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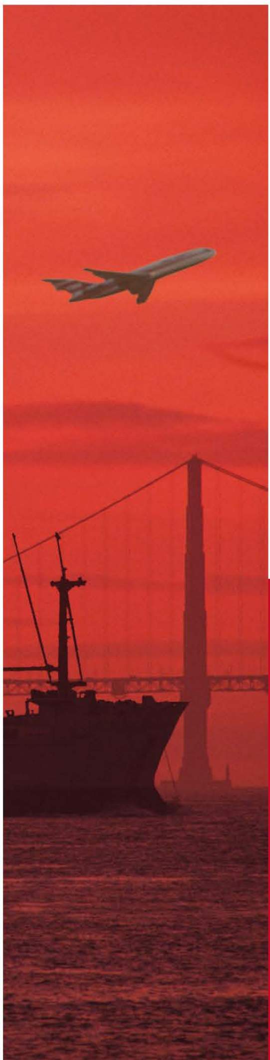

Research for TRAN Committee - Battery-powered electric vehicles: market development and lifecycle emissions

STUDY

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2018

DIRECTORATE-GENERAL FOR INTERNAL POLICIES
Policy Department for Structural and Cohesion Policies

TRANSPORT AND TOURISM

Research for TRAN Committee - Battery-powered electric vehicles: market development and lifecycle emissions

STUDY

Abstract

As 2018 gets under way, there are probably more than three million electric cars in circulation in the world. There are also more than six hundred million electric bikes, scooters and motorcycles. Plus a few hundred thousand electric buses and other types of quadricycles having an electric motor. The first part of this paper traces the fast evolving market of electric road vehicles.

The second part shows that the production of hundreds of millions of battery packs requires a lot of energy and plenty of scarce resources, which affects the real impact of electric vehicles on the climate and the environment and make it necessary to consider the recovery and recycling of used batteries.

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TABLE OF CONTENTS

PART 1:
THE DEVELOPMENT OF BATTERY–POWERED ROAD VEHICLES MARKET **5**

PART 2:
RESOURCES, ENERGY, AND LIFECYCLE GREENHOUSE GAS EMISSION ASPECTS OF ELECTRIC VEHICLES **15**

PART 1

The development of battery–powered road vehicles market

Abstract

This paper traces the fast evolving market of electric road vehicles worldwide with focus on the European Union at the end of 2017.

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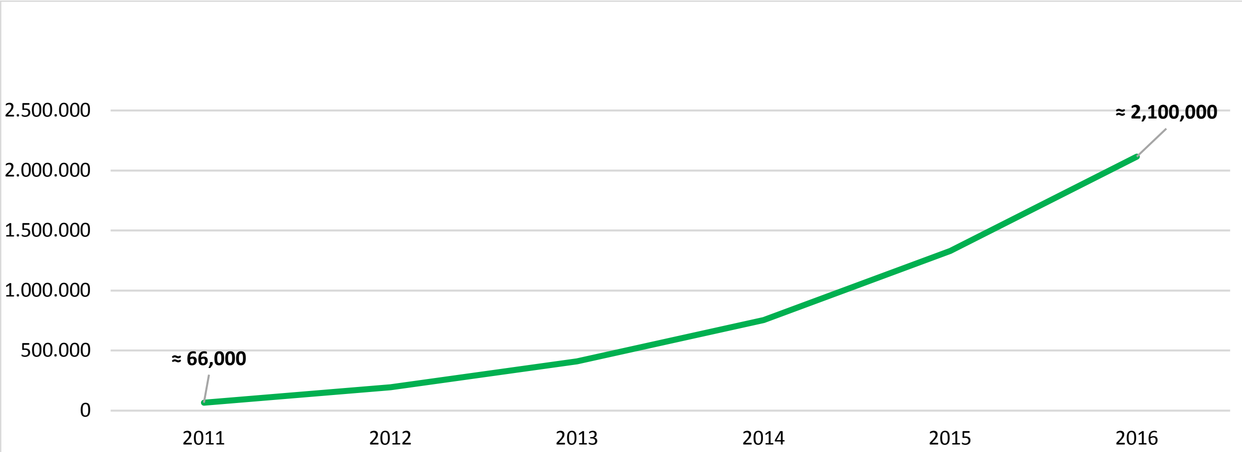
1. WORLD MARKET DEVELOPMENT[1]

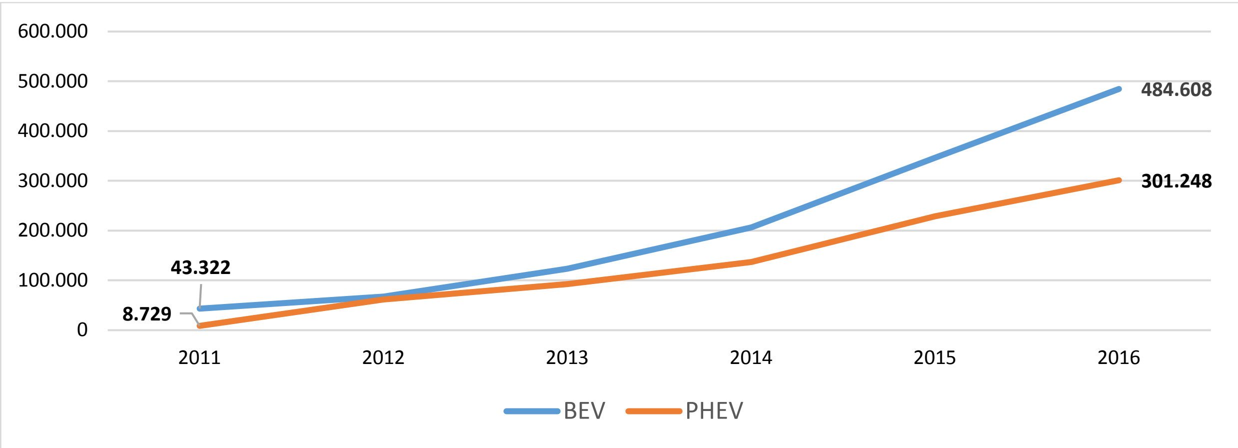
At the end of 2016, there were just over 2 million electric cars[2] in the world. This was less than 0.2% of the global car fleet[3], but this level was achieved in about seven years (the sales of electric vehicles before 2010 are considered insignificant).

Figure 1: Global stock of electric cars (M1+N1), by year (BEV+PHEV)



In 2016, over 780,000 new electric cars were registered worldwide (of which 62% battery-electric vehicles[4]). Annual sales growth, however, dropped below 50% for the first time (+37% compared to 2015), the growth rate being on a downward trend since 2011.

Figure 2: Global sales of electric cars (M1+N1), by year (BEV and PHEV)



[1] Sources: For the EU and Norway: European Alternative Fuel Observatory and EC, Joint Research Centre, Electric vehicles in the EU from 2010 to 2014. For the USA: Hybridcars and Electric Drive Transportation Association. For China, Japan and the rest of the world: International Energy Agency, Global electric vehicles outlook 2017.

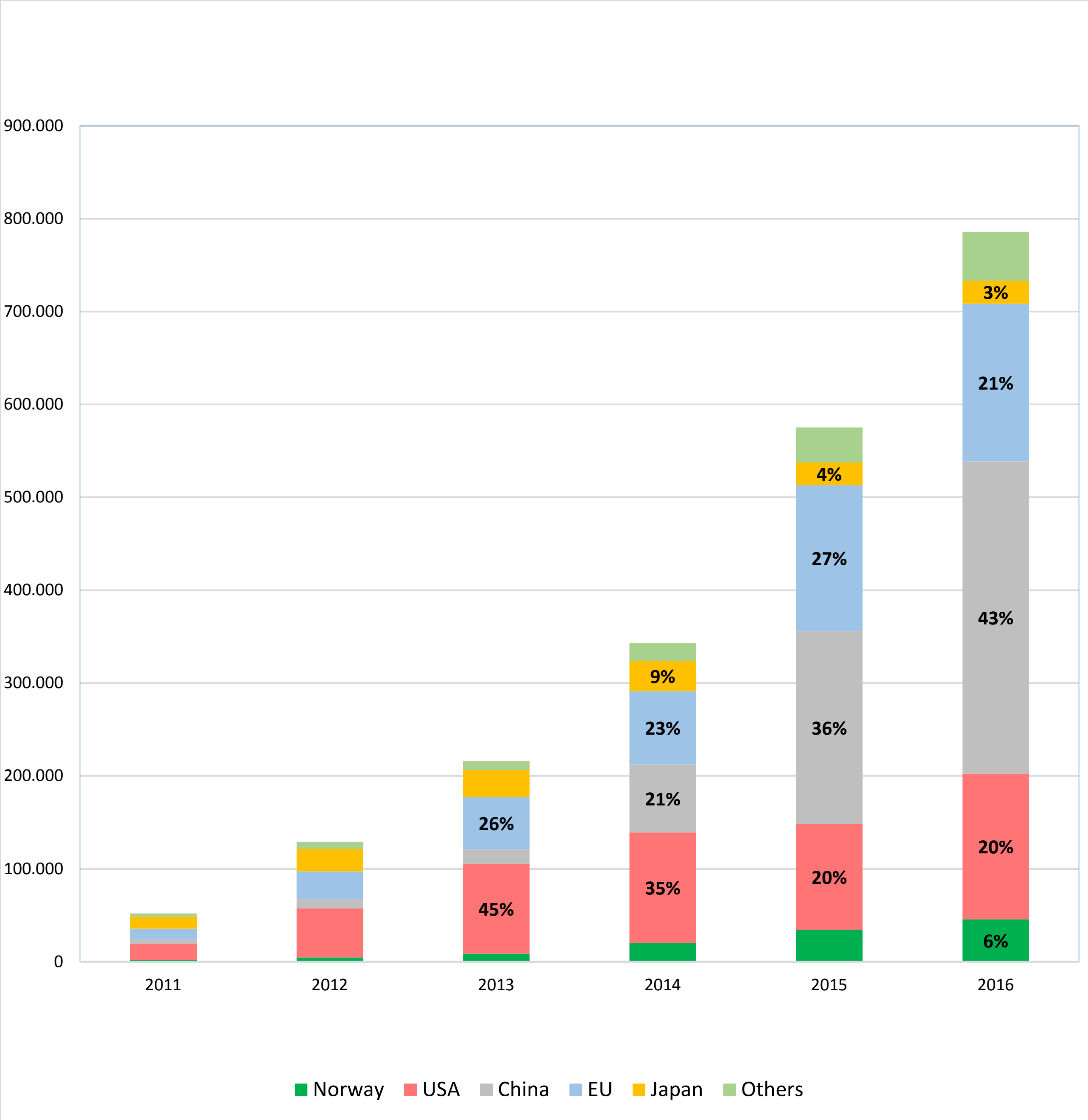
[2] "Cars" means passenger cars, including pickups and vans, and light commercial vehicles (LCV). **Vehicles currently on the market match EU categories M1 and N1**. The categories of vehicles are those defined by Directive 2007/46/EC and Regulation EU No 168/2013.

[3] According to the International Organization of Motor Vehicle Manufacturers (OICA), there were around 1.3 billion (passenger and commercial) vehicles in use in the world at the end of 2016 (including heavy trucks, coaches and buses).

[4] **Battery-electric vehicles (BEV)** derive their power only from their rechargeable battery packs. **Plug-in hybrid electric vehicles (PHEV)** derive their power from their rechargeable battery packs and from an internal combustion engine.

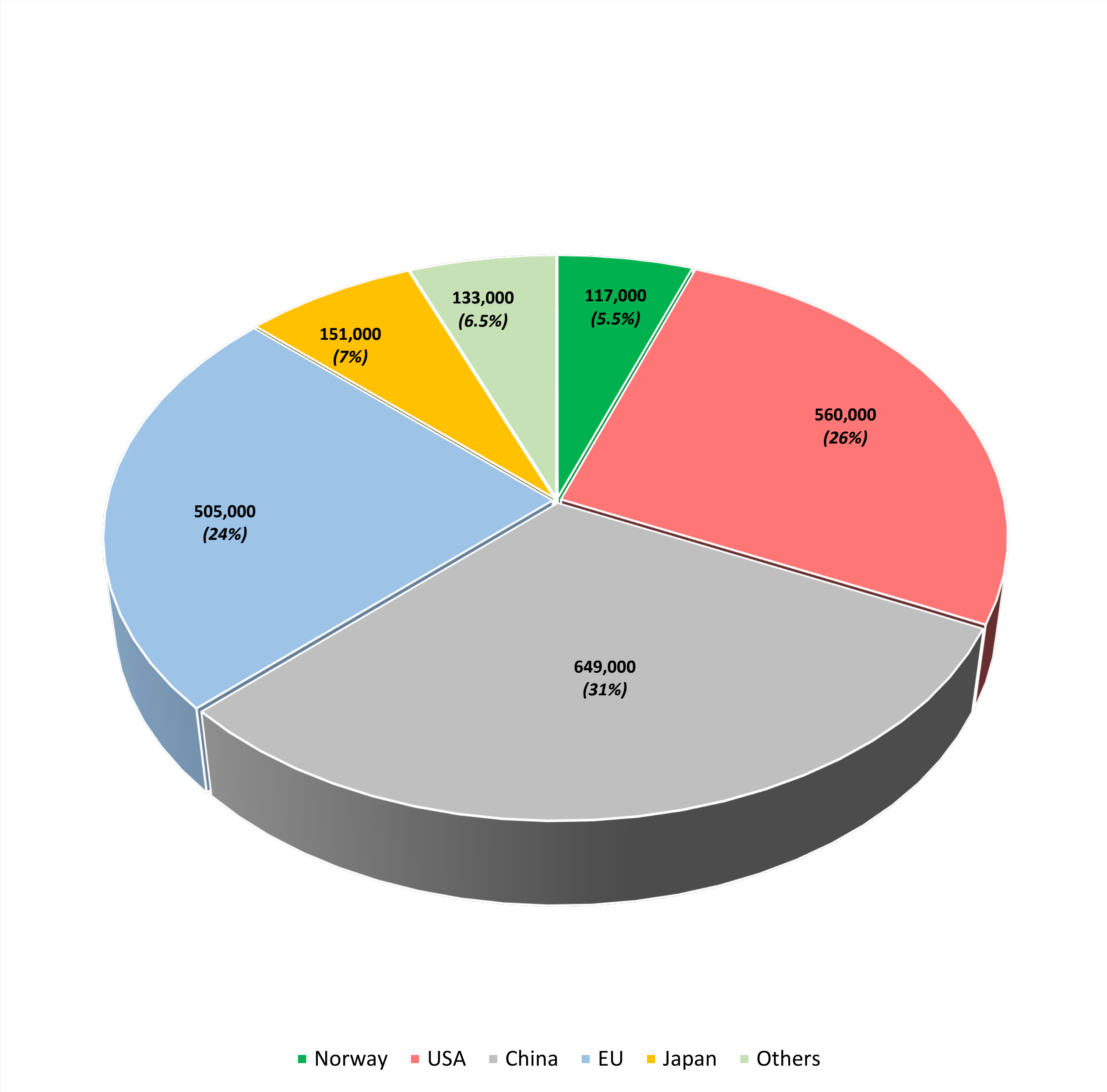
The marketplace is highly concentrated. Together, the five largest markets accounted for 93% of 2016 global sales of electric cars. The same also accounted for almost 94% of the world's fleet of such vehicles. Furthermore, according to the *International Council on Clean Transportation*, nearly a third of global electric vehicle sales would be in just 14 metropolitan areas[5]. Since 2015, **China is by far the main sales place for electric cars**, followed by the EU and the US.

Figure 3: Global sales of electric cars (M1+N1), by year (BEV and PHEV)



[5] ICCT, [Electric vehicle capitals of the world]() (March 2017). Namely, in descending electric vehicle share: Oslo, Utrecht, Shanghai, Shenzhen, Amsterdam, San Jose, San Francisco, Copenhagen, Beijing, Stockholm, Zürich, Los Angeles, Paris and London.

Figure 4: Global stock of electric cars (M1+N1) in 2016, by volume and market share (Total BEV + PHEV ≈ 2,100,000)



Electric cars represent a tiny fraction of car sales. In 2016, this fraction was above 1% only in Norway (29% of car sales), in China (1.4%) and, by a narrow margin, in the EU (1.1%). In 2017, it reached 1.35% in the EU[6].

Globally, battery electric cars dominate the market (around 61% of world electric car stock at the end of 2016). In 2016, this was the case in China (75%), Norway (74%), Japan (57%) and the USA (52%). In the EU, BEV and PHEV are at the same level, but the situation differs greatly between Member States.

[6] Based on 16.6 million new vehicles registered in the EU in 2016, and 17.1 million in 2017 (categories M1 and N1). Source: European Automobile Manufacturers' Association (ACEA). In 2017, this share reached a peak of 32.4% in Norway.

The number of fuel cell vehicles is still very negligible[7]. Similarly, **heavy electric vehicles are still rare**. According to the *International Energy Agency* (IEA), there were around 345,000 electric buses in the world (mainly BEV) at the end of 2016. Almost all of them were in China (343,500), about 1,200 were registered in the EU (M2 and M3 categories - this number rose to just over 1,500 in 2017) and 200 in the United States. So far, the number of electric heavy-goods vehicle is derisory.

It is worth mentioning that there are significant data classification issues for buses and coaches. There are also no precise classifications/figures for "low-speed electric vehicles", "two-wheelers" and "three-wheelers" - for which international comparisons are unreliable. Moreover, it is impossible to know precisely the global number of electric bicycles (e-bikes): while most of them are made in China, in general importing countries (including the EU, the US and Japan) do not distinguish them from their imports of conventional bicycles. In addition, the lifetime of e-bikes in circulation is unknown.

According to the IEA, 200 to 230 million electric two-wheelers were in circulation in China in 2016 - this country being, by far, the world leader. In the same year, the *European Alternative Fuel Observatory* listed about 14,000 electric motorbikes (categories L1 to L5) in the EU (this number rose to approximately 18,000 in 2017).

Identically, China dominates the market of low-speed electric vehicles. It is estimated that three to four million units were in circulation in this country in 2016, compared to around 20,000 in the EU (categories L6 and L7 - this number rose to about 21,000 in 2017).

China also strongly leads on the electric bike market. The most reliable estimates say that, **in 2016, there were around 250 million e-bikes (all types) in circulation in the country**, and 30 million new electric bicycles are expected to be sold there every year. In the EU, 8.2 million e-bikes (all types) were sold between 2006 to 2016. In Japan (where e-bikes emerged in the early 1980s and, unlike other markets, local production largely outweighs imports) 550,000 e-bikes were sold in 2016, but the national stock of electric bike cannot be accurately assessed. The US situation is equally unclear. While all e-bikes or parts thereof are imported from China, the absence of a specific customs code makes these imports difficult to identify. The highest estimates say that between 211,000 and 251,000 e-bikes were sold throughout the country in 2016. On the other hand, a study by the consultancy *NAVIGANT* estimates this number to be little more than 137,000[8].

[7] **Fuel cell vehicles** use a fuel cell to generate electricity to power their electric motor.

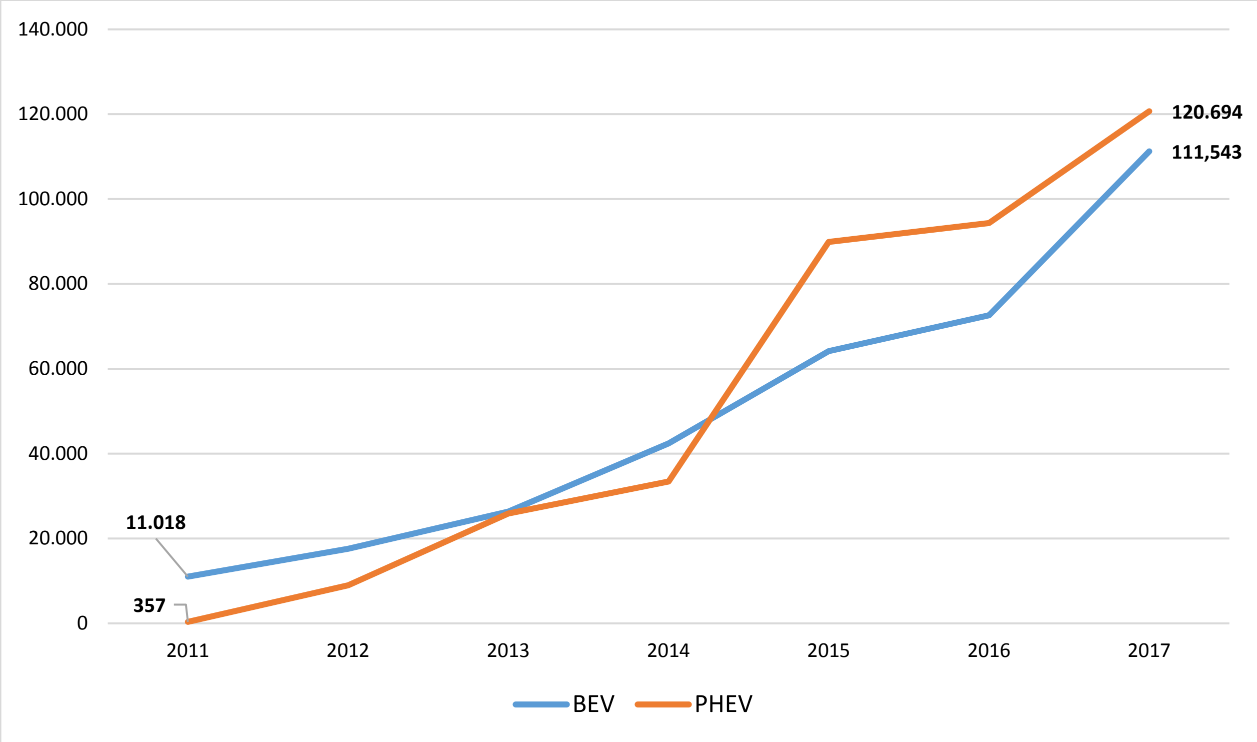
[8] NAVIGANT; electric bicycles

2. EU MARKET DEVELOPMENT

At the end of 2017, there were around 738,000 electric cars (M1 and N1 categories) in the EU, i.e. roughly 0.2% of the Union's car fleet[9].

Year 2017 broke a sales record with about 232,000 new electric cars registered (+37% compared to 2016). Since 2015, PHEV sales exceed BEV sales (in 2017, the ratio was 48-to-52).

Figure 5: Annual sales of electric cars (M1+N1) in the EU (BEV and PHEV)



As at the global level, the EU marketplace is very concentrated. In 2017, four Member States accounted for 74% of EU sales of electric cars (Germany, UK, France and Sweden). In the same year, the same countries plus the Netherlands concentrated 80% of the EU fleet of such vehicles[10].

---

9 In 2016 (most recent available data), there were about 290,000,000 passenger cars (M1) and light commercial vehicles (N1) in use in the EU (Source: ACEA).

10 Note that in the Netherlands the sales of electric cars peaked in 2015 (at about 45,000 vehicles), and then halved in 2016 and halved again in 2017 (at less than 10,000 vehicles). Conversely, in Germany the sales increased more than twofold from 2016 to 2017 (from around 27,500 to 58,000).

Figure 6: Sales of electric cars (M1+N1) in the EU, by volume and market share (BEV+PHEV)

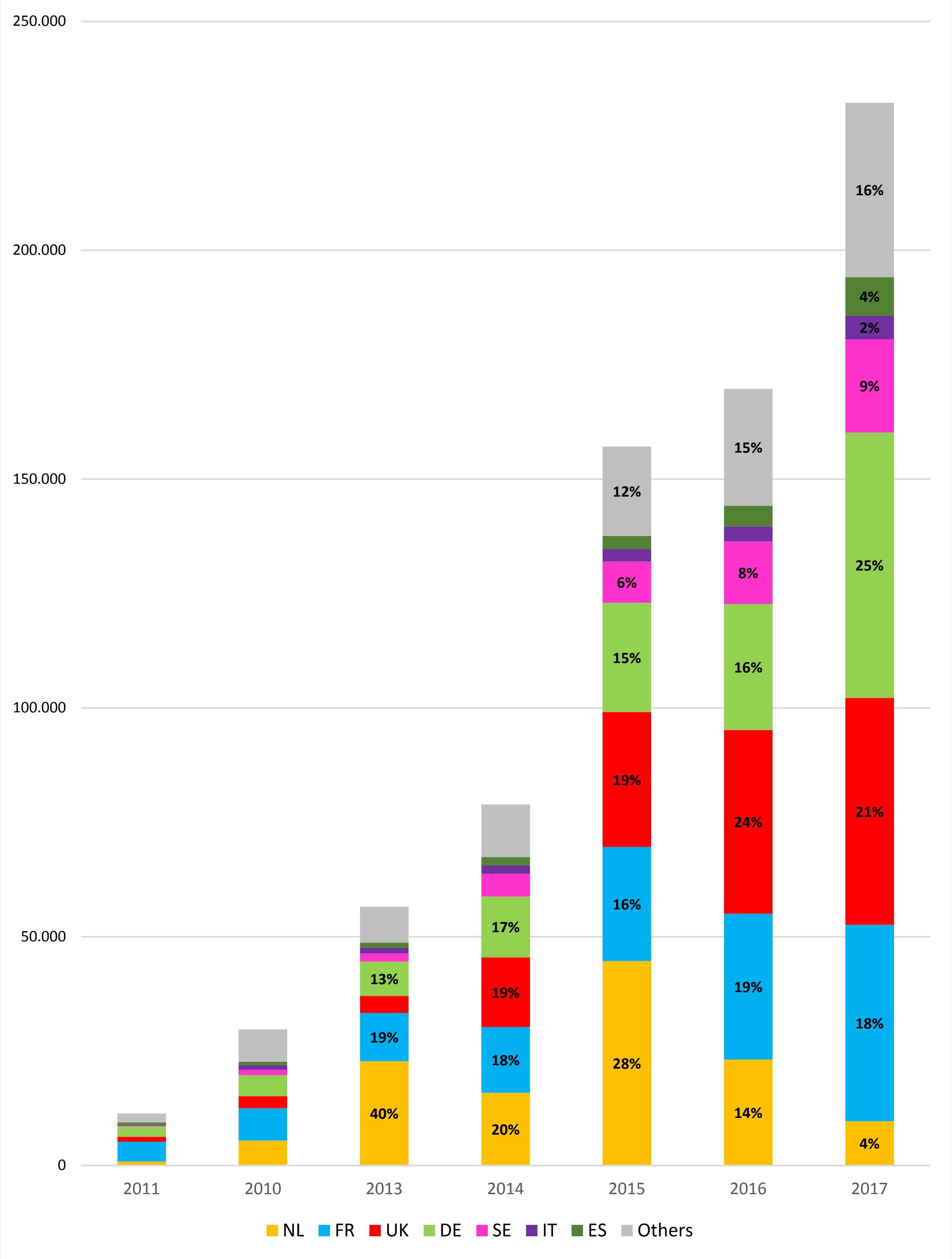
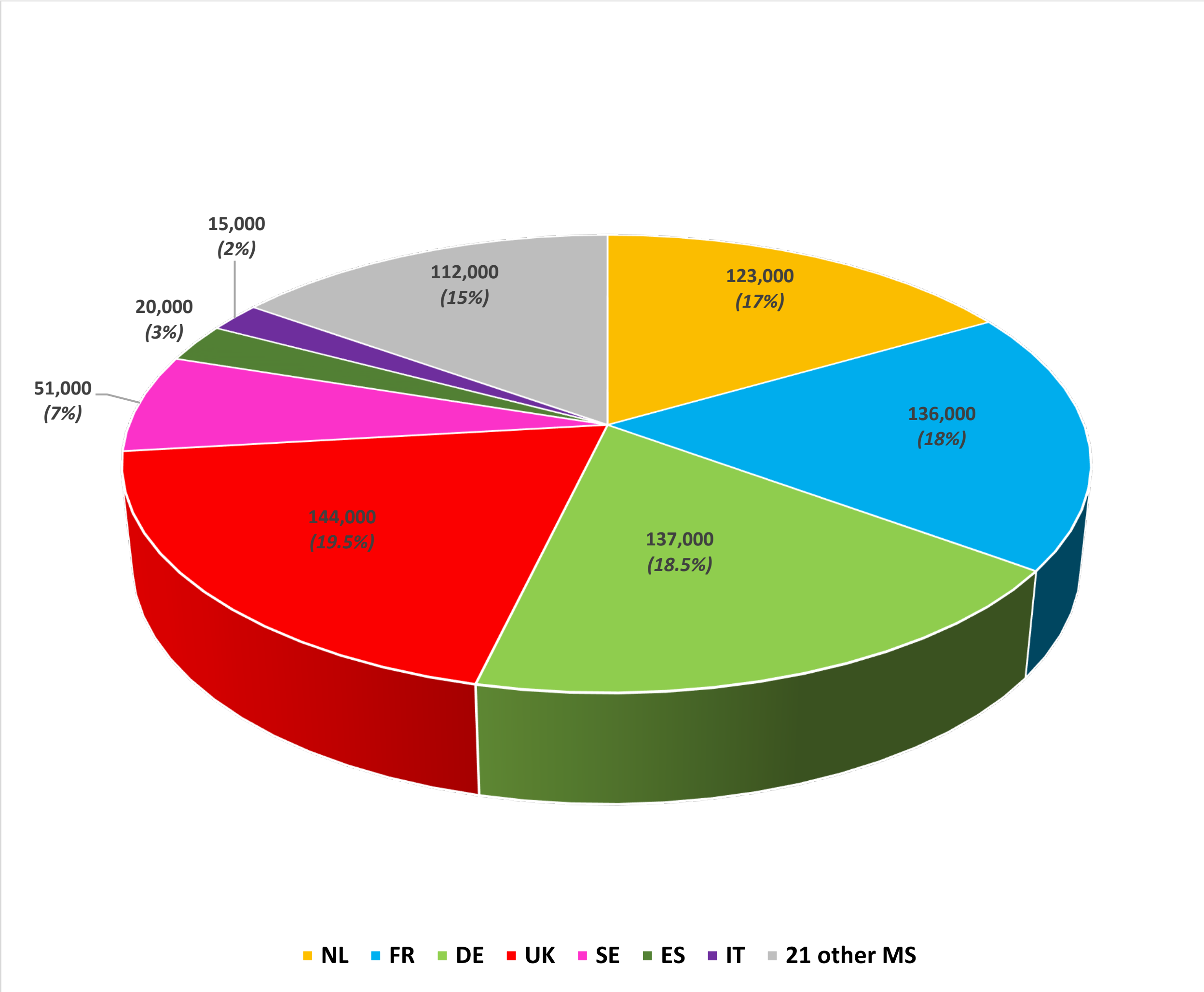
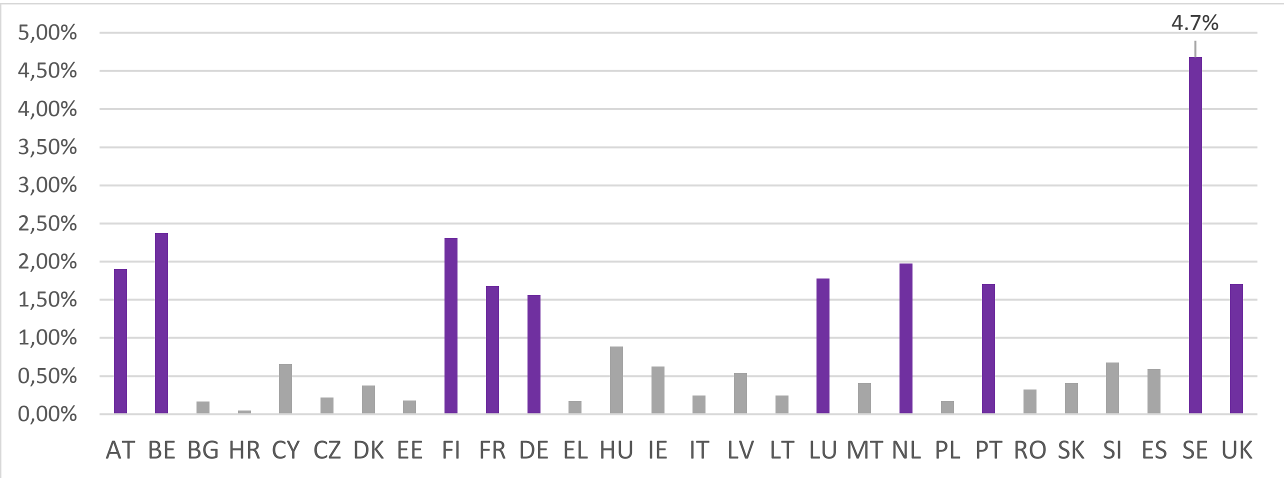


Figure 7: Stock of electric cars (M1+N1) in the EU in 2017, by volume and market share (Total BEV + PHEV ≈ 738,000)



Electric cars represent only a tiny fraction of EU car sales (1.35% in 2017). In 2017, this fraction was above 1% in ten Member States - among which Sweden (4.7%) particularly stood out.

Figure 8: Market shares of electric cars (M1+N1) in the EU Member States in 2017 (BEV+PHEV) (in % of new cars registered that year)



Plug-in hybrid electric cars (PHEV) slightly dominate the market (about 51% of EU electric car stock at the end of 2017) but the situation differs greatly between Member States. For instance in the Netherlands, 80% of electric cars in circulation in 2017 were PHEV, whereas this proportion was 73% in Sweden and 64% in the UK. By contrast, battery electric cars (BEV) dominated in France (79%), Spain (67%), Italy (61%) and Germany (55%).

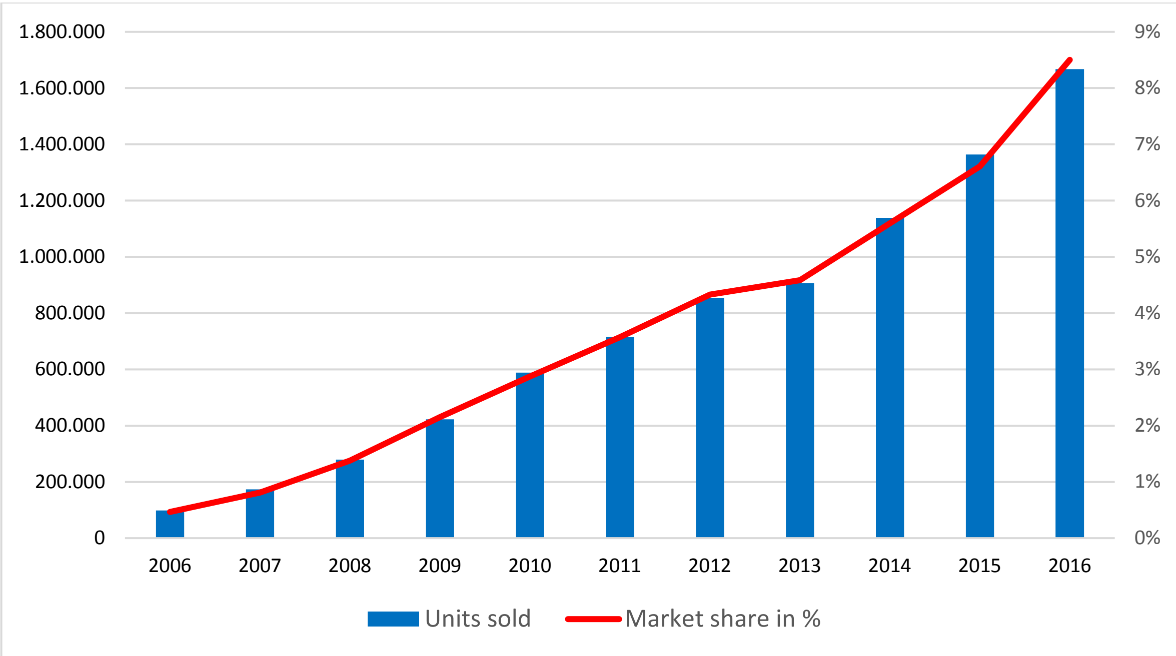
Fuel cell (FC) vehicles have been commercially available since 2013. Their numbers are still very negligible. Less than 500 FC cars and 200 FC LCVs[11] are currently registered in the EU, as well as 65 FC buses.

Identically, the number of heavy electric vehicles is very small, at around 2,200 buses (M2 and M3 categories).

At the end of 2017, there were also about 18,000 (+20% year on year) electric motorbikes (categories L1 to L5) and 21,000 low-speed electric vehicles (categories L6 and L7) registered in the EU.

Finally, according to the *Confederation of the European Bicycle Industry* (CONEBI), over 8.2 million (all types) of electric bikes were sold in the EU between 2006 and 2016 (i.e. about 3.7% of total bike sales over the period). In 2016, sales totalled 1.7 million e-bikes which represents a 22% increase compared to 2015. In the same year, Germany (36%), the Netherlands (16%) and Belgium (10%) together accounted for 62% of the EU market. (2017 figures not yet available)

Figure 9: Annual sales of electric bikes in the EU and related bike market share



[11] Of the 194 FC LCVs registered in the EU at the end of 2017, 182 were in France.

PART 2

Resources, energy, and lifecycle greenhouse gas emission aspects of electric vehicles

Abstract

This report covers topics associated with the production and recycling of lithium-ion traction batteries and lifecycle greenhouse gas emissions of fully battery-powered electric vehicles. It describes in particular material and energy use in battery production, and the climate change impacts of electric vehicles. The effects of the battery, vehicle size and the sources of charging electricity are discussed. Prospective developments in battery production and the power sector and their effect on lifecycle greenhouse gas emissions are also discussed.

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CONTENTS

GLOSSARY AND LIST OF ACRONYMS **19**

LIST OF FIGURES **21**

EXECUTIVE SUMMARY **23**

1. BATTERY ELECTRIC VEHICLES WITH LITHIUM-ION TRACTION BATTERIES **25**

2. BATTERY PRODUCTION **27**

2.1. Cell materials and manufacture 28

2.2. Battery pack components and assembly 29

2.3. Energy demands in battery production 30

3. MINERAL USE, SUPPLY RISKS AND BATTERY RECYCLING **33**

3.1. Supply risks associated with Li-ion batteries 36

3.2. End-of-life treatment of lithium-ion batteries 37

4. LIFECYCLE GREENHOUSE GAS EMISSIONS OF BATTERY ELECTRIC VEHICLES **41**

4.1. Lifecycle climate change impact 42

4.2. Other environmental impacts 46

5. CONCLUSIONS **49**

REFERENCES **51**

GLOSSARY AND LIST OF ACRONYMS

BEV Fully battery-powered electric vehicle.

BMS Battery management system.

EOL End-of-life.

FDP Fossil depletion potential. Environmental impact category associated with the consumption of fossil-based resources such as oil, natural gas and coal.

FEP Freshwater eutrophication potential. Environmental impact category associated with the oversupply of nutrients to freshwater environments, causing oxygen deprivation and the death of larger life forms.

FETP Freshwater ecotoxicity potential. Environmental impact category associated with the release of toxic compounds into freshwater environments.

GHG Greenhouse gas.

Gravimetric energy density Energy density with respect to mass (as opposed to volume), measured in kWh/kg.

GWP Global warming potential. Environmental impact category associated with climate change.

HTP Human toxicity potential. Environmental impact category associated with the release of toxic compounds.

Hydrometallurgy Recycling method using water-based solvents to recover metals.

ICEV Internal combustion engine vehicle.

LCA Lifecycle assessment. Method for determining the environmental impact of products or technologies.

LFP Lithium iron phosphate. Active cathode material in lithium-ion batteries.

LIB Lithium-ion (traction) batteries.

LMO Lithium manganese oxide. Active cathode material in lithium-ion batteries.

MDP Mineral resource depletion potential. Environmental impact category associated with the consumption of mineral resources.

MEP Marine eutrophication potential. Environmental impact category associated with the oversupply of nutrients to marine environments, causing oxygen deprivation and the death of larger life forms.

METP Marine ecotoxicity potential. Environmental impact category associated with the release of toxic compounds into marine environments.

NCA Lithium nickel-cobalt-aluminium oxide. Active cathode material in lithium-ion batteries.

NCM Lithium nickel-cobalt-manganese oxide. Active cathode material in lithium-ion batteries.

NEDC New European Driving Cycle. Used to assess emissions and the fuel economy of light duty vehicles.

NMP N-methylpyrrolidone. Solvent used in lithium-ion battery production.

ODP Ozone depletion potential. Environmental impact category associated with the release of compounds that damage the ozone layer.

PMFP Particulate matter formation potential. Environmental impact category associated with the release of small compounds which may be detrimental to human health. Indicator for local air quality.

POFP Photochemical oxidant formation potential. Environmental impact category associated with the release of smog-forming compounds. Indicator for local air quality.

PHEV Plug-in hybrid electric vehicle. Hybrid electric vehicle that has both an internal combustion engine and a battery-powered electric motor that can be recharged by plugging it into an external source of power.

Pyrometallurgy Recycling method using high temperatures to recover metals.

TAP Terrestrial acidification potential. Environmental impact category associated with the release of compounds that cause a change in the acidity of soils.

TETP Terrestrial ecotoxicity potential. Environmental impact category associated with the release of toxic compounds into terrestrial environments.

WTW Well-to-wheel cycle. Describes the value chain for energy carriers (liquid fuels, electricity, natural gas or hydrogen) for vehicles, from resource extraction, refining, transmission/transport and energy conversion.

LIST OF FIGURES

FIGURE 1
Production flow diagram for lithium-ion traction battery 27

FIGURE 2
Generic material composition of a Li-ion traction battery 33

FIGURE 3
Relative supply risk indicators and overall supply riskfor each element 37

FIGURE 4
The complete lifecycle of vehicles 42

FIGURE 5
Lifecycle greenhouse gas emissions of battery electric and internal combustion engine vehicles 43

FIGURE 6
Lifecycle environmental impacts of battery electric and internal combustion engine vehicles 47

EXECUTIVE SUMMARY

Battery electric vehicles with lithium-ion traction batteries

Fully battery electric vehicles (BEVs) have repeatedly been considered a promising alternative to conventional internal combustion engine vehicles (ICEVs), but have only recently attracted wider consumer interest and gained acceptance due to progress in battery technology. As a result of their higher gravimetric energy density, lithium-ion traction batteries permit longer driving ranges than other battery technologies. They are therefore the battery of choice for automobile manufacturers.

Battery production

Battery production can be subdivided into cell manufacture and pack assembly**. Cell manufacture is a complex and protracted process with stringent requirements in relation to ambient indoor conditions and cleanliness in building zones as well as a high demand for energy.** Current battery cell manufacture primarily takes place in South Korea, Japan, and China. In comparison to cell manufacture, pack assembly is a far less complex and energy-intensive process. The battery packs are either assembled by a cell manufacturer and then delivered to the automobile manufacturer or are assembled by the automobile manufacturers themselves.

Mineral use, supply risks and recycling

Lithium-ion traction batteries are complex products containing a variety of materials, some of which are or may become susceptible to supply risk. Minerals that are particularly susceptible to supply risk in producing current lithium-ion traction batteries are: lithium (Li), aluminium (Al), manganese (Mn), iron (Fe), cobalt (Co), nickel (Ni), copper (Cu), and graphite (C). **Lithium and cobalt are likely to be susceptible to the highest supply risk**, while aluminium involves the lowest risk. Manganese, iron, nickel, copper, and natural graphite have medium supply risk. The improvement of recovery processes for these elements at the vehicle's end-of-life can, to some degree, alleviate these supply risks. As we move from fuel-intensive ICEVs to materials-intensive BEVs, it becomes **increasingly important to have efficient recycling processes in place** to ensure the optimal recovery of finite minerals as well as energy- and pollution-intensive materials.

Lifecycle emissions of battery electric vehicles

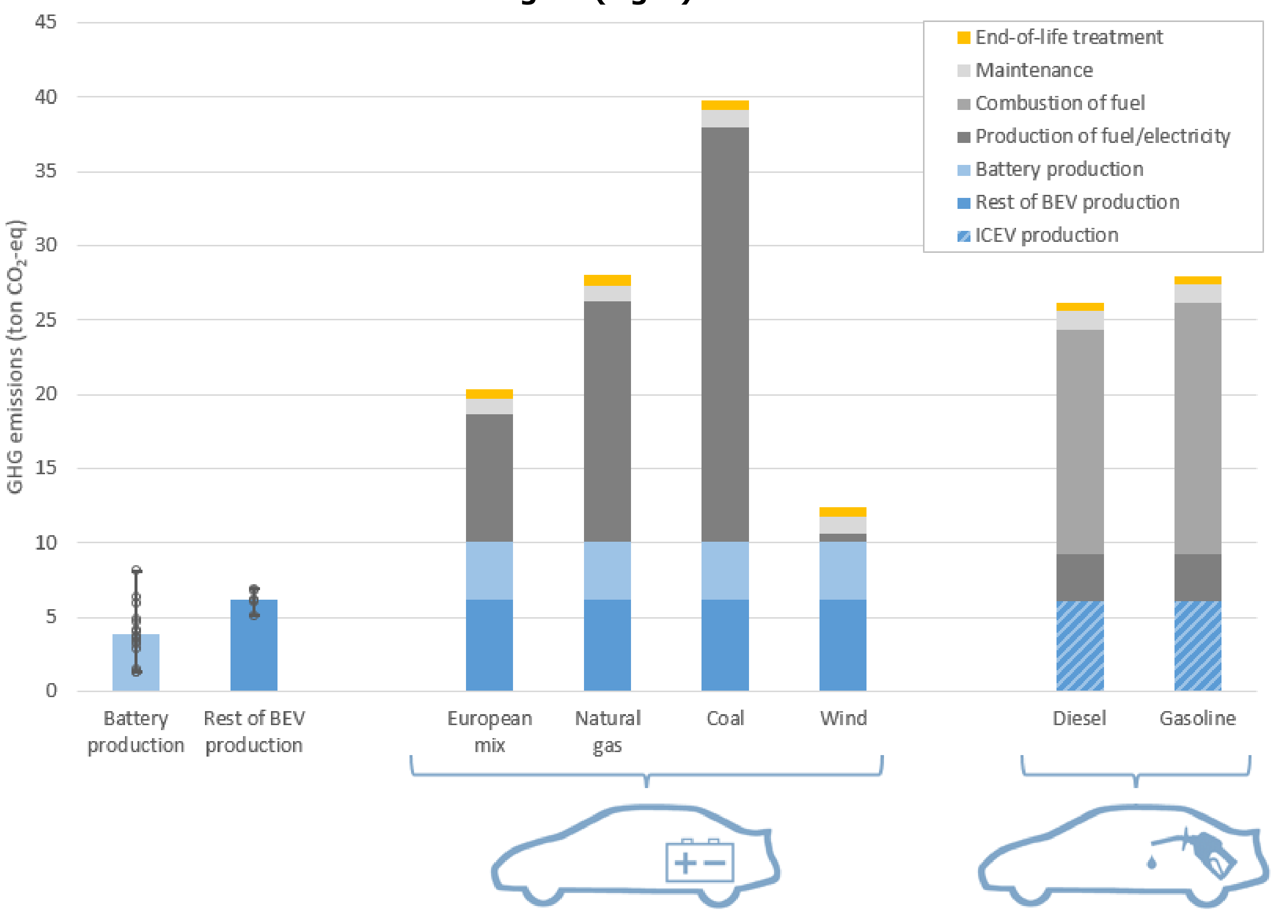
In order for BEVs to provide a climate change mitigation alternative to ICEVs, they must have lower lifecycle greenhouse gas emissions. The **BEV production phase is more carbon-intensive than that of ICEVs,** but BEVs can compensate for the higher production emissions through lower use phase emissions. **The carbon intensity of the electricity consumed in the operation phase greatly influences the advantages – or disadvantages – of adopting BEVs in preference to ICEVs**. Charging BEVs with electricity from renewable energy sources offers significantly lower lifecycle emissions compared to ICEVs. In contrast, BEVs charged using coal-based electricity yields higher lifecycle emissions than ICEVs. Thus the potential climate benefits of BEVs compared to ICEVs cannot be harnessed everywhere and under all conditions. Lifecycle assessment studies consistently **report moderate climate benefits for BEVs powered by the average European electricity mix** compared to ICEVs of a similar size.

Regardless of powertrain configuration, smaller vehicles tend to be more energy efficient during operation and generally have lower GHG emissions than larger ones. The trend towards increasing the size and range of BEVs is unfavourable from both a climate mitigation and resource use perspective. Thus**, given the current state of the technology, striking the right balance between battery and vehicle size and charging infrastructure is important** in maximising the climate change mitigation potential of BEVs.

The climate benefit of BEVs compared to ICEVs is expected to increase in the near future due to changes in the European power sector and developments within cell manufacturing. The **expected decarbonisation of the European power sector should lead to an increase in the environmental benefits of BEV use over time**, particularly from a climate change perspective. Decarbonisation of the electricity mix in cell-manufacturing countries would, furthermore, be beneficial in terms of the battery production process. The most significant change in cell manufacture in the near future is the large production volume of cells expected from Tesla Gigafactory 1, which claims to be self-sufficient in renewable energy. Consequently, GHG emissions from both production and use are expected to decrease in the coming years.

Achieving the goal of more sustainable transport solutions requires good environmental understanding of our technologies. **Battery producers and recyclers have to be more transparent about the environmental implications of their product lines**. An open channel of communication between industry, government and researchers is essential if BEVs are to succeed as a climate change mitigation initiative within the transport sector.

Figure E1: Lifecycle GHG emissions of mid-sized 24 kWh battery electric (left) and internal combustion engine (right) vehicles.



Note: The vehicle's operational lifetime is assumed to be 150 000 km.

1. BATTERY ELECTRIC VEHICLES WITH LITHIUM-ION TRACTION BATTERIES

KEY FINDINGS

- In the EU, **transport represents about 25 % of total GHG emissions** and is a major cause of air pollution in cities.
- **Passenger vehicles contribute the most** to total emissions from road transport.
- In spite of greater vehicle efficiency, **emissions are still rising due to** increasing travel activity by passenger vehicles.
- **Battery electric vehicles are more efficient** than conventional internal combustion engine vehicles and produce zero tailpipe emissions, which makes them a strong candidate for mitigating transport-related GHG emissions.

In spite of increased awareness of climate change and governments' goals to reduce anthropogenic greenhouse gas (GHG) emissions, the strong trend of increasing fossil GHG emissions (Gabriel et al., 2014) remains. In the EU, transport represents roughly a quarter of GHG emissions and is the main cause of air pollution in cities. Road transport is by far the biggest contributor to transport-related GHG emissions in the EU, accounting for more than 70 % thereof. Currently, passenger vehicles contribute the most to total emissions from road transport, and this trend is projected to continue.

Thus far, the emissions reductions arising from improvements in vehicle efficiency have not been able to outstrip the additional emissions from increasing travel activity. Reducing GHG emissions from passenger vehicles is expected to be particularly challenging, yet new technologies have the potential to make substantial contributions to climate change mitigation in the transport sector (Sims et al., 2014). Reducing the energy and fuel carbon intensities of light-duty vehicles represents the greatest opportunity for achieving a large reduction in GHG emissions (Sims et al., 2014). The energy efficiency of conventional vehicles is inherently limited by the thermodynamics of the internal combustion engine to 20–35 % (Sims et al., 2014). By contrast, fully battery-powered electric vehicles (BEVs) operate with a powertrain efficiency of around 80-90 % (Herrmann and Rothfuss, 2015; Sims et al., 2014). Furthermore, BEVs may offer benefits in terms of local air quality as they have no tailpipe emissions. These favourable characteristics make BEVs a strong candidate for mitigating transport-related GHG and air pollutant emissions.

Although BEVs have been considered a promising technology on several occasions over the last century, they have failed to achieve the same commercial success as internal combustion engine vehicles (ICEVs) (Steinhilber et al., 2013). Limited technological progress, particularly in relation to batteries, has been an important reason for the past failures to achieve large-scale commercialisation. Until the 1990s, BEVs were mainly powered by lead-acid batteries, resulting in very limited lifetimes and driving ranges (Dijk et al., 2013). In the late 1980s and early 1990s, the increase in portable consumer goods resulted in a demand for new battery technologies with higher energy density. By the late 1990s, one of these new technologies, the nickel-metal hydride battery, was used to power BEVs. Even though nickel-metal hydride battery packs offered improved energy density (Omar et al., 2012), the driving range was still prohibitively short and the battery cost too high to gain consumer acceptance (Dijk et al., 2013). It was only when lithium-ion batteries (LIBs) were introduced in the new generations of BEVs (first in the Tesla Roadster in 2008 and the Nissan Leaf in 2010), that consumer interest in and acceptance of BEVs increased.

This report is structured as follows: Section 2 describes the current battery production process, while section 3 addresses mineral use, supply risk and recycling issues related to LIBs. Section 4 discusses the GHG emissions associated with BEVs and important factors that contribute to the overall emissions. This section also briefly considers other environmental impacts. Finally, section 5 summarises and concludes the report.

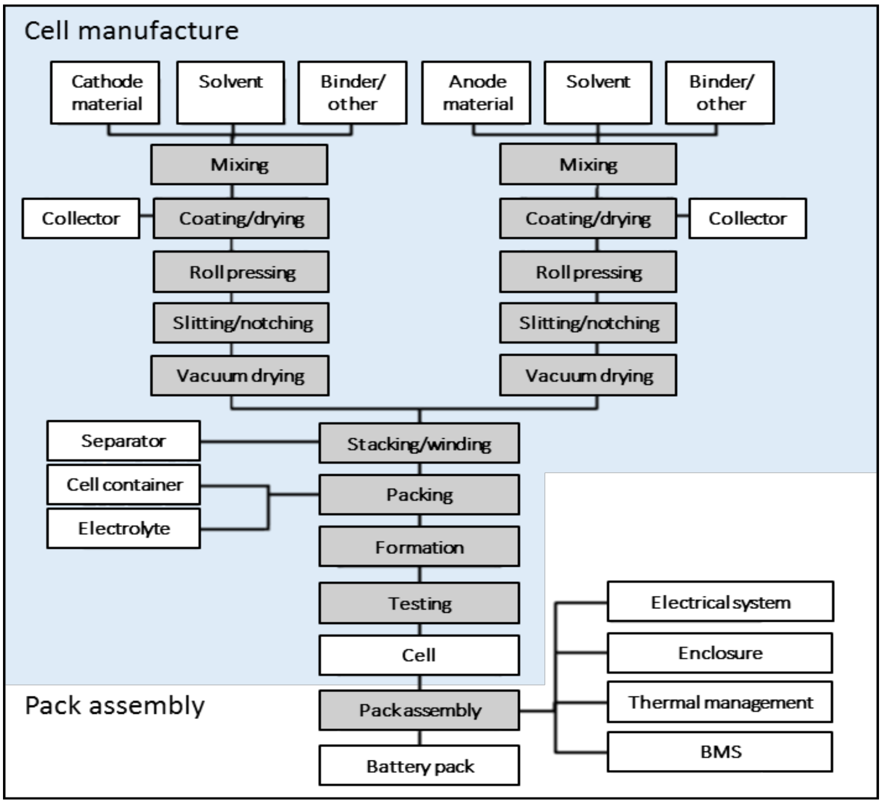
2. BATTERY PRODUCTION

KEY FINDINGS

- Battery production can be subdivided into **cell manufacture** and **battery pack assembly**.
- **Cell manufacture** is a **complex and protracted process** with stringent requirements in relation to ambient indoor conditions and cleanliness in building zones as well as a high demand for energy.
- In comparison to cell manufacture, **pack assembly** is a much **less complex and energy-intensive process**.

This section describes the battery production processes together with their material and energy use. The processes in battery production, including their material and energy use, must be transparent for researchers in order to identify concretely and to understand the related burdens on the environment. The main component of traction batteries is the battery cells, which make up about 55-60 % of the total weight in a battery pack used in mid-sized BEVs (e.g., Nissan Leaf, VW e-Golf, electric Ford Focus) (Ellingsen et al., 2014; Kim et al., 2016). The remaining battery components are: the module and pack enclosure (32-38 % of the total battery weight), the thermal management system (3 %), the battery management system (BMS; 3 %) and the electrical system (1 %) (Ellingsen et al., 2014; Kim et al., 2016). The processes associated with battery production are shown in Figure 1 and described below. Battery production can be subdivided into cell manufacture and pack assembly processes. In comparison to pack assembly, cell manufacture is a much more complex and protracted process.

Figure 1: Production flow diagram for a lithium-ion traction battery.



Note: The white and grey boxes represent part/material flow and the manufacturing process, respectively. The blue background indicates cell manufacture flows and processes, white is pack assembly. BMS = battery management system.

Source: Figure adapted with permission from (Kim et al., 2016). Copyright 2016 American Chemical Society.

2.1. Cell materials and manufacture

Cell manufacture is a complex and protracted process with stringent requirements in relation to ambient indoor conditions and cleanliness in building zones since temperature, humidity, and cleanliness have a significant impact on the quality, safety, performance, and lifetime of the cells (Schönemann, 2017).

The first step in cell manufacture is the production of the cathode and anode. These two components are produced in separate cleanrooms to avoid contamination (Väyrynen and Salminen, 2012). The cathode (positive electrode) consists of an aluminium current collector coated with a cathode material. The cathode material consists of the active material, a polymer binder (usually polyvinylidene diflouroide), and a conductive additive (carbon black). Although there are several active cathode materials available for use in LIBs, only a few of them are currently used in BEVs. The currently used cathode materials are lithium nickel-cobalt-manganese oxide (NCM) and lithium nickel-cobalt-aluminium oxide (NCA). In addition, these cathode materials are also blended with lithium manganese oxide (LMO), forming LMO-NCM and LMO-NCA. In some plug-in hybrid electric vehicles (PHEVs), lithium iron phosphate (LFP) is also used, but it has an inadequate energy density for BEVs.

The anode (negative electrode) consists of a copper current collector coated with the anode material. For traction batteries, the anode material consists of graphite, which is the active anode material, and binders (e.g. carbon methyl cellulose, polyacrylic acid, and styrene butadiene rubber). For both the cathode and the anode, the active material, binders, and additives are mixed with solvents to obtain a slurry that can be coated on to the current collectors. In Japan and South Korea, anodes are almost exclusively produced using a water-based solvent (Wood et al., 2015). Use of water-based solvents with the cathode materials, however, means that it remains difficult to thoroughly disperse these materials in water (Wood et al., 2015). Thus cathodes are mainly produced using the N-methylpyrrolidone (NMP) solvent (Li et al., 2011; Wood et al., 2015). The resulting active-material slurries are fed into coating machines to be spread on the metallic current collector foils (Li et al., 2011; Väyrynen and Salminen, 2012). Many coating techniques exist, which means that there may be differences in coating practices among battery cell manufacturers (Li et al., 2011; Schönemann, 2017). After the coating process, the electrodes are dried at high temperatures to evaporate the solvents (Schönemann, 2017; Väyrynen and Salminen, 2012). The coated and dried electrodes are roll-pressed to ensure homogenous thickness and particle size (Daniel, 2008; Schönemann, 2017). The roll-pressed electrodes are then cut to size and notched to prepare them for tabbing and further assembly (Daniel, 2008; Kim et al., 2016; Väyrynen and Salminen, 2012). After cutting, the finished electrodes are vacuum-dried to remove any residual moisture (Kaiser et al., 2014; Schönemann, 2017; Väyrynen and Salminen, 2012).

In a dry room, the separator, which is made of a porous plastic composite of polyethylene and polypropolyene, is placed between the anode and cathode electrodes and the three layers are stacked or wound. Next, conductive tabs are attached to the current collectors by ultrasonic spot welding or laser welding (Schönemann, 2017). After this, the tabbed assemblies are packed in cell containers. Battery cells used in commercial BEVs are available in three different cell containers: pouch, cylindrical, and prismatic (Anderman, 2016a; Birke, 2014; Omar et al., 2012; Warner, 2014). Pouch cells are the preferred cell format for many major automobile manufacturers as they can be assembled in different sizes and because the cell format itself makes for a cell with high gravimetric energy density (kWh/kg) (Warner, 2014). The pouch material is an aluminium-polymer composite that makes a soft cell container. Cylindrical and prismatic cell containers are hard cases. Cylindrical cell containers are made of steel or aluminium (Birke, 2014), while prismatic cell containers are made of polymers or aluminium (Warner, 2014). In the next step, the dry cells are filled with a liquid electrolyte (Schönemann,

2017) that consists of lithium salts ($LiPF_6$ is preferred) dissolved in organic carbonate solvents (ethylene carbonate, dimethyl carbonate, and ethyl methyl carbonate are the most commonly used) (Li et al., 2011). This process occurs under a vacuum to ensure complete wetting of the electrodes and separator pores (Schönemann, 2017; Wood et al., 2015). After the wetting, the cells are sealed.

The final step of cell manufacture is conditioning. The first stage of conditioning is formation cycling to prepare the cell for commercial use. Before module assembly, the cell performance is checked and tested (Schönemann, 2017).

Before pack assembly, cells are combined to form battery modules. Pouch cells require an additional enclosure that provides structural stability and support. The enclosures may be made of plastic and composite materials (Ellingsen et al., 2014; Kim et al., 2016) or metals (Anderman, 2016b), and form a cassette around the cells. These cassettes are part of the module packaging. Cylindrical and prismatic cells require fixtures or frames to make modules (Schönemann, 2017). Commercial traction batteries therefore differ in cell and module packaging.

2.2. Battery pack components and assembly

In pack assembly, the modules are combined with the BMS, electrical system, and the thermal management system within a battery tray, which is made of steel or aluminium. The battery packs are either assembled by a battery manufacturer and then delivered to the automobile manufacturer (Anderman, 2016c; Kim et al., 2016) or are assembled by the automobile manufacturers themselves (Anderman, 2016c; General Motors, n.d.). In the pack manufacturing process, cell tabs are connected to busbars by welding, modules are tested, and an initial charge creates a uniform state-of-charge across all cells (Schönemann, 2017).

All LIBs require a BMS as the cells fail if they are overcharged, completely discharged, or operate outside of their safe temperature window (Hauser and Kuhn, 2015a; Omar et al., 2012; Vezzini, 2014a). The BMS and electrical system includes battery module boards, a high-voltage system including a battery interface system, and a low-voltage system, as well as cables, fuses, connectors, and gaskets (Ellingsen et al., 2014). The combined weight of the BMS and electrical system makes up about 3 to 4 % of the total battery weight in studies based on primary industry data (Ellingsen et al., 2014; Kim et al., 2016).

Most traction batteries also require a thermal management system to equalise temperature gradients between the cells, to cool the batteries, and to prevent thermal runaways and the destruction of the battery by overheating (Hauser and Kuhn, 2015b; Warner, 2014). There are two main types of thermal management system: liquid-based (glycol cooling fluid) and air-based (fans and heat sinks) (Hauser and Kuhn, 2015a; Warner, 2014). The thermal management system is typically made of aluminium, which has the advantage of being a good thermal conductor and lightweight metal, but steel may also be used. Similarly to the BMS and electrical system, the thermal management system makes up about 3 to 4 % of the total battery weight in studies based on primary data (Ellingsen et al., 2014; Kim et al., 2016).

2.3. Energy demands in battery production

Energy demand in battery production has been much discussed in the lifecycle assessment (LCA) literature (Dunn et al., 2015, 2012; Ellingsen et al., 2015, 2014; Kim et al., 2016; Majeau-Bettez et al., 2011). Assumptions about energy use in the battery production process greatly influence the estimates of the overall GHG emissions of this process. While most studies agree that the energy demand related to pack assembly is relatively low (Dunn et al., 2012; Ellingsen et al., 2014; Kim et al., 2016; Li et al., 2014; Notter et al., 2010; Yuan et al., 2017), the studies have two opposing views on cell manufacture.

Studies that rely on secondary energy data[12] from industry reports find high energy use for cell manufacture (326–1060 MJ/kWh cell) (Bauer, 2010; Majeau-Bettez et al., 2011; Zackrisson et al., 2010). Studies that make their own estimates, however, assume either no (United States Environmental Protection Agency, 2013) or very low energy use (2.3–10 MJ/kWh cell) (Dunn et al., 2012; Notter et al., 2010; United States Environmental Protection Agency, 2013). Recent LCA studies based on primary energy data[13] report high energy use in the range of 530–1670 MJ/kWh cell (Ellingsen et al., 2014; Kim et al., 2016; Yuan et al., 2017), which is largely in agreement with the secondary data from industry reports. The higher end of the range is representative of an industrial pilot scale manufacturing facility (Yuan et al., 2017), while the lower end is representative of a state-of-the-art, world-leading supplier of LIB cells (Kim et al., 2016). The energy demand for cell manufacture can be met by electricity alone (Ellingsen et al., 2014), or by a combination of electricity and heat (Kim et al., 2016). Thus far, no primary data sources have identified the shares of electricity and heat used in the cell manufacturing process. However, studies based on industry reports have assumed 52 % and 57 % electricity (Bauer, 2010; Majeau-Bettez et al., 2011; Zackrisson et al., 2010).

While more primary energy data is becoming available, detailed information regarding the energy consumption of the process is still limited. A breakdown of energy use from energy consumption of the process in cell manufacturing provided by an industrial pilot scale manufacturing facility has identified electrode drying as the most time- and energy-consuming process (Yuan et al., 2017). Because the NMP solvent has a higher boiling point (204.3°C) than the water-based solvent (100°C), the cathode requires more energy during the process to evaporate the solvent than the anode (Wood et al., 2015). The operation of dry rooms has also been pointed out as being particularly energy-intensive (Ellingsen et al., 2014; Yuan et al., 2017). As electrode drying and the operation of dry rooms have been identified as particularly energy-intensive processes, it is likely that of all the battery production processes, these two contribute the most to overall battery production GHG emissions.

Several factors may influence a plant's energy requirements in manufacturing cells (Ellingsen, 2017). One of these is production location, which has implications for energy demand and energy sources. Currently, lithium-ion cells are primarily produced in South Korea, China, and Japan (Bernhart, 2014; pv magazine, 2015; Vezzini, 2014b). Since part of the cell manufacturing energy inputs can only be met by electricity, the environmental impact of cell manufacture is partly dictated by the energy sources used to generate the electricity. These are countries that all use high shares of fossil fuels to generate electricity. Of the three countries, China has the most carbon-intensive electricity mix. Thus, in spite of equal energy use in cell manufacture, cells produced in China will have higher GHG emissions than cells produced in South Korea or Japan. Moreover, certain regions in these countries are affected to a great extent by the East Asian Monsoon, which is characterised by a warm, rainy monsoon

[12] 'Secondary energy data' refers to quantitative energy data that was originally collected by a third party for different purposes.

[13] 'Primary energy data' refers to quantitative energy data collected by the cell manufacturer specifically for this study.

season lasting from early May to September (Yihui and Chan, 2005). The humid monsoon summer might conceivably affect the energy use in dry room operations. As a result, there may be differences in energy demand among the producing countries, depending on the season and region.

Moving cell manufacture to areas that are richer in renewables or installing renewable energy sources for use in manufacture has advantages in terms of energy use and GHG emissions. Once completed and operational, it is claimed that Tesla Gigafactory 1, which is located in the dry Nevada climate, will be self-sufficient, using energy from solar, wind, and geothermal sources in its cell manufacturing. It is therefore likely that Tesla Gigafactory 1 will use less and cleaner energy to produce cells than current cell operations.

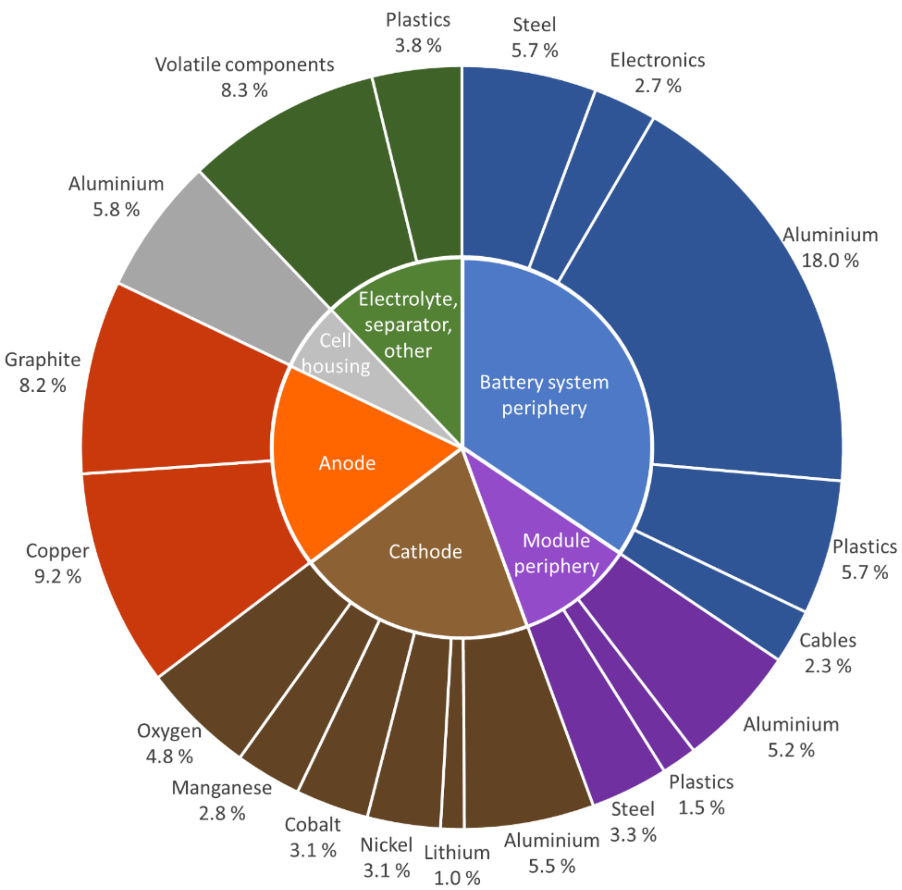
3. MINERAL USE, SUPPLY RISKS AND BATTERY RECYCLING

KEY FINDINGS

- LIBs are amalgams of materials and many of these materials are based on **minerals in finite supply**.
- Of the minerals used in lithium-ion traction batteries, it is likely that **lithium and cobalt** have the **highest overall supply risk**, while aluminium has the lowest risk. Manganese, iron, nickel, copper, and natural graphite have a medium supply risk.
- Current **recycling processes focus on the recovery of cobalt and nickel** because of the price of these metals. Copper and iron are typically also recovered. Only a few recycling routes recover aluminium, lithium, and manganese. Graphite is not recycled in current industrial recycling processes.

The growing use of BEVs powered by LIBs has resulted in an increasing demand for minerals. This section examines mineral use associated with LIBs, the end-of-life (EOL) treatment of batteries by recycling and supply risk evaluations for a number of minerals relevant to LIBs. LIBs are amalgams of materials and many of the materials are based on minerals with a finite supply. Minerals that are susceptible to supply risk for Li-ion traction batteries are: lithium (Li), cobalt (Co), graphite (C), manganese (Mn), nickel (Ni), iron (Fe), copper (Cu), and aluminium (Al). Figure 2 presents the material composition of a generic Li-ion traction battery.

Figure 210: Material composition of a generic Li-ion traction battery with an NCM cathode.



Note: The module periphery includes module enclosure, while the battery system periphery includes the pack enclosure, the battery management system, the electrical system, and the thermal management system.

Source: Reproduced from (Diekmann et al., 2017).

In LIBs, lithium, cobalt, manganese, and nickel are primarily used in the cathode material, while synthetic and natural graphite are used as the anode material. Copper and aluminium are used as current collectors in the cells, but also have uses in other battery components. Steel is not used in the cells themselves, but can be used in several battery components. The text below examines the individual minerals in further detail.

In spite of their name, LIBs contain only small amounts of *lithium*. Lithium is primarily used in the active cathode materials and, in smaller amounts, in the liquid electrolyte. In 2014, LIBs accounted for 35 % of global lithium consumption (Sun et al., 2017), with about a quarter of this share used in electric vehicles (BEVs and PHEVs) (Ziemann et al., 2012). Due to increasing demand for lithium, particularly for batteries, major lithium producers are planning to increase their extraction levels (U.S. Geological Survey, 2017a; Weil and Ziemann, 2014). The geographic distribution of lithium deposits is very uneven. The United States Geological Survey (USGS) estimates that the majority of the global lithium reserves, currently estimated at 14 million tonnes, are located in Chile (52 %), China (22 %), Argentina (14 %), and Australia (11 %). These countries are also the largest producers of lithium globally (U.S. Geological Survey, 2017a; Weil and Ziemann, 2014). Europe is a major consumer of lithium products, but has less than 3 % of world lithium resources (Grosjean et al., 2012). Through its spodumene mining operations, Australia is the largest single producer of lithium (41 %) (U.S. Geological Survey, 2017a; Weil and Ziemann, 2014). Through their salt brine operations in the Andes, Chile (34 %) and Argentina (16 %) are the second and third largest producers of lithium, respectively (Weil and Ziemann, 2014). Global brine resources are mainly (70 %) located in high altitude locations such as the Andes (Argentina, Bolivia, and Chile) and south-western China (Tibet) (Kesler et al., 2012; Vikström et al., 2013). Although lithium reserves are unlikely to be exhausted in the near future, shortages in supply may arise mainly because half of the annual production volume originates from brines. Lithium production from brines is a relatively slow process that cannot increase supply rapidly in response to sudden increases in demand. Furthermore, lithium extracted from brines is a co-product in potassium production; a decrease in demand for potassium might therefore affect the supply of lithium from brines. The market for secondary lithium is practically non-existent at the present time; less than 1 % of lithium is recycled globally (Graedel et al., 2011; Helbig et al., 2017). Lithium has high substitutability as other materials can replace it in non-battery applications (Gruber et al., 2011) and other battery chemistries can replace LIBs (Helbig et al., 2017). Nevertheless, the current BEV market is heavily dependent on LIBs as they offer an as yet unmatched combination of high power and energy densities (Reuter et al., 2013). A number of studies have explored the relationship between lithium availability and LIB demand from the electrification of vehicles, ranging from pessimistic studies suggesting that the demand cannot be met, to optimistic studies finding no significant constraints to ambitious fleet development projections (Speirs et al., 2014). Differences in study outcomes originate from differing assumptions about both future lithium demand (e.g. size of vehicle fleet, degree of electrification, battery size, quantity of lithium per unit of battery capacity, time horizon and other lithium applications) and supply (e.g. estimates of lithium reserves, future lithium production rates, and future lithium recycling rates) (Speirs et al., 2014).

Cobalt is used in the cathode active material. While LIB cathode materials without cobalt do exist, they have lower energy density and are not currently used in BEVs. Research efforts are being made in an attempt to reduce the share of cobalt in cathode material, mainly because of its high price and the political challenges associated with supply. Cobalt is mainly produced as a co-product of nickel or copper. The vast majority of identified terrestrial cobalt reserves are in the Democratic Republic of the Congo (49 %), with smaller shares in Australia (14 %), Philippines (4.1 %), Canada (3.8 %), Zambia (3.8 %), and Russia (3.6 %) (U.S. Geological Survey, 2017b). In 2016, the Democratic Republic of the Congo supplied more than half (54 %) of world cobalt mine production (U.S. Geological Survey, 2017b). As a result, the political risks that might potentially affect the supply of cobalt are particularly high (Helbig et al., 2017). As cobalt is a relatively expensive metal, it has high recycling rates (over 50 % globally in 2008) (Graedel et al., 2011), and it is always recovered in battery recycling processes (Hanisch et al., 2015a). Cobalt recovered from battery recycling is of high enough quality for reuse in batteries.

The carbonaceous active anode material used in LIBs can generally be divided into synthetic and natural *graphite* (Gulbinska, 2014; Yoshino, 2014). While large reserves of natural graphite are found in Turkey (36 %), Brazil (29 %), and China (22 %), the largest-scale mine production takes place in China (65 %) and India (14 %) (U.S. Geological Survey, 2017c). Synthetic graphite is produced from petroleum coke and coal tar pitch (Olivetti et al., 2017). Graphite has an overall 0 % recycling rate (Helbig et al., 2017), and is similarly not recycled from LIBs (Accurec, 2017; Umicore, 2016). Silicon-based anode materials in nanoform are likely to be used in LIBs in the near future and may partly or fully substitute graphite.

In LIBs, *manganese* is only used in NCM and LMO cathode materials. As a general rule, manganese is predominantly used in the production of steel, while manganese consumption for use in batteries is marginal (Ziemann et al., 2013). Major manganese reserves are located in South Africa, Australia, Brazil, and Ukraine. South Africa is the largest single producer with about 30 % of the total market share (U.S. Geological Survey, 2017d). Although manganese generally has high recycling rates in excess of 50 % (Graedel et al., 2011; Helbig et al., 2017), manganese is typically not recovered in LIB recycling schemes at present (Georgi-Maschler et al., 2012; Gratz et al., 2014; Hanisch et al., 2015b).

Like cobalt, *nickel* is used in NCM and NCA cathode active materials. Compared to cobalt, nickel is an inexpensive metal and increases the gravimetric energy density of LIBs. For these reasons, the share of nickel is expected to increase to replace cobalt in battery cells. Nickel is fairly widely distributed geographically with most continents having reserves. The largest reserves are found in Australia (24 %) and Canada (13 %), while European reserves are mainly found in Russia (10 %) (U.S. Geological Survey, 2017e). In general, nickel has a recycling rate of 58 % (Helbig et al., 2017) and is usually recovered in battery recycling as it is considered to be one of the profitable metals to be recovered in the LIB recycling process (Hanisch et al., 2015a).

Iron in the form of steel can be used in a variety of battery components, but, due to its weight, aluminium is often preferred for various uses in BEVs. In LIBs, steel is used in fixings (e.g. screws and bolts) in particular, but can also be used in the battery enclosure (e.g. cell housing, battery tray) and in the thermal management system. China produces more than half of the world's pig iron and raw steel, while Russia is the largest producer in Europe (about 4.5 % of global totals) (U.S. Geological Survey, 2017f). Global recycling rates for iron are close to 70 %. As steel products are typically found in battery peripherals rather than the cells proper, steel is often recovered through battery disassembly as opposed to the processing of cells (Georgi-Maschler et al., 2012; Gratz et al., 2014; Hanisch et al., 2015b).

In LIBs, *copper* is used primarily in the negative current collector (anode), but also in electronics and cables. As a result of its properties, such as its thermal and electrical conductivity and its resistance to corrosion, copper has become a major industrial metal, ranking third after iron and aluminium in terms of the quantities consumed (Glöser et al., 2013). Copper reserves are widely distributed geographically, with the largest reserves located in Chile (29 %), Australia (12 %), and Peru (11 %). The largest European reserves are to be found in Russia (4.2 %) (U.S. Geological Survey, 2017g). The largest-scale mine production takes place in Chile (28 %), Peru (11 %), China (9.0 %), and the US (7.2 %) (U.S. Geological Survey, 2017g). Copper has a global recycling rate of 68 % (Helbig et al., 2017) and is recycled in most LIB recycling schemes (Hanisch et al., 2015a).

Aluminium has many applications in LIBs, the total share of aluminium in a battery depending on material choices and design. Although much of the aluminium used in LIBs can be replaced by other materials such as steel, the cathode current collector is always made of aluminium. Globally, aluminium is the second most used metal after iron and steel. The main raw material for aluminium production is bauxite, which is primarily mined in China and Australia (Liu and Müller, 2013). China is the largest producer of aluminium with more than half of global production (U.S. Geological Survey, 2017h). Aluminium has a relatively high recycling rate of 60 % (Helbig et al., 2017). While battery disassembly can lead to aluminium being recycled from packaging or thermal management systems, the recycling processes for the cells rarely do so (Georgi-Maschler et al., 2012; Gratz et al., 2014; Hanisch et al., 2015b).

3.1. Supply risks associated with Li-ion batteries

Shortages of the above minerals can arise for various reasons – wars, embargoes, cartels and other market manipulations, natural disasters, accidents, cyclical booms in global demand, inadequate investment in new mines and processing facilities and resource depletion (Yaksic and Tilton, 2009). Shortages due to resource depletion differ from the other causes, as they are likely to arise slowly and persistently and to be permanent or at least of very long duration. In the text below, we provide some relevant information on supply risk affecting the various minerals used in traction LIBs.

Various assessments have evaluated the supply risk of minerals. Here, we report on the contents of a recent study that uses a semi-quantitative assessment scheme to evaluate the relative supply risks associated with minerals used specifically in LIBs (Helbig et al., 2017). The main reasons for relying on this evaluation is that it is recent and focuses specifically on LIB minerals. The scheme evaluates overall relative supply risk based on four criteria: (i) risk of supply reduction, (ii) risk of demand increase, (iii) concentration risk and (iv) political risk. Each risk criterion consists of two or three indicators (see Figure 3), the sources of which include scientific literature, reports and various databases (Helbig et al., 2017). The weighting factors indicate the relative importance of each indicator to the overall supply risk score. They are determined by external experts within the framework of an Analytical Hierarchy Process (Helbig et al., 2017).

Figure 3 presents the relative evaluation for the various indicators as well as the overall relative supply risk evaluations (bottom row). Red denotes high risk, yellow intermediate risk, and green low risk.

Figure 3: Relative supply risk indicators and overall supply risk for each element.

Criterion	Indicator	Weighting	Li	Co	C	Mn	Ni	Fe	Cu	Al
Risk of supply restriction	Static reach reserves	8.9 %								
	Static reach resources	5.2 %								
	End-of-life recycling rate	9.2 %								
Risk of demand increase	By-product dependence	3.9 %								
	Future technology demand	14.1 %								
	Substitutability	14.2 %								
Concentration risk	Country concentration	9.7 %								
	Company concentration	13.0 %								
Political risk	Political stability (Worldwide governance indicator)	11.2 %								
	Policy perception index	5.2 %								
	Regulation risk (Human development index)	5.3 %								
Relative overall supply risk										

Note: Comparisons are made across rows, i.e. for each indicator.

Data source: (Helbig et al., 2017).

The study reports that lithium and cobalt have the highest overall supply risk, while aluminium has the lowest risk. Manganese, iron, nickel, copper, and natural graphite have a medium supply risk. Note that the evaluation only pertains to natural graphite, not synthetic graphite, as the carbonaceous anode material. The study also compared its results with those of previous supply risk studies and found that they were similar (Helbig et al., 2017). The main difference was that previous studies had reported a higher supply risk for natural graphite and a lower risk for lithium. This difference may be accounted for by a variety of factors, such as the scope of the study, indicator weighting, and evaluation year. As silicon is a widely abundant material, replacing graphite with nanostructured silicon anodes may reduce the supply risk associated with batteries. However, the production of nanostructured silicon involves high energy requirements (Ellingsen et al., 2016a), which may have implications for GHG and other emissions.

3.2. End-of-life treatment of lithium-ion batteries

Recycling of materials can be advantageous from both a resource conservation and an environmental perspective. Secondary (recycled) materials can replace primary (virgin) materials and thus preserve natural resources. Furthermore, recycled materials generally have a lower environmental impact than virgin materials (Bigum et al., 2012). In principle, metals are infinitely recyclable, but in practice recycling is often inefficient or essentially non-existent because of the limits imposed by social behaviour, product design, recycling technologies, and the thermodynamics of separation (Reck and Graedel, 2012).

In the EU, environmental regulations such End-of-Vehicle directives require automobile manufacturers to take extended responsibility for their vehicles and components after use (Ramoni and Zhang, 2013; Sakai et al., 2014). Under this extended responsibility, automobile manufacturers are financially or physically responsible for either taking back their products with the end goal of reusing, recycling or remanufacturing, or alternatively are obliged to delegate this responsibility to a third party (Ramoni and Zhang, 2013). The recycling flow of vehicles is almost identical in many countries (Sakai et al., 2014). The end-of-vehicle life treatment process starts with dismantling. At this juncture, components containing hazardous substances, such as batteries and refrigerant gases are collected, followed by recyclables and valuable materials for secondary use, including engines, tyres, and bumpers. In the EU, 55-70 % of the original vehicle weight remains after the dismantling process for conventional ICEVs. The vehicle shells left after the dismantling process are put into shredders. The shredded materials are separated using an air classifier and the light shredding residues are taken out. Subsequently, magnetic separators or non-ferrous metal collectors separate iron from non-ferrous materials. The remnants of these processes are heavy shredding residues. The combined amounts of the light and heavy shredding residues are reported to be 12-32 % of the original vehicle weight in the EU, where these shredding residues are landfilled (Sakai et al., 2014).

Battery recycling is driven by legislation and metal value. Directive 2006/66/EC[14] stipulates recycling rates to minimise the negative impact of batteries and waste batteries on the environment, thus aiming to protect, preserve and improve the quality of the environment. Specifically, 45 % of spent LIBs must be collected, and, as stated by the Directive, at least 50 % of the average weight of LIBs should be recycled, excluding energy recovery. Because the Directive stipulates that the recycling processes of batteries must achieve a minimum recycling efficiency by average weight, it favours the recovery of base metals that are widely used in commercial and industrial applications and are relatively abundant in nature (e.g. iron, copper). As a result, the Directive does not promote the recycling of scarce or specialty metals, or metals that place a particularly high burden on the environment. Battery recyclers decide themselves what materials to recover and recycle. The main economic driver of the LIB recycling is the metal value of batteries. Because the metal value is driven by the price of cobalt and nickel, current recycling processes focus on the recovery of these metals (Gratz et al., 2014; Hanisch et al., 2015b; Reuter et al., 2013). Other transition metals, such as copper and iron, are typically also recovered in the course of the current industrial LIB recycling processes. Only a few recycling routes recover aluminium, lithium, and manganese (Georgi-Maschler et al., 2012; Gratz et al., 2014; Hanisch et al., 2015b). Graphite is normally not recycled in the framework of the current industrial recycling processes (Accurec, 2017; Hanisch et al., 2015a; Umicore, 2016).

While there are several industrial LIB recyclers, these companies severely restrict the information available about their recycling processes. The various recycling pathways recover different materials, require different material and energy inputs and achieve different yields. In Europe, there are four large-scale industrial LIB recycling companies: Accurec (Germany), Umicore (Belgium), Recupyl (France), and BatRec (Switzerland). Current industrial battery recycling processes typically involve a combination of different unit operations: mechanical separation, pyrometallurgical treatment and hydrometallurgical treatment (Diekmann et al., 2017). The text below summarises the limited information that is available for these recycling processes.

Accurec combines mechanical pre-treatment with hydro- and pyrometallurgical process steps. In battery disassembly, cells are separated from other battery components, such as the BMS, the thermal management system and the pack and module enclosures. In a vacuum thermal pre-treatment process, the electrolyte and hydrocarbons evaporate and the cells are deactivated. The deactivated cells then go through various mechanical treatment processes, resulting in four fractions. The material generated contains an iron-nickel fraction, an aluminium fraction, an electrode foil fraction and a fine material fraction, which contains the electrode material (Georgi-Maschler et al., 2012). The fine fraction is put into a reducing smelting furnace that produces a cobalt alloy, lithium-containing slag and flue dust and an off-gas from the

[14] Directive 2006/66/EC of the European Parliament and of the Council of 6 September 2006 on batteries and accumulators and waste batteries and accumulators and repealing Directive 91/157/EEC, OJ L 266, 26.9.2006, p. 1.

combustion of electrolytes (Accurec, 2017; Hanisch et al., 2015a). Cobalt, manganese, nickel, and iron are recovered from the alloy at present (Vezzini, 2014b). The slag and flue dust can be treated in additional hydrometallurgical steps to recover lithium, but this is not currently done due to the lack of economic profit. Today, the slag is typically reused in construction, while the small amount of flue dust (about 0.2 tonnes of flue dust per 1 000 tonnes of waste input) is landfilled (Accurec, 2017). After the smelting furnace, the off-gas is cleaned in a treatment process, which generates some waste water (Accurec, 2017).

Umicore applies pyrometallurgical treatment with a subsequent hydrometallurgical process. Its pyrometallurgical treatment produces slag, a liquid metal alloy, flue dust, and gas emissions. The slag fraction, which contains aluminium, lithium, and manganese, can be used in the construction industry or further processed for additional metal recovery. Lithium recovery from the slag, in cooperation with an external partner, started in 2017 (Umicore, 2017). The liquid metal alloy is further refined in hydrometallurgical processes to recover copper, nickel, and cobalt by solvent extraction. Umicore claims close-to-zero waste generation as only the flue dust is landfilled, and all gases produced in their process are cleaned, significantly reducing harmful emissions into the air (Umicore, n.d.).

Recupyl employs mechanical crushing followed by hydrometallurgical treatment. The LIBs are crushed in an enclosure with a defined and controlled atmosphere and pressure (Hanisch et al., 2015a). The crushed materials are then put on a vibrating screen and divided into four fractions depending on size, density, and behaviour in a magnetic field. Only one fraction, a fine fraction that is rich in in metal oxides and carbon, is further processed. This fraction is sieved to reduce the copper content. The remaining fine powder is further treated in hydrometallurgical process steps to derive solutions of cobalt and lithium salts. The cobalt solution is electrolysed to obtain cobalt, while the lithium salts are oxidised to obtain lithium carbonate or lithium phosphate (Vezzini, 2014b).

BatRec runs a mainly mechanical processing plant. The first step is crushing of the batteries in an inert CO_2 atmosphere. The next processing step is mechanical separation that leads to a metal fraction containing non-ferrous metals, a metal fraction containing nickel, one fine fraction containing cobalt and lithium and a plastic fraction. The first two metal fractions can be sold on to other metal recyclers, while the fine fraction is sold on to cobalt and nickel refining companies. The plastic fraction can be partially used for energy recovery in a pyrolysis processes (Hanisch et al., 2015a).

4. LIFECYCLE GREENHOUSE GAS EMISSIONS OF BATTERY ELECTRIC VEHICLES

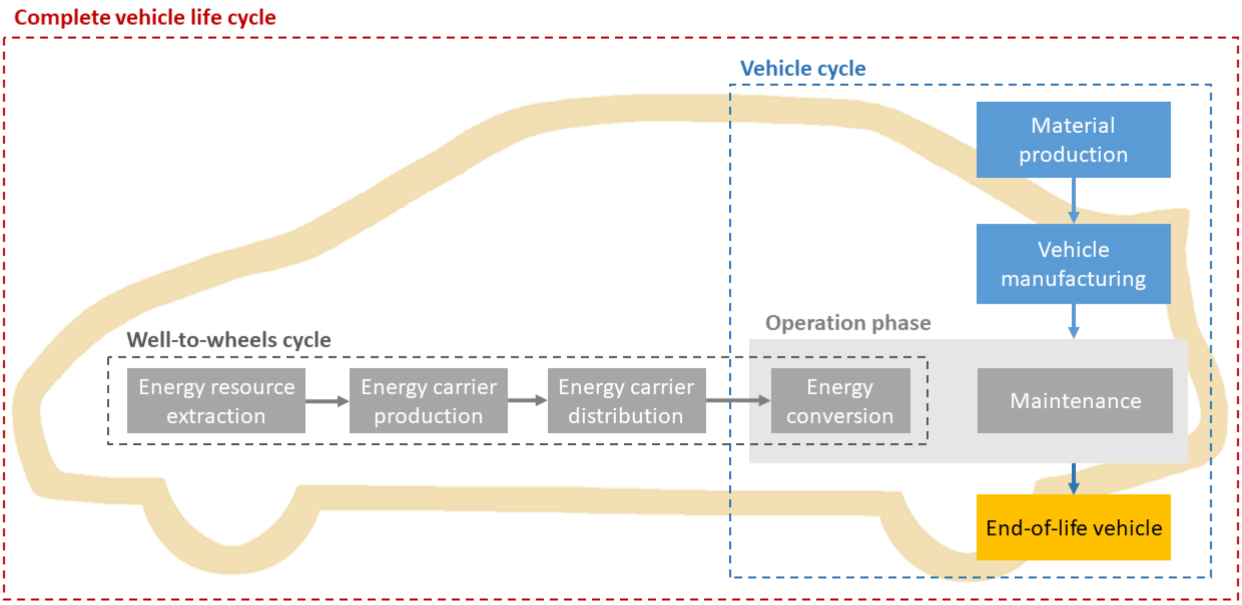
KEY FINDINGS

- BEVs have **higher production emissions** than ICEVs. The difference in production emissions between the two powertrain technologies can be attributed to the traction battery. Decarbonising the energy sources used in battery production is important in relation to total lifecycle BEV GHG emissions.
- The BEV use phase GHG emissions are a product of the operational energy use and the carbon intensity of the electricity sources used to charge the battery. The **electricity sources used in battery production and charging** are a major determinant of whether or not BEVs provide a climate benefit compared with ICEVs.
- BEVs powered by average European electricity are found to **moderately reduce GHG emissions** compared to both the diesel and petrol ICEVs assuming a 150 000 km vehicle lifetime.
- The expected decarbonisation of the European power sector, coupled with improvements in electrochemical and powertrain performances, should lead to an **increase in the environmental benefits of BEV use over** time, particularly from a climate change perspective.
- Larger battery packs can increase driving range and consumer acceptance, but their production also requires more resources and results in higher emissions. Given the current state of the technology, **striking the right balance between battery and vehicle size and charging infrastructure is an important element** in maximising the climate change mitigation potential of BEVs.

As a measure to reduce the combustion of fossil fuels and thereby mitigate climate change, many governments have introduced policies to promote market uptake of BEVs. BEVs have no tailpipe emissions, but the indirect GHG emissions resulting from electricity generation can be significant depending on energy sources (Bauer et al., 2015; Ellingsen et al., 2016b; Faria et al., 2012; Hawkins et al., 2012; Notter et al., 2015). BEVs therefore have the potential to reduce GHG emissions, but a lifecycle perspective is required to accurately assess the sustainability and impacts of BEVs.

Lifecycle assessment (LCA) is a formalised analytical method for quantifying the environmental impacts associated with the production and consumption of products and services. LCA considers the material and energy requirements as delivering a function and strives to take stock of all environmental exchanges that arise as a result of these requirements. These environmental exchanges might consist of emissions into water bodies, air or land, or the removal of resources from the environment, such as crude oil, biomass or metal ores. The LCA method considers all stages, from raw material extraction, through processing and manufacturing, to product use, recycling, and final disposal. LCA thus provides a useful 'whole system' perspective over entire supply chains where the total emissions and resource use associated with the delivery of a 'functional unit' (e.g. transporting one person over one kilometre). Examples of such emissions include GHGs such as carbon dioxide and methane. These emissions are inventoried and linked to potential environmental impacts, such as climate change, or human toxicity. Similarly, resource use is inventoried, and can be used to assess resource depletion effects. This report focuses on the climate change impacts associated with BEVs, while other environmental impact categories will only be touched upon briefly.

Figure 4: The complete lifecycle of a vehicle.



Note: Background colours for each box correspond to those of the bars in Figure 5.

Source: Redrawn from (Nordelöf et al., 2014).

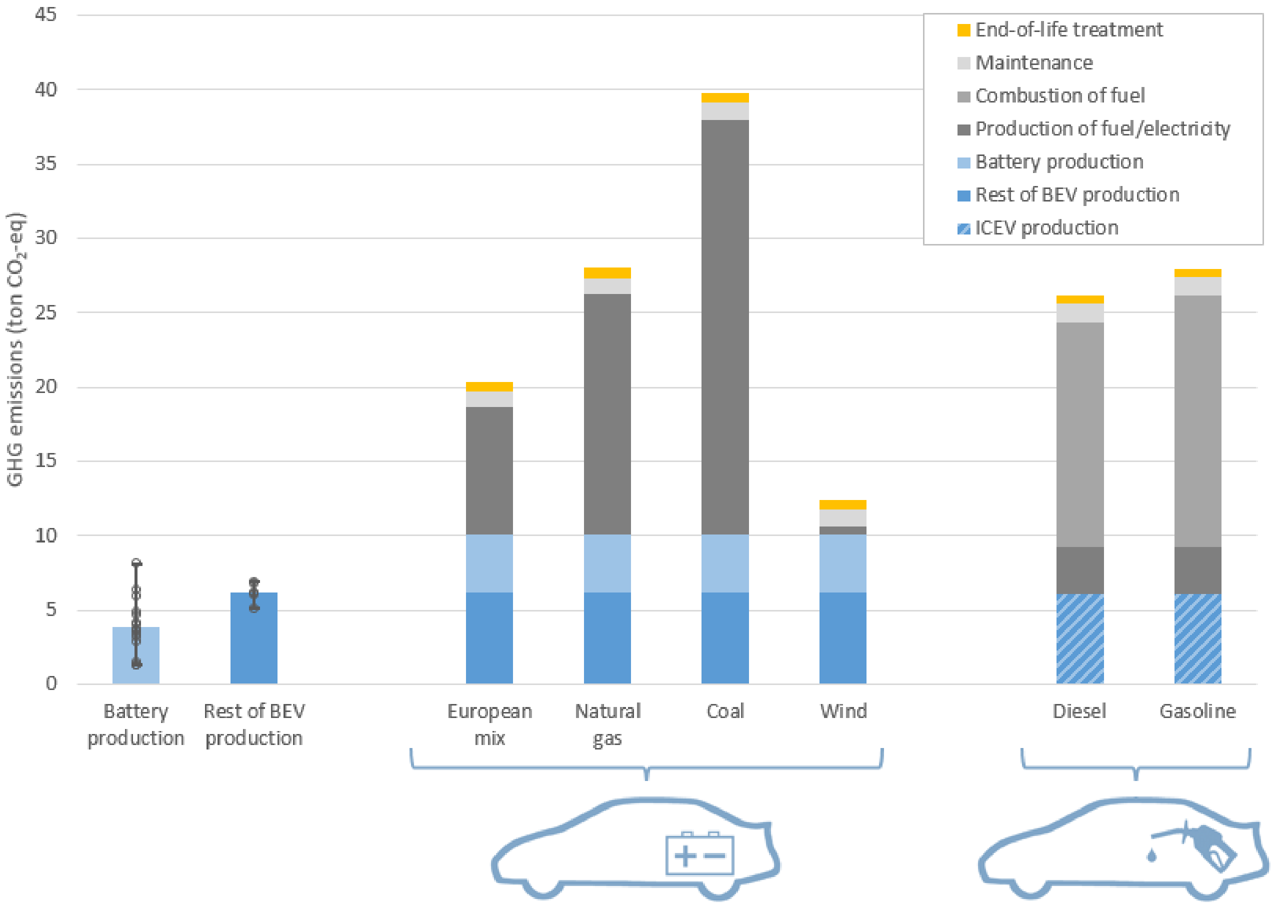
As implied by the term 'lifecycle assessment', it is important to consider the complete vehicle lifecycle, including the vehicle cycle and the well-to-wheel (WTW) cycle (see Figure 4). The first stage of the vehicle cycle is production, which includes resource extraction and the manufacturing and final assembly of components. The second stage is vehicle operation, which also includes the conversion of fuel or electricity to movement, and maintenance. The third and final stage is end-of-life (EOL) treatment, and involves dismantling and the recovery of certain parts, as well as shredding, recycling, and the disposal of residues. The well-to-wheels cycle includes the entire value chain of the fuel or energy source, including extraction of resources, energy carrier production and distribution and the conversion of the energy source to kinetic energy to move the vehicle. This cycle can be the value chain for petrol, diesel, biofuels, electricity or hydrogen, depending on the vehicle powertrain technology being considered.

4.1. Lifecycle climate change impact

Figure 5 presents the lifecycle GHG emissions of a mid-sized BEV powered by a 24 kWh NCM battery[15] and mid-sized ICEVs over a total mileage (lifetime) of 150 000 km. GHG emissions are measured in tonnes of carbon dioxide equivalents (CO_2-eq). Darker blue denotes the production of vehicles without batteries; lighter blue, the production of batteries; grey, the operation phase (driving, including the WTW cycle and maintenance) and yellow, EOL treatment. The contribution from each lifecycle phase is discussed in further detail in the following sections. It is important to note that differences in vehicle lifetime between the two powertrain technologies may affect the overall environmental ranking of the vehicles. That is, to ensure and maximise the environmental benefits of the electric powertrain, it is imperative that the batteries should last as long as the vehicles they power. The production of replacement cells may compromise the ability of BEVs to compensate for the higher production impact.

[15] The capacity of batteries in present-day BEVs ranges between 14.4 and 100 kWh. The smallest battery packs (14.4 – 16 kWh) are used in mini vehicles, such as the Peugeot iOn and the Mitsubishi i-MiEV. Most mid-sized vehicles, such as the electric Ford Focus, Nissan Leaf, and Volkswagen e-Golf, have battery packs with a capacity of around 24 kWh. The largest battery packs (90 to 100 kWh) are primarily used in larger vehicles such as the Tesla Model S and Model X.

Figure 5: Lifecycle GHG emissions of mid-sized 24 kWh battery electric (left) and internal combustion engine (right) vehicles.



Note: The average production emissions were selected for the battery and rest of the BEV. Vehicle operational energy use is assumed to be 16.0 kWh/100 km for the BEVs (assuming the efficiency of the battery to be at 95 %, the electric motor at 95 % and the inverter at 97 %), furthermore assuming 3.8 l/100 km for the diesel vehicle, and 4.8 l/100 km for the petrol-driven vehicle over a lifetime of 150 000 km. The carbon intensity for electricity, including transmission losses, is assumed to be 353 g CO_2/kWh for the average European electricity mix, 671 g CO_2/kWh for world average natural gas-based electricity, 1160 g CO_2/kWh for world average coal-based electricity and 23 g CO_2/kWh from wind.

Data sources: Production, LIBs (Bauer, 2010; Dunn et al., 2015; Ellingsen et al., 2014; Kim et al., 2016; Majeau-Bettez et al., 2011; Miotti et al., 2015; Notter et al., 2010; United States Environmental Protection Agency, 2013; Volkswagen AG, 2012a; Zackrisson et al., 2010). Production, rest of electric vehicle (Dunn et al., 2015; Hawkins et al., 2013; Kim et al., 2016; Miotti et al., 2015; Notter et al., 2010; Volkswagen AG, 2012a). Production, conventional vehicles (Daimler AG, 2014, 2012; Dunn et al., 2015; Hawkins et al., 2013; Kim et al., 2016; Miotti et al., 2015; Notter et al., 2010; Volkswagen AG, 2012b). Operation, energy use ('Ford Focus', 2017, 'Ford Focus Electric', 2016; Volkswagen AG, 2014a). Electric powertrain efficiency (Nordelöf et al., 2017; Ellingsen et al., 2016b; Mahmoudzadeh Andwari et al., 2017). Carbon intensity, electricity production (Bruckner et al., 2014; IEA, 2016). Carbon intensity, electricity transmission (Ecoinvent Centre, 2010). Carbon intensity, fuel production and combustion (WTW) (Edwards et al., 2013). Maintenance and EOL (Hawkins et al., 2012).

There are significant variations in GHG emissions among and between the different powertrain technologies. The lifecycle emissions of the BEVs depend on the electricity sources used to charge the battery (Bauer et al., 2015; Ellingsen et al., 2016b; Faria et al., 2012; Hawkins et al., 2012; Miotti et al., 2015; Notter et al., 2015). When powered by the average European mix, BEVs are found to moderately reduce GHG emissions compared to both diesel and petrol-driven ICEVs assuming a 150 000 km vehicle lifetime (Ellingsen et al., 2016b; Hawkins et al., 2012; Miotti et al., 2015; Notter et al., 2015). When powered by coal-based electricity, BEVs are found to cause an increase in GHG emissions compared with the ICEVs (Ellingsen et al., 2016b; Hawkins et al., 2012; Notter et al., 2015). By contrast, BEVs powered by a renewable energy source, such as wind, have significantly reduced lifecycle GHG emissions compared to the ICEVs (Bauer et al., 2015; Ellingsen et al., 2016b; Faria et al., 2012; Hawkins et al., 2012;

Miotti et al., 2015; Notter et al., 2015). Therefore, even though the BEV production phase is more carbon-intensive than that of ICEVs, the BEVs can, under the right conditions, compensate for the higher production emissions by lower use phase emissions. Diesel ICEVs tend to have somewhat lower lifecycle GHG emissions than petrol-driven ICEVs (Hawkins et al., 2012). The text below provides further information and insights on the GHG emissions associated with vehicle and battery production, vehicle operation and EOL treatment.

4.1.1. Contributions from vehicle and battery production

The underlying sources of BEVs' lifecycle emissions differ from ICEVs. The production intensity for a mid-sized BEV is around 6.0–7.4 tonnes CO_2-eq/tonne of car (Daimler AG, 2014; Hawkins et al., 2013; Kim et al., 2016; Miotti et al., 2015; Volkswagen AG, 2012a), while for ICEVs, the intensity is around 4.2–5.5 tonne CO_2-eq/tonne of car (Daimler AG, 2014, 2012; Dunn et al., 2015; Hawkins et al., 2013; Kim et al., 2016; Miotti et al., 2015; Notter et al., 2010; Volkswagen AG, 2012b). The difference between BEVs and ICEVs in production emissions is mainly due to the traction battery, which accounts for around 33–44 % of total BEV production emissions (Ellingsen et al., 2014; Hawkins et al., 2012; Kim et al., 2016; Miotti et al., 2015; Volkswagen AG, 2012a). Commercial battery packs can vary in design, materials, production techniques, and pack size, so some variations in production impact are to be expected.

LCA studies report a large range in overall battery production emissions, with results between 38–356 kg CO_2-eq/kWh, corresponding to 0.9–8.6 tonnes of CO_2-eq per 24-kWh battery pack (see Figure). As battery producers have generally not been very forthcoming with information or production data, the majority of existing studies base their inventories on assumptions or secondary data available in the literature. Although there are many discrepancies in the reported results, two recent studies based on primary industry data find similar emissions of approximately 140–170 kg CO_2-eq/kWh (3.4–4.1 tonnes of CO_2-eq per 24 kWh battery pack) (Ellingsen et al., 2014; Kim et al., 2016). Confidence in these two results is relatively high for three reasons: the data is primary, originates from two independent battery producers, and the two studies report both similar total production emissions and contributing factors. The same two industry-based studies find that for a 24 kWh battery pack, about half of the production-related GHG emissions derive from the energy use required in cell manufacture (Ellingsen et al., 2014; Kim et al., 2016). The remaining emissions are distributed between the cell materials (about 20 %), enclosure (12–18 %), and other battery components (7–14 %) (Ellingsen et al., 2014; Kim et al., 2016). According to these results, the most promising approach for reducing production-related GHG emissions of traction batteries is to reduce the emissions associated with cell manufacture. This can be achieved through lower energy use or use of cleaner energy sources (Ellingsen et al., 2017). Thus, Tesla Gigafactory 1, which claims to be self-sufficient in energy from renewable sources, may reduce battery cell production impacts by up to 50 % compared to cells produced in South Korea, China, and Japan. To a lesser extent, using recycled metals in the battery, particularly copper, aluminium, nickel, and cobalt, might also reduce production-related GHG emissions by avoiding the use of virgin materials (Ellingsen et al., 2017).

4.1.2. Contributions from vehicle operation

For both ICEVs and BEVs, the use phase is responsible for much of the GHG emissions. The WTW cycle is responsible for the majority of operational emissions, whether directly through fuel combustion or indirectly through electricity production, while maintenance contributes very little to the operational emissions (Hawkins et al., 2012).Consequently, the BEV use phase contributions mainly depend on the carbon intensity of the electricity and can range from about 5–75 % of total lifecycle emissions (Ellingsen et al., 2016b; Hawkins et al., 2012; Miotti et al., 2015; Notter et al., 2015). For diesel and petrol vehicles, the use phase contributes about 80 % of total emissions (Ellingsen et al., 2016b; Hawkins et al., 2012; Miotti et al., 2015; Notter et al., 2015). For both electric and conventional vehicles, energy consumption and the sources of energy are important factors in the use phase. Several studies have found that the carbon intensity of the electricity in the operation phase determines the advantage – or disadvantage – of adopting BEVs over ICEVs with regard to GHG emissions. While relatively similar lifecycle emissions are achieved by charging BEVs with electricity from natural gas (Bauer et al., 2015; Ellingsen et al., 2016b; Hawkins et al., 2012), wind-based electricity results in significantly lower lifecycle emissions compared to ICEVs (Ellingsen et al., 2016b; Miotti et al., 2015; Notter et al., 2015). By contrast, coal-based electricity yields higher lifecycle emissions for BEVs than ICEVs (Bauer et al., 2015; Ellingsen et al., 2016b; Hawkins et al., 2012). Hence, in areas where electricity is primarily produced from thermal power plants fired by lignite, coal or heavy oil, BEVs serve as a means of improving urban air quality and shifting emissions away from the densely populated urban centres, rather than reducing GHG emissions overall (Hawkins et al., 2012).

The importance of the operation phase depends on the duration of operation; the longer the vehicle operates, the greater the importance of the GHG emissions stemming from the use phase, with production emissions contributing a smaller share of the overall emissions.

4.1.3. Contributions from end-of-life treatment

Irrespective of powertrain technology, LCA studies have found that the emissions stemming from EOL treatment are small compared to the emissions stemming from production and operation (Daimler AG, 2014; Hawkins et al., 2012; Miotti et al., 2015; Notter et al., 2010; Volkswagen AG, 2014b). The same studies report that BEVs have somewhat higher EOL emissions than the ICEVs due to the battery, but there is a great deal of uncertainty associated with the reported results of battery recycling because of the low data availability for various industrial recycling schemes. Because of the poor availability of data, only very few LCA studies have evaluated the final lifecycle stage of batteries, and they have assessed different treatment options. The studies report relatively low impacts from battery recycling (Ellingsen et al., 2016b; Hawkins et al., 2012; Li et al., 2014; Notter et al., 2010). Battery recycling schemes that are more reliant on energy use (pyrometallurgical treatment) are likely to have higher GHG emissions, while recycling schemes that rely more on the use of solvents (hydrometallurgical treatment) are likely to have higher loads in other environmental impact categories (e.g. eutrophication). Contrary to the operation phase, EOL emissions associated with battery recycling are unlikely to be highly influenced by changes in the electricity mix as only a small share of the energy input in the EOL treatment processes is from electricity. Improved inventory data is required to provide more certain answers regarding GHG emissions due to battery recycling.

4.1.4. The role of battery electric vehicles in reducing greenhouse gas emissions

Although BEVs have potential GHG benefits, these benefits cannot be harnessed everywhere and under all conditions. Because the carbon intensity of the electricity used for charging significantly influences the lifecycle emissions of BEVs, the potential benefit or disadvantage of BEVs compared to ICEVs will vary among different countries. In countries where there is a large share of renewable or nuclear power, one can expect a substantial benefit from adopting BEVs. However, in countries with a large share of fossil fuels in the electricity mix, using conventional ICEVs is beneficial over BEVs; the larger production emissions for BEVs cannot be overcome in the driving phase when the electricity is carbon intensive. For some countries, the difference in lifecycle emissions between the two powertrain technologies is likely to be relatively small. In many countries, the electricity is a dynamic mix of various energy sources. Electricity mixes are susceptible to change over time as power plants are built and decommissioned.

The expected decarbonisation of the European power sector, coupled with improvements in electrochemical and powertrain performances, should lead to an increase in the environmental benefits of BEV use over time, particularly from a climate change perspective. This anticipated decarbonisation of the power sector combined with the expected increase in impacts associated with the extraction of dwindling oil reserves would result in increased environmental benefits of BEVs compared to ICEVs. Thus, the trend towards increased use of renewables in the power sector is a positive development for BEVs as it reduces their lifecycle GHG emissions. These effects are a result of not only the electricity used for charging becoming cleaner, but also the electricity being used to produce the batteries, which corresponds to about half of the climate change impacts from battery production shown in the light blue series in Figure .

Due to its relative novelty as a technology, the LIBs used in BEVs have considerable potential for improvements in manufacturing efficiencies and technological improvements such as increased gravimetric energy density and lifetime (Hao et al., 2017; Sauer et al., 2016). Such improvements may further decrease the lifecycle impacts of BEVs. By contrast, the energy efficiency of ICEVs is inherently limited by the thermodynamics of the internal combustion engine and are approaching a plateau in technological improvements (Sims et al., 2014). Decreases in the lifecycle emissions from this vehicle technology would therefore have to be achieved through other means, such as lightweighting, reduced aerodynamic drag, and rolling resistance. These improvements, however, are not unique to ICEVs and can also be applied to BEVs to improve their lifecycle impacts even further.

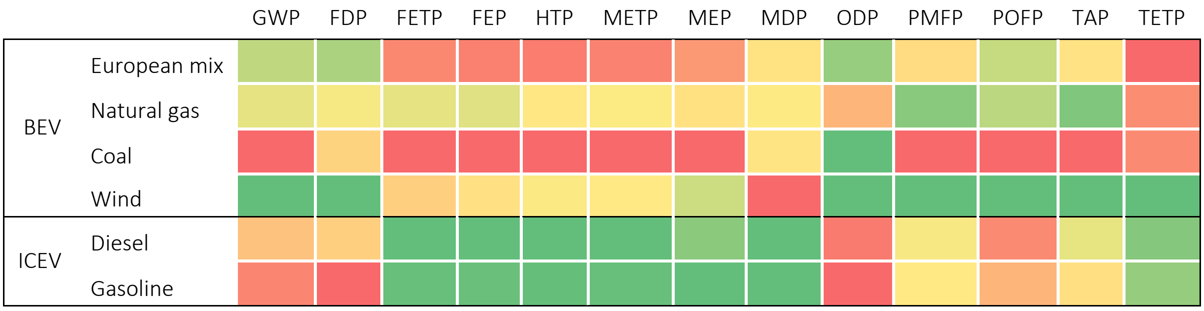
Regardless of powertrain configuration, smaller vehicles are generally more energy efficient than larger ones, and have lower GHG emissions (Ellingsen et al., 2016b; Miotti et al., 2016). Although increasing the size and range of batteries make BEVs more marketable and acceptable for consumers, the implications here are that the BEV production emissions increase, and operation emissions also increase (to move the heavier vehicle). As a result, any advantages over their ICEV counterparts (i.e. vehicles of similar size) decrease when larger batteries are installed in BEVs of a given segment. Developing batteries with higher gravimetric energy density is therefore a priority as it can allow for longer driving ranges without making the battery packs heavier. So far, the gravimetric energy density has mainly been increased by fabrication improvements, by using progressively lighter cases (e.g. using aluminium instead of stainless steel), or by the optimisation of cell design so as to increase the loading of active electrode materials (Scrosati and Garche, 2010). The gravimetric energy density of LIBs is mainly determined by the properties of the active electrode materials (Liu et al., 2010), particularly cathode material as its practically achievable energy is much inferior to that of the anode material (Li et al., 2015; Rosenman et al., 2015). Developments expected in the near future include the increased use of nickel in the cathode materials and the introduction of silicon-based anode materials. However, these developments are expected only to result in incremental improvements of the gravimetric energy density. New battery technologies such as lithium-sulphur and lithium-air are being researched, but such technologies face substantial challenges (e.g. lifetime) that must be resolved before potential commercialisation (Ellingsen et al., 2016a). Therefore, we cannot expect any revolutionary improvements in gravimetric energy density any time soon.

4.2. Other environmental impacts

Climate change is but one of many environmental impact types considered in LCA studies. When assessing the lifecycle impact of product or a service, however, one should also consider other relevant environmental impacts, as trade-offs often exist. Figure 6 presents the relative lifecycle environmental performance of BEVs (charged with the average European mix, natural gas, coal, and wind) and ICEVs (fuelled with diesel and petrol) for an operational use of 150 000 km. The environmental performance of the vehicles in each environmental impact category is ranked by colour, where red denotes the poorest, yellow middling, and green the best relative performance in each environmental impact category.

Figure 6 shows trade-offs between lifecycle benefits and disadvantages for BEVs compared to ICEVs. As a general rule, this is because BEVs are not always able to compensate for the higher environmental impacts from production with lower use phase impacts. This is particularly the case for the environmental impact categories (eutrophication and toxicity) affected by metal use.

Figure 6: Lifecycle environmental impacts of battery electric and internal combustion engine vehicles.



Note: Rankings are made down columns, i.e., for each impact category[16].

Source: Data source: (Ellingsen et al., 2014; Hawkins et al., 2013)

[16] Impact category abbreviations as follows: global warming potential (GWP), fossil depletion potential (FDP), freshwater ecotoxicity potential (FETP), freshwater eutrophication potential (FEP), human toxicity potential (HTP), marine ecotoxicity potential (METP), marine eutrophication potential (MEP), metal depletion potential (MDP), ozone depletion potential (ODP), particulate matter formation potential (PMFP), photo oxidation formation potential (POFP), terrestrial acidification potential (TAP) and terrestrial ecotoxicity potential (TETP).

5. CONCLUSIONS

Battery production can be subdivided into cell manufacture and pack assembly, of which the former is considerably more energy-intensive than the latter. This translates into a considerable contribution to the greenhouse gas emissions associated with BEV manufacturing.

LIBs are complex products containing many different materials, some of which may already be subject to supply risk, or may become so in the future. Lithium and cobalt are likely to pose the highest supply risk, while aluminium poses the lowest risk. Manganese, iron, nickel, copper, and natural graphite have a medium supply risk. The improvement of recovery processes for these elements at the vehicle's end-of-life can, to some degree, alleviate these supply risks. As we move from fuel-intensive ICEVs to materials-intensive BEVs, it becomes increasingly more important to have efficient recycling processes in place to ensure the optimal recovery of finite minerals as well as energy- and pollution-intensive materials.

The shift in powertrain technology leads to a change in lifecycle GHG emissions of passenger vehicles. In order for BEVs to provide a climate change mitigation alternative to ICEVs, they must have lower lifecycle GHG emissions. As a result of the battery, BEVs have higher production intensities than ICEVs. Therefore, BEVs must have lower use phase emissions than ICEVs to achieve lower lifecycle emissions. Lifecycle assessment studies find that whether the BEVs manage to compensate for their higher production emissions or not largely depends on the carbon intensity of the electricity sources used to charge the battery. Thus BEVs do not have unconditional climate benefits over ICEVs; these benefits only exist under certain circumstances. Lifecycle assessment studies consistently report moderate climate benefits for BEVs powered by the current average European electricity mix compared to similarly sized ICEVs.

Regardless of powertrain configuration, smaller vehicles tend to be more energy efficient during operation and generally have lower GHG emissions than larger ones. The trend in increasing the size and range of BEVs is unfavourable from both a climate mitigation and resource use perspective. Thus, given the current state of the technology, striking the right balance between battery and vehicle size and charging infrastructure is an important element in maximising the climate change mitigation potential of BEVs. While improving the gravimetric energy density of LIBs can allow for longer driving ranges without making the battery packs heavier, only incremental improvements can be expected in the near future.

The climate benefit of BEVs compared to ICEVs is expected to increase in the near future due to changes in the European power sector and developments within cell manufacturing. The expected decarbonisation of the European power sector should lead to an increase in the environmental benefits of BEV use over time, particularly from a climate change perspective. Decarbonisation of the electricity mix in cell-manufacturing countries would additionally be beneficial with respect to the battery production process. The most significant change in cell manufacture in the near future is the expected large production volume of cells by Tesla Gigafactory 1, which claims to be self-sufficient in renewable energy. Consequently, GHG emissions from both production and use are expected to decrease in the coming years.

Achieving the goal of more sustainable transport solutions requires a good environmental understanding of our technologies. To avoid potential environmental pitfalls in the adoption of BEVs, the automotive industry and their suppliers, particularly battery producers and recyclers, have to be more transparent about the environmental implications of their product lines. An open communications channel between industry, government and researchers is essential if BEVs are to succeed as a climate change mitigation initiative within the transport sector.

REFERENCES

- Accurec, 2017. Personal communication.
- Anderman, M., 2016a. Extract from The Tesla Battery Report.
- Anderman, M., 2016b. Extract from the Battery Pack of Modern xEVs Report.
- Anderman, M., 2016c. Extract from The xEV Industry Report.
- Bauer, C., 2010. Ökobilanz von Lithium-Ionen Batterien, Analyse der Herstellung von Energiespeichern für den Einsatz in Batteriefahrzeugen. PSI, Technology Assessment / GaBE, Villigen, Switzerland.
- Bauer, C., Hofer, J., Althaus, H.-J., Del Duce, A., Simons, A., 2015. The environmental performance of current and future passenger vehicles: Life Cycle Assessment based on a novel scenario analysis framework. Applied Energy 1–13. doi:10.1016/j.apenergy.2015.01.019
- Bernhart, W., 2014. Lithium-ion batteries : advances and applications, in: Pistoia, G. (Ed.), Lithium-Ion Batteries: Advances and Applications. Elsevier, Amsterdam, pp. 553–565.
- Bigum, M., Brogaard, L., Christensen, T.H., 2012. Metal recovery from high-grade WEEE: a life cycle assessment. Journal of Hazardous Materials 207–208, 8–14. doi:10.1016/j.jhazmat.2011.10.001
- Birke, P., 2014. Lithium-Ion Accumulators Discussion of different cell housings. ATZelektronik worldwide 9, 16–21. doi:10.1365/s38314-014-0286-y
- Bruckner, T., Bashmakov, I.A., Mulugetta, Y., Chum, H., Navarro, A. de la V., Edmonds, J., Faaij, A., Fungtammasan, B., Garg, A., Hertwich, E., Honnery, D., Infield, D., Kainuma, M., Khennas, S., Kim, S., Nimir, H.B., Riahi, K., Strachan, N., Wiser, R., Zhang, X., 2014. Energy Systems., in: Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner, S., Seyboth, K., Adler, A., Baum, I., Brunner, S., Eickemeier, P., Kriemann, B., Savolainen, J., Schlömer, S., Stechow, C. von, Zwickel, T., Minx, J.C. (Eds.), Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 511–598.
- Daimler AG, 2014. Environmental Certificate Mercedes-Benz B-Class Electric Drive. Stuttgart, Germany.
- Daimler AG, 2012. Environmental Certificate Mercedes-Benz A-Class. Stuttgart, Germany.
- Daniel, C., 2008. Materials and processing for lithium-ion batteries. Journal of The Minerals, Metals and Materials Society 60, 43–48. doi:10.1007/s11837-008-0116-x
- Diekmann, J., Hanisch, C., Froböse, L., Schälicke, G., Loellhoeffel, T., Fölster, A.-S., Kwade, A., 2017. Ecological Recycling of Lithium-Ion Batteries from Electric Vehicles with Focus on Mechanical Processes. Journal of the Electrochemical Society 164, A6184–A6191. doi:10.1149/2.0271701jes
- Dijk, M., Orsato, R.J., Kemp, R., 2013. The emergence of an electric mobility trajectory. Energy Policy 52, 135–145. doi:10.1016/j.enpol.2012.04.024
- Dunn, J.B., Gaines, L., Kelly, J.C., James, C., Gallagher, K.G., 2015. The significance of Li-ion batteries in electric vehicle life-cycle energy and emissions and recycling's role in its reduction. Energy and Environmental Science 8, 158–168.
- Dunn, J.B., Gaines, L., Sullivan, J., Wang, M.Q., 2012. Impact of recycling on cradle-to-gate energy consumption and greenhouse gas emissions of automotive lithium-ion batteries. Environmental Science and Technology 46, 12704–12710. doi:10.1021/es302420z
- Ecoinvent Centre, 2010. Ecoinvent data and reports v.2.2. Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories.

- Edwards, R., Larive, J.-F., Rickeard, D., Weindorf, W., 2013. Well-to-Wheels analysis of future automotive fuels and powertrains in the European context: Well-to-Tank Appendix 4 - Version 4.0, Joint Research Centre of the European Commission, EUCAR, and CONCAWE. doi:10.2788/40526
- Ellingsen, L.A.-W., 2017. Life cycle assessment of lithium-ion traction batteries. Norwegian University of Science and Technology.
- Ellingsen, L.A.-W., Hung, C.R., Majeau-Bettez, G., Singh, B., Chen, Z., Whittingham, M.S., Strømman, A.H., 2016a. Nanotechnology for environmentally sustainable electromobility. Nature Nanotechnology 11, 1039–1051. doi:10.1038/nnano.2016.237
- Ellingsen, L.A.-W., Majeau-Bettez, G., Singh, B., Srivastava, A.K., Valøen, L.O., Strømman, A.H., 2014. Life cycle assessment of a lithium-ion battery vehicle pack. Journal of Industrial Ecology 18, 113–124. doi:10.1111/jiec.12072
- Ellingsen, L.A.-W., Majeau-Bettez, G., Strømman, A.H., 2015. Comment on 'The significance of Li-ion batteries in electric vehicle life-cycle energy and emissions and recycling's role in its reduction' in Energy & Environmental Science. Journal. of Industrial Ecology 19, 518–519. doi:10.1111/jiec.12309
- Ellingsen, L.A.-W., Singh, B., Strømman, A.H., 2016b. The size and range effect: lifecycle greenhouse gas emissions of electric vehicles. Environmental Research Letters 11, 1–8. doi:10.1088/1748-9326/11/5/054010
- Ellingsen, L.A., Hung, C.R., Strømman, A.H., 2017. Identifying key assumptions and differences in life cycle assessment studies of lithium-ion traction batteries with focus on greenhouse gas emissions. Transportation Research Part D: Transport and Environment 55, 82–90.
- European Parliament, C., 2006. Directive 2006/66/EC of the European Parliament and of the Council of 6 September 2006 on batteries and accumulators and waste batteries and accumulators and repealing Directive 91/157/EEC (Text with EEA relevance). doi:12.12.2013
- Faria, R., Moura, P., Delgado, J., de Almeida, A.T., 2012. A sustainability assessment of electric vehicles as a personal mobility system. Energy Conversion and Management 61, 19–30. doi:10.1016/j.enconman.2012.02.023
- Ford Focus, 2017.
- Ford Focus Electric, 2016.
- Gabriel, B., Gerlagh, R., Suh, S., Barrett, J., Coninck, H.C. de, Morejon, C.F.D., Mathur, R., Nakicenovic, N., Ahenkora, A.O., Pan, J., Pathak, H., Rice, J., Richels, R., Smith, S.J., Stern, D.I., Toth, F.L., Zhou, P., 2014. Drivers, Trends and Mitigation, in: Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner, S., Seyboth, K., Adler, A., Baum, I., Brunner, S., Eickemeier, P., Kriemann, B., Savolainen, J., Schlömer, S., Stechow, C. von, Zwickel, T., Minx, J.C. (Eds.), Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 351–412.
- General Motors, n.d. Brownstown Battery [WWW Document]. URL http://media.gm.com/media/us/en/gm/company_info/facilities/battery-assembly/brownstown.html (accessed 12.31.16).
- Georgi-Maschler, T., Friedrich, B., Weyhe, R., Heegn, H., Rutz, M., 2012. Development of a recycling process for Li-ion batteries. Journal of Power Sources 207, 173–182. doi:10.1016/j.jpowsour.2012.01.152
- Glöser, S., Soulier, M., Tercero Espinoza, L.A., 2013. Dynamic analysis of global copper flows. Global stocks, postconsumer material flows, recycling indicators, and uncertainty evaluation. Environmental Science and Technology 47, 6564–6572. doi:10.1021/es400069b

- Graedel, T.E., Allwood, J., Birat, J.-P., Buchert, M., Hagelüken, C., Reck, B.K., Sibley, S.F., Sonnemann, G., 2011. What Do We Know About Metal Recycling Rates? Journal of Industrial Ecology 15, 355–366. doi:10.1111/j.1530-9290.2011.00342.x
- Gratz, E., Sa, Q., Apelian, D., Wang, Y., 2014. A closed loop process for recycling spent lithium ion batteries. Journal of Power Sources 262, 255–262. doi:10.1016/j.jpowsour.2014.03.126
- Grosjean, C., Miranda, P.H., Perrin, M., Poggi, P., 2012. Assessment of world lithium resources and consequences of their geographic distribution on the expected development of the electric vehicle industry. Renewable and Sustainable Energy Reviews 16, 1735–1744. doi:10.1016/j.rser.2011.11.023
- Gruber, P.W., Medina, P.A., Keoleian, G.A., Kesler, S.E., Everson, M.P., Wallington, T.J., 2011. Global Lithium Availability A Constraint for Electric Vehicles? Journal of Industrial Ecology 15, 760–775. doi:10.1111/j.1530-9290.2011.00359.x
- Gulbinska, M.K., 2014. Lithium-ion Battery Materials and Engineering. Springer-Verlag London. doi:10.1007/978-1-4471-6548-4
- Hanisch, C., Diekmann, J., Stieger, A., Haselrieder, W., Kwade, A., 2015a. Recycling of Lithium-Ion Batteries, in: Handbook of Clean Energy Systems. pp. 1–24. doi:10.1002/9781118991978.hces221
- Hanisch, C., Loellhoeffel, T., Diekmann, J., Markley, K.J., Haselrieder, W., Kwade, A., 2015b. Recycling of lithium-ion batteries: a novel method to separate coating and foil of electrodes. Journal of Cleaner Production 108, 301–311. doi:10.1016/j.jclepro.2015.08.026
- Hao, H., Cheng, X., Liu, Z., Zhao, F., 2017. China's traction battery technology roadmap: Targets, impacts and concerns. Energy Policy 108, 355–358. doi:10.1016/j.enpol.2017.06.011
- Hauser, A., Kuhn, R., 2015a. High-voltage battery management systems (BMS) for electric vehicles, in: Scrosati, B., Garche, J., Tillmetz, W. (Eds.), Advances in Battery Technologies for Electric Vehicles. Woodhead publishing series in energy, Cambridge, UK, pp. 265–282. doi:10.1016/B978-1-78242-377-5.00011-X
- Hauser, A., Kuhn, R., 2015b. Cell balancing, battery state estimation, and safety aspects of battery management systems for electric vehicles, in: Scrosati, B., Garche, J., Tillmetz, W. (Eds.), Advances in Battery Technologies for Electric Vehicles. Woodhead publishing series in energy, Cambridge, UK, 283–326. doi:10.1016/B978-1-78242-377-5.00012-1
- Hawkins, T.R., Singh, B., Majeau-Bettez, G., Strømman, A.H., 2013. Corrigendum to: Hawkins, T. R., B. Singh, G. Majeau-Bettez, and A. H. Strømman. 2012. Comparative environmental life cycle assessment of conventional and electric vehicles. Journal of Industrial Ecology DOI: 10.1111/j.1530-9290.2012.00532.x
- Hawkins, T.R., Singh, B., Majeau-Bettez, G., Strømman, A.H., 2012. Comparative Environmental Life Cycle Assessment of Conventional and Electric Vehicles. Journal of Industrial Ecology 17, 53–64. doi:10.1111/j.1530-9290.2012.00532.x
- Helbig, C., Bradshaw, A.M., Wietschel, L., Thorenz, A., Tuma, A., 2017. Supply risks associated with lithium-ion battery materials. Journal of Cleaner Production doi:10.1016/j.jclepro.2017.10.122
- Herrmann, F., Rothfuss, F., 2015. Introduction to hybrid electric vehicles, battery electric vehicles, and off-road electric vehicles, in: Advances in Battery Technologies for Electric Vehicles. pp. 3–16. doi:10.1016/B978-1-78242-377-5.00001-7
- IEA, 2016. World Energy Outlook 2016. Int. Energy Agency 1–684. doi:http://www.iea.org/publications/freepublications/publication/WEB_WorldEnergyOutlook2015ExecutiveSummaryEnglishFinal.pdf

- Kaiser, J., Wenzel, V., Nirschl, H., Bitsch, B., Willenbacher, N., Baunach, M., Schmitt, M., Jaiser, S., Scharfer, P., Schabel, W., 2014. Prozess- und Produktentwicklung von Elektroden für Li-Ionen-Zellen. Chemie Ingenieur Technik 86, 695–706. doi:10.1002/cite.201300085
- Kesler, S.E., Gruber, P.W., Medina, P.A., Keoleian, G.A., Everson, M.P., Wallington, T.J., 2012. Global lithium resources : Relative importance of pegmatite, brine and other deposits. Ore Geology Reviews 48, 55–69. doi:10.1016/j.oregeorev.2012.05.006
- Kim, H.C., Wallington, T.J., Arsenault, R., Bae, C., Ahn, S., Lee, J., 2016. Cradle-to-Gate Emissions from a Commercial Electric Vehicle Li-Ion Battery: A Comparative Analysis. Environ. Sci. Technol. 50, 7715–7722. doi:10.1021/acs.est.6b00830
- Li, B., Gao, X., Li, J., Yuan, C., 2014. Life Cycle Environmental Impact of High-Capacity Lithium Ion Battery with Silicon Nanowires Anode for Electric Vehicles. Environmental Science and Technology 48, 3047–3055. doi:10.1021/es4037786
- Li, J., Daniel, C., Wood, D., 2011. Materials processing for lithium-ion batteries. Journal of Power Sources 196, 2452–2460. doi:10.1016/j.jpowsour.2010.11.001
- Li, Q., Li, G., Fu, C., Luo, D., Fan, J., Xie, D., Li, L., 2015. Balancing stability and specific energy in Li-rich cathodes for lithium ion batteries: a case study of a novel Li–Mn–Ni–Co oxide. Journal of Materials Chemistry A 3, 10592–10602. doi:10.1039/C5TA00929D
- Liu, C., Li, F., Ma, L.-P., Cheng, H.-M., 2010. Advanced materials for energy storage. Advanced Materials 22, E28–E62. doi:10.1002/adma.200903328
- Liu, G., Müller, D.B., 2013. Mapping the global journey of anthropogenic aluminum: A trade-linked multilevel material flow analysis. Environmental Science and Technology 47, 11873–11881. doi:10.1021/es4024404
- Mahmoudzadeh Andwari, A., Pesiridis, A., Rajoo, S., Martinez-Botas, R., Esfahanian, V., 2017. A review of Battery Electric Vehicle technology and readiness levels. Renewable and Sustainable Energy Reviews 78, 414–430. doi:10.1016/j.rser.2017.03.138
- Majeau-Bettez, G., Hawkins, T.R., Strømman, A.H., 2011. Life cycle environmental assessment of lithium-ion and nickel metal hydride batteries for plug-in hybrid and battery electric vehicles. Environmental Science and Technology 45, 4548–54. doi:10.1021/es103607c
- Miotti, M., Hofer, J., Bauer, C., 2015. Integrated environmental and economic assessment of current and future fuel cell vehicles. The International Journal of Life Cycle Assessment. doi:10.1007/s11367-015-0986-4
- Miotti, M., Supran, G.J., Kim, E.J., Trancik, J.E., 2016. Personal Vehicles Evaluated against Climate Change Mitigation Targets. Environmental Science and Technology 50, 10795–10804. doi:10.1021/acs.est.6b00177
- Nordelöf, A., Messagie, M., Tillman, A.-M., Ljunggren Söderman, M., Van Mierlo, J., 2014. Environmental impacts of hybrid, plug-in hybrid, and battery electric vehicles—what can we learn from life cycle assessment? The International Journal of Life Cycle Assessment. doi:10.1007/s11367-014-0788-0
- Nordelöf, A., Grunditz, E., Tillman, A.-M., Thiringer, T., Alatalo, M., 2017. A scalable life cycle inventory of an electrical automotive traction machine-part I: design and composition. The International Journal of Life Cycle Assessment 1–19. doi:10.1007/s11367-017-1309-8
- Notter, D.A., Gauch, M., Widmer, R., Wäger, P., Stamp, A., Zah, R., Althaus, H.-J., 2010. Contribution of Li-ion batteries to the environmental impact of electric vehicles. Environmental Science and Technology 44, 6550–6. doi:10.1021/es903729a
- Notter, D.A., Kouravelou, K., Karachalios, T., Daletou, M.K., Haberland, N.T., 2015. Life cycle assessment of PEM FC applications: electric mobility and μ-CHP. Energy and Environmental Science 8, 1969–1985. doi:10.1039/C5EE01082A

- Olivetti, E.A., Ceder, G., Gaustad, G.G., Fu, X., 2017. Lithium-Ion Battery Supply Chain Considerations: Analysis of Potential Bottlenecks in Critical Metals. Joule 1, 229–243. doi:10.1016/j.joule.2017.08.019
- Omar, N., Daowd, M., van den Bossche, P., Hegazy, O., Smekens, J., Coosemans, T., van Mierlo, J., 2012. Rechargeable energy storage systems for plug-in hybrid electric vehicles-assessment of electrical characteristics. Energies 5, 2952–2988. doi:10.3390/en5082952
- pv magazine, 2015. Report: Panasonic largest li-ion battery cell producer [WWW Document]. URL http://www.pv-magazine.com/news/details/beitrag/report--panasonic-largest-li-ion-battery-cell-producer_100020516/#axzz4ECva3DUx (accessed 7.12.16).
- Ramoni, M.O., Zhang, H.C., 2013. End-of-life (EOL) issues and options for electric vehicle batteries. Cleaner Technologies and Environmental Policy 15, 881–891. doi:10.1007/s10098-013-0588-4
- Reck, B.K., Graedel, T.E., 2012. Challenges in metal recycling. Science. 337, 690–695. doi:10.1126/science.1217501
- Reuter, M.A., Hudson, C., van Schaik, A., Heiskanen, K., Meskers, C., Hagelüken, C., 2013. UNEP (2013) Metal Recycling: Opportunities, Limits, Infrastructure, A Report of the Working Group on the Global Metal Flows to the International Resource Panel.
- Rosenman, A., Markevich, E., Salitra, G., Aurbach, D., Garsuch, A., Chesneau, F.F., 2015. Review on Li-Sulfur Battery Systems: An Integral Perspective. Advanced Energy Materials 5, 1–21. doi:10.1002/aenm.201500212
- Sakai, S. ichi, Yoshida, H., Hiratsuka, J., Vandecasteele, C., Kohlmeyer, R., Rotter, V.S., Passarini, F., Santini, A., Peeler, M., Li, J., Oh, G.J., Chi, N.K., Bastian, L., Moore, S., Kajiwara, N., Takigami, H., Itai, T., Takahashi, S., Tanabe, S., Tomoda, K., Hirakawa, T., Hirai, Y., Asari, M., Yano, J., 2014. An international comparative study of end-of-life vehicle (ELV) recycling systems. Journal of Material Cycles and Waste Management 16, 1–20. doi:10.1007/s10163-013-0173-2
- Sauer, A., Thielmann, A., Isenmann, R., 2016. Modularity in Roadmapping - Integrated foresight of technologies, products, applications, markets and society: The case of 'Lithium Ion Battery LIB 2015.' Technological Forecasting and Social Change 125, 321–333. doi:10.1016/j.techfore.2016.08.017
- Schönemann, M., 2017. Battery Production and Simulation, in: Herrmann, C., Kara, S. (Eds.), Multiscale Simulation Approach for Battery Production Systems. Springer International Publishing, 11–37. doi:10.1007/978-3-319-49367-1
- Scrosati, B., Garche, J., 2010. Lithium batteries: Status, prospects and future. Journal of Power Sources 195, 2419–2430. doi:10.1016/j.jpowsour.2009.11.048
- Sims, R., Schaeffer, R., Creutzig, F., Cruz-Núñez, X., D'Agosto, M., Dimitriu, D., Meza, M.J.F., Fulton, L., Kobayashi, S., Lah, O., McKinnon, A., Newman, P., Ouyang, M., Schauer, J.J., Sperling, D., Tiwari, G., Bruckner, T., Bashmakov, I.A., Mulugetta, Y., Chum, H., Navarro, A. de la V., Edmonds, J., Faaij, A., Fungtammasan, B., Garg, A., Hertwich, E., Honnery, D., Infield, D., Kainuma, M., Khennas, S., Kim, S., Nimir, H.B., Riahi, K., Strachan, N., Wiser, R., Zhang, X., 2014. Transport, in: Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner, S., Seyboth, K., Adler, A., Baum, I., Brunner, S., Eickemeier, P., Kriemann, B., Savolainen, J., Schlömer, S., Stechow, C. von, Zwickel, T., Minx, J.C. (Eds.), Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 511–598.
- Speirs, J., Contestabile, M., Houari, Y., Gross, R., 2014. The future of lithium availability for electric vehicle batteries. Renewable and Sustainable Energy Reviews 35, 183–193. doi:10.1016/j.rser.2014.04.018
- Steinhilber, S., Wells, P., Thankappan, S., 2013. Socio-technical inertia: Understanding the barriers to electric vehicles. Energy Policy 60, 531–539. doi:10.1016/j.enpol.2013.04.076

- Sun, X., Hao, H., Zhao, F., Liu, Z., 2017. Tracing global lithium flow: A trade-linked material flow analysis. Resources, Conservation and Recycling 124, 50–61. doi:10.1016/j.resconrec.2017.04.012
- U.S. Geological Survey, 2017a. Mineral Commodity Summaries: Lithium, USGS Mineral Commodity Summaries.
- U.S. Geological Survey, 2017b. Mineral Commodity Summaries - Cobalt, USGS Mineral Commodity Summaries. doi:http://dx.doi.org/10.3133/70140094.
- U.S. Geological Survey, 2017c. Mineral Commodity Summaries: Graphite (natural), USGS Mineral Commodity Summaries.
- U.S. Geological Survey, 2017d. Mineral Commodity Summaries: Manganese, USGS Mineral Commodity Summaries.
- U.S. Geological Survey, 2017e. Mineral Commodity Summaries: Nickel, USGS Mineral Commodity Summaries.
- U.S. Geological Survey, 2017f. Mineral Commodity Summaries - Iron and steel, USGS Mineral Commodity Summaries.
- U.S. Geological Survey, 2017g. Mineral Commodity Summaries: Copper, USGS Mineral Commodity Summaries.
- U.S. Geological Survey, 2017h. Mineral Commodity Summaries: Aluminum, USGS Mineral Commodity Summaries.
- Umicore, 2017. Lithium recycling.
- Umicore, 2016. Personal communication.
- Umicore, n.d. Our recycling process [WWW Document]. URL http://pmr.umicore.com/en/batteries/our-recycling-process (accessed 10.30.17).
- United States Environmental Protection Agency, 2013. Application of Life-Cycle Assessment to Nanoscale Technology: Lithium-ion Batteries for Electric Vehicles.
- Vezzini, A., 2014a. Lithium-Ion Battery Management, in: Pistoia, G. (Ed.), Lithium-Ion Batteries: Advances and Applications. Elsevier, Amsterdam, pp. 345–360. doi:10.1016/B978-0-444-59513-3.00015-7
- Vezzini, A., 2014b. Manufacturers, Materials and Recycling Technologies, in: Pistoia, G. (Ed.), Lithium-Ion Batteries: Advances and Applications. Elsevier, Amsterdam, 529–551. doi:10.1016/B978-0-444-59513-3.00023-6
- Vikström, H., Davidsson, S., Höök, M., 2013. Lithium availability and future production outlooks. Applied Energy 110, 252–266. doi:10.1016/j.apenergy.2013.04.005
- Volkswagen AG, 2014a. Nye Volkswagen e-Golf.
- Volkswagen AG, 2014b. The e-Golf Environmental Commendation - Background Report.
- Volkswagen AG, 2012a. The e-mission. Electric mobility and the environment. Wolfsburg, Germany.
- Volkswagen AG, 2012b. The Golf Environmental Commendation – Background Report.
- Väyrynen, A., Salminen, J., 2012. Lithium ion battery production. The Journal of Chemical Thermodynamics 46, 80–85. doi:10.1016/j.jct.2011.09.005
- Warner, J., 2014. Lithium-Ion Battery Packs for EVs, in: Pistoia, G. (Ed.), Lithium-Ion Batteries: Advances and Applications. Elsevier, 127–150. doi:10.1016/B978-0-444-59513-3.00007-8
- Weil, M., Ziemann, S., 2014. Recycling of Traction Batteries as a Challenge and Chance for Future Lithium Availability, Lithium-Ion Batteries: Advances and Applications. Elsevier. doi:10.1016/B978-0-444-59513-3.00022-4

- Wood, D.L., Li, J., Daniel, C., 2015. Prospects for reducing the processing cost of lithium ion batteries. Journal of Power Sources 275, 234–242. doi:10.1016/j.jpowsour.2014.11.019
- Yaksic, A., Tilton, J.E., 2009. Using the cumulative availability curve to assess the threat of mineral depletion: The case of lithium. Resources Policy 34, 185–194. doi:10.1016/j.resourpol.2009.05.002
- Yihui, D., Chan, J.C.L., 2005. The East Asian summer monsoon: an overview. Meteorology and Atmospheric Physics 89, 117–142. doi:10.1007/s00703-005-0125-z
- Yoshino, A., 2014. Development of the Lithium-Ion Battery and Recent Technological Trends, in: Lithium-Ion Batteries. Elsevier, 1–20. doi:10.1016/B978-0-444-59513-3.00001-7
- Yuan, C., Deng, Y., Li, T., Yang, F., 2017. Manufacturing energy analysis of lithium ion battery pack for electric vehicles. CIRP Annals - Manufacturing Technology 8–11. doi:10.1016/j.cirp.2017.04.109
- Zackrisson, M., Avellan, L., Orlenius, J., 2010. Life cycle assessment of lithium-ion batteries for plug-in hybrid electric vehicles - Critical issues. Journal of Cleaner Production 18, 1519–1529. doi:DOI 10.1016/j.jclepro.2010.06.004
- Ziemann, S., Grunwald, A., Schebek, L., Müller, D. b., Weil, M., 2013. The future of mobility and its critical raw materials. Revue de Métallurgie 110, 47–54. doi:10.1051/metal/2013052
- Ziemann, S., Weil, M., Schebek, L., 2012. Tracing the fate of lithium––The development of a material flow model. Resources, Conservation and Recycling 63, 26–34. doi:10.1016/j.resconrec.2012.04.002

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