CHALMERS
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Environmental concerns of metals in Li-ion batteries

Implications for recycling

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Background

This study is conducted within the project *Recycling of metals from spent Li-ion batteries* (LIBs) running 2014-2016 and funded by the Swedish Energy Agency (Energimyndigheten)[1]. The main aim of the project is to generate fundamental knowledge for resource efficient thermal recycling of current and future Li-ion batteries.

The use of LIBs is rapidly increasing and in addition there is a rapid development of different Li-ion technologies (e.g. Scrosati and Garche 2010). Consequently, spent batteries with varying specific battery technologies in the waste stream have been steadily increasing and is expected to continue to increase. The increasing interest in LIB technology is also evident from the increasing number of studies on environmental impacts from batteries. The metals and materials in batteries are valuable resources that can be recovered in recycling processes. However, current recycling processes recover only few elements from LIBs and also when designing new processes different element are more or less easily recovered. In addition, the precise amounts and composition of future spent batteries cannot be known today which makes designing recycling processes for this growing waste stream difficult.

Assessing the environmental impacts of LIB technologies in a life cycle perspective can provide guidance in regards to what technologies are viable in a long-term sustainability perspective and thereby guide the design of recycling processes. Also this provides knowledge to safeguard that any direct impacts of a recycling process itself isn't environmentally harmful.

Purpose of study

This study provides an overview of the current knowledge of environmental concerns from the use of metals in LIBs, with the purpose to identify what elements are most important to recycle from an environmental and resource perspective. Impacts are assessed along the life cycle of batteries.

Other known hazards associated with LIB such as risk of thermal runaway, fire or explosions due to chemical instability of the active components have not been evaluated here. See e.g. Lisbona and Snee (2011) for a review.

Method

The subject has been approached through the identification of concerns related to environmental impacts from metals in LIBs as evident from different perspectives, and then doing a combined evaluation. The perspectives are market data on metal resources, reserves, supply and demand, material inventories of LIB technologies, life cycle assessment, and criticality studies. The main source of information for this study has been the existing literature. Scientific articles and other publications have been found with search phrases relevant to Li-ion battery recycling, life cycle assessment, criticality, material content, etc. Economical information on metals has also been sought from metal market websites and surveys on resources and reserves. Some data presented here has been calculated e.g. by updating previous published results with more recent data.

The results have been presented to the project group and at reference group meetings and feedback provided from the audience at these meetings has been considered.

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[1] *Recycling of metals from spent Li-ion batteries.*
Energimyndigheten project no: 39060-1, Dnr: 2014-001958

Metal elements in Li-ion batteries

Lithium-ion batteries are a collection of several technologies to store energy that have different qualities useful to different applications. Common denominators are the use of lithium in the cathodes and electrolytes and also (in general) regarding metals used for casing and current collectors including aluminium, iron, and copper. The main differences are found in the material chemistry used in the cathodes and anodes.

Examples of cathode compositions:

$LiCoO_2$	Lithium Cobalt Oxide
$LiNiO_2$	Lithium Nickel Oxide
$LiNiCoAlO_2$	Lithium Nickel Cobalt Aluminium Oxide (NCA)
$LiNi_xCo_yMn_zO_2$	Lithium Nickel Manganese Cobalt Oxide (NMC/NCM)
$LiNi_xMn_yO_4$	Lithium Nickel Manganese Oxide
$LiNi_xMn_yAl_zO_2$	Lithium Nickel Manganese Aluminium Oxide (NCA)
$LiMn_2O_4$	Lithium Manganese Oxide (LMO)
$LiCoPO_4$	Lithium Cobalt Phosphate (LCP)
$LiFePO_4$	Lithium Iron Phosphate (LFP)
Li-S	Lithium Sulphur
Li-air	
Al (Current collector)	

Examples of anode compositions:

Li_2TiO_3 or $Li_4Ti_5O_{12}$	Lithium Titanate (LTO)
C	Carbon
C/Si	
Li/C/Sn	
Cu (Current collector)	

Minor quantities of doping elements including niobium and zirconium can be used in the electrode material matrix adjust the battery performance.

Electrolyte compositions:

The most common electrolyte salt in current LIBs is $LiPF_6$ (lithium hexafluorophosphate) in an organic solvent, such as ethylene carbonate, dimethyl carbonate, and diethyl carbonate (Younesi 2015). Several other Li-salts are applied with varying anion composition of non-metal elements: B, C, N, O, F, P, S, Cl. In early LIB technologies in 1970-80's AsF_6^- (hexafluoroarsenate) and ClO_4^- (perchlorate) was used but have been phased out of commercial applications for toxicity and safety concerns.

The metals used in LIBs that has been further assessed in this study are:

Cathode: Al Co Fe Li Mn Ni

Anode: Li Ti Cu Sn

Electrolyte: Li

Casing/collector/other: Fe Al Cu

Quantities of metals in Li-ion batteries

There is a rapid development of materials used in LIBs which makes it difficult to assess the actual quantities of the different elements in current and future batteries. For the purpose of this project which is focussing on recycling, it would be very valuable to have data on the specific composition of batteries in current and coming products, to be able to forecast future compositions and quantities of waste for which efficient recycling methods needs to be to developed and applied.

At present, used LIBs arriving to the recycling facilities rarely have any information on chemical content. There is no standardised way to document and communicate content of batteries through the supply chain, i.e. manufacturing of batteries and products using the batteries. This makes it difficult to quantify current amounts of different metals in batteries and even more difficult to forecast future compositions and quantities.

Table 1. Composition of materials and components in a selection of LIBs for PHEV (Gaines et al 2011).

	BATTERY	NCA-Graphite	LFP-Graphite	LMO (Spinel)-Graphite	LMO (Spinel)-TiO	AVERAGE
	Cathode	LiNi0.8Co0.15Al0.05O2	LiFePO4	LiMn2O4	LiMn2O4	
	Anode	Graphite	Graphite	Graphite	Li4Ti5O12	
	Battery mass (kg)	75.9	81.6	62.6	106.2	**81.575**
	Cathode active material	24.8%	22.2%	24.4%	28.3%	**24.93%**
	Anode active material	16.5%	15.3%	16.3%	18.9%	**16.75%**
	Electrode elements:					
	Lithium (Li)	1.9%	1.1%	1.4%	2.8%	**1.80%**
	Nickel (Ni)	12.1%	0.0%	0.0%	0.0%	**3.03%**
	Cobalt (Co)	2.3%	0.0%	0.0%	0.0%	**0.58%**
	Aluminum (Al)	0.3%	0.0%	0.0%	0.0%	**0.08%**
Material Composition (mass %)	Oxygen (O)	8.3%	9.0%	12.4%	22.3%	**13.00%**
	Iron (Fe)	0.0%	7.8%	0.0%	0.0%	**1.95%**
	Phosphorus (P)	0.0%	4.4%	0.0%	0.0%	**1.10%**
	Manganese (Mn)	0.0%	0.0%	10.7%	12.4%	**5.78%**
	Titanium (Ti)	0.0%	0.0%	0.0%	9.8%	**2.45%**
	Graphite (C)	16.5%	15.3%	16.3%	0.0%	**12.03%**
	Carbon	2.4%	2.1%	2.3%	4.5%	**2.83%**
	Binder	3.8%	3.4%	3.7%	4.5%	**3.85%**
	Copper parts	13.3%	13.8%	13.5%	2.6%	**10.80%**
	Aluminum parts	12.7%	13.3%	12.5%	13.7%	**13.05%**
	Aluminum casing	8.9%	9.4%	9.2%	8.8%	**9.08%**
	Electrolyte solvent	11.7%	14.2%	11.8%	13.4%	**12.78%**
	Plastics	4.2%	4.6%	4.5%	3.6%	**4.23%**
	Steel	0.1%	0.1%	0.1%	0.1%	**0.10%**
	Thermal insulation	1.2%	1.3%	1.2%	1.2%	**1.23%**
	Electronic parts	0.3%	0.3%	0.4%	0.2%	**0.30%**

**AVERAGE COMPOSITION
OF MAIN COMPONENTS
IN A SELECTION OF PHEV BATTERIES
MASS-%**

Other
8%

Plastics
4%

Copper parts
11%

Cathode active
material
25%

Aluminium casing
and parts
22%

Anode active
material
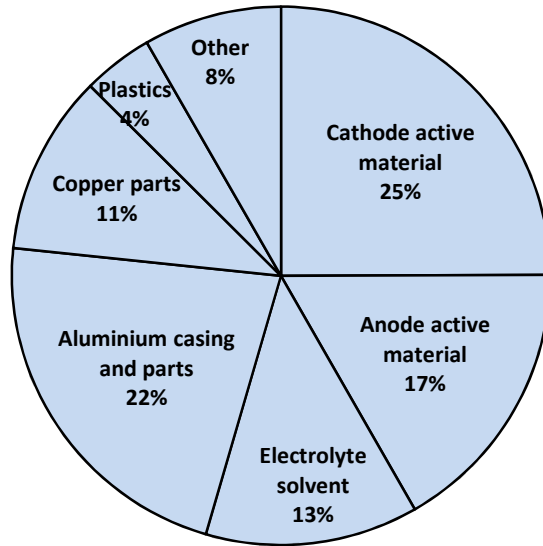17%

Electrolyte
solvent
13%

Figure 1. Average composition of a selection of PHEV batteries based on Gaines et al. (2011).

**AVERAGE COMPOSITION
OF ELEMENTS
IN A SELECTION OF PHEV BATTERIES
MASS-%**

C (incl.
electrolyte);
31,9

Al; 22,2

other; 5,4

Co; 0,6
P; 1,1
Li; 1,8
Fe; 2,1
Ti; 2,5
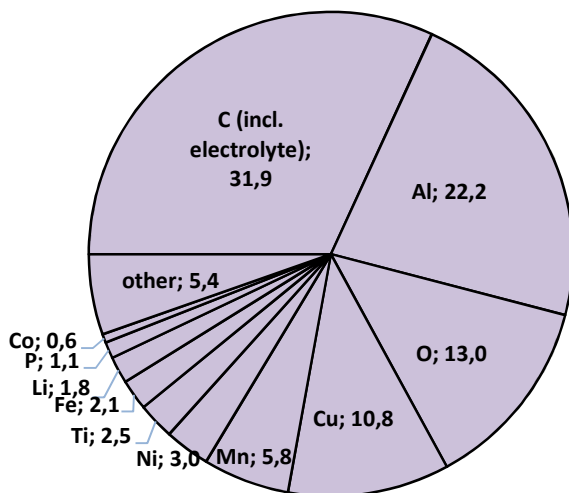Ni; 3,0

Mn; 5,8

Cu; 10,8

O; 13,0

Figure 2. Average composition of elements in a selection of PHEV batteries based on Gaines et al. (2011).

The bulk of the expected increased use of LIBs is due to forecasts of growth of the market for electric vehicles (EV). The amount of spent LIBs in future waste streams therefore largely depends on the future EV market.

A study by Gaines et al. (2011) provides a distribution of mass% of metals in a selection of typical Li-ion batteries used in plug-in hybrid electric vehicles (PHEV), as shown in table 1. In general these batteries are similar in composition except for the cathode active material. It should be noted that these are all batteries for the same application in PHEV. The average composition of components are shown in figure 1. The "other" category contains carbon, binder, steel, thermal insulation, and electronic parts. Figure 2 shows the average elemental composition of the same batteries. The large proportion of carbon content is partly due to that this includes electrolyte and plastics whose elemental composition is not further specified. The lithium salt content and additives of the electrolyte is thereby counted as carbon. The elements in the binder, electronic parts, thermal insulation is also not known and constitute the 8 % of the "other" category.

Richa et al. (2014) made an estimation of the development of future total quantities of main elements in electric vehicle (EV) battery cells waste in the United States, as shown in figure 3. This estimation includes only the cells and excludes casing, thermal insulation, electronics, etc. Furthermore it includes PHEV, hybrid electric vehicles (HEV), and battery electric vehicles (BEV) and is thereby a wider range of battery applications than by Gaines (2011).
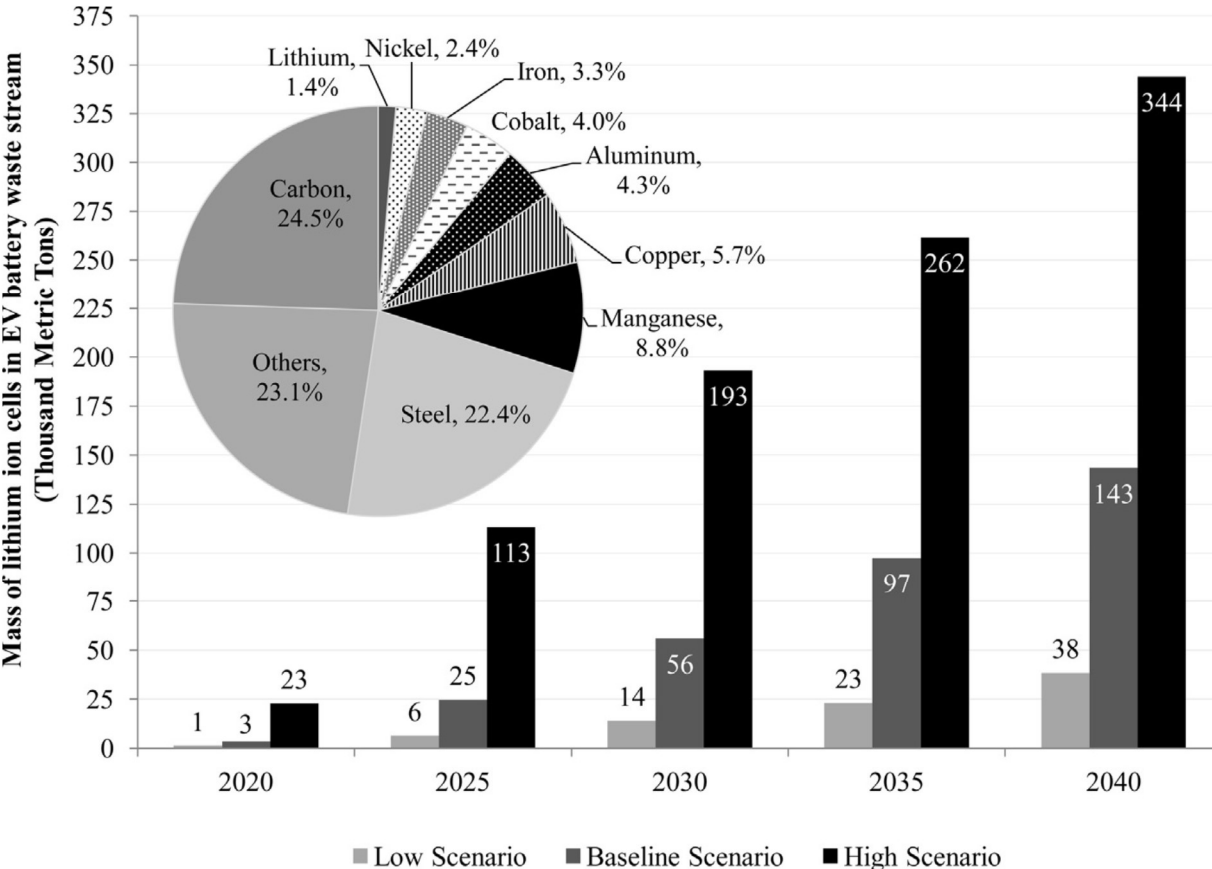

Figure 3. Estimated future EV battery mix and total mass in future US waste. From Richa et al. (2014).

Comparing these two estimates shows a similar large proportion of non-metals, mainly C, O, and H (ca. 45% mass) in both the full battery and in the cells. Notable also is the large difference in Al and (Fe) between the two estimates. This may be due to the specific assumptions and selection of batteries, but also that the bulk part of the complete battery casing and structural part are in general made of aluminium, while the structural components of the cells are mostly made of steel. A significant part (6-11%) is copper both in the full battery and cells. The remainder is the metals of the active materials in the electrodes, i.e. Al, Co, Li, Fe, Ni, Mn, Ti, ca 19% of the whole battery and ca 25% of the battery cells.

The Li content of the electrolyte is not included in these percentages but counted as carbon. Assuming a typical $LiPF_6$ salt concentration of 1M/kg (Younesi 2015), the mass of Li is ca 6.9 g/kg electrolyte. This corresponds to less than 0.1 mass% Li in solvent per kg battery in addition to the estimated 1.8 mass% (fig. 2).

The forecast of future amounts of spent batteries in the waste has a spread of one order of magnitude between a low and a high scenario of LIB market expansion.

Production and demand of metals in Li-ion batteries

Global data on resources of and demand for metals in Li-ion batteries have been collected to provide a snapshot overview of current or potential future supply (table 2). The data was derived from several sources (USGS 2016, EC 2013ab, EC 2014, Steen 2013, UNEP 2011, current metal prices online e.g. infomine.com) with varying scopes and assumptions since no single source could provide all data sought. No source could provide specific data for future demand in LIBs. As a proxy the demand of metals to produce 10M electric vehicles has been estimated based a combination of the above estimates of LIB elemental composition. The recycling rates in table 2 are calculated as three different values as illustrated in figure 4.
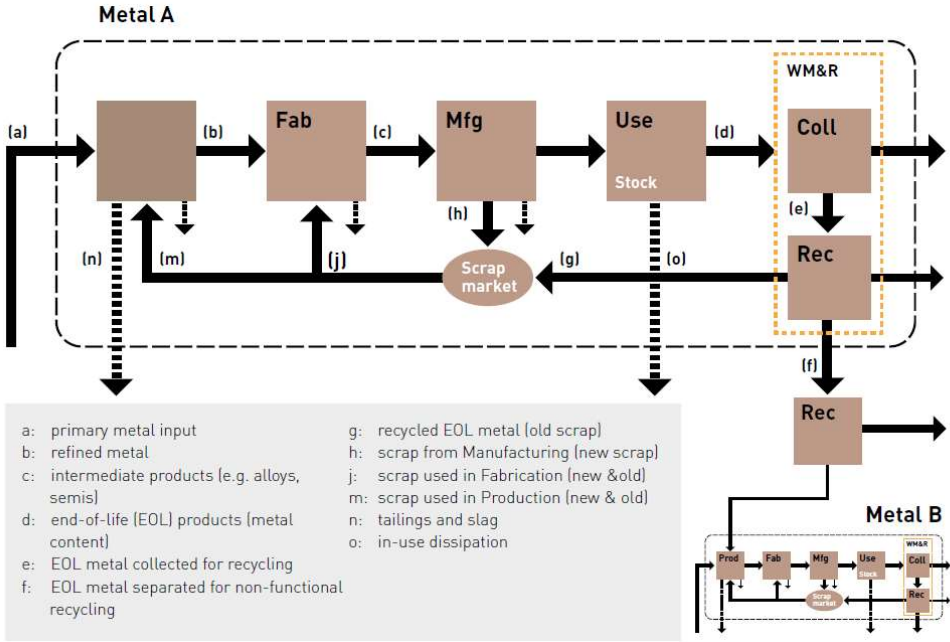

Figure 4. Flows in the life cycle of metals. Source UNEP (2011). Recyling rates in table 2 are given as:
EOL-RR - global end-of-life recycling rates i.e., the percentage of a metal in spent products that is recycled as the same metal = g/d
RC - recycled content in metal production = (j+m)/(a+j+m)
OSR - old scrap ratios i.e., the share of old scrap in the total scrap flow. = g/(g+h)

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Table 2. Resources and demand data on metals in Li-ion batteries. Data collected from USGS 2016, EC 2013ab, EC 2014, Steen 2013, UNEP 2011, current metal prices online e.g. infomine.com.

Element	Resources identified global (tonne)	Reserves current global (tonne)	Production primary 2015 (tonne/yr)	Recycling rate current (%)	Current demand in LIB (% of current prod)	Demand for 10 M EV. (% of current prod)	Price primary US$/tonne (5 yr history trend Apr 2016)
Al	5.5-7.5E+10 (bauxite)	2.8E+10 (bauxite)	5.8E+7	EOL-RR 42-70 RC 34-36 OSR 40-50	< 1% (EV-LIBs)	< 1%	1600 (decr)
Co	2.5E+7 terrestrial 1.2E+8 ocean floor	7.1E+6	1.2E+5	EOL-RR 68 RC 32 OSR 50	30% (all batteries EU 2011)	35%	25 000 (decr)
Cu	2.1E+9	7.2E+8	1.87E+7	EOL-RR 43-53 RC 20-37 OSR 24-78	<1% 10% of prod (EU 2011 automotive)	1.3%	5 000 (decr)
Fe	2.3E+11	8.1E+10 (ore iron content)	1. 178+9 (pig iron)	EOL-RR 52-90 RC 28-52 OSR 52-65	<1% 15% of prod (EU 2011 total automotive)	< 1 %	300 (decr)
Li	3-4E+7 (excl. oceans)	1.4E+7	3.3E+4	EOL-RR <1 RC <1 OSR <1	20%	125%	6 000 (Li2CO3) (steady)
Mn	"large" 75% in South Africa	6.2E+8	1.8E+7	EOL-RR 53 RC 37 OSR 33-67	2% (all batteries EU 2012)	2% tonne/yr	1 700 (decr)
Ni	1.3EE+8	7.9E+7	2.53E+6	EOL-RR 57-63 RC 22 OSR 66-88	<5% (all batteries EU 2012)	2.75%	8 500 (decr)
Sn	"extensive"	4.8E+6	2.7E+5	EOL-RR 75 RC 22 OSR 50	?	?	17 000 (slight decr)
Ti	>2E+9	7.9E+8	6.09E+6	EOL-RR 91 RC 52 OSR 11	< 1%	< 1%	13 000 (decr)

While this data is of varying reliability it does indicate that for many of the metals there is no acute concern in terms of supply for current or future expanded LIB market. For cobalt and lithium a large proportion of total production is used for batteries (30-35%). Currently there are ca 1,3M electric vehicles in use globally (statista 2016). Estimations of future EV car production differ e.g. between 5M to 20M in 2020 (Richa et al 2014 and references therein). Another recent study indicate that production will pass 10M cars/year around year 2030 (Bloomberg 2016). Future demand of Li does likely overshoot current production which indicate a relatively near term supply issue where production must increase. If the trend continues there must also be a significant recycling unless more Li reserves are secured. Current recycling of Li is very small. See also "Focus on lithium" p11. The

production of tin is also relatively small which may pose supply shortage if LIB technologies include Sn-based materials. Data for demand for tin in LIB was not found.

Criticality assessment of metals

To assess the risk of material shortages a number of criticality studies have been conducted (e.g. EC 2014, NRC 2008, USDoE 2010, NEDO 2009, Hatayama and Tahara 2015, Öko-Institut 2009). The criticality concept is a combination of risk of supply shortage and the importance of negative economic impact of shortage as illustrated as in figure 4. Criticality assessments are mostly done with regards to the supply risks associated with a specific geographical jurisdiction such as a region or country or to the supply needs of an organisation or an industrial sector. The assessments' methodologies and scopes differ substantially and are lacking in transparency and clear policy recommendations are rare.
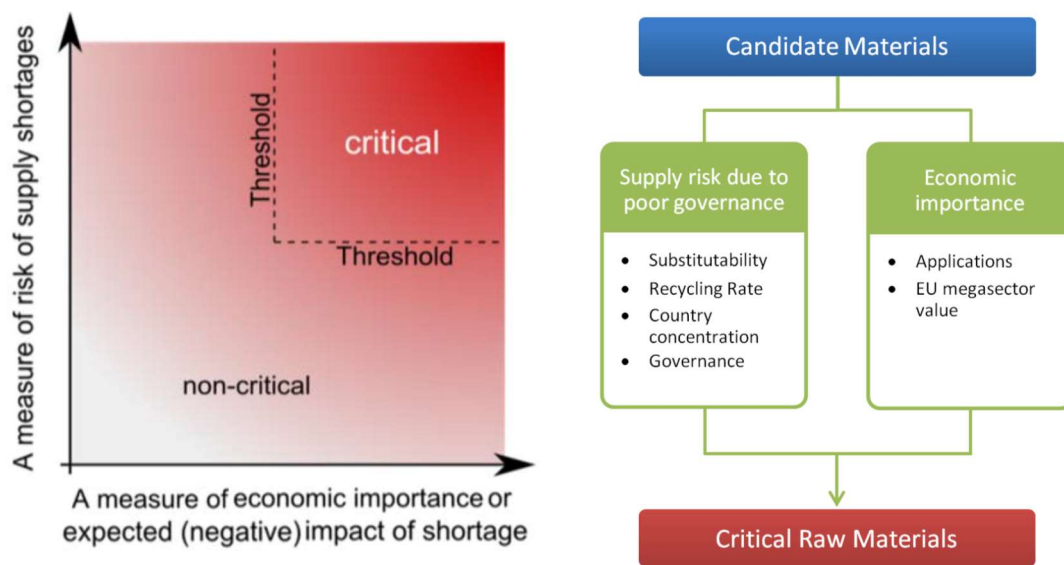

Figure 5. Left: Illustration of the criticality concept as a combination of risk of supply shortages and economic negative impact of shortage. Right: method pathway applied by EC to assess criticality. From EC (2014).
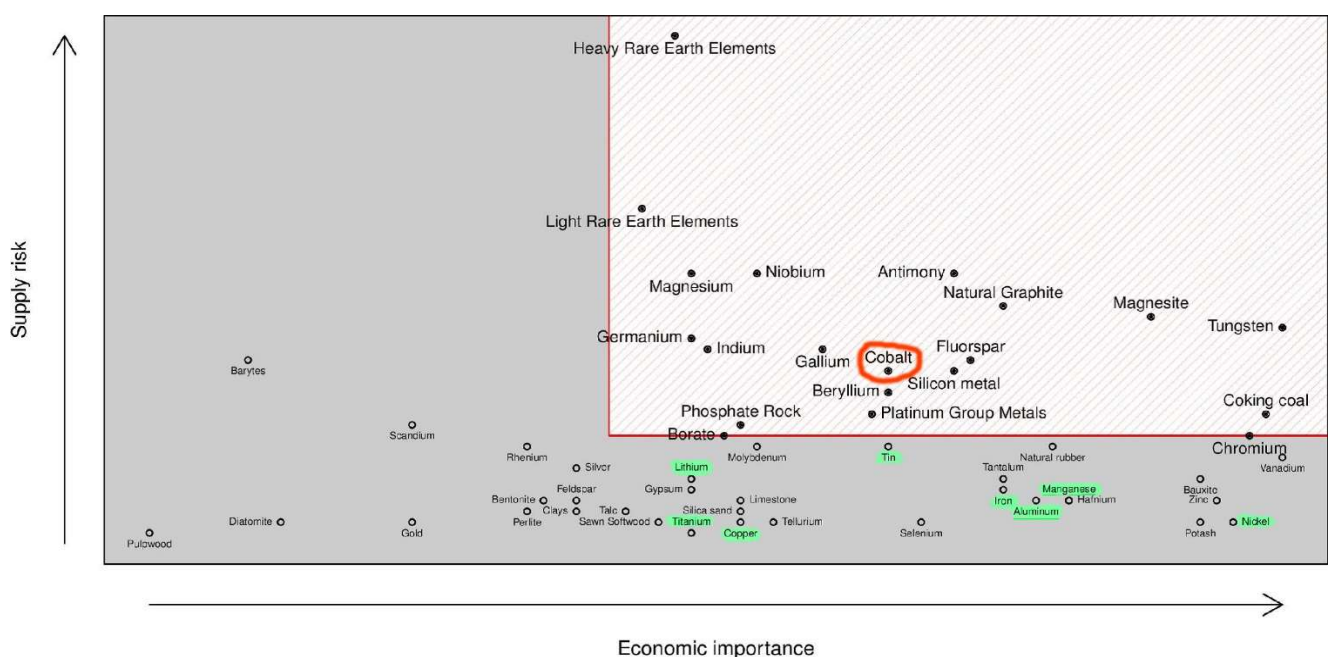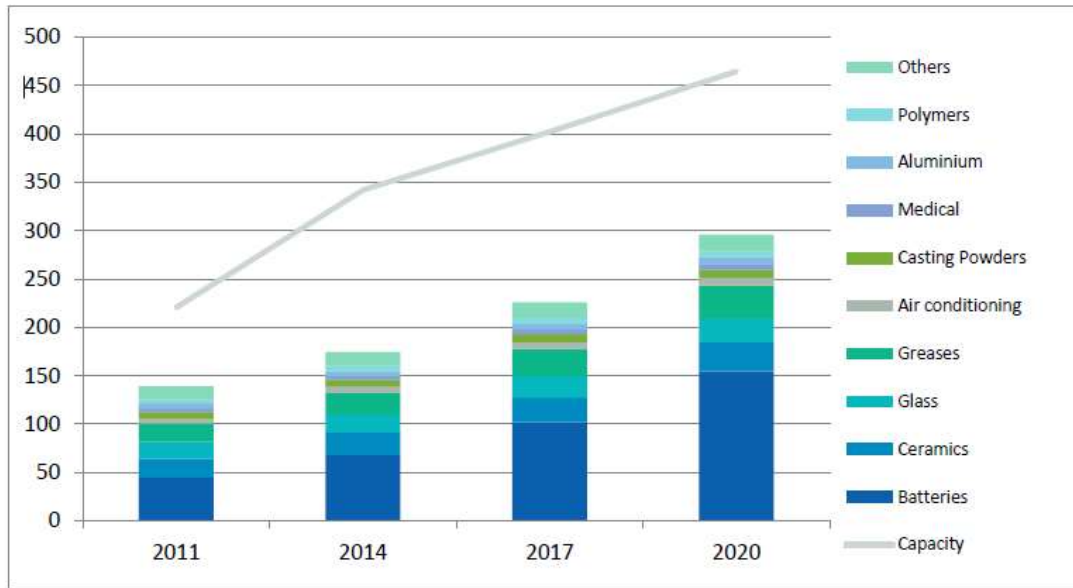

Figure 6. Criticality characterisation from EC (2014), LIB elements highlighted.

The resulting designation of elements from the EC (2013) study is given in figure 6. Of LIB metal elements only cobalt was categorised as critical (in the white area of figure 6). All LIB elements scored higher than the threshold of economic importance. The thresholds are however somewhat vaguely defined based on a "clustering" of resources scoring high in a previous study (EC 2010).

A review of 7 criticality studies was done by Erdmann and Graedel (2011). They compiled which elements were designated as "critical" by the studies as shown in figure 7.

Figure 7. Designations of critical elements from a review of seven criticality studies. From Erdmann and Graedel (2011). Elements related to LIBs are indicated by a star.

None of the elements related to LIBs were consistently categorised as critical in all studies. Li, Co, and Sn was considered critical in half of the studies where these elements were assessed; Mn was considered critical in two of seven assessments; Al, Ti, and Ni by one of six; Cu and Fe was not critical by any assessment. While this doesn't provide conclusive data on criticality for metals in LIBs it does indicate concern for Li, Co, and Sn which is consistent with what is indicated by the supply and demand data in table 2.

Focus on Lithium

The issue with supply risk of lithium has been studied mostly in the context of whether Li production capacity could meet any foreseeable up-scaling of the EV market or not. There is a general consensus from studies that in the short term, ca 10-15 years ahead, there are no foreseen supply problems as long as lithium production capacity increases steadily to meet the increasing demand (e.g. EC 2013b), (fig 8).

Sources: Li3Energy & Roskill Presentations (2013), International Lithium Alliance & Chemetall Presentations (2012)

Figure 8. Short term forecast of lithium demand and supply capacity. From EC (2013b).

In the longer perspective the supply risk depends to a large degree on whether extraction of lithium from ocean water will become viable. If so, Li-reserves would increase tremendously. However Kushnir and Sandén (2012) conclude that in either case virgin production capacity will not grow sufficiently fast to meet plausibly large future increases in lithium demand. E.g. in a scenario of 3 billion HEV or EV in 2100 with LIB technology is only possible with a substantial increase in lithium recycling (fig. 9). Current recycling of lithium is less than 1%. EC (2010) also reached similar conclusions and recommends that a recycling system for lithium is set up to avoid exhaustion of known lithium resources (fig. 10).
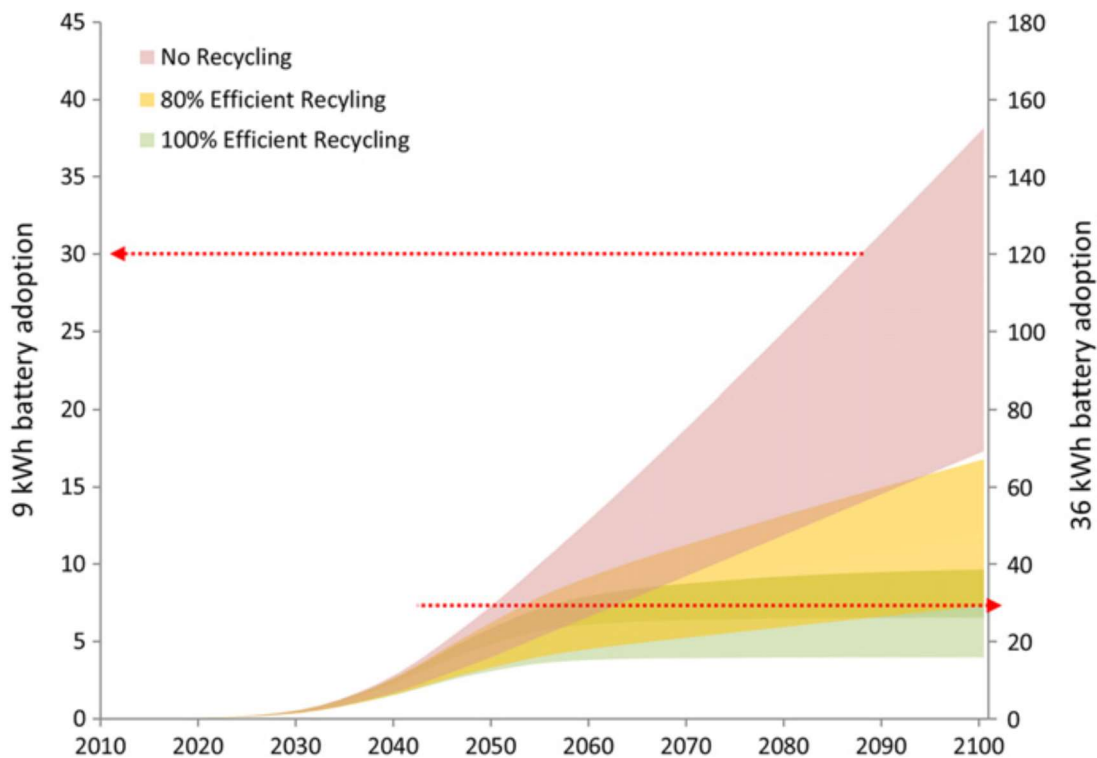
Figure 9. Long term forecast of virgin lithium demand from EV (scale Mt). Assuming ca 3 billion cars in 2100 all HEV or EV. Dashed arrows indicate assumed 30Mt Li-resources. From Kushnir and Sandén (2012).

Uses of lithium cumulated [in t Li] – 50% penetration of electric cars in 2050
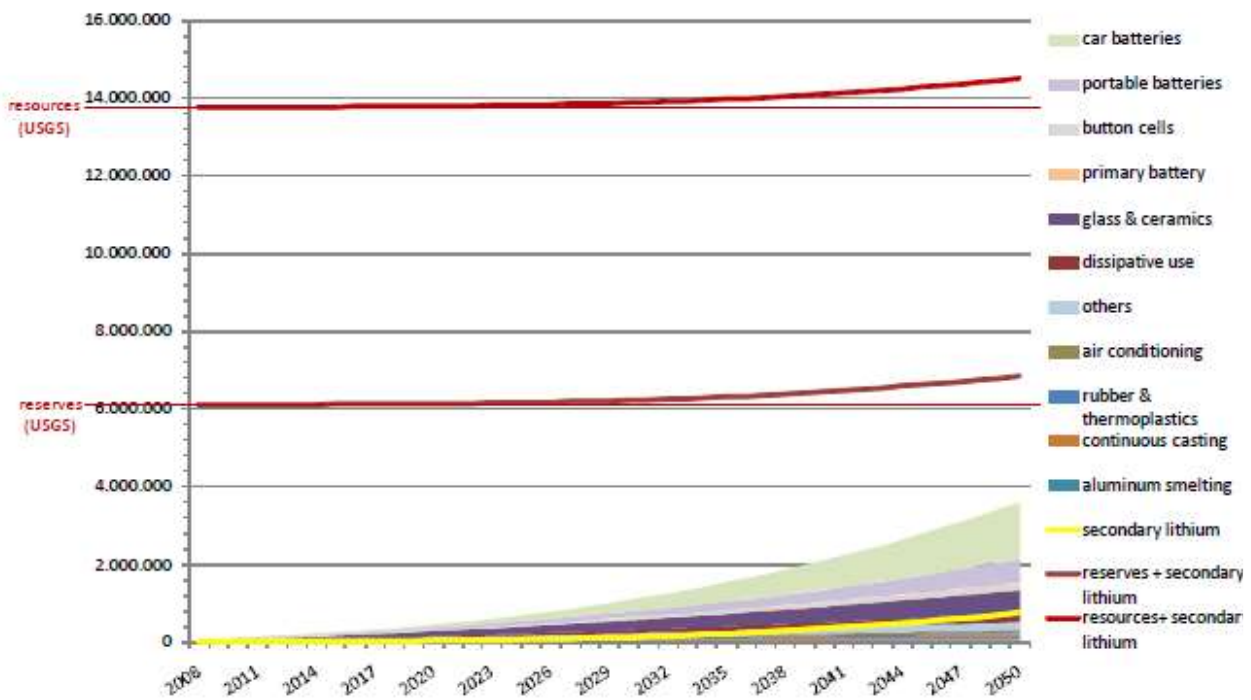
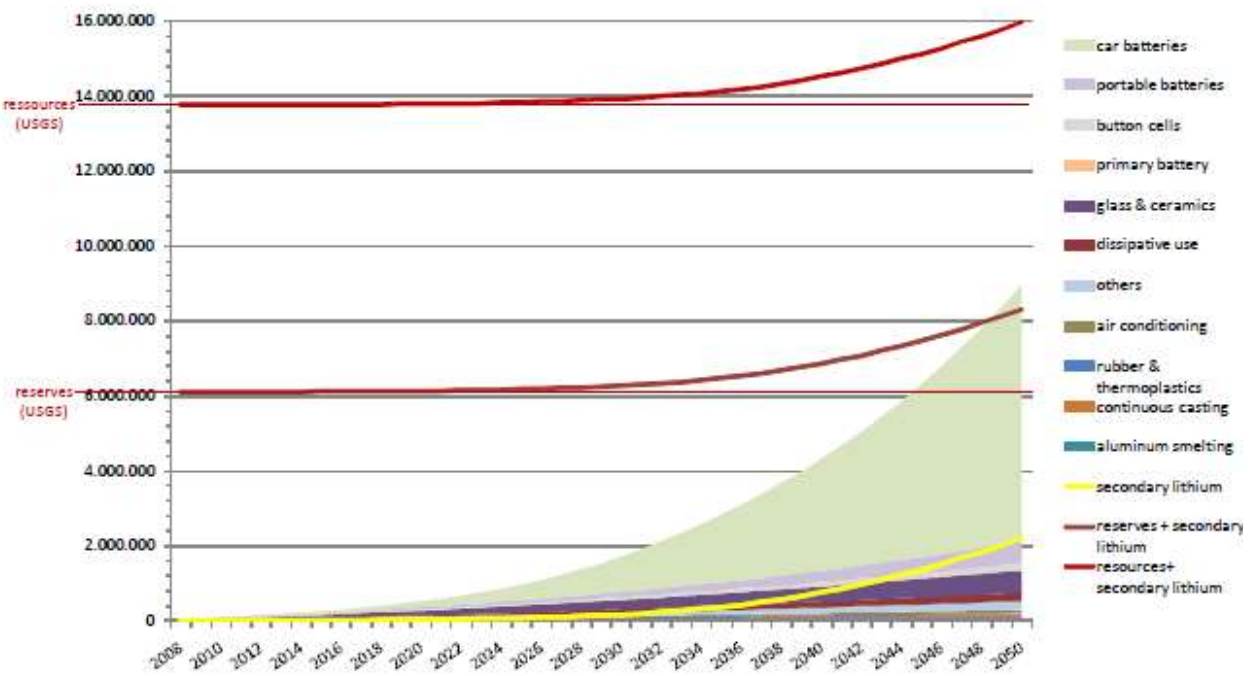**Uses of lithium cumulated [in t Li] – 85% penetration of electric cars in 2050**

Figure 10. Forecast to 2050 on Li demand resources and reserves. From EC (2010).

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Life cycle assessment

Characterisation factors for metals in LIBs

There are several different life cycle impact assessment (LCIA) methods available, each with their own coverage and definitions of environmental impacts and resource depletion, and how to calculate the impacts. All commonly used LCIA methods are however equal in that they have a set of characterisation factors (CF) to calculate environmental impacts. A CF is applied as a linear multiplier of quantified flows of use of natural resources and emissions (in case of metals given as mass flows in kg) to and from the technical system being assessed. This yields quantified impact scores according to environmental indicators as defined by the specific LCIA method.

For most flows of elemental metals including the LIB metals the mainstream LCIA methods provide CFs only for impacts on resource depletion and human and ecological toxicity. Other environmental problems are considered to have negligible impacts from elementary metal flows.

Figure 11 shows diagrams of CFs for the impact category resource depletion from a selection of commonly used LCIA methods. The figure should be read as an indication of the severity of resource depletion per kg of metal extracted from nature, as assessed by commonly used LCIA methods. The comparison of CFs shows that extraction of tin is consistently given high resource depletion scores in different LCIA methods. For lithium no CF is given in most methods this may be due to the high degree of uncertainty of future demand of lithium or as in the case of the EPS method (Steen 2015) a comparably low value is assigned to depletion of Li because a very long time perspective is adopted and extraction from ocean resources is assumed to be available.
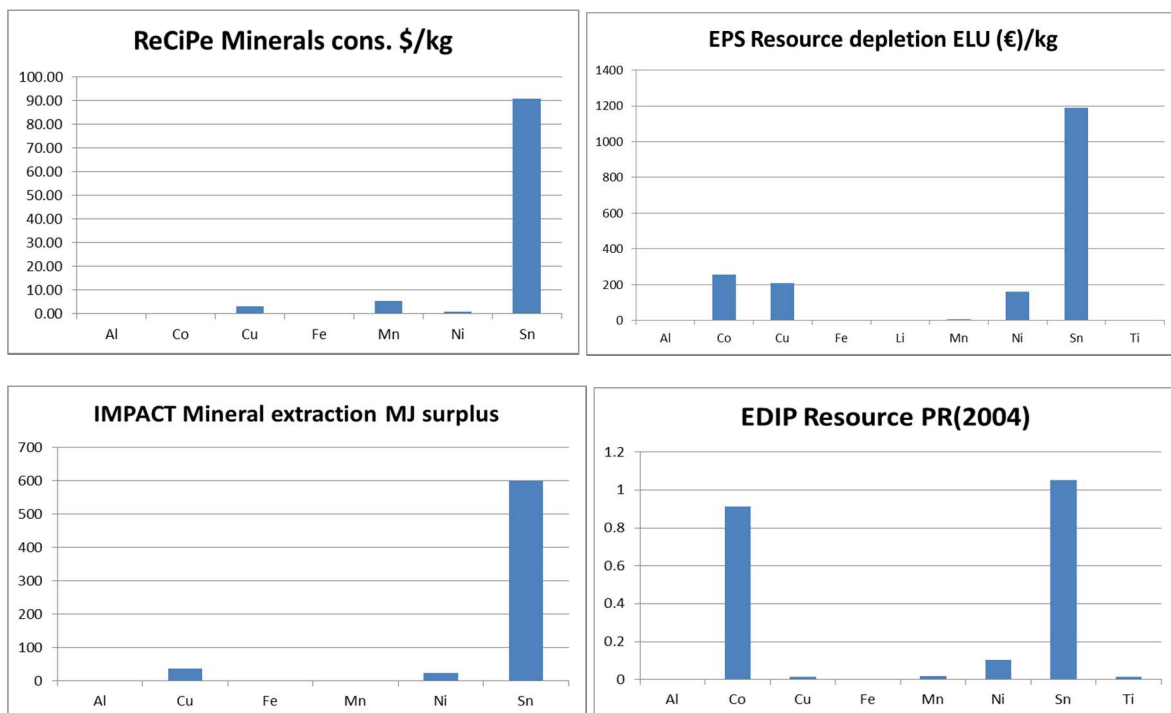

Figure 11. Comparison of characterisation factors for impact on resource depletion from four LCIA methods:

- ReCiPe "Minerals consumption" unit: USD/kg extracted material. (Goedkoop et al. 2013)
- EPS "Resource depletion" unit: ELU/kg (ELU – environmental load unit comparable to 1€) (Steen 2015)
- IMPACT "Mineral extraction" unit: MJ/kg (Jolliet et al. 2003)
- EDIP "Resources" unit: PR2004 (Hauschild 2005)

Metals are not only a resource issue, many of them are also toxic when emitted to the environment, which is an aspect considered in LCA. Figures 12-15 show the CFs for toxicity impact categories from the commonly used LCIA methods ReCiPe (Goedkoop et al. 2013) and CML (Guinée et al. 2002). The impact category human tox indicates toxic impact on humans whereas ecotox indicates toxic impact in the ecosystem. The figures should be read as an indication of the severity of toxic impact per kg of metal if emitted to the environment. As can be seen Co, Cu, and Ni are characterized as problematic relative to other metals in both methods. Mn stands out as having the largest human toxicity impact by ReCiPe while data is missing in CML.
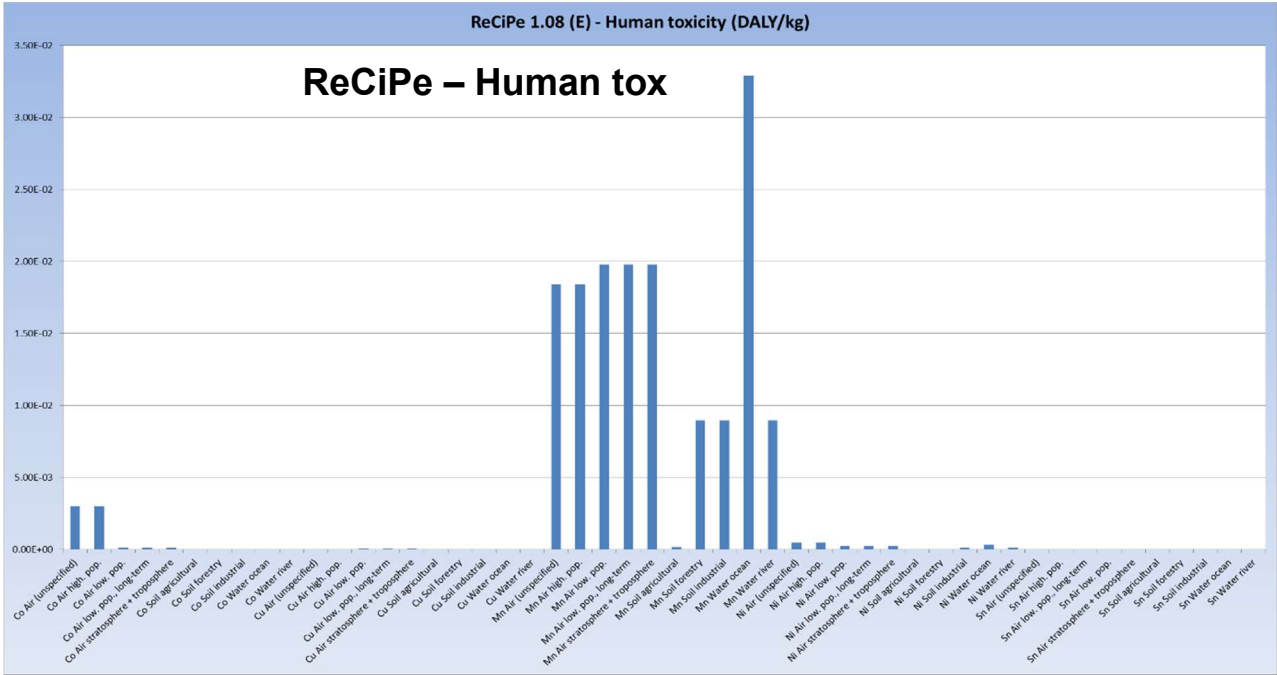
Figure 12. Impact on human toxicological effects per kg of the pure elemental metal emitted into different environmental compartments. Based on ReCiPe 1.08 (Goedkoop et al. 2013).
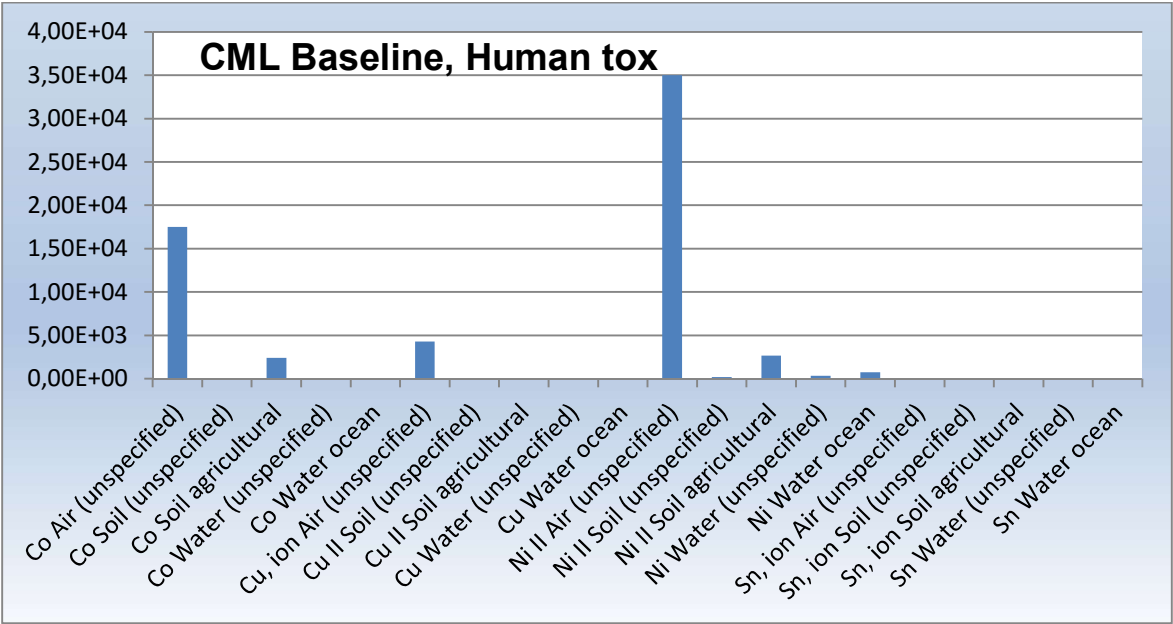

Figure 13. Impact on human toxicological effects per kg of the pure elemental metal emitted into different environmental compartments. Unit 1,4 DB-eq/kg. Based on CML 2002 (Guinée et al. 2002).
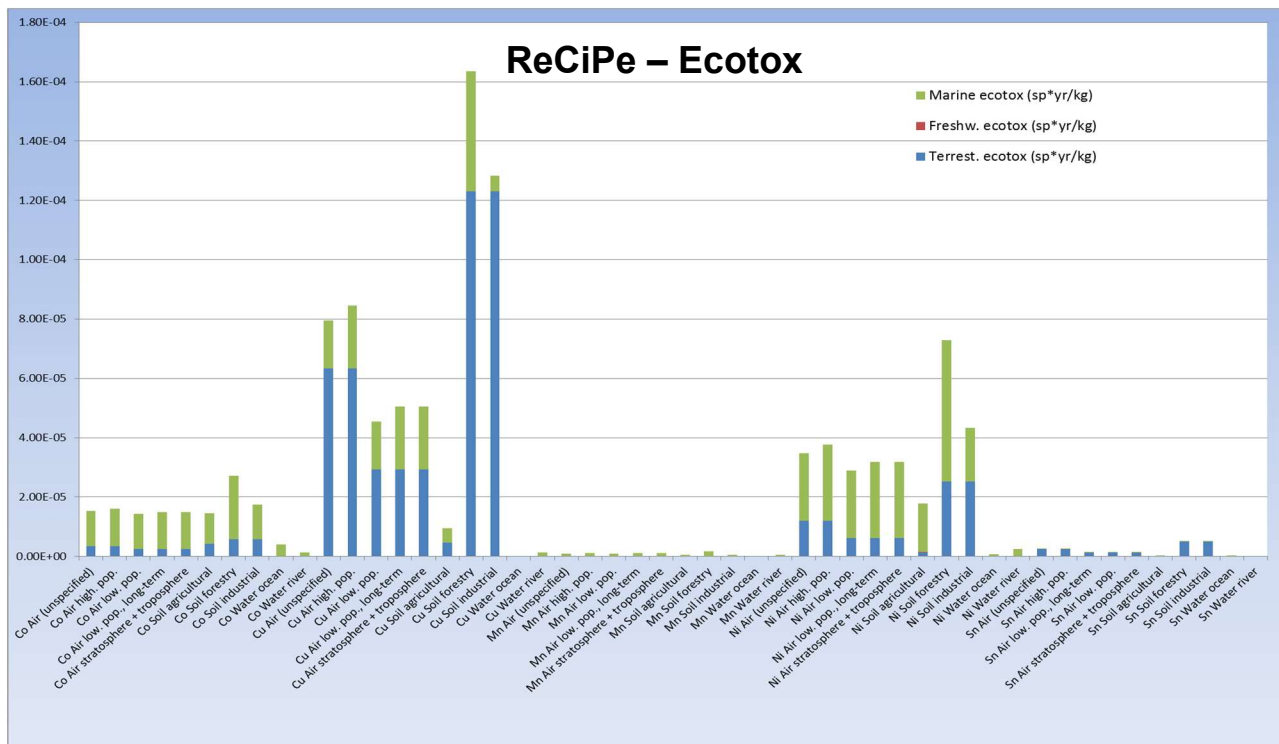
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Figure 14. Impact on ecotoxicological effects per kg of the pure elemental metal emitted into different environmental compartments. Unit: affected species*year/kg Based on ReCiPe 1.08 (Goedkoop et al. 2013)
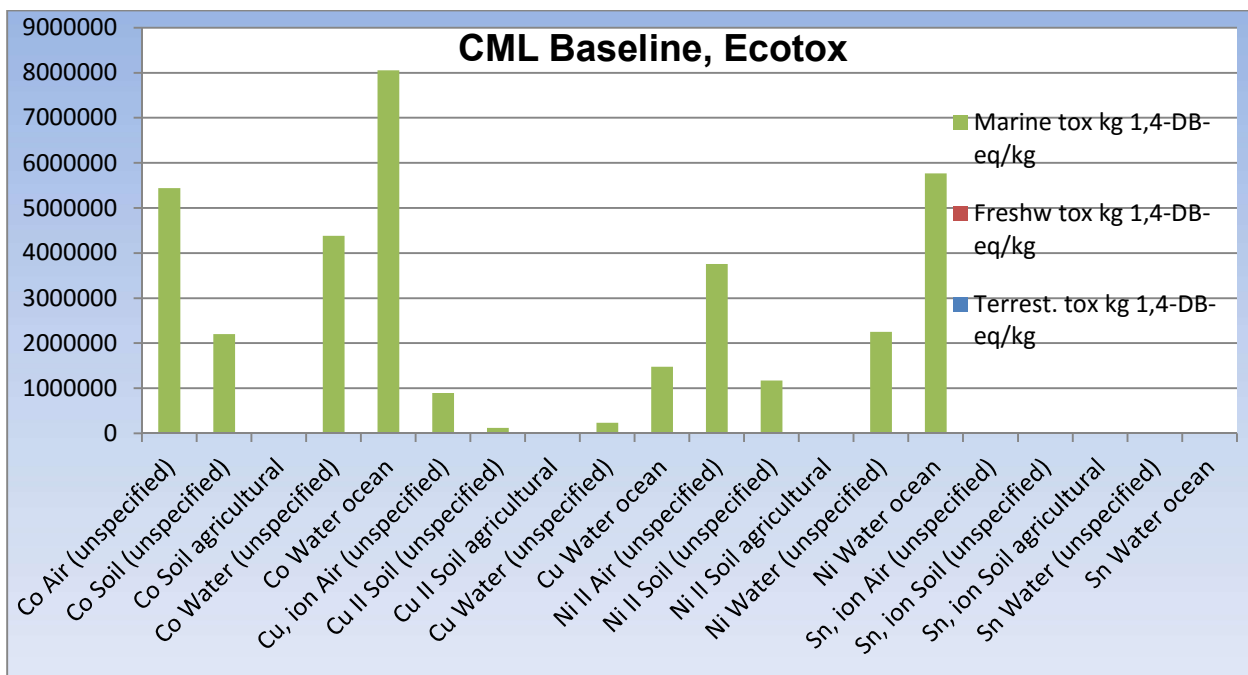

Figure 15. Impact on human toxicological effects per kg of the pure elemental metal emitted into different environmental compartments. Unit: 1,4-dichlorobenzene-eq/kg. Based on CML 2002 (Guinée et al. 2002).

Within LCA there are also methods available which puts different kinds of environmental impact on a common scale through weighting. Figure 16 show impacts per kg metal weighted to a common unit (dimensionless points) according to one such method, ReCiPe, as an indication of overall environmental impact, including both toxicity and resource depletion.
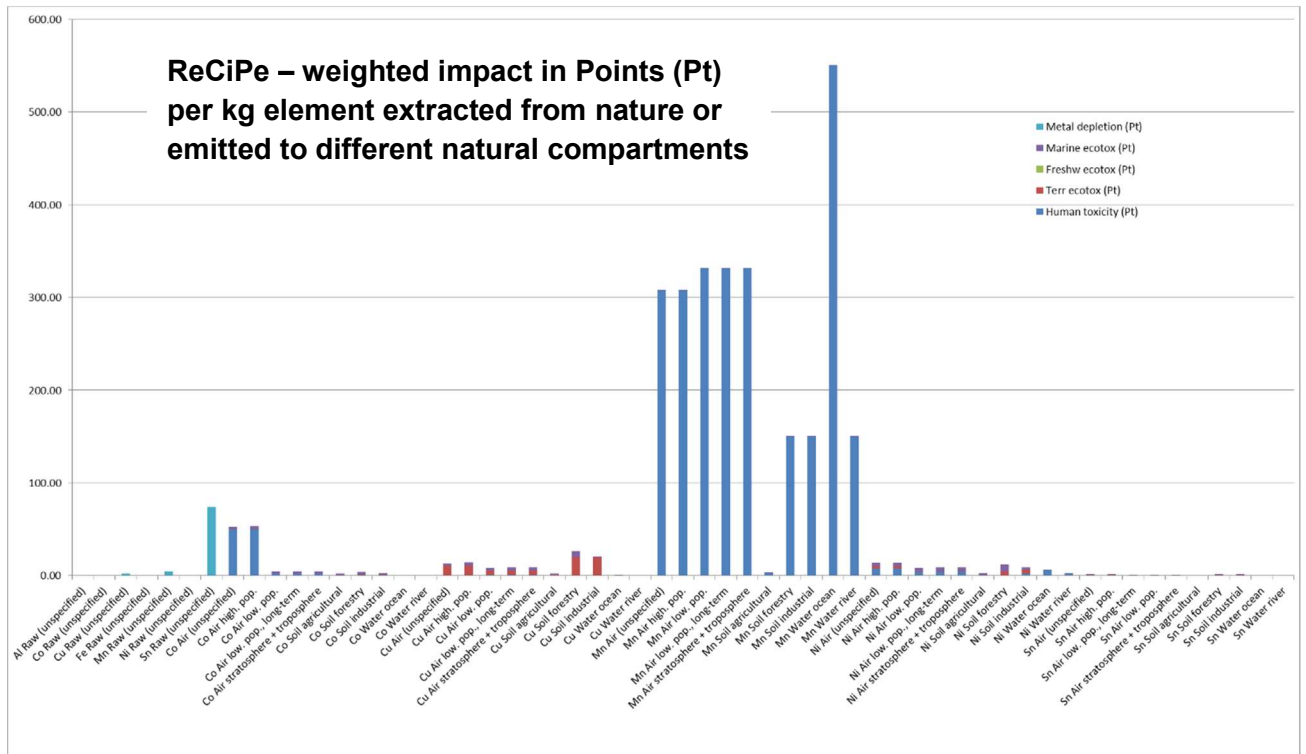
Figure 16. Weighted impact on resource depletion and toxicological effects per kg of the pure elemental metal emitted into different environmental compartments. Unit: Points (dimensionless) / kg. Based on ReCiPe 1.08 (Goedkoop et al. 2013)
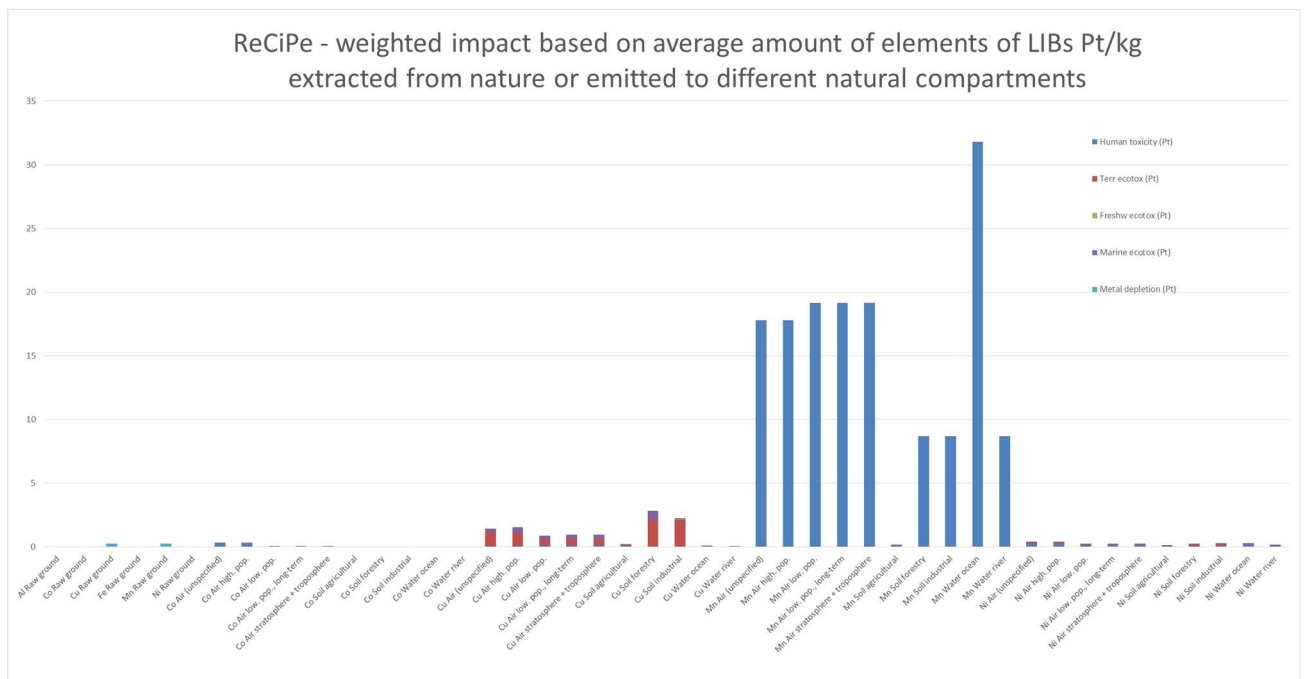

Figure 17. Weighted impact based on average amount of elements in LIBs according to table 1 according to ReCiPe 1.08. Unit: Points (dimensionless) / kg. Not all elements shown due to incomplete data.

In figure 17 the weighted impact factors per kg element (fig. 16) have been multiplied with the average composition of elements in LIBs (table 1). This gives an indication of total potential impacts of the element in batteries, although it rests in the unrealistic scenario that all battery material is extracted from nature and eventually emitted into different natural compartments. The result shows

that according to the ReCiPe impact method toxicity may be a bigger concern than resource depletion, in particular toxic impact on humans (again, under the unrealistic assumption all metals in used batteries are released to the environment). The result also show that the inclusion of Mn in LIBs has the highest potential total impacts compared to other elements in LIBs. Potential Cu emissions from LIBs also have a significant impact. Note that the emission impact values are not additive per element; e.g. if all Mn in LIBs was emitted to ocean water the total weighted impact from Mn emissions is ca 32 Points/kg and thus zero from Mn emission into other compartments (since there is no more Mn that could be emitted into other compartments). Note also that data for Li and Sn is missing as either impact factors or data on battery content is missing.

Taken together, the available data of characterisation factors show consistently that raw material extraction of tin should be avoided. Extraction of cobalt, nickel, copper, and manganese is also problematic according to more than one method. Emissions of elemental manganese and cobalt and nickel to air have potential human toxic impacts. Manganese, cobalt, copper and nickel have potential ecotoxicological impacts.

Life cycle assessment of Li-ion batteries and electric cars

The growth of electric vehicle (EV) technologies has spurred several LCA studies. Some studies are focussed on the function of a LIB alone but mostly LIBs are assessed in an EV use context. LCA of vehicles can be schematically described as the equipment life cycle, combined with the energy supply life cycle, see figure 18. The equipment life cycle includes the production, use, and end-of life of the actual car including the battery. The energy supply chain is often called well-to-wheel (WTW) originating from extracting fuel from "wells" for conventional internal combustion engine vehicles (ICEV). This in turn is divided into well-to-tank (WTT) to deliver the energy to the car, and tank-to-wheel (TTW) to utilize the energy for propulsion. For EVs the well-to-tank correspond to electricity production and battery charging and well to tank correspond to discharging.
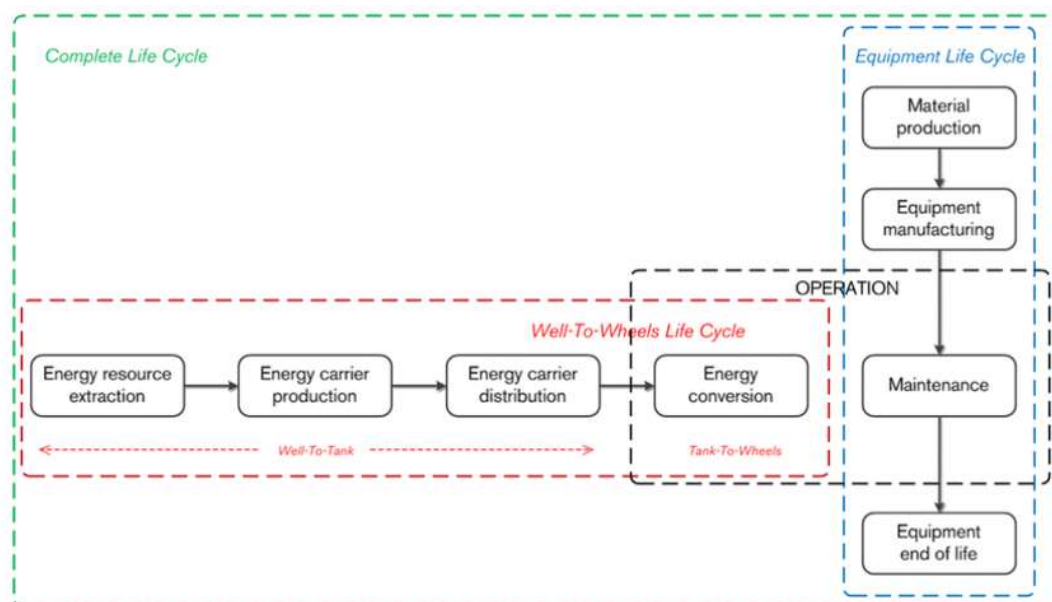

Figure 18. Overview of main life cycle processes related to vehicles. Recycling options excluded. From Nordelöf (2014).

Well to wheel lifecycle

Most LCAs on vehicles focus on energy use and GHG emissions. A general conclusion is that the WTW life cycle contributes the most to overall impact for all ICEV and hybrid-EV. For ICEV as well

as semi electrified cars (hybrid and plug-in hybrid) the TTW phase dominates, i.e. fuel combustion in the vehicle. For BEV it is the WTT phase i.e. the electricity production that heavily influences the environmental performance of the EV (Nordelöf 2014). If the electricity is produced from fossil fuels such as coal, oil or natural gas the overall impact is often greater than from conventional ICEV. On the other hand if the electricity is produced from low CO_2-emission technologies including renewables and nuclear the emissions are substantially lower. Figure 19 shows in blue well-to-wheel GHG emissions for battery EVs (BEV) with different types of electricity production (Nordelöf 2014, Swedish Energy Agency 2016, Vattenfall 2016). For reference, displayed in red are EC legislation on emission limits for newly manufactured cars 2015 and 2021 (ICCT 2014) as well as Swedish legislation (Vägtrafikskattelag SFS 2006:227) emission limit to qualify as 'environmental car' (in Swedish: 'miljöbil') for a 2015 average passenger car weight of 1491 kg (Trafikanalys 2016). A value of 13 g CO_2-eq have been added to the references to account for the WTT as the legislation only considers TTW (Nordelöf 2014).
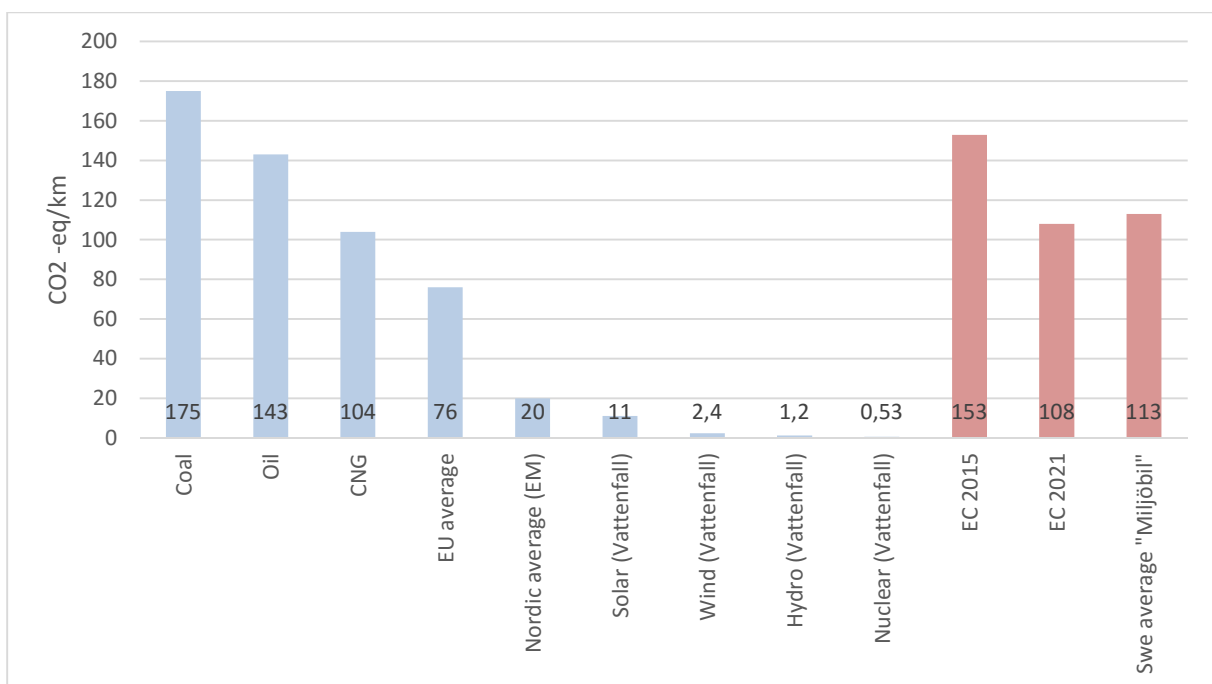

Figure 19. BLUE: Well to wheel GHG emissions for a mix of BEV for different types of electricity production. (Based on Nordelöf 2014, Swedish Energy Agency 2016, Vattenfall 2016). RED: EC legislation emission limit for new cars 2015 and 2021 (ICCT 2014), and Swedish classification of "Environmental car" ("Miljöbil") (Vägtrafikskattelag SFS 2006:227) for 2015 average weight of cars of 1491 kg (Trafikanalys 2016).

Equipment life cycle

The equipment life cycle is excluded from the comparison in figure 19. Including the equipment life cycle shows how the impact of the manufacturing stages increase with degree of electrification from hybrid, plug-in hybrid to pure BEV (Nordelöf 2014). Also, better performing electricity production leads to that the share of the impact that comes from the equipment increases.

As study by Volkswagen (2012) of their BEV provides data for the upstream equipment life-cycle, see figure 20. The total GHG emissions per km to manufacture is 74 g CO_2-eq/km; approximately the same as the WTW emissions of 76 g CO_2-eq/km for an average BEV running on average EU electricity (fig. 19).

Vehicle manufacturing
74 g CO$_2$/km

Battery production
33 g CO$_2$/km

Basic vehicle and electric
components 41 g CO$_2$/km

Battery cells
27 g CO$_2$/km

Peripherals
6 g CO$_2$/km

Battery-related systems account for a large proportion of overall vehicle CO$_2$ emissions at the manufacturing stage. So we need to focus on the battery cells and the raw materials and processes that go into producing them. The associated emissions are largely due to the use of materials such as cobalt and copper, where the extraction process is highly energy- and CO$_2$-intensive.

Figure 20. Life cycle GHG emissions to manufacture a BEV. From Volkswagen (2012).

A significant part of the equipment emissions, 33 g CO$_2$-eq/km, is due to manufacturing of the LIB. Specifically it is the extraction of elements and preparation, including grinding into very fine particles, of the cathode material that is very energy intensive (Dunn et al. 2012, Kushnir and Sandén 2011).

Battery life cycle including recycling

When considering recycling of LIBs it is in a life cycle perspective not only the yield of elements from recycling processes that matters, but also what species result from the recycling process. This is due to that much of the energy needed to produce batteries are used to refine the active materials. Dunn et al. (2012) made a comparative LCA of three different recycling technologies for automotive LIB: hydrometallurgical resulting in lithium as a salt; a commercial intermediate physical recycling process resulting in Li$_2$CO$_3$; and a direct physical process (under development) that recovers LiMn$_2$O$_4$, i.e. the active material is conserved. LCA results, as illustrated in figure 21, show that the production of LiMn$_2$O$_4$ via the hydrometallurgical process is nearly as energy intensive as virgin production while the direct recycling only requires approx. 25% energy. They also estimated that recycling the active cathode component, aluminium and copper, reduce the total cradle to gate energy requirement to produce a battery by 48% compared to virgin production. Several studies show similar significant differences in energy demand and environmental impact depending on assumptions about battery recycling technology and capacity (Nordelöf 2014 and references therein). While such results points to a large potential for energy and environmental savings from recycling, they rest on assumptions as highly efficient recycling processes for LIBs are not yet in commercial practice.
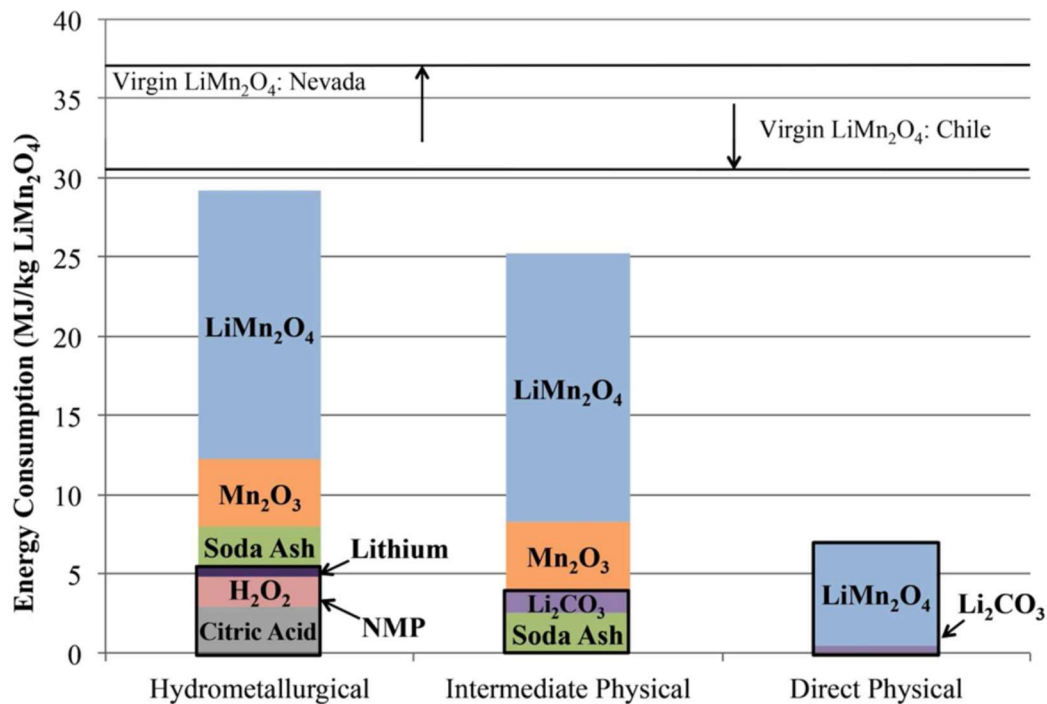
Figure 21. Life cycle energy consumption to manufacture LiMn$_2$O$_4$ via three different recycling technologies (from Dunn et al. 2012).

Other impact categories

As mentioned most LCAs on LIB applications focus primarily on energy consumption coupled with GHG emissions. An exception is Kang et al. (2013) who looked at other effects of the direct extraction of elements and emissions from smartphone batteries. Results point out cobalt and copper as contributing the most to resource depletion and potential toxic effects from leakage from mine tailings and spent batteries in landfills, which is fairly consistent with the above analysis of life cycle characterisation factors.

The potential enlarged toxic effects from an increased use of batteries and other electric components have raised concerns. A comparison by Messagie (2013), show a nearly double human toxicity potential from a BEV compared to two conventional ICEVs. The production of LIBs alone contribute almost as much to the toxic impact as the complete propulsion system of the assessed ICEV. The toxic emissions are mostly connected to mining processes (Nordelöf 2014), see figure 22.
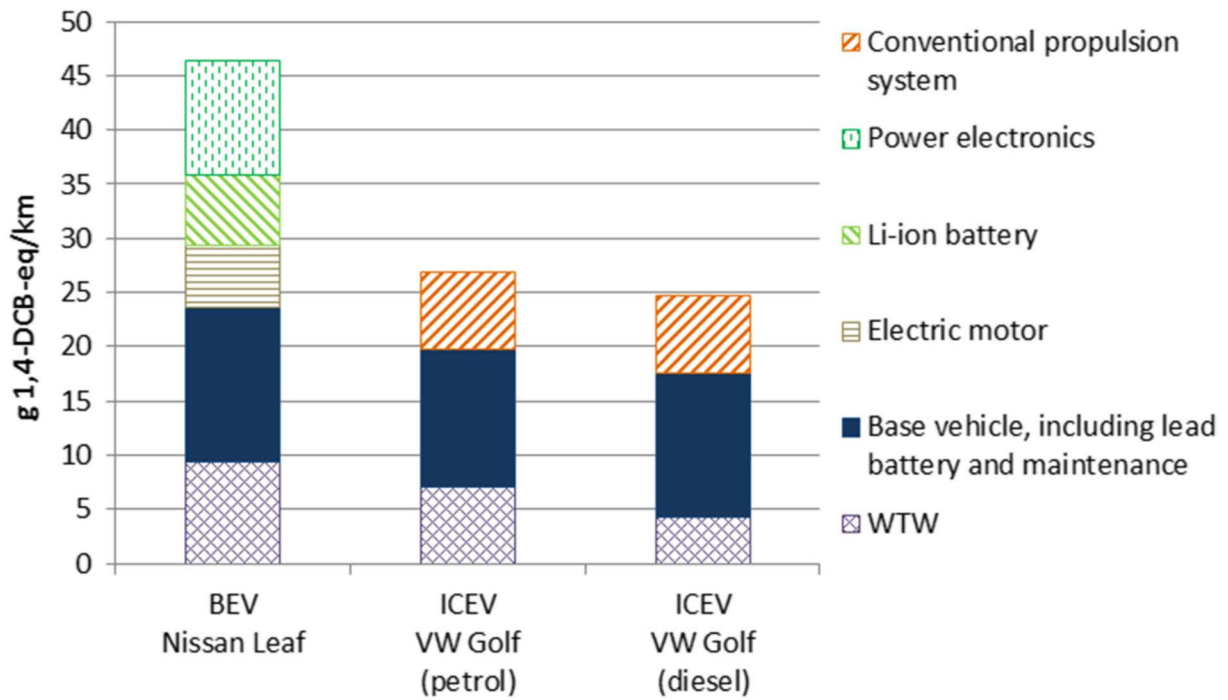
Figure 22. Human toxicity potential for three passenger cars (From Nordelöf 2014, Messagie 2013).

Contribution to acidification and eutrophication, while only rarely quantified in LCAs for LIB and EV, is, similarly to GHGs, highly correlated to combustion of fuel either in the vehicle or in fuel based electricity production plants.

Discussion and conclusions

A combined evaluation of criticality studies, supply and demand prognoses, and LCA studies point out the potential future supply risk of lithium. In order to meet future Li demands, largely driven by an expected rapid growth of EV production, Li production capacity must increase and the limited reserves implies recycling of Li must increase in a medium (decadal) time perspective. Li is currently not recycled to any significant extent as there is little commercial value of lithium species resulting from current recycling practices.

Cobalt is identified as a critical element. Total Co resources and reserves are limited and a large part of total Co demand is projected to come from batteries. Recycling processes for Co are however in commercial use.

The production of batteries and especially the manufacturing of active material matrices are generally pointed out as high contributor to environmental impact in battery life cycles. In LCAs of EVs LIB production contributes significantly to the total impact per km accounting both for WTW and equipment life cycle of the vehicle. Assumptions about electricity production as well as recycling scenario has a large impact on results regarding the overall performance of the EV per km.

The refinement of elements including Li to active anode and cathode material is energy intensive and expensive. Hence, energy savings and corresponding avoided environmental burden significantly depend on in what form and at what stage the recycled metal products are fed back in the production of new batteries.

Assessment of impact factors for extraction and emission of elements from LCIA methods combined with average elemental composition of LIBs indicate that high potential impact from emissions of Mn and Cu. This suggests that efforts to avoid Mn and Cu emissions from end-of-life processes including recycling should be prioritized.

The rapid development of LIB technology makes it difficult to assess. Scrosati and Garche (2010) conclude that the pace of LIB development is so fast that efforts to evaluate the progress may easily become obsolete. Recycling technologies are also under development. Much ongoing development of technologies, policies and markets, scarce and often inconclusive data makes it difficult to predict the future.

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