Comparative Social Life Cycle Assessment between Battery Electric Vehicles and Internal Combustion Engine Vehicles

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Abstract

Battery electric vehicles (BEV) as opposed to internal combustion engine vehicles (ICEV) are seen as a viable solution for reducing transportation related environmental impacts. There are however advantages and disadvantages to both alternatives. Environmental life cycle assessments (LCA) are used to quantify the environmental lifecycle impact of these vehicles. The social impact for people in the value chain of these alternatives has not yet been tested. For this purpose, a social lifecycle assessment (S-LCA) can be conducted. This paper compares BEVs and ICEVs in a S-LCA according to the UNEP/SETAC guidelines, using the Product Social Impact Life Cycle Assessment database (PSILCA). It was found that overall, the ICEV seems to have lower levels of social risk related to the life cycle than the BEV has. These differences are smaller or opposite when the lifetime of both vehicles is assumed to be longer. The raw material extraction seems to be a hotspot for social risks, especially the extraction of cobalt in the Democratic Republic of Congo has high risk levels. The paper includes the assessment of a future end of life scenario where BEV batteries are recycled. Recycling could reduce the amount of social risk related to the BEV. The utility of the PSILCA for this case and the general cost sensitivity of this method was discussed. It was concluded that while the PSILCA is useful to provide insights in social risks related to a products lifecycle and uncovering hotspots, it is not very suitable to compare product systems where a high number of assumptions are made. The results of this study should therefore not be used for generalisations.

Keywords: Social Life Cycle Assessment (S-LCA); PSILCA; Battery Electric Vehicle (BEV); Internal Combustion Engine Vehicle (ICEV); social risk, industrial ecology

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1. Introduction

Battery electric vehicles (BEV's) are seen as one of the primary solutions to solve the problem of traffic related emissions, such as carbon dioxide and nitrogen oxides (Andwari et al., 2017). The main advantage of BEV's is the fact that they have zero tailpipe emissions. The main disadvantages, however, can be found in the battery (Steward, Mayyas, & Mann, 2019), which is related to high levels of human toxicity and emissions related to the production process (Verma, Dwivedi, & Verma, 2021). Batteries also are expensive, take a long time to charge, and are hard to recycle (Dai et al., 2019). Especially lithium and cobalt, two of the main components of electric vehicle (EV) batteries, are thought to have a big environmental impact (Dolganova et al., 2020), mainly due to the mining processes involved. A large amount of research already goes into finding environmental hotspots in the production of batteries and its necessary resources (Dai et al., 2019; Majeau-Bettez, Hawkins, & Strømman, 2011; Verma, Dwivedi, & Verma, 2021). However, the social factors related to BEV production, even though they are mentioned in many studies, are often not quantified.

A recent study (Omahne at al., 2021) has found that most papers related to EV's focus on at least one of the Sustainable Development Goals (SDG;UN General Assembly, 2015). SDG 11 and 13, which are both related to environmental factors are the goals most commonly included in the studies. On the other hand, SDG 3, Good health wellbeing, 5, Gender equality, and 6, Clean water and sanitation, are all related to (social)wellbeing and healthy conditions. These SDG's are rarely included in studies. These goals might be especially interesting to include, since the production of EV batteries has been linked to detrimental social effects related to water use and worker health and wellbeing (Omahne et al., 2021).

The social performance of a product or service, as opposed to the environmental performance, can be assessed by conducting a social life cycle assessment (S-LCA). Guidelines for conducting an S-LCA were first presented in 2009 and were later refined and expanded by a United Nations Environment Programme/Society of Environmental Toxicology and Chemistry (“UNEP/SETAC”) workgroup (Benoît et al., 2009: UNEP, 2020). An S-LCA will show how the alternatives perform or how an organisation performs on social indicators and can assess the amount of risk on several different social aspects related to the life cycle of a product, such as the risk of child labour involved in the product lifecycle. For example, an S-LCA was performed on the supply chain of lithium-ion batteries (LiB; Thies et al., 2019); This study showed that the production of LiB’s involves a considerable risk of child labour, as well as risks of occupational toxics and hazards. The study further showed that these risks were considerably lower for production in Germany as compared to China. From this study, we can thus see which social factors play a role, and where potential hotspots for these risks are. In this case, it shows that batteries as energy storage system are not without their drawbacks, but that risks could be reduced by using production in Germany.

Still, batteries are an essential component in EV’s and since the EV market has been projected to grow considerably over the coming years (Gersdorf et al., 2020) it is important to have a complete view of the impacts related to BEV’s. Hotspots for social issues in the value chain could be identified which could help mitigate social issues that arise during the transition to more sustainable means of transport. It is therefore also interesting to see how BEV’s compare to the currently more conventional internal combustion engine vehicles (ICEV’s), since this could help legislators in making more socially sustainable decisions.

While the comparison between BEV’s and ICEV’s has been made regarding the environmental life cycle impact of these vehicles (Verma et al., 2021), a comparison of the social risks involved in these two alternatives is currently not available. Therefore, this study will compare ICEV’s and BEV’s in an S-LCA. The results of this study can contribute to a more complete insight in the pros and cons of electrical vehicles compared to internal combustion engine vehicles and can help reduce the social risks involved in the vehicle production value chain.

To assess the social risks related to the ICEV and BEV production, use, and end-of life, an S-LCA will be conducted using the Product Social Impact Life Cycle Assessment database (PSILCA; Eisfeldt & Ciroth, 2016; 2017). The functional unit (FU) of this S-LCA will be 150.000 km driven by car. The PSILCA database, with an initial introduction in 2016 is relatively new, but has already seen limited use in the context of a comparative S-LCA. The PSILCA was for example used to assess the social risks related to the production of rare earth magnets (Werker et al., 2019). This study however, only compared different production locations, and not different products. Different products have been compared using the PSILCA. For instance, vanadium redox flow batteries were compared with lithium-ion batteries (Koese et al., 2022). In this study interesting insights into potential social hotspots related to the production and use of the different types of batteries were provided, but the end-of-life phase of the alternatives was not taken into consideration. The current study will try to fill this research gap by including the EOL in a near future scenario. Additionally, it was mentioned that the assessment of emerging technologies, such as the recycling of batteries, might prove difficult when using the PSILCA. Therefore, this study will also discuss the applicability of the PSILCA for the purpose of comparative S-LCA's on complex systems, including emergent technologies, like a vehicle lifecycle, and try to identify potential upsides or downsides related to the PSILCA.

2. Methods

The PSILCA database (Eisfeldt & Ciroth, 2016; 2017) used in the current study builds on the Eora Multi-Regional Input/Output database (Lenzen et al., 2013). The Eora contains data of over 15.000 sectors in 189 countries. In the PSILCA, these sectors are harmonised for all countries, so processes can more easily be compared between regions. The PSILCA contains

88 social indicators, such as ‘Living wage per month’ and ‘Violations of mandatory health and safety standards.’ These indicators can be summarised in 25 groups of social-, and socio-economic subcategories, like ‘Respect of Indigenous Rights’ and ‘Corruption.’ The indicator subcategories cover five different stakeholder categories. The stakeholders are workers, local community, society, consumers, and value chain actors. After the life cycle inventory has been established of a certain product, the social life cycle impacts can be calculated. This is done by quantifying the flows in the inventory into so called ‘medium risk hours’. These indicator results can then be translated into a risk level based on a scoring table that is different for every indicator. There are six increasing risk levels, from ‘no risk’ to ‘very high risk.’ In Table 1, which is taken from the PSILCA v3 manual, the Child labour risk assessment can be seen. This indicator value (y) is based on the percentage of children involved in economic activity for at least one hour in the reference week of the survey.

Table 1: Child labour risk assessment.

Indicator value y, %	Risk level
0	no risk
$0 < y < 2.5$	very low risk
$2.5 \leq y < 5$	low risk
$5 \leq y < 10$	medium risk
$10 \leq y < 20$	high risk
$20 \leq y$	very high risk
-	no data

This indicator risk level can then be used to identify and describe possible hotspots for social risks in the product life cycle, that can consecutively be targeted for improvement. The analysis can also show the differences between regions for certain processes. So, the PSILCA might be a very suitable database to use for comparative S-LCA's.

LCA's are usually conducted according to ISO standards 14040 and 14044 on LCA methodology (ISO, 2014). These standards are not fully applicable to the S-LCA methodology; therefore, this paper will follow the guidelines set by the UNEP/SETAC (UNEP, 2020). This means that the S-LCA will be conducted following four steps: 1) Definition of Goal and Scope, 2) Life Cycle Inventory Assessment, 3) Life Cycle Impact Assessment, 4) Life Cycle Interpretation. To assess the utility of the PSILCA, the results from the S-LCA will be compared to existing literature, and the results of the sensitivity analyses will be further interpreted.

2.1 Definition of Goal and Scope

The goal of this S-LCA is to compare the social life cycle impacts of BEV's to ICEV's. The scope of this study will entail the production, the use, and the end-of-life phase of both vehicles. The EOL phase will be assessed in a near future, scenario. This is because there is still very limited knowledge on the large-scale recycling of LiB's (Beaudet et al., 2020; Wang et al., 2020). In the scenario that will be assessed it is assumed that 60 percent of the vehicles is recycled, and that there is sufficient capacity for the large-scale recycling of vehicle batteries. The battery in this scenario is recycled through a hydrometallurgy process. The FU in this study is mainly determined by the expected lifetime of the alternatives. A vehicle lifetime of 150.000 km is assumed for both alternatives in this study, which is commonly assumed in other studies regarding this subject (Hawkins et al., 2013; Tagliaferri et al., 2016). The assumed lifetime however, is an interesting subject for a sensitivity analysis, since it has been shown in an environmental LCA, that the expected lifetime can be an influential factor when comparing BEV's and ICEV's (Majeau-Bettez et al., 2011). Thus, the FU of this S-LCA will be 150.000 driven kilometres. The two alternatives are then 150.000 kilometres driven by BEV, and

150.000 kilometres driven by ICEV. For the further interpretation, several sensitivity analyses will be conducted.

2.2 System Flowcharts

In the following two figures the product system for both alternatives are presented. The different components, as well as the sectors in the PSILCA that provide the worker hours related to those components are shown. As can be seen, The EOL phase is included in the form of vehicle recycling. This process also produces recycled material, which is counted as avoided cost for the corresponding sector, as will be further explained in the life cycle inventory section.

Figure 1: ICEV System Flowchart

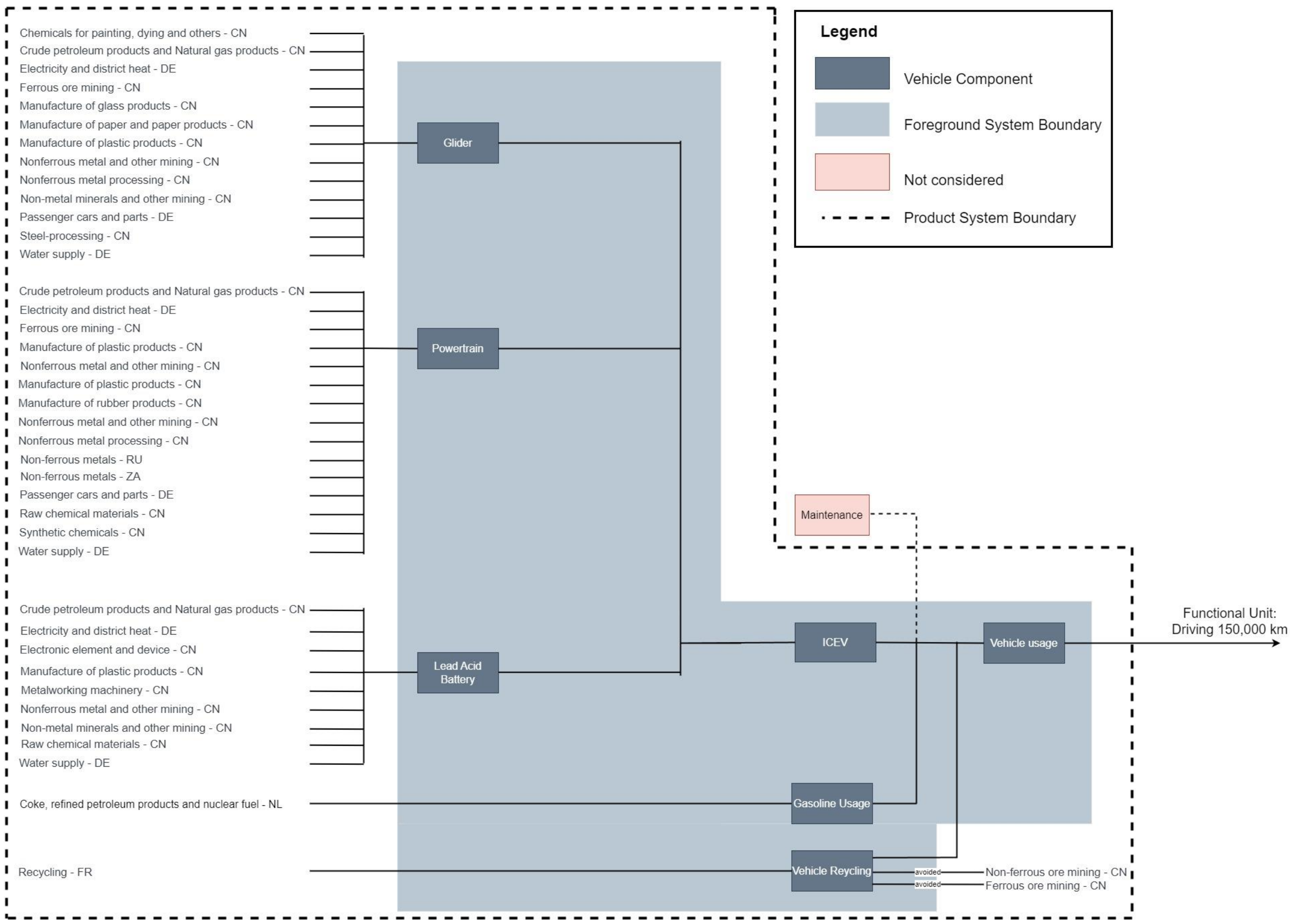
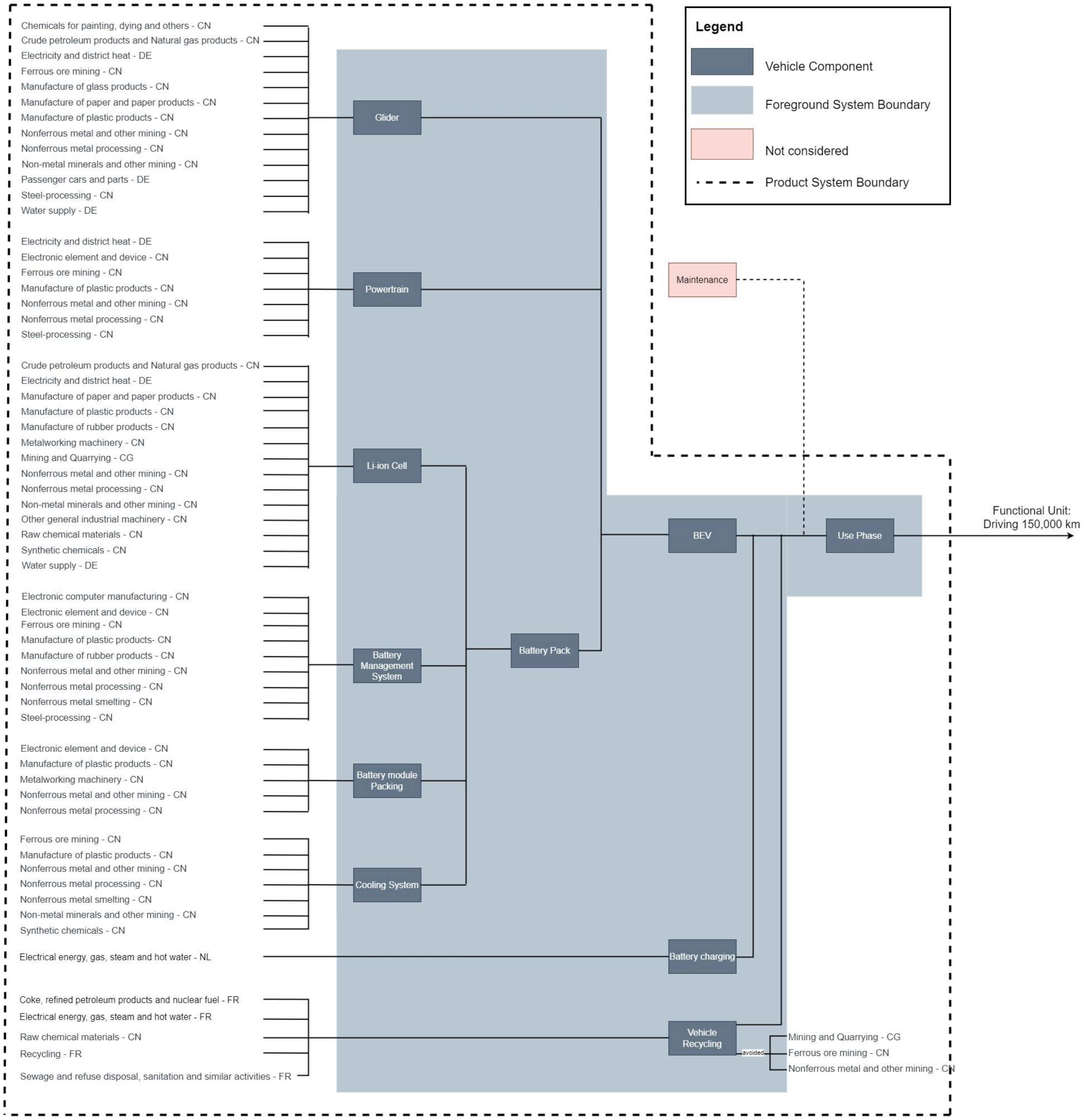


Figure 2: BEV System Flowchart



2.3 Life Cycle Inventory Assessment

For both alternatives, the life cycle inventories in this study are mainly based on data from the Ecoinvent 3.8 database (Ecoinvent, 2021). Additionally, several sources were used to determine the composition of conventional BEV battery packs (Ellingsen et al., 2013; EVANNEX, 2023; Zheng et al., 2019). The products in these inventories, that are still mainly in line with the

Ecoinvent database, need to be translated into a format that is fitting to the PSILCA. This, because the PSILCA does not have a list of products or resources, but it has a list of sectors from different regions, such as ‘non-ferrous metal industry in China’, or ‘recycling in France’. If a product system contains 1kg of aluminium, it has to be accounted for not by mass, but as money going to a corresponding sector. For example, 1 kg of wrought alloy aluminium is worth $2.98. This is then accounted for as $2.98 from the non-ferrous metal industry in China. The PSILCA automatically translates a monetary value going to a sector to worker hours in that sector. This translation is done based on a certain factor for every process. Hence, one dollar going to a process in the financial sector in the Netherlands, might be related to fewer worker hours than a dollar going to the mining sector in Chile. These worker hours are then translated into medium risk hours, again by using a factor based on the risks related to that specific process and sector. For this study, the producer of each product will be assumed to be the biggest global producer.

For the current inventory assessment, the value of every product was indexed to match the reference year that is used in the PSILCA database, which is 2015. Matching the correct value of a product or service is especially important for products with highly fluctuating prices. For example, the Ecoinvent 3.8 database mainly contains data of the reference year 2011, often even older. To correctly match this data to the PSILCA environment, St. Lous federal reserve database was used (*FRED, 2023*). This database contains indices on price fluctuations for a high number of sectors and products. For example, the Ecoinvent database lists aluminium, wrought alloy as 1.44 EUR per kg in 2005. The FRED database on non-ferrous metal prices shows that the price has increased since 2005. To correct for this, the 1.44 EUR price in 2005 will be multiplied by the same factor as the sector prices have increased so the price matches the price of the PILCA reference year. After this, the prices in Euros were converted to US dollars, since that is the designated currency of the PSILCA. This conversion was done by assuming an exchange rate of 1.05 USD/EUR. It is expected that both the indexing and the conversion to USD will add a certain level of error to the calculations, but both steps are essential, so this uncertainty must be accepted.

2.3.1 Production phase

For both alternatives, to make a fair comparison, the glider was assumed to be identical. The glider is the part of a vehicle that consists only of the rolling body and interior. So this is essentially the vehicle without the powertrain and the batteries. In this study the inventory for the glider was largely based on the Volkswagen Golf A4 (Habermacher, 2011), as a representation for a modern compact car. This was a 800kg glider which would be part of a 1200kg to 1400kg compact car. This is also in line with other literature on this topic (Naranjo et al., 2021). To this glider, for the ICEV, a gasoline ICE powertrain and a lead acid battery was added, and for the BEV, a battery and a Li-ion battery was added.

The ICE powertrain that was chosen in this study was also from a Golf A4 (Haberman, 2011). This represents a modern vehicle and is also in line with similar research as was the case with the glider (Naranjo et al., 2021). The data on the lead acid battery was taken from the Ecoinvent database as a standard 15 kg lead acid battery.

The inventory for the BEV required a few more steps. Just like the ICEV, the 800kg glider was taken as the base for this vehicle. The powertrain in this case was divided into the electric engine, and the Li-ion battery pack. The data on the electric engine was found in the Ecoinvent database and was assumed to be a 53kg unit. The battery pack was assumed to be the most widely used Tesla battery pack (EVANNEX, 2023). There is however no direct inventory available on this battery pack, so this had to be determined by combining several sources. First the composition of the battery pack needs to be determined. A battery pack consists of several parts: battery cells, a battery management system (BMS), the battery module packing, and the cooling system. The most popular Tesla battery pack consists of 16 battery modules that all contain 444 cells, for a total of 7104 Li-ion cells of the 18650 model. The Li-ion cell in this battery is an NCA cell with the following chemical composition: $LiNi_{0.8}Co_{0.15}Al_{0.05}O_2$ (Zheng et al., 2019). The total weight of this battery pack is 540 kg. The Ecoinvent database has inventories on the battery pack parts per kg so for the full inventory on the battery pack, the mass of all the individual parts needed to be determined. An LCA study on vehicle battery packs (Ellingsen et al., 2013) provided a detailed description of the battery pack composition and mass of the parts, described in a way that directly matches the Ecoinvent 3.8 database. The percentage of mass of the sub parts derived from the 2013 study was multiplied with the total mass of the Tesla battery pack to find the individual mass of the sub parts. The results from this calculation, which can be found in the table below, where also in line with the total cell mass calculated from individual cell weight. For these sub parts, an Ecoinvent entry exists, containing information on the composition of for example 1kg of battery packaging. This is then multiplied with the mass of the respective parts to get a detailed inventory of the full battery. The full inventory of the vehicles can be found in the appendix.

Table 2. Battery pack composition

Part	**Mass percentage**	**Part mass (kg)**
Battery packaging	32.1	173.34
BMS	3.7	19.98
Cooling system	4.1	22.14
Cells	60.1	324.54
Total	100	540

The comparison between these two vehicles should be fair because of the glider being identical. Additionally, the selection of the respective drivetrains should ensure that this study is both representative, and uses more robust data, since both drivetrains are highly popular and thoroughly tested.

2.3.2 Use phase

The use phase in this study was simplified into solely the energy usage of the vehicles. Maintenance was excluded from this study. Maintenance on the glider can be assumed to be the same for both alternatives since the glider is the same. The difference would be in the maintenance of the powertrain and the batteries (Burchart-Korol et al., 2018), but there is still very limited data available on battery maintenance, so data quality would differ too much between the alternatives. For the ICEV the gasoline usage over 150.000km was calculated using data on average C-segment vehicle fuel economy (Rijksdienst voor Ondernemend Nederland, 2020). For the BEV, the total electricity consumption was calculated using the average energy consumption of a Tesla Model S (EV Database, 2020). Although this vehicle might be of a higher segment than the other alternative, it was still considered the best choice, since it performs equally to average electric C-class vehicles (EV Database, 2023). Additionally, the Tesla Model S uses the same battery as has been assumed in the rest of the study, which increases the consistency. The results of these calculations can be found in the appendix.

2.3.3 End-of-life phase

The EOL phase in this study included some assumptions. Since, for the ICEV data is widely available, but for the BEV data is scarce, some assumptions needed to be made. This study considers a more prospective future scenario for the EOL phase. For both alternatives, the complete recycling process was chosen to be situated in France, even though in reality most vehicles are exported outside the EU for further use and eventually dismantling and recycling (Simic, 2015). The reason for this, is that France already has a relatively big battery recycling industry, and in the future could expand this towards car batteries and develop to become one of the main EV recyclers. Furthermore, to keep as many factors the same between the alternatives, France was chosen for both alternatives. The recycling of the gliders and the ICE was simplified to the recycling of the metals, which constitutes almost 80% of the vehicles' weight (Ferrão, et al., 2006). The costs of recycling were calculated using the study on passenger car recycling (Ferrão, et al., 2006), as can be seen in the appendix. The materials that are recycled were counted as avoided burden. This means that the materials that are won back through recycling were counted as negative values to their respective sectors. Effectively, this means that the risk hours related to that amount (or value) of material is subtracted from the total amount of risk hours related to the corresponding sector. The recycling of the Li-ion battery was assumed to be done using the process of hydrometallurgy, which is relatively new in the context of large-scale battery recycling (Joulie, et al., 2014). Hydrometallurgy has been shown to be able to recover up to 99% of the cathode precursor material, and 80% of the anode material. These efficiencies were applied to the Tesla battery, and the results were again counted as avoided loss in the EOL modelling. The resources needed for the battery recycling were taken from a study on battery recycling in China, and then applied to the current study (Shu, et al., 2021).

3. Sensitivity analyses

Apart from the comparison between BEV and ICEV. A number of sensitivity analyses were also conducted. These sensitivity analyses help gaining more insights in the value chains and possible hotspots. The first sensitivity analysis switches the production location of cobalt to a different region, the second sensitivity analysis investigates the impact of a reduced price for rare earth elements (REE), and the last sensitivity analysis doubles the life expectancy in mileage for the two vehicles.

3.1 Production location of cobalt

An important part of BEV production is the extraction of precursor resources for the production of the lithium-ion battery. One of these resources is cobalt. More than half of the worlds' cobalt is produced in the Democratic Republic of Congo (Pistilli, 2023). Australia, however has the second biggest reserve of cobalt in the world and has considerably better working conditions compared to Congo (Köllner, 2018; Hermanus, 2007). In this sensitivity analysis, the production location for cobalt is therefore moved from Congo to Australia.

3.2 Technology critical elements

Both ICEVs and BEVs contain technology critical elements (TCE). These are elements such as cobalt, lithium, and neodymium. They are critical for automotive technology, both existing and emerging. These elements have seen high fluctuations in price due to rapid changes in demand and availability. TCE's provide a high risk for the emergence of resource related conflicts (Ali, & Katima, 2020). In ICEVs, TCE's are present in the form of platinum and palladium, which are found in the exhaust where they function as catalysts. In BEVs TCE's are mainly found in the batteries in the form of lithium, and cobalt, but are also present in the electric engine in the form of neodymium. The prices of TCEs have greatly increased over the last few years (Mancheri et al., 2019). Since the PSILCA database makes use of the monetary value of products, it might be sensitive to price fluctuations, in the real world, since data that is put in might not use the same valuation of various resources as the PSILCA does. Therefore, this sensitivity analysis investigates the effect of an REE price reduced by 25%. The sensitivity analysis looks at reduced prices, because lower prices might better fit the data. This is because the PSILCA uses data from the last few years, whereas some TCE prices have more than doubled over the last two years and the rapid increase in pricing might not scale well with the PSILCA database.

3.3 Vehicle Life Expectancy

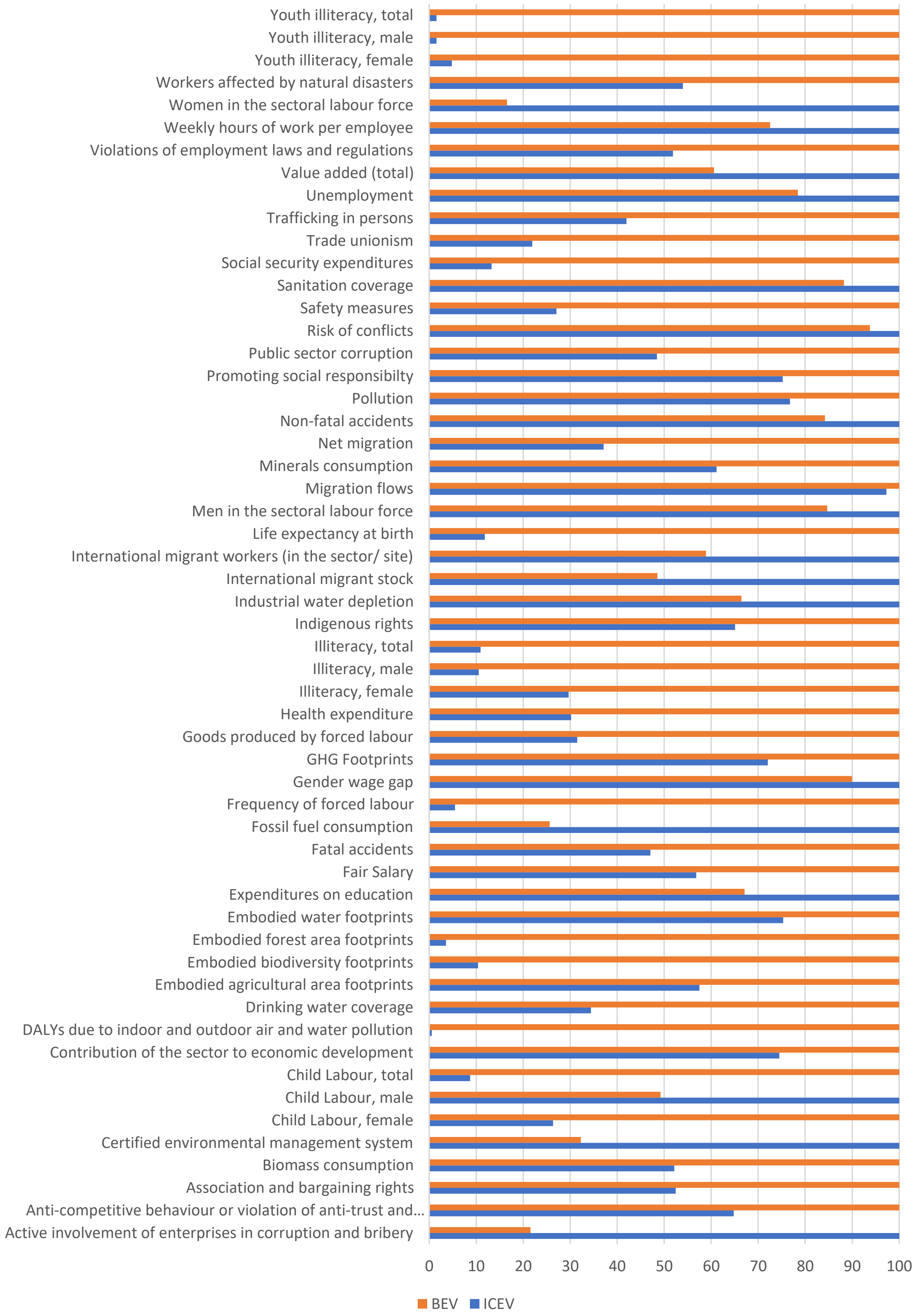
In most LCA's conducted on vehicles, the assumed life expectancy of a vehicle is 150.000 km (Majeau-Bettez et al., 2011). Therefore, this study also assumes a life expectancy of 150.000km. Literature suggests however, that the lifespan of a vehicle might be considerably longer than 150.000km, even up to 300.000km (Velandia et al., 2019; Borlaug et al., 2020). Therefore, this sensitivity analysis investigates the effect of an assumed vehicle lifespan of 300.000km instead of 150.000km.

4. Results

4.1 General comparison

The two alternatives were compared on the 55 categories included in the PSILCA. The results of this comparison can be found in figure x: *Compared Social Life Cycle Impact.* At first glance the ICEV seems to perform considerably better compared to the BEV on the social indicators. Especially on the impact categories related to worker safety and wellbeing (*workers affected by natural disasters, weekly hours of work per employee, violations of employment laws and regulations, safety measures, frequency of forced labour, fatal accidents, child labour)* the BEV seems to perform worse than the ICEV. Contrarily, the environmental performance of the two vehicles was more closely tied over the categories that were included in this social lifecycle analysis, such as greenhouse gas emissions. Obviously, the fossil fuel consumption for the ICEV was way higher compared to the BEV, but for example the minerals consumption of the BEV was higher than of the ICEV. The result that stands out the most from this comparison, is that the disability adjusted life years (DALY's) that can be attributed to air and water pollution was almost 200 times as high for the BEV compared to the ICEV. The most probable reason for this, is that the tailpipe emissions of the ICEV alternative have not been included in the calculations. Only the production and transportation of the fuel was considered in the analysis. This means that the DALY's that can be attributed to air and water pollution are likely highly underestimated for the ICEV alternative. Contrarily, this indicator could be slightly overestimated for the BEV alternative. From the contribution tree, it can be found that the DALY's that can be attributed to air and water pollution for the BEV alternative are almost entirely related to the mining activities in Congo. More specifically, mining related transport seems to contribute considerably to the indicator results. This might explain an overestimation of the air pollution, since the local mining industry produces more products than solely cobalt, such as copper. Copper for example, has a much lower value per kg than cobalt, however the mining related transportation costs are averaged for the Congo mining sector. This gives an overestimation of the transportation related emissions for cobalt, because due to the higher price per kg, cobalt needs less tonne-km of transportation per dollar of product compared to the average of the Congo mining sector. On the other hand, the results seem in line with current literature on water pollution in the Democratic Republic of Congo. Namely, it was found that mining effluent discharges in Congo have contaminated rivers and sediments with levels exceeding health standards. Additionally, agricultural products from the region have been found to contain elevated levels of heavy metals. There is even evidence that parental involvement in the mining industry can increase the chance of birth defects. (Atibu et al., 2013; Muimba-Kankolongo et al., 2022; Van Brusselen, 2020).

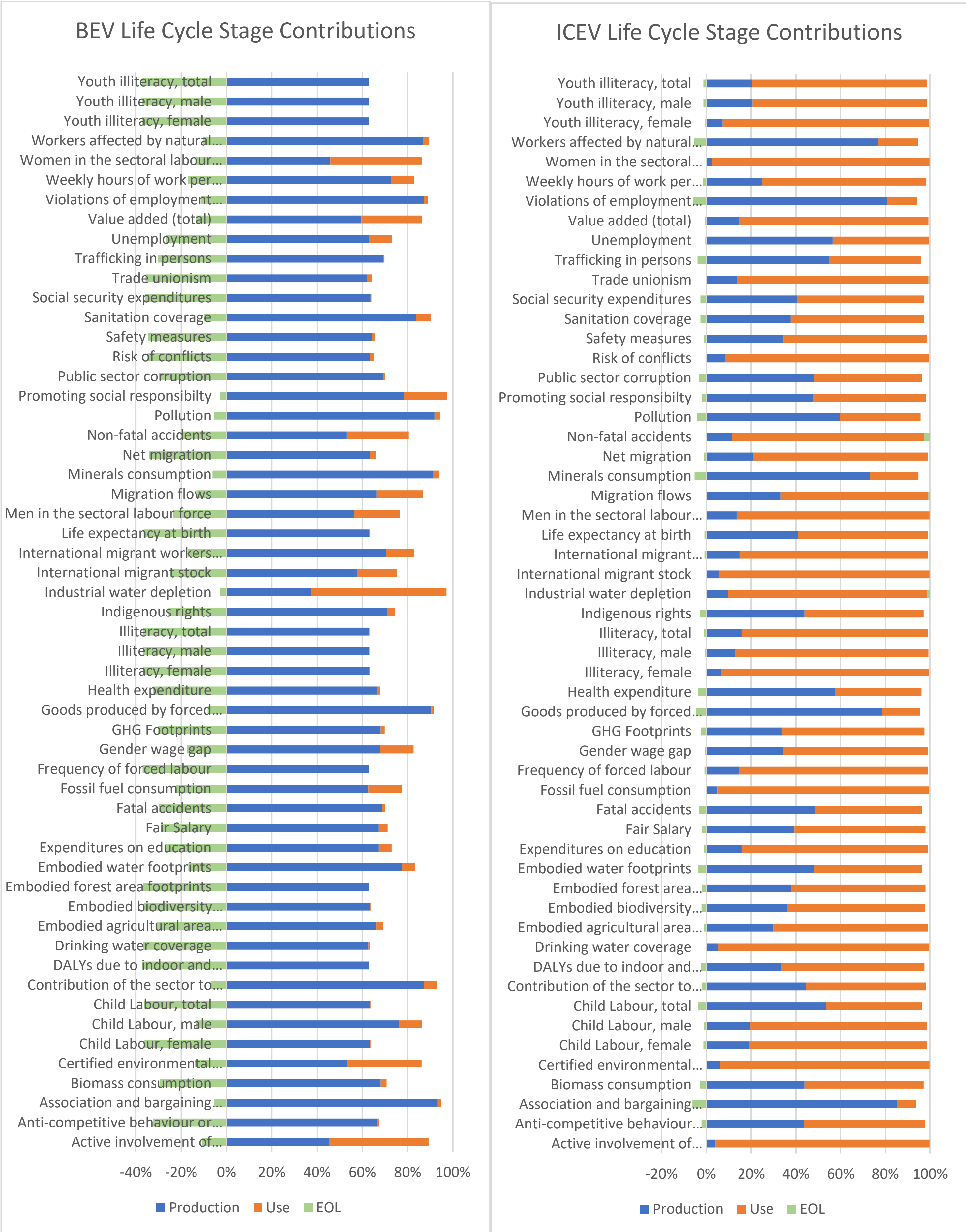
Figure 3: Compared indicator results relative to biggest alternative



4.2 Life cycle stage contributions

To gain insight in the impact of the different life cycle stages of the alternatives, the life cycle was cut into three stages: vehicle production, vehicle use, and vehicle recycling. The first stage contains every step from raw material extraction to a completed usable vehicle. The use stage contains the impacts related to driving, e.g. fuelling and charging the vehicles. The last stage, recycling, contains the dismantling, discarding, and recycling of the materials and parts. The results are plotted in figure x and figure x. As can be seen, the recycling stage, for the most impact categories, has a negative impact. This is because recycling in this study was treated as avoided cost. From these figures it is most evident that for the ICEV the use phase is generally related to the biggest impacts, whereas for the BEV the production phase is responsible for the biggest impacts. Subsequently, since the materials that are used are mainly responsible for these impacts, the recycling stage has a bigger impact for the BEV, because the products that are recycled account for a higher level of avoided impact. However, these results must be viewed critically, since in this study a recycling rate of 60% was assumed for both alternatives. In reality the recycling rate of ICEVs might be higher whereas the recycling rate of BEVs, especially if you consider the battery, might be lower. This is partly because the systems to recycle ICEV's have been around longer and have had gradual improvement over the years. The recycling of BEV's is still in a very early stage, hence it is not strange that the recycling processes are still less optimised (Nurdiawati, & Agrawal, 2022; Ferrao, & Amaral, 2006). Additionally, in this study, the recycling efficiency for the battery assumed best available technology. This could create a distorted image since the best available technology in this case is very recent and far from widely used (Velázquez-Martínez et al., 2019).

Figure 4. Life cycle stage contributions as percentage of total contributions per indicator



4.3 Sensitivity analyses results

4.3.1 Production location Cobalt

For the BEV it was assessed what the impact would be of shifting the cobalt production from Congo to Australia. For most impact categories, a shift to Australia would reduce the impacts related to the BEV production. This is to be expected because of the generally better safety standards and regulations in Australia compared to Congo (Köllner, 2018; Hermanus, 2007). However, some categories stand out because of an opposite effect. Namely the categories for corruption, international migrant workers, and fossil fuel consumption show higher risks in the Australia scenario. This result might not be entirely unexpected since Australia has been involved in multiple corruption scandals revolving around mining and fossil resources (White, 2017), but it seems inaccurate since Congo has been linked to much bigger corruption problems than Australia (Transparency International, 2022). This discrepancy between previous studies and the current results will be discussed further in the reflections on the methods.

The higher level of migrant workers in the sector in Australia is also well documented (Goel & Goel, 2009). Australia already had a shortage of skilled workers for the mining industry. With the rapid growth of this sector Australia has looked across its borders to solve this problem. Higher levels of migrants in a certain sector are in itself not a problem but in reality, higher levels of migrants are accompanied by a higher level of discrimination. This still needs to be put in perspective however, since it is not unlikely that even if treated unfairly in Australia, migrant workers might still be in a better environment compared to their country of origin, so overall migration could lead to positive social outcomes for migrants involved in the value chain. The higher fossil fuel consumptions that are related to mining in Australia, from looking at the contribution table, appear to be the result of services to the mining industry. These services are mainly related to logistical services. It might be that the higher fuel consumption is the result of the mines in Australia being situated in more remote areas, therefore more transport kilometres are needed to support the industry.

4.3.2 Technology critical element price

In this sensitivity analysis, the price of TCE’s was lowered by 25% compared to the original scenario. This made a significantly bigger difference for the BEV alternative than for the ICEV alternative. This is due to the BEV having a higher share of REE in its total price compared to the ICEV. The reduction in REE price had a strong impact on the results for the BEV. For example, the 25% price reduction resulted in a reduction of the risk of up to 60% for several impact categories such as illiteracy, life expectancy at birth, forced labour, and child labour. From these results it is hard to make conclusions regarding the comparison between ICEV and BEV impacts, but it does give some insights in the applicability of the PSILCA and possible weaknesses of this method, which can be valuable for future research.

4.3.3 Vehicle life expectancy

This sensitivity analysis compared two different scenarios for the vehicle life expectancy. The original 150.000km life expectancy was compared to a life expectancy of 300.000km. As literature suggests, the life expectancy of 150.000km which is commonly used in LCA’s might not be the best reflection of the real world, since longer life expectancies are very commonly reported (Velandia et al., 2019; Borlaug et al., 2020). Interestingly, the results of this sensitivity analysis show that the chosen life expectancy can have a crucial impact on the results of a study. In this case, the higher life expectancy scenario tipped the scale in favour of the BEV alternative for a number of categories, such as GHG footprint, and migration flows. The risk of these phenomena was higher for the BEV in the 150k scenario, but were lower for the BEV in the 300k scenario. These indicators are more impacted by lifetime for the BEV, because these are mostly related to the material extraction in the production stage, so if a vehicle has twice the life expectancy, it requires half the production resources per driven kilometer. For the ICEV, these indicators are mostly tied to the usage stage, that means that per driven kilometre, they do net really change if the lifetime increases. This shows how important making the right assumptions regarding life expectancy is. Future research might focus more on this problem to determine a more realistic life expectancy, because this could greatly improve the accuracy of LCA results.

Figure 5: Sensitivity analyses heatmap

Indicator	Cobalt production		REE price				Vehicle lifetime			
	Congo	Australia	BEV low price	BEV normal price	ICEV low price	ICEV normal price	BEV 300k	BEV 150k	ICEV 300k	ICEV 150k
Active involvement of enterprises in corruption and bribery	39.5	100.0	20.2	21.5	99.8	100.0	16.8	21.5	97.9	100.0
Anti-competitive behaviour or violation of anti-trust and monopoly legislation	100.0	47.1	60.9	100.0	65.3	65.4	51.5	100.0	51.0	65.4
Association and bargaining rights	95.4	100.0	99.6	100.0	53.9	53.9	50.9	100.0	29.5	53.9
Biomass consumption	100.0	57.0	71.0	100.0	52.7	52.8	53.1	100.0	41.1	52.8
Certified environmental management system	100.0	85.1	29.6	32.2	99.8	100.0	23.4	32.2	97.0	100.0
Child Labour, female	100.0	10.5	44.9	100.0	26.4	26.5	50.5	100.0	24.0	26.5
Child Labour, male	100.0	90.2	45.7	49.0	99.7	100.0	27.9	49.0	90.5	100.0
Child Labour, total	100.0	10.0	44.3	100.0	8.8	8.9	50.2	100.0	6.5	8.9
Contribution of the sector to economic development	100.0	96.2	96.3	100.0	75.0	75.1	53.3	100.0	58.2	75.1
DALYs due to indoor and outdoor air and water pollution	100.0	0.6	38.8	100.0	0.6	0.6	50.0	100.0	0.5	0.6
Drinking water coverage	100.0	4.6	41.8	100.0	34.4	34.4	51.1	100.0	33.5	34.4
Embodied agricultural area footprints	100.0	49.7	66.6	100.0	57.7	57.7	54.0	100.0	49.1	57.7
Embodied biodiversity footprints	100.0	9.3	43.9	100.0	10.4	10.5	50.8	100.0	8.6	10.5
Embodied forest area footprints	100.0	3.8	40.3	100.0	3.6	3.6	50.2	100.0	2.9	3.6
Embodied water footprints	100.0	96.3	91.6	100.0	76.4	76.5	54.2	100.0	57.8	76.5
Expenditures on education	100.0	63.7	50.7	66.9	99.9	100.0	37.6	66.9	92.2	100.0
Fair Salary	100.0	65.1	72.0	100.0	57.2	57.3	54.7	100.0	46.0	57.3
Fatal accidents	100.0	53.6	69.6	100.0	47.7	47.8	51.8	100.0	36.0	47.8
Fossil fuel consumption	37.2	100.0	21.3	25.6	100.0	100.0	16.3	25.6	97.6	100.0
Frequency of forced labour	100.0	2.5	39.8	100.0	5.5	5.5	50.3	100.0	5.1	5.5
Gender wage gap	100.0	92.0	79.8	89.7	96.7	100.0	54.9	89.7	82.7	100.0
GHG Footprints	100.0	51.6	69.1	100.0	72.8	72.8	52.2	100.0	60.6	72.8
Goods produced by forced labour	100.0	96.9	96.7	100.0	32.2	32.2	50.7	100.0	19.0	32.2
Health expenditure	100.0	41.1	62.3	100.0	30.7	30.7	51.4	100.0	21.7	30.7

Illiteracy, female	100.0	5.5	42.0	100.0	29.7	29.7	50.9	100.0	28.8	29.7
Illiteracy, male	100.0	3.7	40.8	100.0	10.6	10.6	50.6	100.0	9.9	10.6
Illiteracy, total	100.0	4.7	41.3	100.0	11.0	11.0	50.6	100.0	10.1	11.0
Indigenous rights	93.0	100.0	79.1	100.0	65.8	65.9	53.4	100.0	51.3	65.9
Industrial water depletion	100.0	39.0	65.5	66.8	100.0	100.0	54.6	66.8	94.8	100.0
International migrant stock	26.3	100.0	38.4	48.6	99.9	100.0	32.8	48.6	97.1	100.0
International migrant workers (in the sector/ site)	8.4	100.0	52.6	58.7	100.0	100.0	34.8	58.7	92.7	100.0
Life expectancy at birth	100.0	7.0	42.5	100.0	11.1	11.9	51.0	100.0	9.5	11.9
Men in the sectoral labour force	100.0	84.9	69.1	84.6	99.9	100.0	58.4	84.6	93.3	100.0
Migration flows	96.6	100.0	92.2	100.0	97.0	97.1	64.0	100.0	80.7	97.1
Minerals consumption	62.9	100.0	99.0	100.0	62.5	62.6	51.6	100.0	38.7	62.6
Net migration	91.0	100.0	55.3	100.0	37.2	37.2	54.1	100.0	33.4	37.2
Non-fatal accidents	100.0	42.3	72.7	85.1	99.9	100.0	61.7	85.1	93.4	100.0
Pollution	96.4	100.0	99.4	100.0	78.2	78.2	51.4	100.0	54.2	78.2
Promoting social responsibilty	93.5	100.0	98.3	100.0	75.8	75.8	60.0	100.0	57.7	75.8
Public sector corruption	100.0	52.6	69.5	100.0	49.1	49.2	51.3	100.0	37.1	49.2
Risk of conflicts	100.0	18.2	48.6	93.6	99.6	100.0	49.8	93.6	96.0	100.0
Safety measures	100.0	26.1	53.1	100.0	27.2	27.2	51.9	100.0	22.5	27.2
Sanitation coverage	100.0	97.4	84.2	87.3	99.9	100.0	47.1	87.3	81.2	100.0
Social security expenditures	100.0	13.6	46.2	100.0	13.4	13.4	50.7	100.0	10.7	13.4
Trade unionism	100.0	19.0	47.3	100.0	21.9	21.9	53.9	100.0	20.3	21.9
Trafficking in persons	100.0	51.4	68.8	100.0	42.7	42.7	50.7	100.0	30.7	42.7
Unemployment	100.0	58.8	59.3	78.3	89.7	100.0	47.7	78.3	71.7	100.0
Value added (total)	100.0	85.6	55.4	60.5	99.8	100.0	41.3	60.5	92.9	100.0
Violations of employment laws and regulations	100.0	99.4	95.9	100.0	53.2	53.2	51.1	100.0	30.5	53.2
Weekly hours of work per employee	89.0	100.0	65.0	72.1	99.8	100.0	41.8	72.1	87.7	100.0
Women in the sectoral labour force	100.0	79.1	15.3	16.6	99.9	100.0	12.9	16.6	98.7	100.0
Workers affected by natural disasters	100.0	99.4	96.3	100.0	55.3	55.3	51.8	100.0	33.0	55.3
Youth illiteracy, female	100.0	1.1	39.1	100.0	4.8	4.8	50.2	100.0	4.7	4.8
Youth illiteracy, male	100.0	1.0	39.0	100.0	1.6	1.6	50.1	100.0	1.4	1.6
Youth illiteracy, total	100.0	1.0	39.0	100.0	1.6	1.6	50.1	100.0	1.4	1.6

5. Discussion

5.1 Initial implications

The results from the original scenario indicates, that from a social risk point of view, the ICEV might perform better than the BEV. The results of this study are partly in line with results of previous studies. A higher human toxicity for the BEV, in line with the results of the current study, was for example found in an environmental LCA (Velma et al., 2021), additionally the higher GHG emissions related to the ICEV that this study found also corresponded with the results of that E-LCA. Social implications found in this study mostly correspond to previous findings. For example, the high risk related to the mining of battery precursor material is well documented in earlier studies on the subject (Van Brusselen et al., 2020; Thies et al., 2019). So, from the results in this study, the social lifetime impacts of ICEV's might, overall, be lower than the social lifetime impacts of BEV's.

The results of this study must however be taken with caution for several reasons. Firstly, the use of the PSILCA has not yet been proven effective for this application, especially regarding the EOL phase. The utility of the PSILCA will therefore also be discussed further in the discussion. Secondly, a great number of assumptions had to be made, both for the sake of simplicity and due to limited data being available. Thirdly, this study of course has some modelling shortcomings. In the LCI for the BEV for example, there was no data on neodymium for example, while neodymium is highly important for the production of electric cars. EV's contain over 2 kg of neodymium, which currently costs almost 200 euro per kg (Filippas et al., 2021; Strategic Metal Invest, 2023). This could lead to an underestimation of the risks related to BEV production. Also, maintenance of both cars was not taken into consideration, due to limited data on battery maintenance. In reality, it seems that maintenance costs per km for a BEV might be as low as half the costs compared to ICEVs (Harto, 2020). This seems to be related to a far more complicated drivetrain for the ICEV. However, it should be noted that data on BEV maintenance is derived from way fewer cases and often does not include data on battery maintenance and repair or replacement. For this study, this could mean that vehicle use phase costs are more strongly underestimated for the ICEV than for the BEV, but due to the limited data, this cannot be said with certainty. Lastly, the exclusion of the ICEV tailpipe emissions from the calculation leads to a considerable underestimation of the DALY's due to carbon dioxide, and particulate matter emissions (Fantke et al., 2019; Gronlund et al., 2015; Tang et al., 2018), as well as an underestimation of environmental impacts such as global warming effect due to GHG emissions (Montzka et al., 2011). A quick calculation based on the producer specified carbon dioxide emissions of 161 gkm^{-1} of the Golf A4 multiplied by $2*10^{-6}$ DALY per kg CO_2 (Yakar, & Kwee, 2020) shows that the DALY's due to carbon dioxide emissions resulting from fossil fuel combustion would be 0.0483 in the 150.000km scenario. This only takes into account the CO_2 emissions, and not the CO, NO_x, and particulate matter emissions. Additionally, the global warming potential related to these emissions would be $2.42*10^4$ $kgCO_2Eq$ (ReCiPe Midpoint method; Huijbregts et al., 2016). Again, this calculation only takes into account carbon dioxide emissions, and not other emissions that are more potential and could lead to considerably higher results. This goes to show that the omittance of tailpipe emissions might contribute to a strongly distorted image of the impacts.

Inaccuracies like these increase the error margin of the current study. The results must therefore be viewed in the light of the assumptions and decisions that were made and might thus differ from the real-life situation. The sensitivity analyses that were performed shed some light on the magnitude of the effect that different assumptions can have on the results and provide valuable insights in the challenges that the PSILCA database faces in its currents state.

While the current study seems to indicate that more social risks are related to BEV's than to ICEV's, this should not mean that the BEV as a solution for vehicle related environmental impacts should be refrained from. Namely, as mentioned earlier, BEV's, although having been around for some years now, are still an emerging technology.

Developments in automotive, and especially battery technology follow each other in a rapid fashion. Even recently for example, scientists found that altering the electrolyte of a Li-ion battery in a specific way, could as much as double the life expectancy of the batteries (Wang et al., 2023). If batteries become more efficient, smaller batteries can be used while maintaining performance at the same level. So, discoveries like these could lead to a rapid decline of the risks related to the production of BEV batteries in the next years. Such factors are hard to anticipate when conducting an S-LCA. This also reduces the value of the PSILCA as an advisory tool for legislators, because it is always behind on current and emerging technologies.

5.2 Reflections on Methods

One crucial aspect of this paper was the assessment of the PSILCA and its use in comparative S-LCA. The current study has given some very interesting new insights into this subject. For example, a potential weakness is the fact that the PSILCA is based on the monetary value of products or services. This gives potential problems when assessing products systems containing sectors or products that are subject to cost price volatility. For example, in the case of this study the prices of some battery precursors and TCE's are highly volatile (Proelss et al., 2020). In a time span of 10 years, the prices of some elements have more than doubled. This volatility could lead to distorted results because in the calculation of the inventory, often a conversion needs to be made from mass to value (e.g. 1kg of cobalt converted to $80.000 in the Congo mining sector). If the price of a product quickly rises, in the results rendered by the PSILCA database, this would lead to increased risks related to that service or product, while in reality not more of the product is used. A way to counter this effect, could be to index the value of products to correspond to the year of the data being used, but this could still lead to some problems, since the PSILCA is based on data from a spread period of time. Thus, prices could be matched better for one indicator than for the other. Additionally, indexed prices might not reflect the true effects of price fluctuations on worker condition, since indexing in relation to the PSILCA only effectively increases or decreases the amount of worker hours related to a certain process.

Strong fluctuations in pricing could lead to negative effects both for the workers as for resource rich countries (Badeeb et al., 2019). This effect is called the resource curse. This effect needs to be taken into consideration when conducting studies using the PSILCA, but currently that is not possible.

Another challenge of the PSILCA is the availability of data. In many cases, data gaps are filled by taking the region average, or by taking data from similar sectors. While in some cases this might be fitting, in many cases this could lead to greatly distorted results. For example, when looking at accidents in the workplace, the PSILCA uses data from the ILOstat database (ILO, 2019). The statistics from the occupational injuries (ILO Department of Statistics, 2022) indicate that countries like Sweden, Finland and the Netherlands top the charts of work-related incidents per 100.000 workers, with Sweden for example having over double the number of injuries compared to Mexico. This image is greatly distorted due to countries like Sweden and the Netherlands having a higher reporting rate than Mexico. The PSILCA however uses this database for its indicator of worker safety. Therefore, the risks related to sectors in developing countries might be highly underestimated. So, the PSILCA database still has some challenges to overcome in order to reduce biases related to reporting differences between countries and regions.

5.3 Future Research

In this paper, several weaknesses and challenges related to the PSILCA and S-LCA methodology were identified. Future research should aim at investigating these challenges and creating a set framework for researchers to address these flaws. This can lead to S-LCA studies being more comparable and useful for legislative purposes. One important aspect that should be assessed is a potential backlash effect related to price fluctuations, but also to LCA based legislation. Because if for example the sale of BEV's is promoted as a result of studies showing the environmental benefits of this alternative, this could lead to ha higher demand of BEV's and the related resources. This could in turn lead to negative effects for people involved in the value chain as previously discussed. It is therefore important that future research assesses this issue in the form of a prospective S-LCA. Additionally, more research is needed into the real lifetime of vehicles, which not only includes the time when a vehicle is registered and driven in the EU, but also when a vehicle is exported and continually used in often developing countries. Because especially when a vehicle is no longer on the radar in the EU, emissions might be higher due to users no longer adhering to rules such as emission standards by for example removing a broken catalyst instead of replacing it.

6. Conclusion

At first sight it seems that the BEV value chain is related to more risks than the ICEV, however this result must be taken with a grain of salt due to the assumptions that had to be made and the scaling system related to the PSILCA. This means that the results of this study are valid only under the conditions that are described in this study, such as the limitations of the PSILCA, the availability of data, and the assumptions made to fill these gaps. Nonetheless, some hotspots for social risks in the vehicle value chains were identified. However, overall data on BEV's is still lacking detail to make a reliable assessment of current and possible future emissions related to BEV use. Conclusively, namely the extraction of resources seems to be related to the highest number of social risks, both for the BEV and the ICEV alternative. In the future, improvements in battery recycling could lead to a better prospect for the BEV due to the extraction of resources being partly negated by the recycling.

The applicability of the PSILCA was also assessed, and altogether, it can be stated that the PSILCA, while a great tool to identify hotspots, is not entirely suitable to make the comparison between too complex product systems such as vehicle usage, where a high number of assumptions must be made. This high number of assumptions greatly increases the error margin of the calculations to a level where it is impossible to draw reliable conclusions from the results. Additionally, the PSILCA makes an assessment of social impacts based on data from a very specific time. This means that results are always behind on reality. If you take into consideration the speed at which developments in global industry and trade happen, the PSILCA will never be able to make an up to date, or even prospective assessment of the situation of a specific product or service. On top of that, does the PSILCA only assess the individual indicators included in the methodology, the interaction between these indicators is not taken into account and probably never will, due to the extreme complexity of their interplay. Effects such as the backlash effect of new technologies, or price fluctuations are therefore hard to consider. Nonetheless, the PSILCA can give an insight into current hotspots in the value chain, and can help people make more informed decisions regarding consumption choices as well as help producers and governments target these hotspots for improvement.

Altogether, this study did provide a great insight in the applicability of the PSILCA which can be used for further improvements to the database. Future studies could use the results of this paper to further assess the assumptions that are made in LCA’s on vehicles and could thus further improve the accuracy of this field of research. This can help making more accurate S-LCA based recommendations and will thus further help fight climate change and improve the wellbeing of people around the globe.

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