

Life cycle assessment of future electric and hybrid vehicles: a cradle-to-grave systems engineering approach

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Abstract

Electric mobility is playing an important and growing role in the context of sustainable transport sector development. This study presents the life cycle assessment of an electric car based on the technology of Lithium-ion battery (BEV) for Europe and compares it to an internal combustion engine vehicle (ICEV). According to a cradle-to-grave approach, manufacturing, use and disposal phases of both vehicles have been included in the assessment in order to identify the hot spots of the entire life cycles. For electric vehicles two manufacturing inventories have been analysed and different vehicle disposal pathways have also been considered. Furthermore, the environmental performances of hybrid vehicles have been analysed based on the life cycle models of the BEV and ICEV. The results of the hot spot analysis showed that the BEV manufacturing phase determined the highest environmental burdens mainly in the toxicity categories as a result of the use of metals in the battery pack. However, the greenhouse gas emissions associated with the BEV use phase were shown to be half than those recorded for the ICEV use phase. The trend of the results have also been investigated for future energy mixes: the electricity and diesel mixes for the year 2050 have been considered for the modelling of the use phase of BEV and ICEV.

Keywords: life cycle assessment of electric vehicles; electric battery disposal; future energy scenarios.

1. Introduction

The transport sector is one of the most appealing and challenging when tackling the target of emissions reduction: currently, the CO₂ emissions in the transport sector are about 23% of the total antropogenic CO₂ emissions worldwide (UNECE 2015). In addition to this, a study commissioned by the World Business Council for Sustainable Development (World Business Council for Sustainable Development 2004) estimates that the number of light-duty vehicles in operation will rise to about 1.3 billion by 2030 and 2 billion by 2050. Hence,

34 there will be a dramatic increase in demand for fuel supplies associated with transport, which raises issues for
35 climate change, urban air quality as well as non-renewable resources depletion. This has pushed towards the
36 development of new technologies in the automotive industry.

37 One possible solution for decreasing the carbon footprint of the transport sector is the use of biogenic carbon
38 content fuels (such as cellulosic ethanol or soy biofuel) in conventional internal combustion engine vehicles,
39 instead of the regular fuel supply (Samaras and Meisterling 2008). However, the production of bio-fuel has well
40 known drawbacks when applied to large scale, mainly associated with pressure on land that would otherwise be
41 used for agricultural purposes. In addition to this, bio-electricity powered electric vehicles can offer higher
42 mileage per unit of biomass than when liquid biofuels such as ethanol are used in a conventional internal
43 combustion engine (IEA Bioenergy 2011)

44 The generic term 'electric vehicle' refers to several types of vehicles that differ for the share of electricity used
45 for traction over conventional fuels: purely battery-driven electric vehicles (BEVs), hybrid electric vehicles
46 (HEVs), plug-in hybrid electric vehicles (PHEVs) and extended range electric vehicles (E-REVs). The use of
47 electricity for the transport sector is also promising for having the potential to reduce greenhouse gas emissions
48 compared to ICEVs thanks to the avoided or reduced requirements of diesel or gasoline as fuel supply.
49 According to Zackrisson et al. (2010) the potential savings in GHG range is between 25% for hybrid EVs, up to
50 50-80% for plug-in hybrid EVs and about 90% for battery EVs. Moreover, a very wide range of road vehicles
51 can use electric power for motion: from heavy duty vehicles- such as hybrid buses and tramways to light duty
52 ones, including city cars, forklift trucks etc. Each type of vehicle can use various battery technologies, such as
53 Lithium-ion battery (Li-ion), Nickel Metal Hydride (NiMH), lead acid, nickel cadmium batteries, each
54 characterised by specific properties (specific power, depth of discharge (DoD), memory effect, number of
55 charges per cycle, etc.).

56 However, the use of heavy metals for battery manufacturing, the electricity mix used for charging the battery
57 and the disposal of the used battery are key aspects in the life cycle of an electric vehicle that need to be
58 carefully considered under a life cycle approach to identify possible sources of increased environmental impacts.
59 In Wietschel et al. (2013), electric cars are reported to be increasingly penetrating the future fleet market but
60 they are also facing the most technological challenges today. Because of that, the environmental impacts
61 associated with the manufacturing, use phase and end of life of electric cars need to be analysed.

62 Some environmental assessment studies on EV, differing in scopes and details of analysis, have already been
63 performed. For examples, many (Samaras and Meisterling 2008; Aguirre et al. 2012; Helmers and Marx 2012;
64 Dunn et al. 2012b; Faria et al. 2013) mainly focused on the analysis of the energy requirements and the
65 greenhouse gas emissions throughout the vehicle life cycle. In particular, the latter calculated the energy inputs
66 and CO₂ equivalents emissions of a conventional gasoline vehicle, a hybrid vehicle, and a battery electric
67 vehicle for California referring to the aggregated inventory data reported in a model previously developed by the
68 US Argonne National Laboratory (Sullivan et al. 2010; Sullivan and L. 2010); however, a detailed description of
69 the inventory data and model parameters was not available. The same is also true for other two studies (Van den
70 Bossche et al. 2006; Matheys et al. 2008) who reported an aggregated environmental impacts of different
71 electric batteries used for motion, calculated according to the grouping phase of the life cycle assessment (LCA)
72 methodology. Ellingsen et al. (2014) and Bettez et al. (2011a) published the environmental assessment of the
73 manufacturing phase of different types of electric batteries but they did not consider the entire life cycle of the

74 vehicles. Conversely, Daimler AG (2009) and Zackrisson et al. (2010) reported the results of LCA studies on
75 batteries used for hybrid vehicles accounting also for the entire life cycle of the vehicles. Finally, the most
76 detailed and complete studies on BEV are those by Notter et al. (2010) and Hawkins et al (2013) that reported
77 the life cycle assessment of the entire life cycle of the vehicle, analysing also different environmental impacts.
78 However, in both studies the results of the disposal phase were entirely based on the Ecoinvent database (Swiss
79 Centre for Life Cycle Inventories 2014).

80 The purpose of this study is to perform an attributional life cycle assessment of the manufacturing, use and
81 disposal phases of BEVs and hybrid vehicles, and compare it to the life cycle of a conventional vehicle, such as
82 diesel ICEVs. All the components of the vehicle, including the battery system, the glider, and the power train
83 are analysed in the hot spot analysis. As shown before, many LCA studies have been conducted on batteries for
84 electric vehicles. However, very few studies have analysed the entire life cycle of vehicles, including both the
85 battery and the rest of the vehicle and have considered the disposal phase based on an industrially developed
86 technology as done in this study. In addition to this, few studies considered the effect of different energy shares
87 on the environmental impacts of the electric vehicle life cycle. Egede et al. (2015) reported that the electricity
88 mix is a crucial parameter for the LCA calculation and this is also supported by the results reported in Faria et
89 al. (2013) where the electricity mix of different countries in 2013 were considered. However, the latter fails to
90 consider the potential development of the electric vehicle fleet within the projected future energy mixes.
91 Conversely, our study considers referenced projections of the future EU electricity share and therefore
92 contextualises the study of the environmental burdens of electric vehicles to more realistic future scenarios.

93 **2.Life cycle assessment methodology**

94 Life cycle assessment is one of the most developed and widely used environmental assessment tools for
95 comparing alternative technologies (Clift et al. 2000; Clift 2013). LCA quantifies the amount of materials and
96 energy used and the emissions and waste over the complete supply chain (i.e. life cycles) of goods and services
97 (Baumann and Tillman 2004). Moreover, it helps to identify the ‘hot spots’ in the system; i.e. those activities
98 that determine the most significant environmental impact and should be targeted in the first instance, thus
99 enabling identification of more environmentally sustainable options (Clift 2006).

100 In the Impact Assessment phase, the emissions and inputs quantified in the Inventory phase are translated into
101 environmental impacts. This study focuses specifically on three impact categories - showed in Table 1 - which
102 are considered the most significant for the purpose of this work. However, other impact categories as suggested
103 by ILCD handbook have been analysed in this study (ILCD 2011). For a full list of categories see
104 Supplementary Information.

105 The global warming potential (GWP) characterises and calculates the impact of greenhouse gases based on the
106 extent to which they enhance radiative forcing. GWP values for specific gases, developed by the
107 Intergovernmental Panel on Climate Change (IPCC), express the cumulative radiative forcing over a given time
108 period following a pulse emission in terms of the quantity of carbon dioxide giving the same effect (IPCC
109 2007). Following common convention, such as the Kyoto Protocol, the 100-year values have been used here.

110 The abiotic depletion (AD) addresses the environmental problem of the diminishing pool of resources. It focuses
111 on the depletion of non-living resources such as iron ore, crude oil, etc. The measurement unit of abiotic
112 depletion is MJ as the majority of non-renewable resources represent energy sources. The human toxicity

113 potential (HTP) reflects the potential harm of chemical species released into the environment, based on both the
 114 inherent toxicity of a compound and the potential human exposure.
 115 Currently more than thirty software packages exist to perform LCA analysis, with differing scope and capacity:
 116 some are specific for certain applications, while others have been directly developed by industrial organisations
 117 (Manfredi and Pant 2011). In this study, GaBi 7 has been used (Thinkstep 2015); it contains databases
 118 developed by ThinkStep that incorporates industry organisations' databases (e.g. Plastics Europe, Aluminium
 119 producers, etc.) and also regional and national databases (e.g. Ecoinvent, US NREL database, etc.).

120 **Table 1** Impact categories and indicators used in this study

<i>Impact categories</i>	<i>Impact Indicator</i>	<i>Acronym</i>	<i>Caracterisation model</i>	<i>Units</i>
Climate change	Global warming potential	GWP	CML 2001 baseline (IPCC 2007)	kg CO ₂ eq
Resources depletion (fossil)	Abiotic depletion	AD	CML 2001 baseline (Guinée et al., 2001)	MJ
Human toxicity	Human toxicity potential	HTP	USEtox model (Rosebaum et al., 2008)	kg DCB ¹ eq

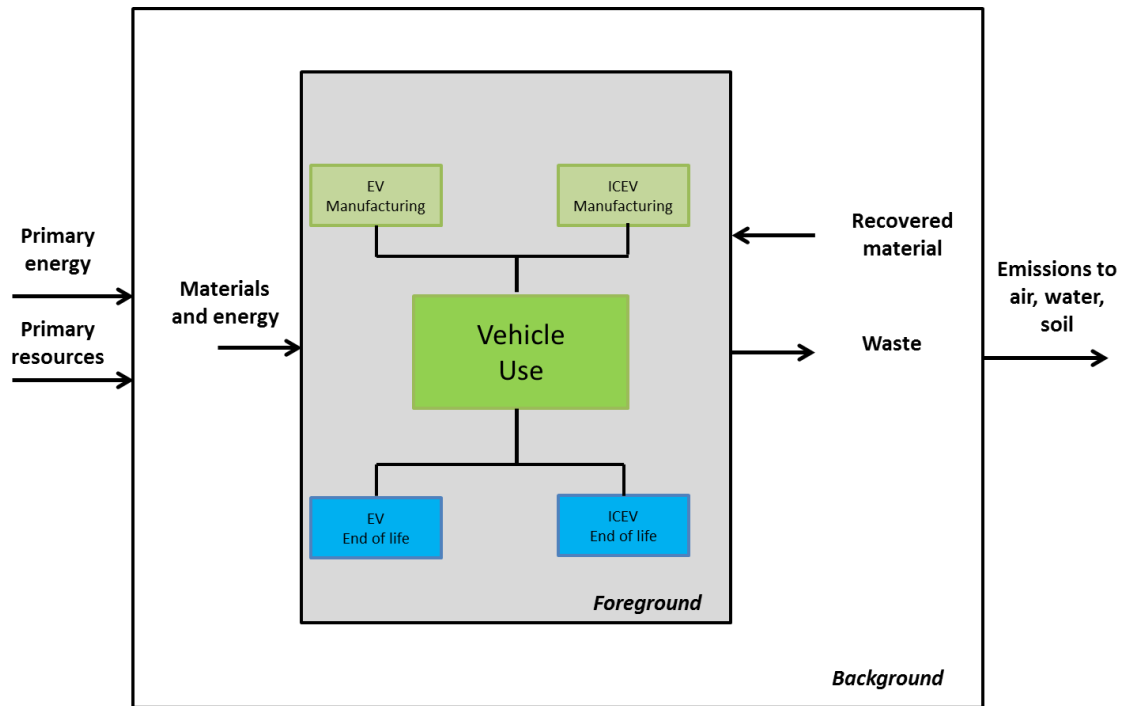
121 **Note:** ¹ DCB: dichlorobenzenes.

122 3. Goal and scope definition and system boundaries

123 The main goal of this paper is to perform an attributional life cycle analysis with a consequential approach of a
 124 battery electric vehicle (BEV), and compare it with the life cycle impacts of a more conventional technology,
 125 such as an internal combustion engine diesel vehicle (ICEV). Moreover, a hot spot analysis to identify the steps
 126 with the highest impacts to the total life cycle is presented. The two models for ICEV and BEV are used to
 127 analyse the environmental impacts of different types of hybrid vehicles. Two different scenarios are also
 128 considered for the end-of-life phase: a 'high recycling rate' scenario – where the total vehicle is assumed to be
 129 recovered in EU at its EoL; and a 'low recycling rate' scenario, where a fraction of the vehicle is assumed to be
 130 landfilled outside EU borders. While the former is more unrealistic, the latter represents the current situation of
 131 the vehicle EoL market in Europe where part of the fleet exits the EU borders (Mehlhart et al. 2011). A final
 132 scenario analysis considers future EU energy technologies and mixes.

133 Figure 1 shows the boundary of the system analysed. Three different phases have been considered in the
 134 analysis: the manufacturing phase – which includes the production of the batteries and all the single components
 135 up to the glider; the use phase – which includes the production of electricity needed to recharge the battery; the
 136 end of life phase – which includes the reprocessing of the vehicle including the battery, up to the recovery of
 137 some metals. The same phases have also been considered for the ICEV. For all the phases, indirect, direct and
 138 avoided burdens are considered in the life cycle models according to EU-site specific inventories (Behrens et al.
 139 2013) and allocation is performed using the method of system expansion. Transport of the different components

140 to the production and dismantling sites- manufacturing and disposal phases - is considered negligible as already
 141 analysed in literature (Hawkins et al. 2013).
 142 The functional unit used in this study is the function of 1 km driven by one vehicle (car). To account for the
 143 manufacturing and the disposal phase, an assumption of the total km driven in the entire life cycle of the vehicle
 144 was made. Based on previous studies (Majeau-Bettez et al. 2011a; Ellingsen et al. 2014), a total life cycle of
 145 150,000 km is considered for both BEVs and ICEVs.



146
 147 **Figure 1** System boundary

148 **4. Life cycle inventory**

149 In this section, the inventory models built for the BEV and ICEV are presented. The inventory was based on a
 150 mix of data coming from several literature studies published in the recent years on BEVs and plug-in vehicles,
 151 existing dataset (Swiss Centre for Life Cycle Inventories 2014; Thinkstep 2015), and reports and presentation
 152 from private companies (Umicore 2015).

153 The vehicle's models were based on the most promising and most popular commercial vehicles currently sold
 154 on the market (see Table 2). For the BEV, a Nissan leaf equipped with a Li-ion battery technology was assumed
 155 as reference while for the ICE a Toyota Yaris was assumed as reference vehicle because of the same category as
 156 the Nissan Leaf.

157 **Table 2** Reference vehicles for the life cycle assessment of BEV, ICEV and hybrid vehicles

<i>Technology</i>	Model	Fuel
Electric(100%) EV	Nissan Leaf	/

Hybrid(30%) HEV	Yaris Hybrid	Diesel
Hybrid(60%) P-HEV	Toyota Prius Plug-in	Diesel
'Hybrid'(90%) E-REV	Toyota Prius	Diesel
Internal Combustion Engine ICEV	Toyota Yaris	Diesel

158 **4.1 Manufacturing**

159 The vehicle is composed by several units, which can be divided in sub-units up to the single component. The
160 two main macro units which have been considered in the vehicle models are the powertrain (electric motor and
161 battery system for the BEV and the internal combustion engine for the ICEV) and the glider. For both ICEV and
162 BEV, the model for the glider was based on Ecoinvent 2.1 (Swiss Centre for Life Cycle Inventories 2014).

163 *Battery electric vehicle*

164 Two different models were considered for the manufacturing of the BEV, in order to test the robustness of the
165 results. The first model (EV I) was based on the study published by Bettez et al. (2011a). In their study, Bettez
166 et al. based the LCA inventory on average literature data for the manufacturing of the battery. The second model
167 (EV II) was based on the study published by Ellingsen et al. (2014). Their inventory was based on an existing
168 battery, and the dataset for this was built using a mix of average and commercial data supplied by the battery
169 manufacturing company (Miljøbil Grenland 2012).

170 The powertrain of the battery electric vehicle includes all the units of the BEV excluding the glider. In total, the
171 weight of the glider and the powertrain excluding the battery was 1307 kg and 1271 kg for EVI and EVII,
172 respectively, in order to match a total weight for the BEV equal to the Nissan leaf. The weight of the Li-Ion
173 battery was 214 and 250 kg for EVI and EVII, corresponding to a specific power of 112 and 106 Wh/kg. Table 3
174 shows the characteristics of the Nissan Leaf assumed in this study.

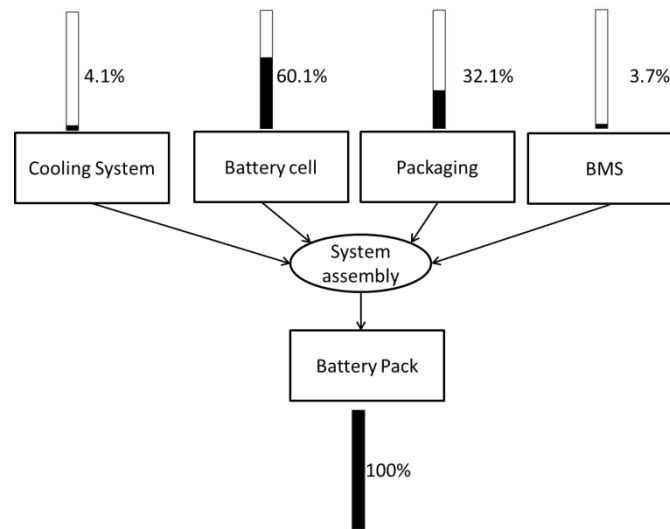
Nissan Leaf		
Curb weight	kg	1521
Length	cm	444.5
Width	cm	177
Height	cm	155
Body style		5-door hatchback
Electric motor	kW	80
Battery (Li-Ion)	kWh	24
Range	km	117

	(EPA ¹)	
	km (NEDC ²)	175
Energy per km	Wh (NEDC)	173

175 **Table 3** Characteristics of the Nissan Leaf assumed as BEV in this study (Nissan 2015)

176 The battery pack is the core of the BEV; this is composed by four units: the cooling system; the battery cell; the
 177 packaging; and the battery management system (BMS) (for an example see Figure 2). The modelled battery
 178 pack consists of 48 modules and each module contains four battery cells for a total of 192 cells. As an example,
 179 the weight distribution of each battery pack as assumed in this study for EVII is reported in Figure 2; for both
 180 models, the main component of the battery pack is the cell. The thermal management of the battery is done by
 181 the cooling system. This is made by six sub-components: radiator, manifolds, clamps & fasteners, pipe fitting,
 182 thermal gap pad, and coolant (Ellingsen et al. 2014). The main component is the aluminium radiator, which
 183 accounts for the 30% of the total aluminium used in the battery system (Ellingsen et al. 2014). It is worth
 184 noticing that originally Majeau-Bettez et al. (2011a) did not include the cooling system as a component of the
 185 battery pack. For a fair comparison between the two models, we included the cooling system as defined by
 186 Ellingsen et al. (Ellingsen et al. 2014) in the EVI model.

187



188

189 **Figure 2** Battery pack weight distribution for EVII. Adapted from the literature (Ellingsen et al. 2014)

190 Several compositions for the cathode of a Li-ion battery were studied in the literature (Goodenough and Park
 191 2013): LiMn2O4 (LMO), LiFePO4 (LFP), Li(NiCoAl)O2, and Li(NixCoyMnz)O2 (NCM), where x, y, and z
 192 denote different possible ratios. However, the models (EVI and EVII) developed in this study are based on a
 193 LiNi0.4Co0.2Mn0.4O2 battery according to the characteristics of the Nissan Leaf. The main differences
 194 amongst the EVI and EVII model are in terms of materials and quantities involved in the manufacturing phase,
 195 and in the energy assumed for the manufacturing of the battery system. The main inventory data and differences
 196 in cell manufacturing are reported in Table 4.

EVI Model		EVII Model	
Material requirements			
Active material positive electrode paste LiNi _{0.4} Co _{0.2} Mn _{0.4} O ₂	0.20184	Active material positive electrode paste LiNi _{0.4} Co _{0.2} Mn _{0.4} O ₂ [kg/kg of battery]	0.218
Carbon black (furnace black) [kg/kg of battery]	0.0116	Carbon black (furnace black) [kg/kg of battery]	0.00464
N-methyl-2-pyrrolidone [kg/kg of battery]	0.064953271	N-methyl-2-pyrrolidone [kg/kg of battery]	0.0952
Tetrafluoroethylene [kg/kg of battery]	0.01856	Polyvinylfluoride [kg/kg of battery]	0.00928
Negative electrode paste		Negative electrode paste	
Graphite, battery grade [kg/kg of battery]	0.0893	Graphite, battery grade [kg/kg of battery]	0.0964
N-methyl-2-pyrrolidone [kg/kg of battery]	0.02632	N-methyl-2-pyrrolidone [kg/kg of battery]	0.094
Tetrafluoroethylene [kg/kg of battery]	0.0047	Acrylic acid [kg/kg of battery]	0.002
		Carboxymethyl cellulose, powder [kg/kg of battery]	0.002
Energy requirements			
Electricity for assembly [MJ/kg of battery]	27	Electricity for assembly [MJ/kg of battery]	100.8014
Heat for assembly and electrode pastes [MJ/kg of battery]	30	Heat [MJ/kg of battery]	/

197 **Table 4** Main inventory for the two BEV models in terms of materials and energy requirements

198 The EVI and EVII models differ basically in chemicals used (except the active material which is the same) and
 199 their quantities and structure complexity. For both Li-ion batteries, the anode is composed primarily of graphite;
 200 acrylic acid and carboxymethyl cellulose in EVII substitute tetrafluoroethylene in EVI. The assumed energy
 201 required for the manufacturing of the battery system can vary greatly amongst the literature, from 3.1 to 1060
 202 MJ/kWh (Ellingsen et al. 2014). In particular, this figure is considerably different for the two battery models
 203 analysed. Ellingsen et al. (2014) assumed an energy requirement of 586 MJ/kWh based on industrial data, while
 204 Majeau-Bettez et al. (2011a) reported an energy consumption between 371 and 473 MJ/kWh based on industry
 205 reports. Moreover, while Ellingsen et al. (2014) refers to the energy required to manufacture the battery cell, the
 206 figure proposed by Majeau-Bettez et al. (2011a) included also the energy for the battery system assembling. The
 207 electricity assumed for the manufacturing of the cell in EVI includes the coating of the electrode pastes to
 208 metallic foils used as current collectors, welding of current collectors to tabs, filling of electrolyte, and initial
 209 charging of the finished cell. However, as reported by Ellingsen et al. (2014) the main consumption is due to the
 210 operation of various dry rooms that are vital to the quality of the battery cells. This explains also the difference
 211 in energy consumption assumed in EVI and EVII.

212 In order to ensure the comparability of the results, the energy accumulated by the battery and then delivered to
 213 the powertrain, was fixed to 24 kWh (Genikomsakis et al. 2013; Nissan 2015) for the two BEV models. The two
 214 batteries have equal charge capacity but different weight (214 kg for the EVI vs 250 kg for EVII).

215 Internal combustion engine vehicle

216 The inventory for the ICEV manufacturing phase is based on Ecoinvent 2.2 database (Swiss Centre for Life
217 Cycle Inventories 2014). The model was built according to a life cycle inventory analysis based on a “Golf A4,
218 1.4 l Otto” (Leuenberger and Frischknecht 2010). The whole life cycle inventory as reported by Ecoinvent was
219 scaled up to match the total weight of the Toyota Yaris, which was 1500 kg. The emissions to air during the
220 manufacturing process are assumed to result from stationary combustion processes at the factory site (Swiss
221 Centre for Life Cycle Inventories 2014).

222 4.2 Use

223 In the use phase we have accounted for the emissions due to the use of the vehicles (direct emissions) and for
224 the emissions due to the production of the fuel (indirect emissions), i.e. electricity for BEV and diesel for ICEV.

225 Battery electric vehicle

226 The energy consumption reported in the literature varies significantly depending on the assumption of battery
227 cycles and lifetime. In this study, we assumed a lifetime of 150,000 km for the BEV, in line with literature
228 (Notter et al. 2010). The electric energy needed to drive 1 km was assumed equal to 0.56 MJ/km, based on
229 Ecoinvent 2.2, with a powertrain efficiency of 80% in a standard driving cycle (New European Driving Cycle,
230 NEDC). This is similar to the consumption reported in the literature (Notter et al. 2010), which shows an
231 electrical consumption of 17 kWh for 100 km, referred to a combination of the urban (12.8 kWh/100km) and
232 extra-urban (16.8 kWh/100km) energy consumption in a NEDC, plus the consumption of heating and air
233 conditioning during one year. A slightly lower electrical consumption was assumed by other authors (Majeau-
234 Bettez et al. 2011a; Ellingsen et al. 2014), corresponding at 3,000 cycles for the battery lifetime, and equals to
235 0.5 MJ/km. No battery package replacement was considered during the vehicle life (Notter et al. 2010).

236 In this study we have assumed an electricity production mix for the BEV use phase which is representative of
237 the European electricity grid. The inventory was based on Thinkstep database (Thinkstep 2015) and it is
238 dominated by nuclear energy (mainly from France), hard coal and natural gas which alone constitute around
239 64% of the total primary energy. The sensitivity of the model to the electricity mix is considered in the future
240 energy scenarios analysis.

241 Internal combustion engine vehicle

242 For the ICEV use phase we have assumed the same lifetime of the BEV, which is 150,000 km. The fuel
243 consumption was 50.04 mL/km, based on Ecoinvent 2.2 (Spielmann et al. 2007; Swiss Centre for Life Cycle
244 Inventories 2014). A EURO 5 vehicle was modelled, in accordance to the most recent European regulations on
245 the subject.

246 The model for the diesel production was based on Thinkstep database (Thinkstep 2015). The data set covered
247 the entire supply chain of the refinery products. Country / region specific downstream (refining) technologies,
248 feedstock (crude oil) and product (diesel fuel, etc.) properties, like sulphur contents, were considered. The
249 sensitivity of the model to the diesel mix is considered in the future energy scenarios.

250 4.3 Disposal

251 In the disposal phase models of the ICEV and BEV, the valuable outputs considered by system expansion are
252 the metals: mainly nickel, cobalt, manganese, aluminium, copper and steel. The latter is the main component, in
253 weight, of the glider; conversely, nickel, cobalt and manganese are mainly found in the battery pack of the BEV.
254 The recycling of each of the previous metal was considered together with the processes required for recovery.
255 For the ICEV disposal, precious metals have been included in the assessment according to Ecoinvent (Swiss
256 Centre for Life Cycle Inventories 2014).

257 A scenario analysis was performed on the disposal phase of BEV and ICEV. In the ‘high recycling rate’
258 scenario the entire vehicle fleet was assumed to be recycled and disposed within the EU borders according to the
259 later described EoL. Conversely, in the ‘low recycling rate’ scenario only 57% of the vehicle fleet was assumed
260 to be disposed in EU (Mehlhart et al. 2011); part of the fleet was considered to be sold outside the EU borders
261 (Mehlhart et al. 2011) and track was lost of it. However, in order to account for the EoL of these vehicles, an
262 assumption was made on their final disposal: in the ‘low recycling rate’ scenario 43% of the fleet was assumed
263 to end up in a material landfill outside EU. To model this, the ‘Landfill of ferro metals’ process was considered.

264 Battery electric vehicle

265 Two steps have been considered for the end of life treatment of BEV. The first refers to the disposal of the
266 glider and the power train excluding the battery and the model for this is based on Ecoinvent v.2.2; the second,
267 involving the battery disposal after dismantling, has been modelled according to currently used technologies in
268 specialised industries (Umicore 2015).

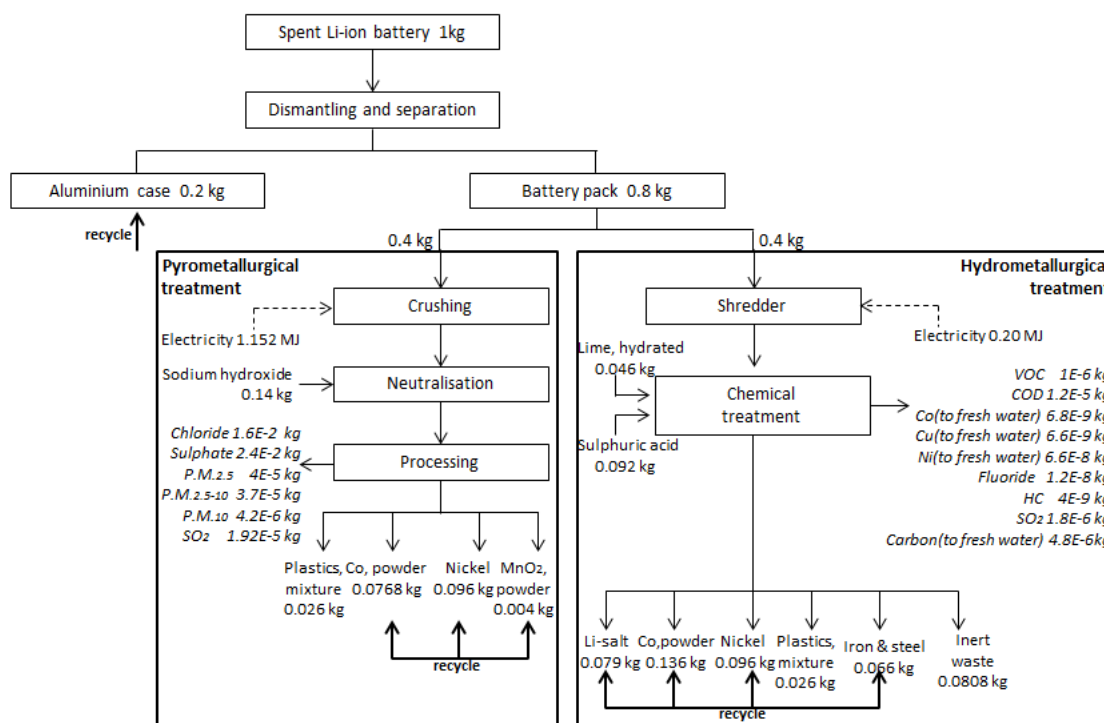
269 The modelled process for the battery disposal firstly involves a single-furnace pyro-metallurgical treatment
270 method for the treatment of Li-Ion batteries and Li-polymer cells, as well as nickel metal hydride (NiMH)
271 batteries (Vadenbo 2009). The main focus of the process is the recovery of cobalt and/or nickel. Cobalt is
272 commonly found in lithium-ion and lithium-polymer batteries, whereas nickel is mainly introduced into the
273 process through the treatment of NiMH batteries. Publicly available data for the pyro-metallurgical process
274 (Vadenbo 2009; Dunn et al. 2012a; Umicore 2015) and the battery disposal process (Hischier and Gallen 2007)
275 have been put together to build a novel battery disposal model. The process for the battery disposal and the
276 relative inventory is reported in Figure 3. Material recycling is also considered.

277 The slag is mainly formed by compounds containing aluminium (Al), silicon (Si), calcium (Ca) and to some
278 extent iron (Fe). In the process, lithium also ends up in the slag in the form of lithium oxide. The slag can be
279 used in the construction or concrete industry (Vadenbo 2009). However, in this model, slag use was not
280 considered because the amount of the slag produced compared to the weight of the entire vehicle is negligible.
281 The alloy fraction is predominantly made up of residual iron, copper, cobalt, and possibly nickel. The alloy is
282 subsequently leached with sulfuric acid in a hydrometallurgical step which extracts metals like cobalt, copper,
283 nickel and iron (Vadenbo 2009). Hence, the model has included two pathways: pyro-metallurgical treatment and
284 hydrometallurgical treatment and the allocation of the two pathways has been done at 50% each.

285 The recovery rate of the metals from the pyro and hydrometallurgical processes was based on data elaborated by
 286 different authors in the literature (Vadenbo 2009; Dunn et al. 2012a). Considering 1 kg of Li-Ion battery, these
 287 recovery rates for the following metals were considered:

- 288 • 7.7% Co (Pyro)
- 289 • 13.6% Co (Hydro)
- 290 • 9.6% Cu (Pyro)
- 291 • 10.8% Steel (Pyro)
- 292 • 6.6% Steel (Hydro)
- 293 • 6% Ni (Hydro)

294 To account for the avoided burdens due to the recovery of these metals, primary production processes were
 295 considered in the system expansion according to the market recycling rates (Graedel 2011; Thinkstep 2015).



296 **Figure 3** Process flow chart for disposal of lithium-ion battery. This model was adapted and built on the base of
 297 the data available in literature (Hischier and Gallen 2007; Vadenbo 2009; Dunn et al. 2012a; Umicore 2015)
 298

299 Internal combustion engine vehicle

300 The disposal of the ICEV is based on Ecoinvent database (Swiss Centre for Life Cycle Inventories 2014). It
 301 accounted for 100% recycling of aluminium, copper and steel contained in the vehicle. The rest of the materials
 302 was assumed to be sent to an incineration plant after dismantling (Swiss Centre for Life Cycle Inventories
 303 2014).

304 **4.4 Scenario analysis: hybrid vehicles and extended range**

305 Different types of hybrid vehicles have been analysed in the scenario analysis on the base of the BEV and ICEV
 306 models. As shown in Table 2, the hybridization factors – that represents the percent of the electric part of
 307 powertrain out of the total weight of the powertrain- has been changed between 30% and 90%: the chosen
 308 values were 30% for the HEV, 50% for the pHEV and 90% for the E-REV according to the different vehicle
 309 technologies reported in Table 2. Table 2 also shows the reference vehicles; Toyota Yaris Hybrid was chosen
 310 for the hybrid vehicle and the Toyota Prius for both plug-in hybrid and extended range electric vehicles.
 311 Those vehicles normally use gasoline when running on the internal combustion engine; the environmental
 312 impacts of gasoline use have been computed even though the results for this case are not reported. This analysis
 313 showed that the environmental impacts associated with the use of gasoline in hybrid car did not significantly
 314 differ from the environmental impacts of hybrid vehicles run on diesel. Hence, to allow a fairer comparison with
 315 the ICEV, we assumed that the required fuel was diesel also when analysing hybrid vehicles.

316 **4.5 Scenario analysis: future energy mix**

317 The use phase environmental impacts of the ICEVs and BEVs are also compared considering future EU energy
 318 mix. For the production of the electricity requirements of the BEV, the EU electricity mix of 2050 was
 319 considered according to the data reported in the literature (Behrens et al. 2013). Conversely, for the ICEV we
 320 assumed that the diesel mix in 2050 will be made up by 72.8% of conventional diesel and 27.2% of biodiesel
 321 according to the IEA (International Energy Agency 2011). Two models were built to calculate the impacts of the
 322 electricity production and diesel production 2050 and then these impacts were added up to the use phase of the
 323 two types of vehicle.

324 The electricity mix from 2015 to 2050 shows a decrease of energy from fossil sources (about -6% lignite, -1%
 325 peat, -9% hard coal, -0.5 % coal gases, -1% heavy fuel oil and -5% natural gas), a 6% increase of nuclear and an
 326 increase of all renewable sources (about +3.5% biomass, +2% biogas, +1.8% waste, +1.6% hydro, +8.5% wind,
 327 +2% photovoltaic, slightly increase solar thermal).

328 The biodiesel production was modelled according to the soybean-biodiesel model reported in Ecoinvent
 329 database (Swiss Centre for Life Cycle Inventories 2014).

330 **5 Results**

331 **5.1 Normalised results**

332 Table 5 shows the normalised results of the ‘high recycling rate’ scenarios (assumed as the baseline) for EVI,
 333 EVII and ICEV according to the functional unit (i.e. 1 km driven per vehicle). The total impacts of the three
 334 cases have been normalised using the regionalised CML European factors reported in the supplementary
 335 information (Thinkstep 2015).

<i>Total normalized impacts</i>	<i>EVI</i>	<i>EVII</i>	<i>ICE</i>	<i>HEV 30%</i>
Abiotic Depletion (ADP elements)	5.24E-13	1.79E-13	5.64E-14	9.31E-14
Abiotic Depletion (ADP fossil)	3.77E-14	3.80E-14	6.65E-14	5.79E-14
Acidification Potential (AP)	3.57E-14	3.76E-14	3.87E-14	3.84E-14

Eutrophication Potential (EP)	9.76E-15	7.64E-15	9.62E-15	9.03E-15
Freshwater Aquatic Ecotoxicity Pot. (FAETP inf.)	1.62E-12	1.56E-12	8.86E-14	5.31E-13
Global Warming Potential (GWP 100 years)	2.30E-14	2.13E-14	3.21E-14	2.89E-14
Human Toxicity Potential (HTP inf.)	1.75E-13	1.42E-13	8.36E-14	2.91E-14
Marine Aquatic Ecotoxicity Pot. (MAETP inf.)	4.06E-12	3.16E-12	8.70E-13	1.01E-13
Ozone Layer Depletion Potential (ODP, steady state)	3.09E-14	2.02E-16	1.28E-16	1.56E-12
Photochem. Ozone Creation Potential (POCP)	3.14E-14	3.25E-14	-5.35E-14	1.50E-16
Terrestrial Ecotoxicity Potential (TETP inf.)	8.36E-15	8.23E-15	1.36E-14	-2.77E-14

336 **Table 5** Normalised results of the ‘high recycling rate’ scenarios for EVI, EVII and ICEV. The normalisation
337 was performed according to the European regionalised impacts reported in the GaBi database (EU25+3, year
338 2000, incl biogenic carbon (region equivalents))

339 The impact indicators related to water (MAETP and FAETP) of EVI and EVII are significantly higher than all
340 other normalised impact indicators; the HTP and the ADP element follow in order of magnitude. The reason of
341 the increased values of those results for the EV models has to be found in the use of precious and non-precious
342 metals during manufacturing. It is common that LCA analysis of processes involving the use of metals show
343 high impacts in the water-related categories (Pizzol et al. 2011). This is related to the extraction and processing
344 of the metal itself. In LCA analysis, the MAETP is sometimes analysed but usually the FAETP is the preferred
345 category for the hot spot analysis as it refers to fresh water impacts; the MAETP is not usually included in the
346 analysis also because the emissions to environment contributing to the FAETP are very similar to those
347 contributing to the MAETP; hence there is no need for giving the same information twice.

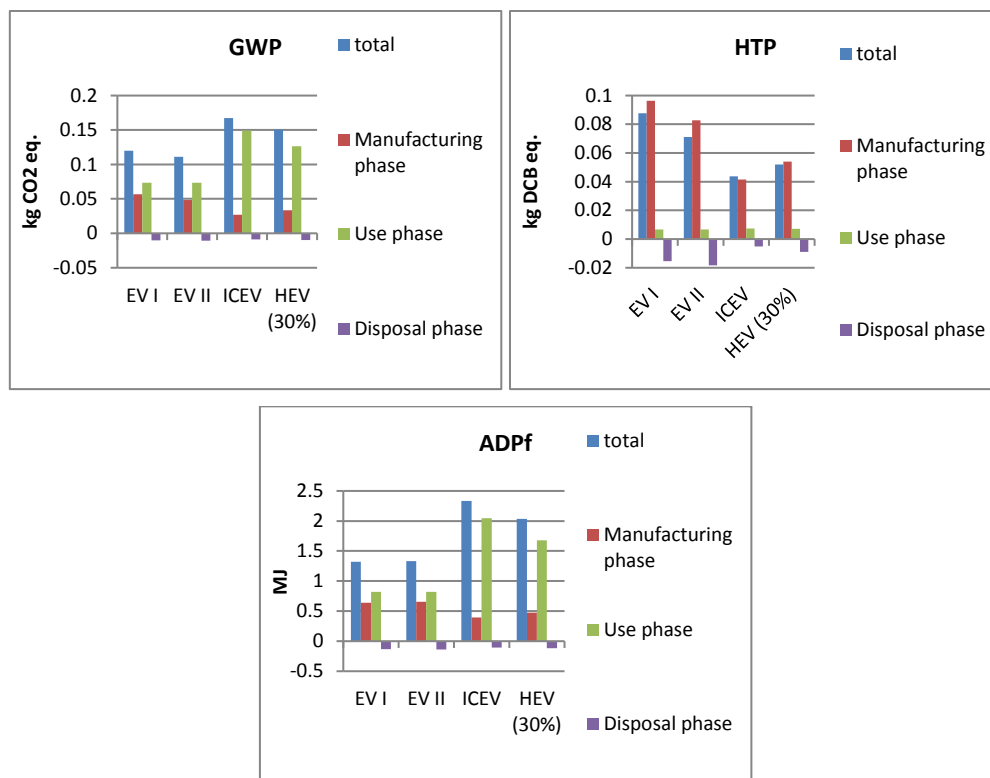
348 In LCA, the normalisation of the results is often used to identify the relatively significance of the impact
349 categories and those that score the highest in the normalised results are usually further discussed in the hot spot
350 analysis. However, according to the following reasons, in this work we decided to analyse the ADP fossil, HTP
351 and GWP categories even though they do not determine the highest normalised impacts (a detailed break-down
352 of all the other impact categories is reported in the supplementary information). In this study, more detailed
353 results are reported for the GWP because in the western countries, policies are rarely constraining impacts on
354 water resources as a result of the increased focus on carbon emissions and global warming; GWP was chosen as
355 primary indicator because the majority of ‘green policy’ and targets for climate change are set on greenhouse
356 gas emissions without specifically looking at other indicators or water impacts. The ADP fossil is also analysed
357 to quantify the impact on depletion of fossil resources and thus on use of primary energy; furthermore, the HTP
358 is considered because of the impact of this category on human health. All other environmental indicators are
359 reported in the supplementary information for further information.

360 **5.2 Base scenario**

361 Figure 4 shows the GWP of the different technologies analysed for the high recycling rate scenario; the total
362 impacts are broken down into the manufacturing, use and disposal phases and the results are reported for the
363 functional unit.

364 The total GWPs of the two EV models are very similar (0.12 kg of CO₂eq for EVI and 0.11 kg of CO₂eq for
 365 EVII) whereas the GWP of the ICEV is 45% higher (0.16 kg of CO₂eq). The higher GWP of the ICEV is due to
 366 the higher impact associated with the use phase: the disposal phase of all models is almost the same; the
 367 manufacturing phase of the ICEV determines the lowest GWP and hence, it is the use phase to determine the
 368 total trend of the results for the ICEV. The higher GWP of the use phase of the ICEV model is associated to the
 369 greater amount of greenhouse gas emitted during the use of diesel as fuel when compared to the production and
 370 use of the current EU electricity mix for electric vehicles. For all models the use phase determines the major
 371 contribution to the total GWP (61% for EVI, 66% for EVII, 89% for ICEV and 80% for the HEV). Furthermore,
 372 for all the scenarios analysed the disposal phase determines a benefit contribution to the environment thanks to
 373 the allocation of avoided burdens according to the system expansion method adopted in this work. However,
 374 this phase accounts for both the ‘disruptive’ burdens associated with the reprocessing and the ‘beneficial’
 375 burden associated with recycling of metal: the association of these two phases results in the values for the
 376 disposal phase reported in Figure 4. Overall the disposal phase does not strongly contribute to the total
 377 environmental burdens of the technologies analysed as the benefit of metal recycling are reduced by the burdens
 378 of the energy intensive reprocessing processes.

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381 **Figure 4** GWP, HTP, ADP of the EVI, EVII, ICEV and HEV (30%) for the high recycling rate scenario

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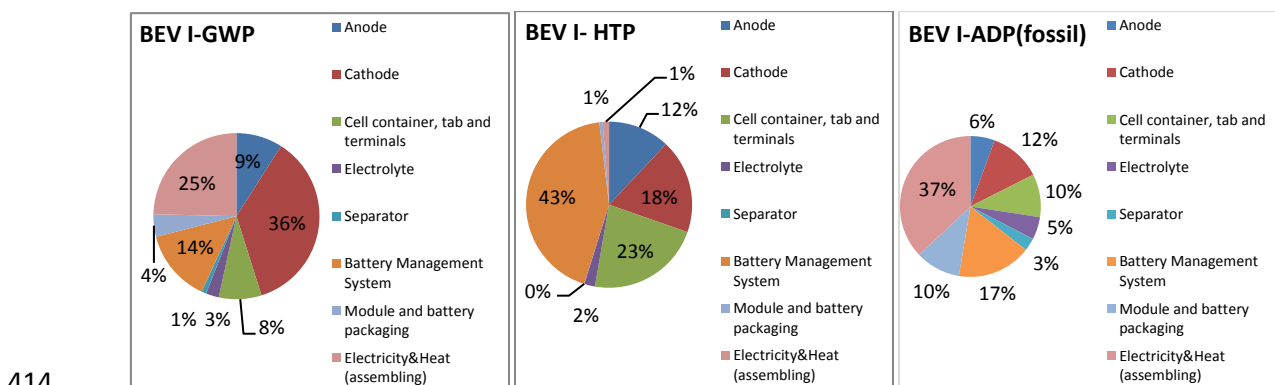
383 The manufacturing phase of the EVs determines a higher GWP than the manufacturing phase of the ICEV.
 384 Therefore, a further analysis has been performed on the manufacturing phases of the two EV models: about half
 385 of the total GWP of the manufacturing phase is due to the manufacturing of the battery pack. A detailed hot spot
 386 analysis of the EV battery pack model is presented in Figure 5 for EVI and EVII. The main contributor to the
 387 GWP of the battery pack for EVI is the manufacturing of the positive electrode paste as also found in previous
 388 works (Majeau-Bettez et al. 2011b). In particular, the indirect burdens associated with the production of the

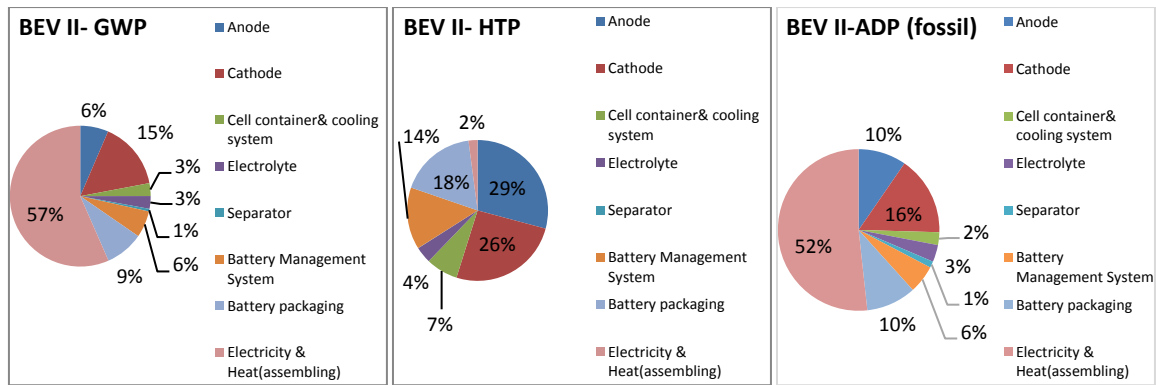
389 tetrafluoroethylene- a chemical used for the manufacturing of this paste- contribute for 78% to the GWP of the
 390 positive electrode paste. A different hot spot analysis is shown for the manufacturing of the battery pack for
 391 EVII (Figure 5). For this case, the energy used for the battery assembly determines more than 55% of the total
 392 GWP of the battery pack. Overall the manufacturing phase is a significant burden of the total GWP of an EV
 393 and it is comparable to the use phase.

394 The total ADP is reported in Figure 4. The trend of the results is the same as the GWP. The main contribution to
 395 the total depletion of energy resources is the use phase; particularly for the case of ICEV, this is associated to
 396 the diesel consumption (hence fossil resources). For the EV models, the impact of the manufacturing phase is
 397 comparable to the impact of the use phase and the burden is almost equally spread among the battery
 398 manufacturing and the manufacturing of the rest of the vehicle. The energy requirements for the battery
 399 assembly are the main contributors to both the ADP of EVI and EVII (the energy requirements for the battery
 400 assembly determine 37% of the total ADP of battery manufacturing for EVI and 52% of the total ADP of battery
 401 manufacturing for EVII) as shown in Figure 5.

402 A different trend of the results is shown for the HTP in Figure 4: the total HTP of the two EV models is higher
 403 than the total HTP of the ICEV model in opposition to what has been shown for ADP and GWP. The
 404 manufacturing phases of the EVs are the main contributors to this indicator. The processes associated with the
 405 chemical and metals production used in the manufacturing phase determine more emissions contributing to the
 406 toxicological impacts than the emissions associated with the production of electricity required during the electric
 407 vehicle use. In opposition to what reported for the other indicators, the HTPs of the EVI and EVII disposal
 408 phases are more than double than the disposal phase of the ICEV. The detailed hot spot analysis on the HTP of
 409 the batteries manufacturing is reported in Figure 5.

410 Further aggregated results for the base scenario of EVI, EVII and ICEV are reported in the supplementary
 411 information where all environmental indicators are shown. The hot spot analysis of the EVI, EVII and ICEV for
 412 all environmental indicators is also reported in the supplementary information and the burdens of the
 413 manufacturing, use and disposal phases are identified.

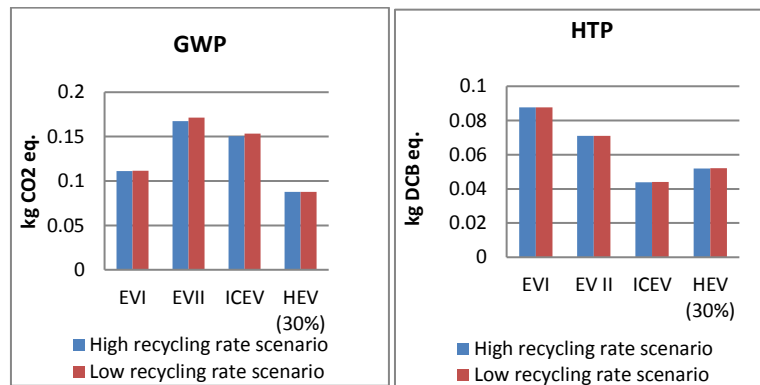




415
416 **Figure 5** Hot spot analysis of the manufacturing phases for EVI and EVII

417 **5.3 Scenario analysis: disposal phase**

418 A scenario analysis has been performed on the disposal phase: in previous results the high recycling rate has
 419 been considered whereas the results according to the modelling of the low recycling rate are reported in Figure
 420 6. The GWP and HTP results for EVI, EVII, HEV (30%) and ICEV are reported for comparison. As shown
 421 previously, the disposal phase is not the major contribution to the total environmental impacts. Therefore, a
 422 change of the modelling assumptions according to the low recycling rate scenarios does not determine a
 423 significant variation of the results. This means that whether the vehicle fleet is entirely disposed in EU countries
 424 or a proportion is disposed outside the EU boundaries, the environmental impacts of the vehicle life cycle does
 425 not change significantly.

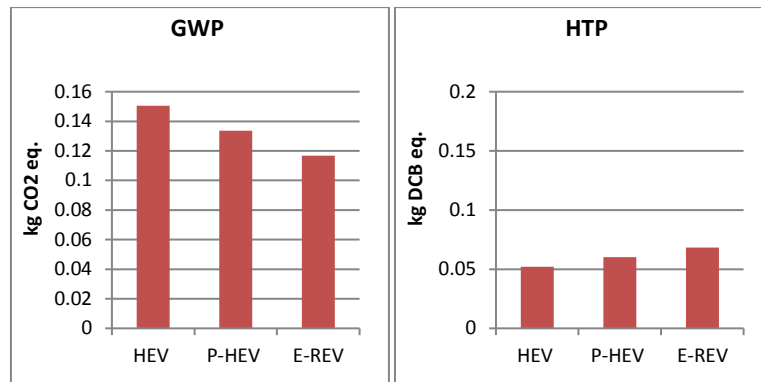


426
427 **Figure 6** Total GWP and HTP for 'high recycling rate scenario' and 'low recycling rate scenario'
428

429 **5.4 Scenario analysis: hybridisation factor**

430 Figure 7 shows the GWP and the HTP for different hybridization factors: the trend for the two indicators is
 431 opposite. An increase in the ratio of the electric motion determines a decrease in the total GWP as opposed to
 432 what happens for the HTP. This result is due to the opposite trend of the HTP and GWP already shown in Figure
 433 4 where the ICEV was the best option for the HTP and the EVs were the best options for the GWP.

434 A decrease in the hybridisation factor determine and increase of the GWP of up to 25% (for the HEV) when
 435 compared to the base scenario of the EVII as reported in Figure 4; on the other hand the HTP decreases up to
 436 28% (for the HEV) with a decrease of the hybridisation factor when compared to the base scenario of EVI.
 437



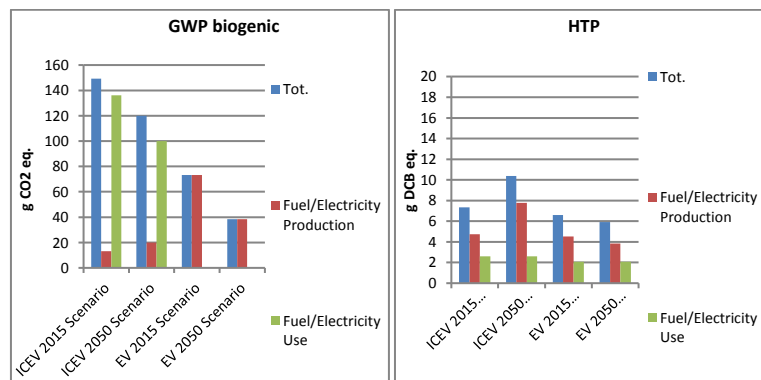
438
 439 **Figure 7** GWP and HTP for different hybridization factors

440 **5.5 Future energy mixes**

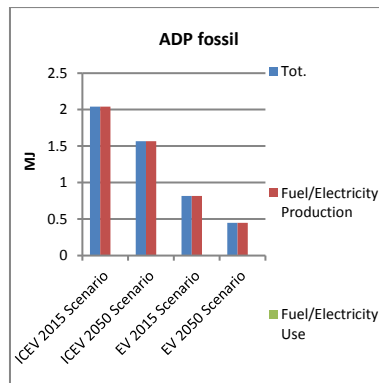
441 Finally, Figure 8 shows the ADP, HTP and GWP of the use phase models analysed for current and future energy
 442 mix in EU. For this case the GWP biogenic –that considers the uptake of the CO₂ from atmosphere during the
 443 growth of the organic matter- is considered in order to analyse the effects of the biodiesel share in 2015 and
 444 2050. For the GWP, both the ICEV and EV improve their performances in time and, therefore, the relative
 445 difference does not change between 2015 and 2050. The decrease of the GWP associated with the ICEV is due
 446 to the increased share of bio-diesel considered in the diesel mix and hence to the biogenic CO₂ emissions.
 447 However, the GWP of the ICEV associated to the production of diesel increases (see red bars of ICEV in Figure
 448 8); this is due to a higher environmental impact of bio-diesel production when compared to the fossil diesel, as
 449 later discussed. For the EV, the decrease of the GWP is due to the increased share of renewable making up the
 450 electricity mix in 2050.

451 In 2050, the EV represents the best option according to the total HTP. In fact, the HTP of the ICEV significantly
 452 increases over time according to an increase share of biodiesel in the mix. This point is further analysed in the
 453 discussion.

454 Future energy scenarios do not alter the relative trend of ICEV and EVs according to the ADP.



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Figure 8 GWP, HTP and ADP of the ICEV and BEVI use phase for current and future energy mix.

458

This analysis only considered the impacts of different energy shares on the use phase, excluding a possible variation of the environmental impacts associated with the manufacturing and disposal phases. The calculations were not reported for the case of the manufacturing and use phase as the variation in the results was less strong than what shown for the use phase. However, lower energy requirements associated with a ‘greener energy mix’ also determined a decrease of the environmental impacts associated with both the manufacturing and disposal phases (lower avoided burdens would therefore be allocated for the recycling of materials in the disposal phase)

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464 6. Discussion

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The GWPs of the BEV for EVI and EVII and high recycling rate scenarios have been compared to the results reported by previous studies, as shown in Figure 9: the results presented in this study are within the range of GWPs already reported. The differences mostly stem from differing assumptions concerning manufacturing energy requirements and system boundaries. Data from older studies are placed in the upper range of the literature results; higher energy production efficiencies, advanced technologies and higher shares of renewable energy have contributed to the decrease of the total GWP of EVs. The same trend has been predicted to continue also for the future years according to the analysis reported for the future energy mix in 2050.

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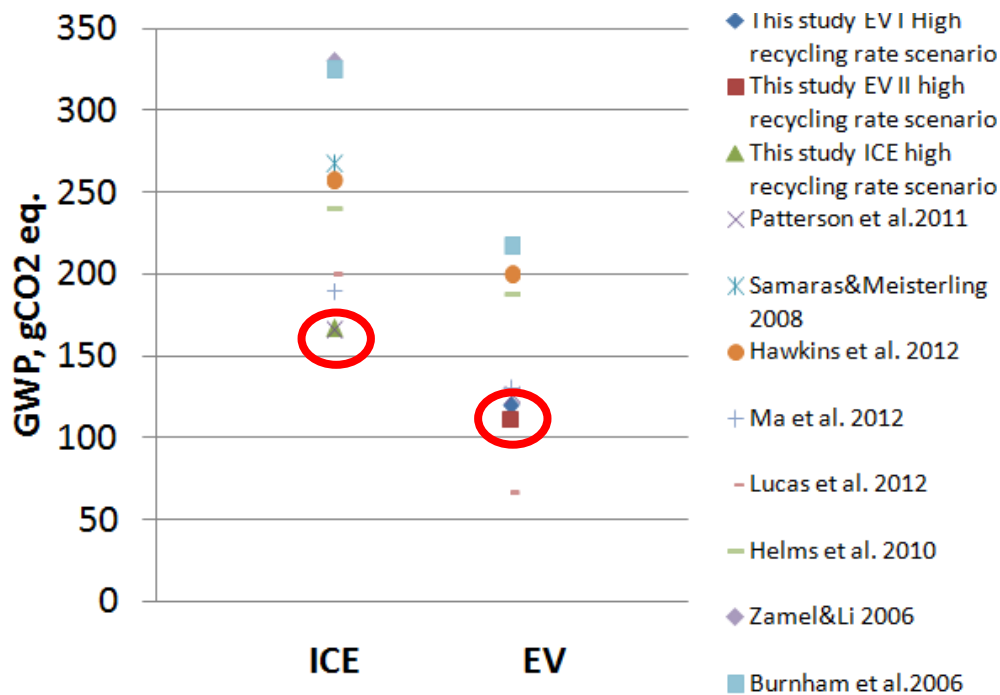
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Figure 9 GWP: comparison with literature

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475 When considering the entire life cycle of passenger vehicles, the gradual substitution of the ICEV fleet with the
 476 BEV fleet will determine a progressive reduction of the greenhouse gas emissions from the automotive sector
 477 thanks to the reduced emissions associated to the use phase. However, more technological developments are to
 478 be considered for the manufacturing phase, especially for the battery manufacturing. The results have shown
 479 that the high toxicological impacts are strictly linked with the exploitation of precious metals and production of
 480 chemical used in the battery manufacturing phase. Therefore, advanced processes and higher efficiencies are
 481 required to limit the impacts on water and human life. Although these are key points for the development of a
 482 BEV fleet, it has to be recognised that the use of conventional internal combustion engine vehicles might
 483 represent an even higher threats in future when considering the changing mix of fuels. Production of diesel from
 484 soybeans is known (Panichelli et al. 2008; Rocha et al. 2014) to determine a high human toxicity impact
 485 associated mainly to pesticide use during crop growth and fossil fuel consumption for oil extraction during
 486 biodiesel production.

487 This study presented a significant improvement to the modelling of the inventory data used to build the
 488 assessment, particularly for the disposal treatment. In fact, although publicly available data have been used for
 489 the inventory, this study has uniquely developed the model of the disposal phase according to a currently used
 490 industrial process. Furthermore, the attributional analysis associated with the consequential approach for the
 491 calculation of the avoided burdens adopted in this study, was able to identify future changes of the energy mix
 492 and project the environmental impacts of developing technologies.

493 **7. Conclusions**

494 The transport sector is one of the most challenging when tackling the targets on emissions reduction: developing
495 technologies in the automotive industry, such as electric and fuel cell vehicles, associated with the use of low-
496 carbon content fuels are appealing solutions to potentially reduce greenhouse gas emissions.

497 This study presented a life cycle assessment of an electric passenger vehicle using a Lithium-ion battery
498 compared to an internal combustion engine vehicle and hybrid vehicles. A hot spot analysis was also performed
499 to identify the phases of the entire vehicle life cycles that are firstly to be addressed to reduce the overall
500 environmental impacts. Three major phases makes up the entire life cycle of the vehicles, the manufacturing,
501 use and disposal. A further break down of the impacts associated to these phases has been reported in the hot
502 spot analysis. Two models for the manufacturing of the EV have been analysed according to different inventory
503 data.

504 The ICEV determines a higher total global warming than the BEVs: this is mostly due to the greater (by almost
505 50%) amount of greenhouse gas emitted in the use phase. Conversely, the manufacturing phase of the BEVs is
506 almost double that of the ICEV: the higher global warming of the EV manufacturing is explained by a more
507 complex propelling system that includes the battery manufacturing. This is associated with the production and
508 use of metals, chemicals and energy required in the systems. The same trend as the GWP was also shown for the
509 ADP whereas the analysis of the HTP has shown that the total burden of EVs is higher than that of ICEVs. This
510 result is linked with the use of metals and chemicals for the battery manufacturing.

511 The LCA methodology has been used to predict the environmental impacts of the ICEV and BEV for future EU
512 energy mix. The trend of the results have shown that the GWP is projected to decrease for both technologies but
513 advanced processes for manufacturing of biodiesel for ICEV and battery for BEV need to develop further to
514 significantly reduce the toxicity impacts of both systems.

515 Two different disposal scenarios have been analysed. In the base scenario, the vehicle fleet was totally assumed
516 to be disposed in EU. Conversely, in the low recycling rate model, the part of the vehicle fleet that leaves the
517 EU was considered to be sent to landfill outside the EU. A negligible variation of the results was shown for the
518 two cases and this highlighted how the disposal phase has a minor impact on the total environmental burdens.

519 Overall this study has shown how the EVs are a potential technology that can contribute to the decrease of GHG
520 emissions when compared with conventional fuel vehicles. However, the manufacturing phase still represents
521 the major impediment to the total performance of the technology. Hence, significant fundamental research has
522 still to be developed on the subject: future energy mix and improvements of the technological efficiencies could
523 contribute to a reduction of the GWP of the BEV manufacturing phase decreasing the difference with the
524 manufacturing phase of the ICEV.

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