exception of manganese (Mn concentrate, Figure 2), of which 14% is contained only in the $LiFeMnPO_4$ battery; however, the proportions of materials composing the Li-ion cells vary greatly between the battery types. (Figure 2). $LiFePO_4$-cells, on the other hand, contain much more Al (Figure 2). Detailed composition data are provided in Tables S9 and S10 in Supplement #2 in Supplementary Materials). We decided for the $LiFePO_4$-cells due to the reasons discussed below.

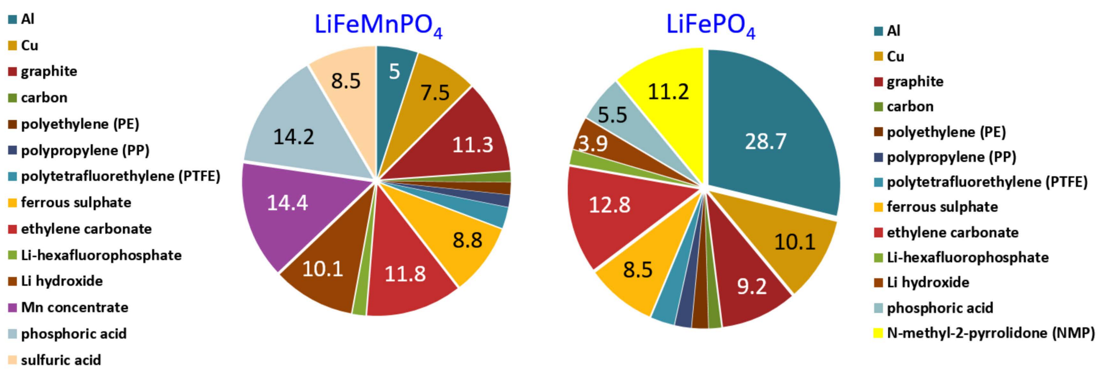


Figure 2. Material cakes of two Li-ion cells examined prior to the electric conversion of the VW Caddy (mass percentages displayed). For detailed composition see Tables S9 and S10 in Supplement #2 in Supplementary Materials.

Prior to the electrification of the VW Caddy in the university's workshop, the composition of the two Li-Ion-cells was modelled in order to identify possible impacts on the vehicle's life-cycle performance (the $LiFePO_4$-cells from CALB were earlier also taken electrifying the SMART, see Helmers et al. [17]). The impact comparison of $LiFePO_4$- and $LiFeMnPO_4$ battery cells exhibited the biggest differences in the midpoint category of mineral resource depletion (MRD), varying over several orders of magnitude. At first glance, this indicates that the choice of minerals/metals should have a strong influence in this category. Battery cell materials like Al, carbon, graphite, ferrous sulphate, and phosphoric and sulphuric acid look non-critical (0.00145–0.29 kg Fe-eq/kg), while there are higher MRD impacts from Li-hexafluorophosphate (2.38 kg Fe-eq/kg) and Cu (6.46 kg Fe-eq/kg), respectively. Mn, however, scores the highest result with 76.6 kg Fe-eq/kg, being responsible for 88% of the overall MRD impact based on $LiFeMnPO_4$-cells. However, this high MRD impact of Mn as indicated by the ReCiPe database is to be questioned. Mn is among the most abundant elements in the Earth's crust [68]. We conclude that the MRD differences identified here are not specific enough to lead a decision for or against one of the two chemical battery compositions. The CC relevant production impacts of the respective minerals are relatively similar (1–8.3 kg CO_2-eq/kg), within one order of magnitude. We conclude, on the basis of both CC and MRD impacts, that there is no urgent need to keep away from any one of the minerals listed here for battery production.

We also checked the list of critical raw materials provided by the EU (2017) [69]. Among battery-relevant materials, only cobalt and phosphorus are specified as critical raw materials (EU 2017). Cobalt is an essential component of prevailing commercial Li-ion cells based on $NiCoMnO_2$ chemistry as commonly used in automotive applications. Co is under criticism as a battery component because about 50% of cobalt on the world market stems from the Democratic republic of Congo, where mines commonly use child labor [70]. Cobalt is the only metal mentioned here on which data about availability and supply risks are extensively available due to an advanced LCA attempt [71]. However, both the $LiFe(Mn)PO_4$ and $LiFePO_4$ cells examined here for possible EV application do not contain Cobalt. The $LiFePO_4$ cells we finally decided for do not contain Mn, and less phosphorus than the $LiFe(Mn)PO_4$ cells available for a comparable price.

2.3.5. Battery Second Use

After use in electric vehicles, and prior to recycling, batteries may be transferred to stationary applications as a storage buffer for fluctuating renewable electricity. According to Casals et al. [72], this "second use" can decrease the vehicle's carbon footprint caused by the battery by 50%. Although not focusing specifically on the vehicle-related battery production impact, Ahmadi et al. [73] and Richa et al. [74] confirmed this finding qualitatively. In a meta study, the reduction of the battery GHG emissions attributable to the vehicle on a per-km basis was quantified to be 42% in case of a second use (ICCT 2018) [75], based on data from Neubauer et al. [76]. Recently May et al. [77] estimated the impact saving potential of a subsequent stationary use of the batteries at around even 50% of the production impact of the vehicle in total. Bobba et al. [78] also identified such savings, but, as in all other studies, very much depending on their specific scenario. We apply the second battery use here as an extrapolation case, based conservatively on the assumption of 50% savings in GHG emissions from battery production adjusted to the electric vehicle in case the battery is transferred to a later stationary use.

2.3.6. Emission Profiles Development

Ei has extensively revised the emission profiles when switching from Ei2.2 to version 3. Many new parameters were added (Table 4), and the data of some parameters were changed. This is documented in detail by Simons [50], from whom we have taken additional emission species and manually added them to the respective Ei2.2 modules to build the "real-world" emission profiles of the electric and conventional vehicles. Also, we evaluated the emission species for correctness, plausible magnitude and, in a few cases, replaced them by more real-world-oriented numbers (Table 5). By modelling these emission inventory alternatives, we can use them as sensitivity parameters and search for possible effects changes.

Table 4. Details of emission profiles advancement as modelled in this work.

Propulsion	Emission Profile/Sensitivity	Applied to Vehicle #, see Table 1	Number of Species Included	Comment/Origin
ICEV	Euro 5 original, Euro 5 scaled	1; 2; 4; 5	emissions to air: 25, to water: 6, to soil: 6	according to Ecoinvent 2.2
	Euro 5 real-world	3; 6	emissions to air: 63, to water: 31, to soil: 31	emission species added according to Ecoinvent 3, four diesel emission species corrected (see Table 5)
BEV	abrasion original	7; 8	emissions to air: 10, to water: 6, to soil: 6	according to Ecoinvent 2.2
	abrasion real-world	9–13	emissions to air: 36, to water: 31, to soil: 31	non-exhaust emission species added/adapted according to Ecoinvent 3

Table 5. Corrections made to diesel exhaust emissions as taken from Ecoinvent 3*. (Added to the emission profile "Euro 5 real-world," diesel vehicle model #6, see Table 1.)

Species Emitted	Ecoinvent 2.2 (Diesel Euro 5), and Applied for Vehicles #4+5 kg/km	Ecoinvent 3 (Diesel Euro 5, Simons 2013) [50], kg/km	Corrected (Diesel Euro 5, Applied for Vehicle #6) kg/km	Comments, Sources
SO_2	1.07×10^{-6}	1.06×10^{-6}	3.19×10^{-6}	corrected value from sulphur content in fuel plus lubrication oil combustion (Helmers 2010) [79]
CO (fossil)	5.09×10^{-4}	6.07×10^{-5}	2.55×10^{-4}	perhaps a data error in Simons (2013) [50]. The corrected value is calculated from the Ecoinvent 2.2 emission which should have been halved due to Simons (2013) [50]
NO_x	2.00×10^{-4}	9.38×10^{-5}	8.63×10^{-4}	updated according to data from remote-sensing campaigns (Tate 2013) [80]
Particu-lates > 10 µm	7.82×10^{-5}	1.19×10^{-5}	3.84×10^{-3}	originally abrasion considered only. We updated this emission according to data from remote-sensing campaigns (Tate 2013) [80]

* Simons (2013) [50].

Simons (2013) [50] reported emission data of 40 species due to non-exhaust emission factors, as employed by the new Ei3 database, divided according to the three sources tires, brakes, and road. We have taken these abrasion emission numbers, kept those for tires and road, and added them manually to the Ei2.2 BEV emission profile, building a new profile "abrasion real-world" (Table 4). An electric vehicle causes the same resuspension of particles from the street, as well as tire and road wear abrasion, compared to a combustion engine vehicle. Abrasion from brakes, however, can approach almost zero in case the EV uses a strong regenerative braking (GreenCarCongress 2016 [81]). On the other hand, recuperation may also be turned off in some EVs, which is why we still considered half of the brakes-related emissions of ICEV within our EV emission profile, now called "abrasion real-world" (Table 4).

Eighteen of the 22 species emitted by EVs due to Ei2.2 are metals (abrasion); particulates (three species) and heat are added as non-metal species. Still, in Ei3 all "non-exhaust" emission species are metals/inorganic species/elements except for particulates and heat.

Species emitted by petrol cars according to Ei3 were manually added to the Ei2.2 module, building the new "Euro 5-real world" emissions inventory (model #3, Table 1). However, when it comes to the emission inventory of the diesel car (model #6, Table 1), additional corrections were made to four emission species employed by Ei3 [50] due to significant real-world deviations noticed. These corrections are partly changing the magnitude of the species' emission and are displayed in Table 5. Slight corrections were made to SO_2 and CO: CO emissions increased by a factor of 4 (for explanation, see Table 5). We base further corrections on new insights derived from long-term remote sensing campaigns. However, vehicle emission remote sensing along streets covered only a few species so far—CO, HC, NO_x, NO_2, and PM_{10} (e.g., Tate 2013) [80]—while just the NO_x and PM_{10} values reported were usable. NO_x, however, increased by a factor of 9 (Table 5). This is in accordance with multiple scientific findings (e.g., references [11,13,82,83]).

The most substantial adjustment was made according to real-world PM_{10}—the number employed here for the corrected inventory is 323 times higher than the original number (Table 5), which considers particles from abrasion only. Particulate emissions in the sizes classes < 2.5 µm and 2.5–10 µm, however, remained unchanged in the adjusted emissions inventory. We also kept the acetaldehyde emissions from Ei2.2, a species missing in Simons [50]. Two important species are generally missing so far in the Ei3 database [50] and are also missing in our updated inventory, namely PN (particle number) and BC (black carbon), although emission data are available (e.g., [84]). PN and BC inhalation poses a considerable health risk to humans (e.g., reference [85]). In conclusion, even with our updated real-world-oriented emission profile it is not possible to fully account for all adverse health impacts of diesel car emissions by LCA.

3. Results and Discussion

3.1. Impact Differences Due to Variations in Emission Profiles

For better clarity, we removed models 3a and 13 from the following impact overview (Figure 3). Model #3a is just an expansion of model #3 based on CNG consumption (Table 1). Model #13 considers wind power instead of PV electricity for battery production, which results in slight impact variations only in comparison to model #12 (see Table S11 in Supplement #2 in Supplementary Materials). Both cases are separately discussed below.

First, the results reveal that in a minority of five of 18 impact categories the electric VW Caddy is having clear advantages over the combustion engine models (climate change, photochemical oxidant formation, fossil resource depletion, natural land transformation, and ozone depletion), while in 10 of 18 impact categories, a vice versa picture appears. The disadvantages for electric vehicles are not reflected in the single score endpoints, which almost mirror the climate change impacts (Figure 3). This goes back to the fact that the ReCiPe endpoint evaluation scheme is very much dominated by the climate change effect [86].

In the impact categories of terrestrial acidification, particulate matter formation, and marine eutrophication, the lowest impacts of both technologies are comparable (Figure 3). This resembles the picture found during life cycle modelling the electric and the combustion engine SMART with the exception that the LCIs in the category of marine eutrophication (ME) of both technologies are balanced at the VW Caddy, while the electric SMART exhibited larger impacts in this category [17].

For the first time, the results depicted in Figure 3 enable a comparison of the different use phase emission inventories. Comparing the impacts of vehicle models #1 and #2 (petrol "Euro 5 original" and "Euro 5 scaled" inventories), there are only small differences (Figure 3): midpoint impacts of climate change (CC), terrestrial acidification (TA), terrestrial ecotoxicity (TET), particulate matter formation (PMF), photochemical oxidant formation (POF), ME and the single score of model 2 ("Euro 5 scaled") are slightly higher (Figure 3). This is also the case when comparing midpoint impacts of the models #4 and #5 (diesel "Euro 5 original" and "Euro 5 scaled" inventories, see Table 1). Model #5 (Euro 5 scaled) shows slightly larger impacts in the midpoint categories of terrestrial acidification (TA), terrestrial ecotoxicity (TET), particulate matter formation (PMF), photochemical oxidant formation (POF), and marine eutrophication (ME). Concluding, adapting LCA inventory modelling to individual mileages did not result in significantly different impacts, as shown when comparing the respective modelling results ("Euro 5 original" vs. "Euro 5 scaled").

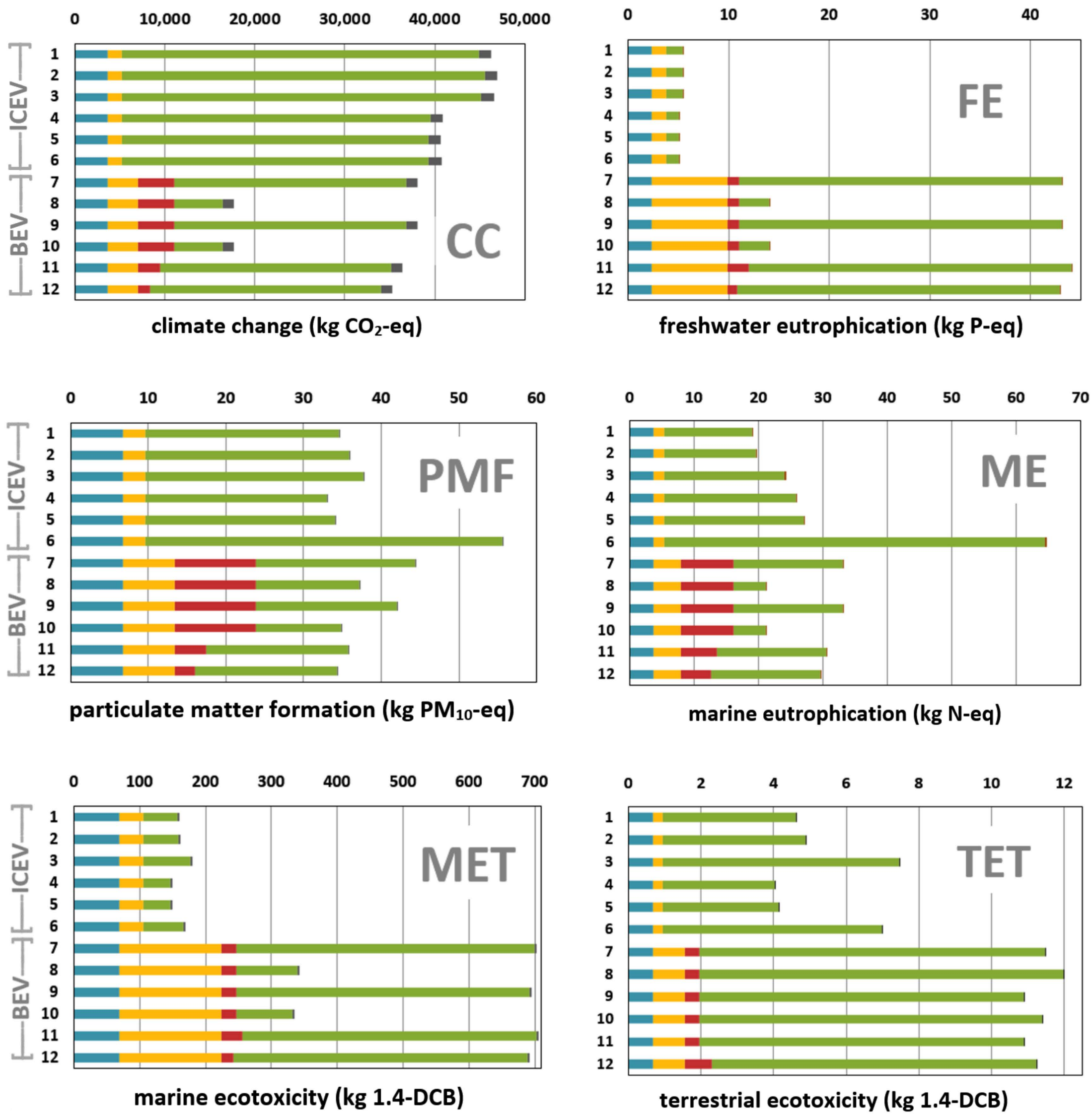
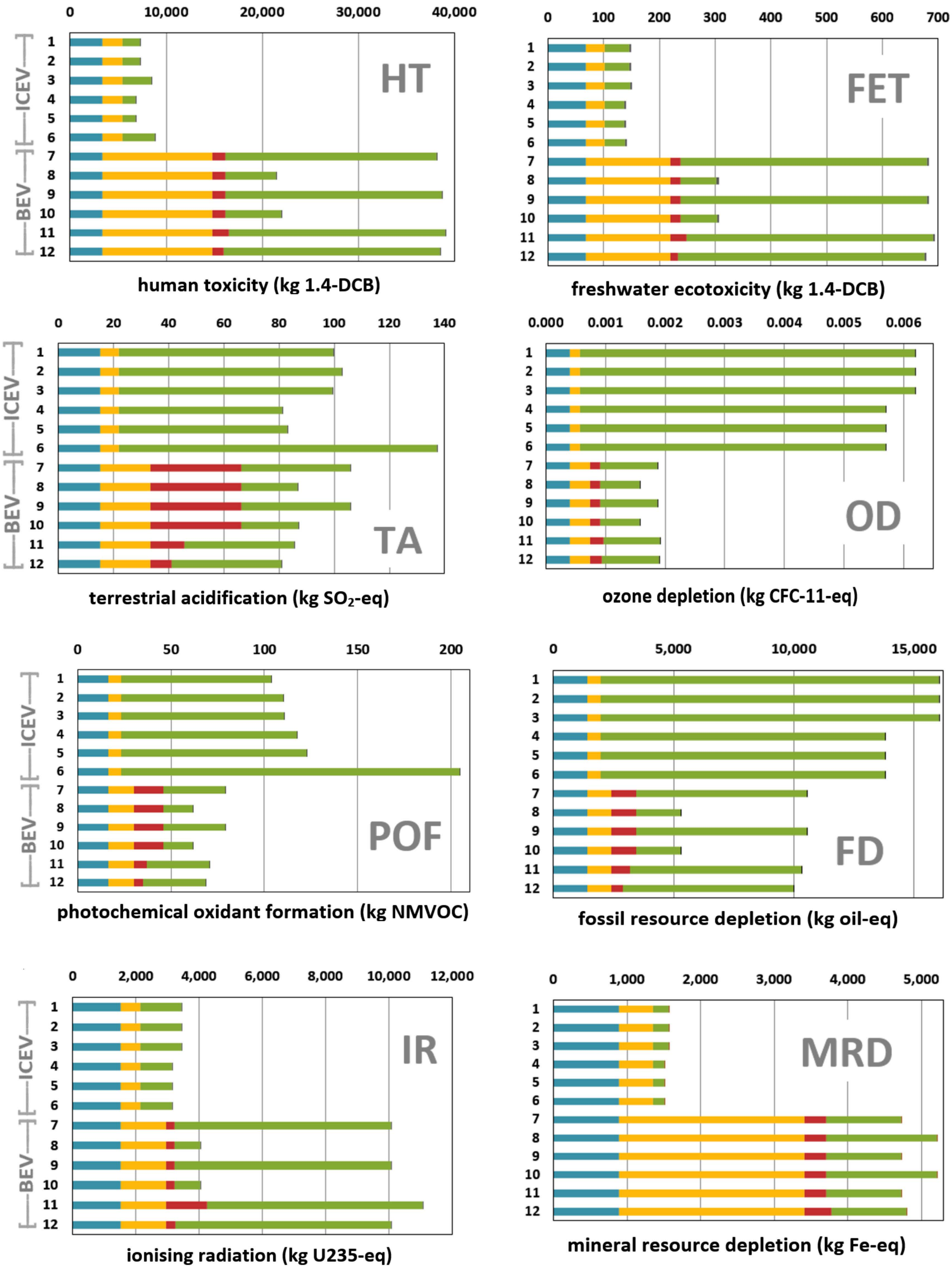


Figure 3. *Cont.*



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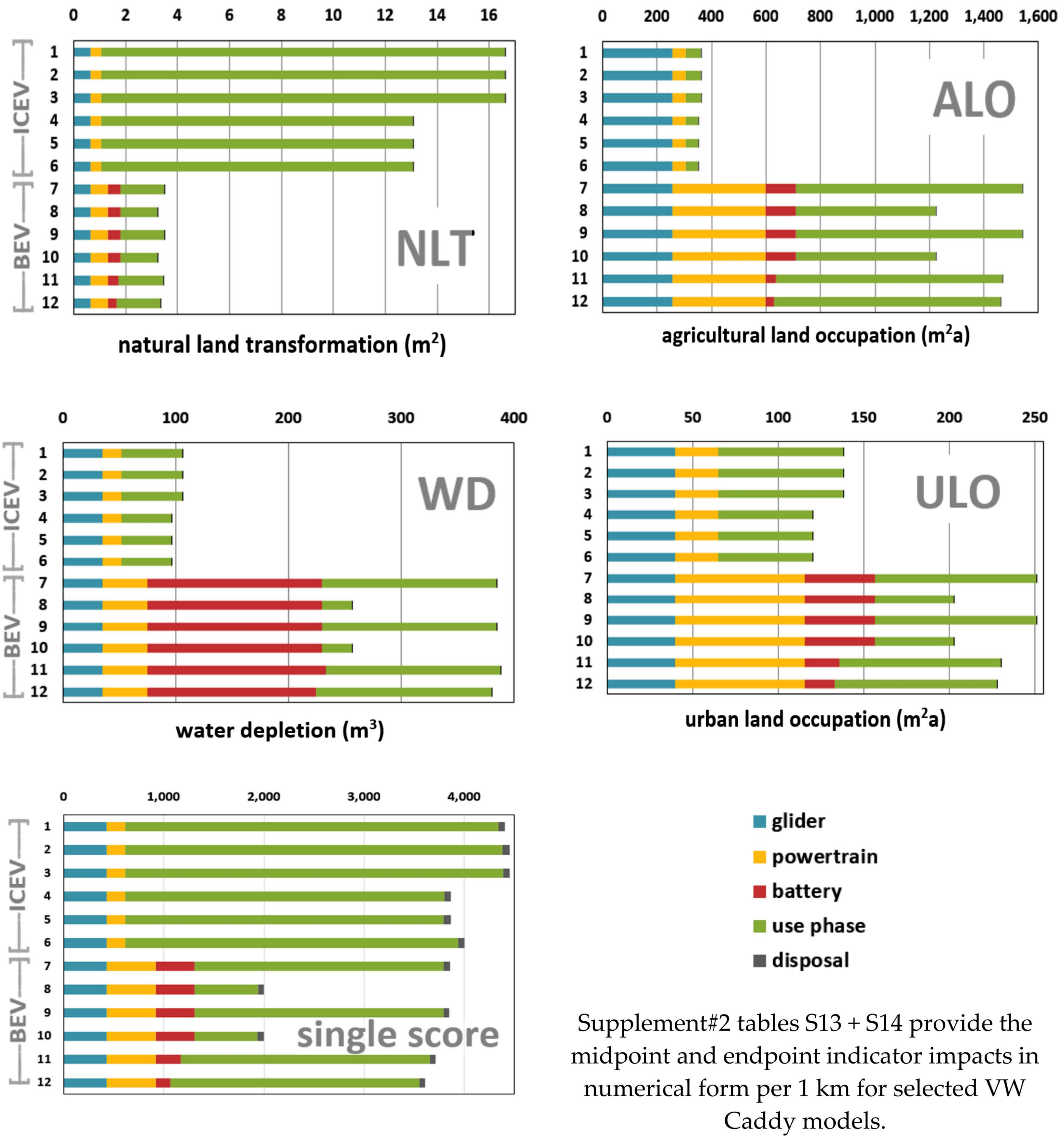

Figure 3. Ecoinvent (Ei) midpoint impacts and single score endpoints of VW Caddy production, use, and disposal. Impacts calculated for models #1–12 (for details, see Table 1). Impacts per 150,000 km of use phase.

Due to Ei3 emission species amendments [50] to the petrol VW Caddy a few midpoint categories scored slightly higher impacts of model #3 (particulate matter formation, PMF; marine ecotoxicity, MET; marine eutrophication, ME), compared to vehicle models 1 and 2 (Figure 3). Adding the new group of emissions (Ei3) to the petrol car model, however, has a significant impact on the terrestrial ecotoxicity (TET, Figure 3): Vehicle model 3 scores 57% higher in terrestrial ecotoxicity (TET)-LCI (Life cycle impact), when compared with petrol models 1 and 2 (averaged). The process module composition of the impact category terrestrial ecotoxicity (TET) of vehicle #3 exhibits that 53% of the LCI of terrestrial ecotoxicity (TET) can be traced back to the impact of the updated Ei modul "operation, passenger car, petrol EURO 5," while another 33% of the LCI of terrestrial ecotoxicity (TET) is due to impacts caused by the Ei module "petrol, low sulphur, at regional storage [CH]." Consequently, the newly added emission species (Ei3) may have resulted this additional impact. This is also the case for the LCI in human toxicity (Figure 3): The LCI in the impact category human toxicity (HT) of vehicle #3 increased by 17%, compared to the average of vehicles models #1 and #2 (both petrol).

Correcting the emissions of four species (Table 5) for diesel vehicle models (#6, Figure 3) led to more complex changes than in the impacts of the petrol vehicles. First, and analogous to petrol model #3, diesel model #6 exhibits an increase of impact in terrestrial ecotoxicity (TET) and human toxicity (HT, Figure 3), compared to the averaged LCIs from diesel vehicle models #4 and #5 (Figure 3). Despite this terrestrial ecotoxicity (TET) impact variations it can be concluded that in case of an unchanged adoption of the Ei3 emission numbers there would be no significant impact alterations, neither between the two attempts of mileage implementation (original vs. scaled), nor between the database advancement from Ei2.2 to Ei3.

Second, updating the emissions of four species in the inventory of diesel model #6 leads to noticeable deviations in the four midpoint categories of marine eutrophication (ME), terrestrial acidification (TA), particulate matter formation (PMF), and photochemical oxidant formation (POF) (Figure 3, see model #6 each), by 143%, 67%, 65%, and 70%, respectively, in comparison to the averaged LCIs of vehicle models #4 and #5.

This is based on the assumption that without adapting the respective emission numbers relative to real-world emissions (Table 5), the impact increases of diesel vehicle #6, compared to vehicle #4 and #5, would have been small or insignificant as it was observed comparing the petrol vehicle #3 with petrol vehicles #1 and #2 (see above). The corresponding original Ei3 emissions are all lower than those of Ei2 (Table 5).

The elevated impacts observed here are due to excess use phase emissions, clearly caused by the corrections applied (Table 5): As it can be calculated from the process modules composition of all four impact categories of marine eutrophication (ME), terrestrial acidification (TA), particulate matter formation (PMF), and photochemical oxidant formation (POF), respectively, the LCI in each is dominated by "operation, passenger car, diesel EURO 5" (to 78%, 53%, 59%, and 67%, respectively). In all these four impact categories, the second largest factor in the respective process modules compositions is "diesel, low sulphur, at regional storage [CH]" (to 12%, 28%, 21%, and 20%, respectively). This results in predominant use phase impacts over lifetime in the respective four impact categories (Figure 3).

To the best of our knowledge, such increased impacts as a result of an updated diesel vehicle emissions inventory was first presented by Bauer (2017) [87] but has not been documented further on in detail. Bauer (2017) [87] pointed to a 35% increase of lifecycle particulate matter formation (PMF) impact as well as to a 64% increase of lifecycle photochemical oxidant formation (POF) impact, when updating the diesel car emission inventory to be closer to real-world results, but did not go into further detail.

We found bigger LCI increases than Bauer (2017) [87] when applying real-world emissions, and detected increased impacts in two more impact categories. However, Bauer (2017) [87] used NO_x emissions elevated by a factor of 6, while our updated NO_x emissions increased by a factor of 9.2, relative to Ei3 (Table 5).

These excess LC impacts of diesel vehicle model #6 are hardly visible in the single score result that is just 4% higher than the single score indicator of models #4 and #5 (averaged). When averaging the elevated LCIs of diesel vehicle model #6 identified in the impact categories of marine eutrophication (ME), terrestrial acidification (TA), particulate matter formation (PMF), terrestrial ecotoxicity (TET), and photochemical oxidant formation (POF) for the 18 impact categories, it should result in an elevated single score LC impact of +18% of model #6. The dominance of the CC impact in the ReCiPe evaluation scheme [86] is blurring this effect (Figure 3).

In conclusion, switching from Ei2.2 to Ei3 emissions does not significantly change the environmental impacts of petrol and diesel cars, despite of the slightly increased impacts in terrestrial ecotoxicity and human toxicity, respectively. The high number of added chemical emission species in Ei3 does not cause significant increases in LCIs despite of the impact categories of terrestrial ecotoxicity (TET) and human toxicity (HT) (Figure 3). However, the real-world close update of a few species, emitted by diesel cars, changes the picture.

Interestingly the switch from Ei2.2 to Ei3 also has very small and negligible effects on the use phase impact in any impact category of the electric BEV (comparing models #7 and #9, Figure 3), although the number of emissions increased from 22 to 98 (Table 4).

3.2. Climate Change Impact Comparison of Diesel and Petrol Cars

Figure 3 displays the main vehicle types (VW Caddy) in order to identify the choices to optimize the climate change impact. The Caddy with the diesel engine, #6, scores 13% lower compared to the petrol propelled Caddy (Figure 3). We do not think this difference can be generalized, because our VW Caddy was not equipped with a particularly modern charged and downsized petrol engine as they are technical standard to date. Statistically, the use phase CO_2 emission of an average gasoline car is only 4% higher compared to a diesel car (T&E 2018 [88]). However, increased emissions of aged cars as they have been measured (e.g., reference [89]) are not considered in any LCA or well-to-wheel (WTW) report. Preliminary calculations on this issue revealed that aged diesel cars without (properly working) particulate filters may statistically add 8 g CO_2-equivalents/km on top of the direct CO_2 emissions on average over lifetime by black carbon emissions (Helmers et al. 2018 [90]). In conclusion, as in the majority of LCA reports, diesel cars also here seem to be favourable over petrol cars, but we believe this is misleading. The lack of representative emission data for aged cars is preventing a refinement of modelling so far. Consequently, we prefer to work with averages calculated from the petrol and diesel vehicle when comparing them to the electrification alternative.

3.3. Effects of Electricity Supply Choices on Battery Production Impacts

Results depicted in Figure 4 highlight the advantages of battery cells made under provision of renewable electricity. Particularly, human health and ecosystem related impacts decrease when substituting coal-based electricity (Figure 4). Human health–related endpoints for the Chinese electricity provision scenario are 3.4 times higher than those of the scenario assuming wind electricity only (Figure 4). The analogue difference in ecosystem related endpoints is a factor of 3.5 (Figure 4). This is reflected in individual midpoint deviations: Coal dominated electricity has particularly high impacts in photochemical oxidant formation (POF), particulate matter formation (PMF), agricultural and land occupation (ALO), terrestrial acidification (TA), fossil resource depletion (FD), and marine eutrophication (ME). UCTE electricity, on the other hand, exhibits larger impacts than the other scenarios regarding human toxicity (HT), ozone depletion (OD), water depletion (WD), and ionizing radiation (IR, the latter due to the nuclear power plants still available in Europe). Individual midpoints are reported as per 1 kWh electricity in Table S11 in Supplement #2 in Supplementary Materials. When switching from Chinese electricity toward 100% PV, the impacts in 14 categories decrease, on average by 43% per impact category (Table S11). On the other hand, there are slight increases in the impacts of the midpoint categories of ionizing radiation, ozone depletion, terrestrial ecotoxicity, and mineral resource depletion, respectively (Table S11). When using 100% wind electricity, on the other hand, even these impacts are lower than those under provision of Chinese electricity, despite of the impact in mineral resource depletion (Table S11).

Battery cells produced with coal-dominated electricity have a 3.8 times higher carbon footprint (156 kg CO_2-eq/kWh) than cells made with 100% wind electricity (41 kg CO_2-eq/kWh) (Figure 4, Table S11). Compared to 100% PV electricity, 100% wind electricity's carbon footprint is still an additional 16% lower. These findings coincide in principle with the 61–106 kg CO_2-eq/kWh reported in a recent review [91], decreasing from the 150–200 kg CO_2-eq/kWh reported earlier [92]. As other reports, also Romare and Dahllöf [92] conclude that the magnitude of battery carbon footprint is "nearly independent of the cell chemistry."

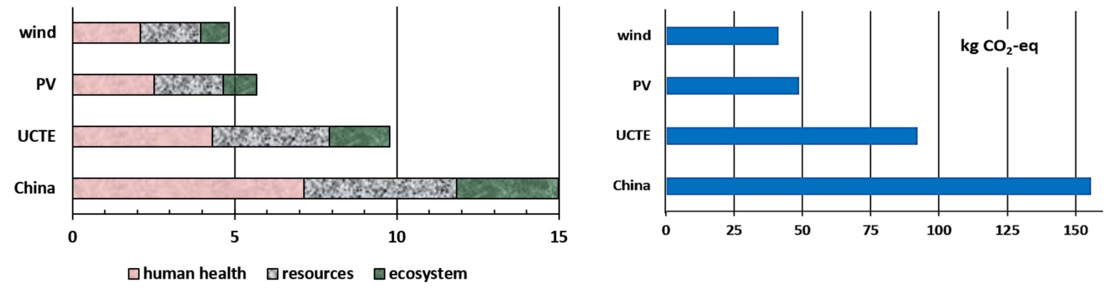

Figure 4. Battery production impacts as per 1 kWh of battery capacity. Left: endpoints. Right: climate change midpoints. Characteristics of the four kinds of electricity production are described in Table 3. (UCTE = European Union for the Coordination of the Transmission of Electricity, now ENTSO-E, see www.entsoe.eu). Numerical impacts for all categories provided in Table S11 in Supplement #2 in Supplementary Materials. PV = Photovoltaics.

Based on UCTE (2004) electricity, the CC impact due to direct electricity consumption during battery production accounts for 57% of the CO_2 equivalent emissions relative to the entire CC impact of Li-Ion cell production, as derived from the process modules composition (the remaining 43% are caused by the provision of the cell components). Ellingsen et al. (2013) [64] reported an average of 78% CO_2-eq emissions, due to direct electricity use during cell production under a comparable electricity mix. Under provision of Chinese electricity as shown in our modelling, direct electricity consumption accounts for 75% of CO_2-eq emissions during cell production, under 100% PV electricity this goes down to 18%, respectively. Under provision of wind electricity, the contribution of direct electricity consumption to the climate impacts of battery cell production is 0.4%, which is negligible. The carbon footprint of battery production under wind electricity consumption is thus almost completely dominated by the provision of the chemical/mineral battery components.

These findings suggest an enormous potential to mitigate greenhouse gas emissions by producing Li-ion battery cells with renewable electricity (Table 3). A battery cell production in a European country with a high proportion of coal-derived energy like Poland (88% coal, 92% fossil in total, according to Frischknecht et al. [60]) would even worsen the production backpack of the battery, while a European production with renewable electricity can deliver the savings quantified here.

Apart from the climate change impacts, the following impact categories particularly benefit from switching to renewable electricity during battery production—terrestrial acidification, particulate matter formation, photochemical oxidant formation, agricultural land occupation, urban land occupation, and marine eutrophication (Figure 3). The high impacts from electric powertrain production (Figure 3) are caused during the production of printed circuits. Details on this analysis can be found in Helmers et al. (2017) [17].

3.4. The Natural Gas Alternative and Effects of Electricity Supply Choices During Battery Production on the Lifetime Impacts

CNG vehicles tend to have lower pollutant emissions (e.g., Khan et al. [93]) than petrol vehicles. They are based on almost the same engine technology. Our approach (modelling the same chemical emissions of the petrol version) thus results in a slight impact overestimation of vehicle 3a, propelled with CNG, in impact categories like photochemical oxidant and particulate matter formation (compare Figure 3), which affects single score impact (Figure 5). The climate change impact of natural gas combustion was as at first glance quantified with an independent LCI model based on the Ei module "natural gas, burned in gas motor, for storage [DE]." This module revealed 214 g CO_2/kWh CNG, which was converted to 2.92 kg CO_2/kg CNG. We regard this as being unrealistically low because this would include only a 6% additional impact along the fuel supply chain. Well-to-tank efficiency, however, is 80.25% on average in CNG provision (reviewed in [39] Helmers & Marx 2012). We accordingly add 19.75% due to fuel chain expenses on top of the 5.99 kg CNG/100 km as measured for the VW Caddy, arriving at 198 g CO_2/km (well-to-wheel) for modelling the use phase.

CNG vehicle #3a (Table 1) illustrates the impacts of combustion engine vehicles propelled with natural gas, which in the use phase produce much lower CO_2 emissions than diesel and petrol cars (for European wide data, see Helmers et al. [8]). The electric VW Caddy cannot compete with the NG version (Figure 5), as long as electricity is provided by a mix close to the European average (Figure 5). This only changes when the BEV is charged with renewable electricity (Figure 5). The electric vehicle, however, comes with an additional production impact both due to the battery and the powertrain. Its powertrain production impact with 3.4 t CO_2-eq is 2.2 times higher compared to that of the conventional combustion engine car, according to the models run here (Figure 5).

Lifetime CC impacts of vehicles #11-13 represent a BEV similar to model #9, with the only difference of a reduced battery production impact, when switching from a coal-dominated electricity mix during battery production (vehicle #9) to average European electricity mix (#11), 100% photovoltaic electricity (#12), and 100% wind electricity (#13) during battery production (and at the same time keeping DE 2013 electricity supply in the use phase, Figure 5). The CC impact from battery production decreases this way from 4.0 t CO_2-eq (vehicle #9), to 2.4 t CO_2-eq (vehicle #11), 1.3 t CO_2-eq (vehicle #12), and 1.1 t CO_2-eq for vehicle #13, respectively, roughly a 50% reduction for every step excluding the last step from PV to wind electricity supply (Figure 5, all battery sizes: 25.9 kWh). This illustrates the relevant impacts during battery production, on the one hand, and the optimization potentials, on the other hand, affecting the whole life cycle of the vehicles. With a 25.9 kWh battery made under provision of PV or wind electricity, the electric Caddy is already advantageous compared to the ICEV alternatives, even when charging DE 2013 electricity mix.

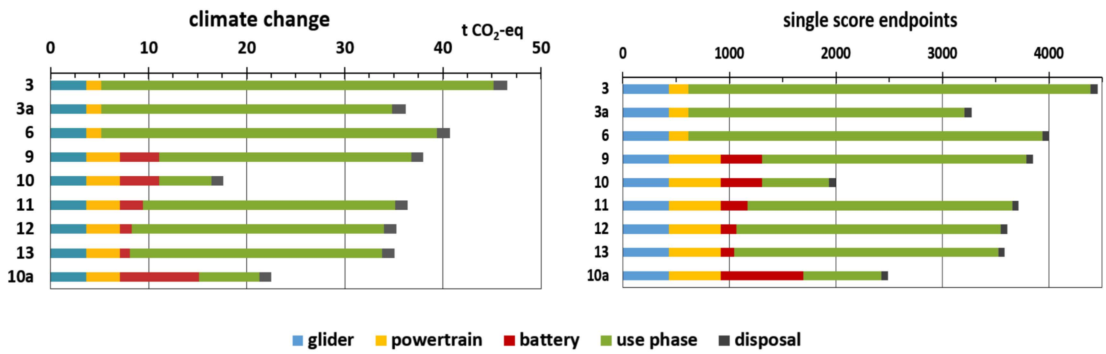

Figure 5. Lifetime climate change impact and single score endpoints results of main vehicle models (VW Caddy). The vehicles included 3–6 ICE and 9–13 BEV (see Table 1). Vehicle 3a propelled with CNG (compressed natural gas). 10a: Vehicle 10 modelled with twice the battery size, increased electricity consumption in the use phase caused by the heavier battery considered. BEV 9, 11–13: DE 2013 electricity mix during use phase. BEV 10, 10a: renewable electricity mix during use phase (DE 2050). Use phase: 150,000 km. Battery production under provision of Chinese electricity (9, 10, 10a), UCTE 2004 (11), PV (12), and wind electricity (13). BEV 9–13 operating with 25.9 kWh battery, BEV 10a with 51.8 kWh of battery. All emission profiles are real-world.

The analysis of single score endpoints reveals a pattern similar to the CC impact, which has been also observed in the earlier modelling of an electrified SMART [16] (Helmers et al. 2017). However, the advantage of natural gas fueling appears even more pronounced in the single score endpoints than in the CC analysis (Figure 5, model #3a). Also, the relative share of impact caused by electric powertrain production, is higher in the single score endpoints, compared with its CC impact (Figure 5).

Battery sizes have grown continuously over the past few years—while the average BEV battery pack in the year 2015 was rated as 30 kWh, there was already a tendency toward an average of 50 kWh in the year 2017 [94]. Accordingly we add modelling version 10a of a BEV, with a 51.8 kWh battery, twice the size of the battery assumed for the other BEV models. In case this BEV is charged from a grid with renewable electricity (130.6 g CO_2-eq/kWh, DE 2013), but the battery is still produced with coal-based electricity (1180 g CO_2-eq/kWh, e.g., China), 36% of the lifetime CC impact will be caused by the battery production only (vehicle 10a, Figure 5). The single endpoints reveal that battery production is responsible for 31% of the lifecycle impact of vehicle 10a (Figure 5). This finding highlights again the necessity to produce vehicle batteries with renewable electricity.

When averaging the life-cycle CC impact of the two Caddy vehicles operating an Otto engine (#3, 3a, propelled with petrol and natural gas, respectively), there is only an insignificant difference of 0.5% between this average and the CC impact of the diesel engine vehicle (vehicle model 6). Accordingly an averaged climate change impact of all the three combustion engine vehicles (petrol, diesel, natural gas) will be used in the following section to quantify the competition with the electric vehicle in better detail.

3.5. Climate Change Impact: Combining Different Mileages, Battery Sizes, and the Battery Second Use Case

In Figure 6, seven variables of the LCA modelling process are evaluated in parallel for their influence on CC impacts, with emphasis on the choices of electricity provision during battery production. Two different battery sizes are considered (25.9 kWh, as originally installed, plus an extrapolated 52.8 kWh of battery capacity). Both battery sizes are as well modelled with and without a subsequent second-life stationary use (reducing these batteries CC impact relative to the vehicle by 50%, see above).

The choice of battery size and post-vehicle battery management (with/without second use) influences the BEV's overall CC lifecycle impact by a factor of 1.2–2.2. When charged with renewable electricity, the lifecycle CC impact of the VW Caddy ranges from 75–166 g CO_2-eq/km (Figure 6, bottom left and right), this increasing to 194–280 g CO_2-eq/km when charging DE 2013 electricity (Figure 6, top left and right). Adding the battery second use case to the vehicle with 51.8 kWh battery reduces its lifecycle impact by 5–27 g CO_2-eq/km (Figure 6). At the utmost, application of a battery second use can remove up to 15% of the lifecycle CC impact from a VW Caddy.

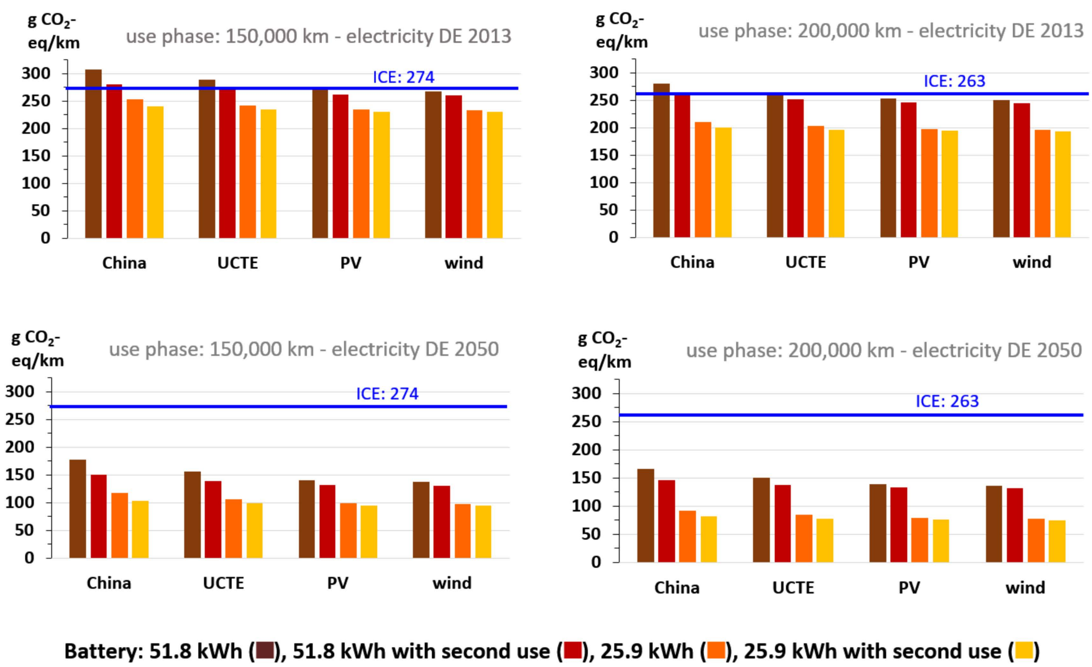

Figure 6. VW Caddy vehicle lifetime climate change impacts under various scenarios for use phase electricity carbon footprint, battery size, battery second use and battery production. X-axis: battery production under different electricity provision (China, UCTE, PV, wind, see Table 3). Different battery sizes and management: 51.8 kWh, 51.8 kWh with second use, 25.9 kWh, 25.9 kWh with second use. In comparison added as blue line: averaged impact of combustion engine cars (ICE). Increased weight of 51.8-kWh battery pack considered in use phase impacts. ICE = integrated combustion engine (vehicle).

If battery production and vehicle use rely on electricity with a considerable carbon footprint, e.g., electricity resembling the European mix (DE 2013), then only electric vehicles with a small battery (25.9 kWh or below) can match the life-cycle carbon footprint of conventional cars (Figure 6). Under the provision of the German electricity mix, the three vehicle combinations with a 51.8 kWh battery emit along their lifetime 281–308 g CO_2-eq/km and thus showing higher climate impacts than the conventional VW Caddy (use phase 150,000 km Figure 6, top left). The BEV also benefits, much more than the ICEV, from a prolonged use phase of 200,000 km (Figure 6).

3.6. The Size Effect: Comparing the Break-Even Mileages of the Electrified SMART and the VW Caddy

The electric conversion projects presented here allowed us to compare the impacts of two electric vehicles which differ distinctly in size—a mini car (by the company SMART, see Helmers et al. [17]) and a midsize car, the VW Caddy, as modelled and presented here.

We find that, except from one case, the electric SMART and the VW Caddy drive to be advantageous in a foreseeable (realistic) life cycle, when it comes to the comparison with the combustion engine vehicles (Figure 7). The exception is the Caddy with 51.8 kWh battery made in China with no battery second use, which needs an unrealistic mileage of 310,063 km to reach the LCI of the ICE Caddy. The break-even mileage, however, is reduced to 207,000 km or 137,000 km, when the battery is produced with electricity of the average European carbon intensity (531 g CO_2-eq/kWh) or with renewable electricity, e.g., generated from PV (92.5 g CO_2-eq/kWh), respectively (Figure 7, use phase DE 2013). The corresponding VW Caddy with smaller battery (25.9 kWh, a size usually modelled in early investigations) provides break-even mileages of less than 75,000 km under all conditions (Figure 7). Early LCA data published, on the contrary, did not report any advantages in CC impact of BEV compared to ICEV when charging average German electricity [95]. While it is known that smaller batteries move EVs faster toward advantages over ICEs, the degree of variation in reaching break-even mileages depends on the use phase electricity supply. Switching from "medium carbon impact" electricity (DE 2013) to a renewable mix (DE 2050, Table 3) reduces the number of km needed for break-even with the ICEV by a factor of 3.1–5.2 in case of the VW Caddy or by 3.8 in case of the SMART (Figure 7).

Assuming a secondary use for battery reduces the number of km necessary to reach break-even point further by factor of 1.2–1.7 for the VW Caddy or 1.2–1.4 for the SMART, respectively (Figure 7). The switch from batteries made in China to batteries made under provision of wind electricity reduces the number of km needed for break even by a factor of 1.6–2.5 (Caddy) or 1.4–1.8 (SMART), respectively (Figure 7).

Under the provision of DE 2013 electricity, both cars' overall predicted CC lifecycle impacts behave much more sensitive to a switch of battery production from higher to lower carbon footprint (Figure 7). The reason is that the lifecycle CC footprint of BEV provided with DE 2013 charging electricity is much closer to that of the ICEV, the difference can even be very small (Figures 5 and 6). Accordingly, under DE 2013 electricity provision the BEV needs to drive many more km to undercut the ICEV's LC CC impact. Under provision of green electricity (DE 2050), the BEV reaches break-even mileages much faster and is thus not so sensitive to increased battery impacts: The electrified VW Caddy then passes the ICE Caddy after 13,900–59,000 km, depending on battery size and application of battery second use (Figure 7). Accordingly, the electrified SMART needs 17,000–35,000 km to reach this target under DE 2050 electricity provision (Figure 7).

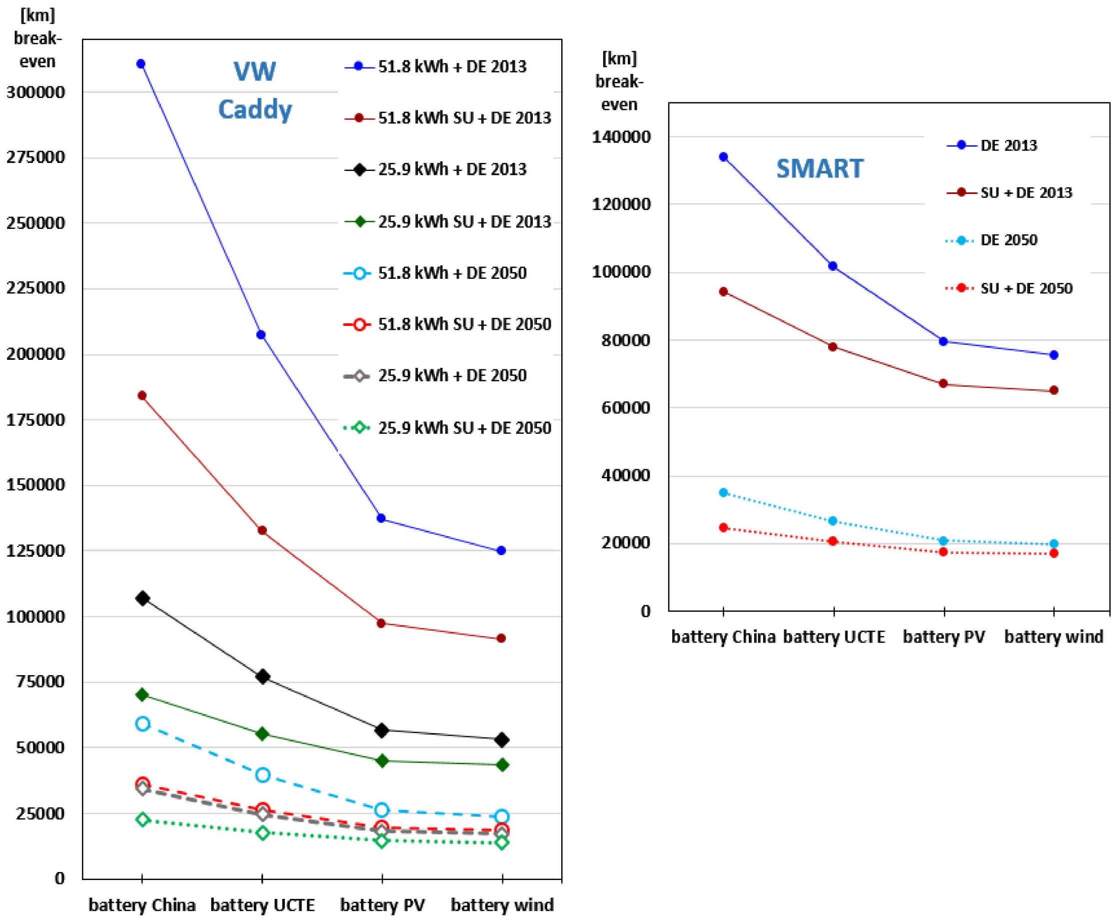

Figure 7. Break-even mileages for life-cycle climate change impacts of electrified vs. combustion engine cars for four electricity mixes during battery production. Solid lines: use phase with DE 2013 electricity; dashed lines: use phase with DE 2050 electricity provision. VW Caddy: 51.8 kWh battery with filled symbols, 25.9 kWh battery with unfilled symbols (left Figure). For Inventory and technical details of the SMART (14 kWh battery, right Figure) see the Supplements #1 and #2 in Supplementary Materials as well as Helmers et al. (2017) [17] and Helmers & Marx (2012) [39]. For electricity CO_2-eq emissions see Table 3. Diesel and petrol ICE use phase averaged for the VW Caddy. The ICE SMART drives with petrol. EOL impacts neglected. SU = battery second use. Impacts modelled per car.

In contrast to findings from Ellingsen et al. [22], who reported that mileages necessary to reach break-even increased with vehicle size, this has not been confirmed here. Although the SMART with 14 kWh of battery capacity and the Caddy with 25.9 kWh of battery size provided a comparable driving range (104 vs 128 km), or in other words, possess a comparable battery in relation to vehicle size, both vehicles reveal pretty much comparable mileages necessary for the electric models to reach lower life cycle carbon impacts (Figure 7). The smaller electric SMART even requires to drive 1.3–1.4 times more km than the electric Caddy (25.9 kWh) to reach the "green zone" based on supply of DE 2013 electricity (Table 3). Charged with renewable electricity (DE 2050, Table 3), both cars reveal relatively similar break-even mileages (Figure 7). The bigger VW Caddy enables the lowest break-even mileage in this comparison: 13,900 km (25.9 kWh battery made with wind electricity, plus battery second use, Figure 7).

While the size of the vehicle is apparently not significantly influencing the break-even mileages as shown here, it matters when it comes to quantifying the lifecycle climate impacts in terms of CO_2-eq/PKT (see below).

3.7. Lifecycle Climate Change Impacts of Electric Cars in Comparison with Competing Transportation Modes

Finally, the impacts relative to passenger kilometers travelled need to be quantified—a step usually missing in LCA of electric cars. This is essential in evaluating whether the electric car can play its role in the future decarbonizing transport. There is a statistically documented occupancy rate for European cars (1.57 persons/car, Castellani et al. [96]); however, this is an average covering all sizes/classes of cars. Only 6% of all European passenger cars registrations count among the segment of mini class cars (ICCT 2018) [97], and the SMART, as a two-seater, is among the smallest. We assume that the occupancy rate of a two-seater will be lower than that of a five-seater, as is the vast majority among the cars on the streets. Reducing the occupancy relative to the number of seats would result in an occupancy of even < 1, so we suggest to calculate with 1 person statistically driving in a SMART.

When dividing the LCI of the VW Caddy by a factor of 1.57 and assuming one person/car for the SMART then both vehicles deliver fairly similar life cycle impacts in terms of g CO_2-eq/PKT (passenger km travelled, Figure 8). Without accounting for occupancy, the electric VW Caddy would have with 85–248 g CO_2-eq/PKT 1.6-1.7times the impact of the SMART. It also turns out that while the electric Smart provides small advantages in g CO_2-eq/PKT over lifetime even when charged with DE 2013 electricity in comparison with the ICE SMART, this is not the case for the electrified VW Caddy. The latter needs a renewable electricity mix to deliver advantages in this comparison (Figure 8).

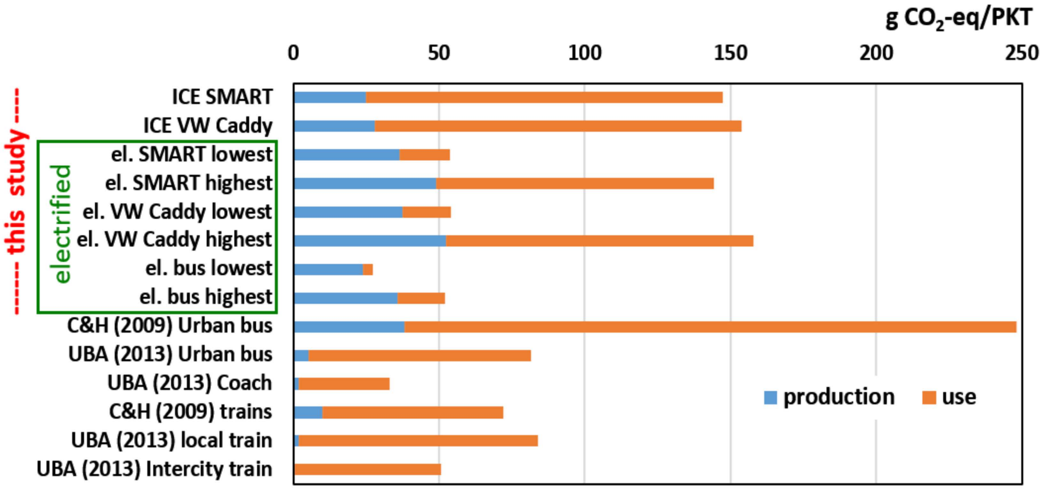

Figure 8. Lifecycle CC impact of ICEVs and BEVs modelled in in comparison with further passenger traffic modes. SMART/Caddy highest: 14 kWh/25.9 kWh battery production in China. Use phase based on DE 2013 electricity supply. SMART/Caddy lowest: Battery production under provision of wind energy, battery second use guaranteed. Use phase based on DE 2050 electricity supply. SMART modelled with 1 person/car, Caddy with 1.57 (Castellani et al. 2017) [96], use phase: 150,000 km. Electric bus: Production impact taken from Cooney et al. (2013) [98], calculation based on 1 Mio km of statistical mileage, a lowest and highest estimation of 14 vs 21 as average occupancy factor (Castellani et al. 2017) [96] and as well on DE 2013 vs DE 2015 electricity supply (see Table 3). Lowest bus el. consumption with 0.561 kWh/km taken from Gao et al. (2018) [99], highest bus el. consumption with 1.75 kWh/km taken from Zhou et al. (2016) [100]. US urban diesel bus and train averages calculated from Chester and Horvath (2009) [101]. German average coach and train impacts taken from UBA (2013) [102]. Infrastructure impacts neglected, EOL impacts included for the SMART/VW Caddy. ICEV = Integrated combustion engine vehicle. PKT = passenger km travelled.

The LC CC impact of the SMART was quantified as 72 g CO_2-eq/km per vehicle in the previous project [17], if recalculated with a 150,000 km lifetime mileage and based on Chinese battery production. Additionally assuming battery production under supply of 100% wind electricity plus a battery second use scenario reduces the lifetime CC impact by 25% on 54 g CO_2-eq/km per vehicle for the SMART

(Figure 8, "el. SMART lowest"). This illustrates again the potentials of reducing the lifetime impact of an electric vehicle by adjusting production and post-use treatment of the battery.

To put the impacts into a wider transportation context, we estimate climate impacts for electric buses from literature sources and added data for alternative passenger traffic modes such as diesel buses and coaches, and trains (Figure 8).

It turns out that both the electrified SMART and the VW Caddy can compete well with the carbon intensity of passenger km travelled with bus, coach and train, assuming the cars are charged with renewable electricity (Figure 8). Electric buses seem to deliver the lowest CC impacts during vehicle use in terms of passenger-kilometers travelled (27–52 g CO_2-eq/PKT, Figure 8). These results are not expected to change significantly when incorporating infrastructure construction and operation costs as these are quite similar when comparing railroads and streets [101,103].

This modelling reveals high savings due to electrification, when it comes to the CC impact in comparison to the cars with combustion engine. Under optimized conditions (battery produced with wind electricity, BEV charged with renewable electricity, battery second use), the BEV delivers 64% (SMART) or 65% (VW Caddy) savings in CC LCI, respectively (Figure 8; the VW Caddy modelled with 1.57 persons travelling on average) in terms of g CO_2-eq/PKT. This would be in accordance to the EU target to reduce the transport related CO_2 emissions by 60% from the 1990 levels to 2050 (EU 2019b) [104]. In other words, the EU CO_2 reduction target might not be achieved by electric vehicles if not reducing the life cycle impact of the batteries in the way described here.

4. Conclusions and Outlook

Our modelling results suggests that producing battery cells with renewable electricity decreases the environmental impacts of electric cars considerably. This is important particularly for Europe, where only 3% of the global cell production is currently located but large-scale manufacturing is being planned (EU 2019) [105].

For further reductions in the impacts of electric vehicles, but also to make better use of mineral resources needed for battery production, a large scale second use application of batteries as local energy storage to buffer fluctuating renewable electricity production can be very advantageous.

The presented modelling of the entire set of ReCiPe impact categories revealed disadvantages to the case of the electric VW Caddy in several impact categories (e.g., human and freshwater toxicity, freshwater eutrophication, mineral resource depletion, agricultural and urban land occupation). However, those disadvantages can be largely tackled ensuring battery cell production with renewable electricity but also by a cleaner production of printed circuits [17], the latter aspect being not yet satisfactorily addressed in the public discussion. Due to the continuous increase of electronic parts in automobiles the environmental impacts of printed circuits are a general sustainability problem in the car industry. The omission of many impact categories from life cycle modelling, as it is practiced mostly, may prevent those problems from being solved. In fact, there are, for example, solutions to reduce the impacts due to the life cycle of electronic parts (e.g., [106–108]).

Altogether, and perhaps due to the real-world-close origin of inventory data in this project, the two electric cars built and analyzed here (SMART and VW Caddy) turned out to be very advantageous with respect to climate change impacts. This is especially true when using all options (green electricity charging, battery production under green electricity supply, battery second use). The lifecycle impacts of diesel cars were shown to be significantly higher than previously described in LCA modelling due to adjusting diesel car emissions more real-world close as suggested here. It has been demonstrated here for diesel cars, that the established modelling is based on emissions which can be magnitudes lower than it is in reality, while at the same time emission species essential for human health are not yet considered (e.g., the PN emissions). Regarding the elevated, real-world close, NO_x emissions as they have been modelled here for diesel cars: Current remote-sensing data exhibited that very recent Euro 6-diesel cars are emitting 30–88% less NO_x than when compared with the 2015 level [14]. As from our data, we cannot discriminate the NO_x effect from the effects of the other elevated species emissions.

However we assume that considering a prolonged mileage (> 200,000 km) would again increase the magnitude of emissions to be considered. In Europe alone, there are millions of aged (> 15 years old) combustion engine vehicles driving on the streets, mainly in the South-Eastern part of the continent, with most probably defective/missing after-treatment systems for emissions. Accordingly, we conclude that the state-of-the-art LCA of combustion engine vs. electric cars is still underestimating the adverse impacts of the combustion engine. At the same time, the environmental impact advantages of electric cars seem to be underestimated today because an impact very important to human health is not included even in the most comprehensive set of impact categories we applied—health damages from road traffic noise. Preliminary modelling revealed that considering this effect may double the magnitude of human health damages due to road transportation [109].

We conclude for further research that LC modelling of such technologies should consider more complete and more real-world close emission inventories and longer life cycles. Real-world-close emissions may be established as a new sensitivity during LCA of electric vs. combustion engine vehicles. Modelling an individual inventory is a weakness of this project, because its results cannot be generalized. At the same time, it is a strength of this report that it highlights the deviation of impacts resulting from modelling a non-standardized, more real-world-close inventory.

Under optimal conditions, the CC life cycle impacts of two electric cars of different sizes investigated here are competitive even with public transport modes such as diesel buses, coaches, and trains. Electric buses alone can be more advantageous than the two electric cars analyzed.

Supplementary Materials: The following are available online at http://www.mdpi.com/2071-1050/12/3/1241/s1. Supplement #1: Detailed life cycle inventories. Supplement #2: Material balances, Li-cell production midpoints, Ecoinvent modules applied, vehicle midpoint indicator impacts.

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