EVS27
Barcelona, Spain, November 17-20, 2013

Challenges facing the development of a credible sustainability assessment

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Abstract

Sustainability assessments of alternative powertrains are rarely clear-cut and often involve complex interplays between factors. These range from vehicle parameters and powertrain details, which manufacturers can broadly influence, to customer choices and usage patterns that are far harder to control and predict. We investigated some of the reasons for variations in assessments of vehicles, focusing on some of the additional parameters that affect parametric comparisons of battery electric vehicles (BEV), and the trade-offs many of these introduce.

The production impacts of alternative vehicles such as BEVs were found to be potentially far more significant than those of current vehicles. Existing data for battery production was found to be highly variable and have the potential to significantly influence the overall results of whole life assessments. Some of the economic considerations of alternative vehicles are also highlighted. These show that, along with direct vehicle costs, taxation revenues, external costs and recycling profitability may also be affected.

We conclude that in order to enhance whole life sustainability assessments, more research is required to better quantify, incorporate and appreciate many additional factors and the effects of various trade-offs.

Keywords: Life cycle assessment, Alternative powertrains, Batteries, Variability, Comparisons

1 Introduction

Light duty vehicle usage is set to more than double and sales treble by 2050 [1]. These huge increases will exacerbate the already significant problems surrounding fuel provision, emissions, materials supply and disposal associated with current vehicle levels.

Alternative powertrains, particularly battery electric vehicles (BEVs) and hydrogen fuel cell vehicles (FCVs), have received considerable attention in recent years, with many life cycle assessments (LCAs) highlighting their potential benefits [1] [2] [3] [4] [5]. Numerous variables can significantly influence the results of such assessments, including component lifetimes, driving patterns, production impacts, fuel production routes and recycling. Even if these considerations can identify a clear 'best choice', these often contradict other vital considerations of cost, utility and materials supply. BEVs and FCVs have the potential to substantially reduce in-use impacts but they are currently costly, offer reduced ranges and place increased demands on some materials which have already been identified as critical, such as rare earths, platinum, and graphite [6].

We examined some of these considerations, focusing on BEVs, with the aims of helping to establish their importance and highlighting how caution must be observed when interpreting the results of assessments.

2 Considerations and their effects

The in-use impacts of BEVs have been extensively modelled, and consistent trends obtained, which highlight the importance of employing low emission electricity sources such as wind and nuclear over conventional fossil fuel plants [4] [5] [7] [8] [9]. The data available for other factors such as BEV production, the influences of battery parameters, end-of-life treatments and economics, are fewer and variable.

The section discusses some of the factors involved in establishing the impacts of vehicles and how the choice of a particular powertrain may impact other areas. Examples include fuel taxation revenues, recycling processes and economics.

2.1 Use phase

At present, internal combustion engines almost exclusively propel the world's road vehicles. The majority of their energy is derived from crude oil, a finite resource, of which transport accounts for approximately 70% of demand [10]. The supply and price of oil is however quite volatile and significant concerns have been raised over potential supply issues and price hikes [11].

Current road vehicles are also a major source of air pollution, which has a variety of detrimental effects on the ecosystem and human health, particularly in urban areas where greater numbers of people are exposed [12].

These factors are driving a need for alternative vehicles which reduce emissions and oil dependency. However these vehicles must achieve this whilst not placing unsustainable demands on energy or materials sources, incurring unacceptable costs or shifting unjustified impacts to other phases of a vehicle's lifetime (e.g. production, see Section 2.2).

Standard driving cycles, such as the New European Driving Cycle (NEDC), are typically used to model the in-use phase of vehicles. These have the advantage of allowing assessments to be more easily verified and compared. However, different values are likely to be obtained during the actual use of the vehicle, e.g. [13] [14]. The in-use lifetime distance assumed in assessments can also affect the impact results [3]. This is due to ramifications on the relative impacts of the other phases which are averaged out over the vehicles life. For example, an increase in the assumed distance covered could result in the production phase appearing less significant, with consequent reductions in the whole life vehicle impacts on a distance basis.

2.1.1 Electricity generation

The lifetime benefits of BEVs are highly dependent upon the impacts of the electricity used to charge them. The impact of grid electricity varies considerably between locations, due to differences in grid mixes and efficiencies. This can lead to the effects of BEVs being location specific. For example, over 1.4kg of greenhouse gas (GHG) emissions are associated with the production of one kWh of electricity using India's grid, while Sweden's grid emits only 0.05kg/kWh [15]. This is due to India's grid consisting mainly of coal power plants, while Sweden's uses mainly hydro and nuclear.

Some assessments have also studied the additional complications arising from the impacts of marginal electricity, which reaffirms this importance [4] [16]. These are the impacts resulting directly from the additional electricity load that will be placed on a grid by BEVs, which may be significantly different to those for the average grid, depending upon the particular source.

For example, [4] showed the well-to-wheel GHG emissions of a BEV increased by over 75%, when marginal electricity was used in place of average values. This results from the fact that the marginal electricity was assumed to be produced predominantly from coal, while the average value included renewables and nuclear used for base loads. However, marginal emissions could reduce substantially as the oldest, often highest emission plants, are replaced with more efficient alternatives [17].

A further perspective could be presented, that current marginal supplies are needed for current variable loads. Therefore, additional loads, e.g. large numbers of BEVs, would require the installation of additional capacity. New installations often have below average emissions, which will lower the values associated with BEVs. To ensure the greatest overall benefits, this scenario should consider what the optimal use of the new capacity is. For example, could larger gains be realised by using the same electricity to first substitute other energy demands, such as existing grid installations or heating oil.

2.2 Production impacts

The large in-use impacts of conventional vehicles have resulted in this phase being the focus of many studies. However, the production phase still adds a significant proportion to the overall lifetime impacts of conventional vehicles, accounting for around 15-20% of GHG emissions and in some cases over half the sulphur dioxide emissions [3] [18] [19]. With the advent of advanced powertrains which offer abated in-use impacts and place increased demands on specialist materials/components with higher impacts, this importance will increase significantly.

The following sub-section uses the battery packs of electric vehicles as a case study to investigate the effects on production, recycling and whole life impacts.

2.2.1 Batteries

The heavy batteries utilised in BEVs lead to large production impacts and variability. A few LCAs have been conducted, but the range of findings places considerable uncertainty on the results of whole vehicle comparisons [3] [20] [21] [22]. To show this, Figure 1 was constructed which compares the impacts of a ‘C’ segment low emission diesel vehicle with those for two BEVs. This indicates how the results for BEVs could dramatically alter depending solely on the battery production impacts employed. The assumptions used in the model are outlined below.

The in-use phase was modelled over the NEDC, using published values for the diesel vehicle and simulations for the BEV, based on the model described in [13]. To minimise discrepancies, the BEVs were modelled using the same non-powertrain factors as the diesel vehicle, with adjustments to account for the higher masses of their powertrains. The well-to-tank phase was based on data from [15] [23], assuming EU grid mix electricity emissions for the BEV and the production phase for the base vehicle employed data from [18]. The BEVs were assumed to incorporate a 24kWh battery pack requiring one replacement in order for them to equal the lifetime of the diesel vehicle. Further details on the assumptions used are discussed elsewhere [1]. The only difference between the two BEVs in Figure 1 is their battery production impacts, with the high and low values being based on results presented in [21] and [3] respectively.

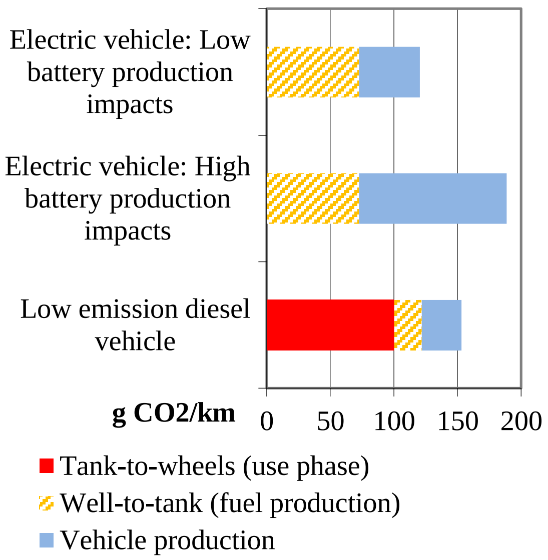

Figure 1: Influences of battery production impacts

The discrepancies found in battery LCAs arise due to many reasons, including:

- Differences in chemistries, even amongst lithium-ion,
- Limited measured production data,
- The assumptions involved in complex LCAs,
- Changes in manufacturing methods,
- Different battery designs and requirements,
- Recycling.

Recycling has been indicated to have the capability to reduce the impacts associated with batteries [22] [24]. However very limited LCA data on the recycling of lithium-ion batteries is available in the existing literature [25]. Quantifying the potential implications of recycling is further complicated by different recycling routes, input materials (i.e. lithium-ion cell types), recovery amounts and types of output materials, e.g. pure metals or carbonates [26] [27]. Figure 2 indicates this with simplified process flows for the two main techniques used for the recycling of lithium-ion cells, along with some of the additional considerations that will influence the method chosen.

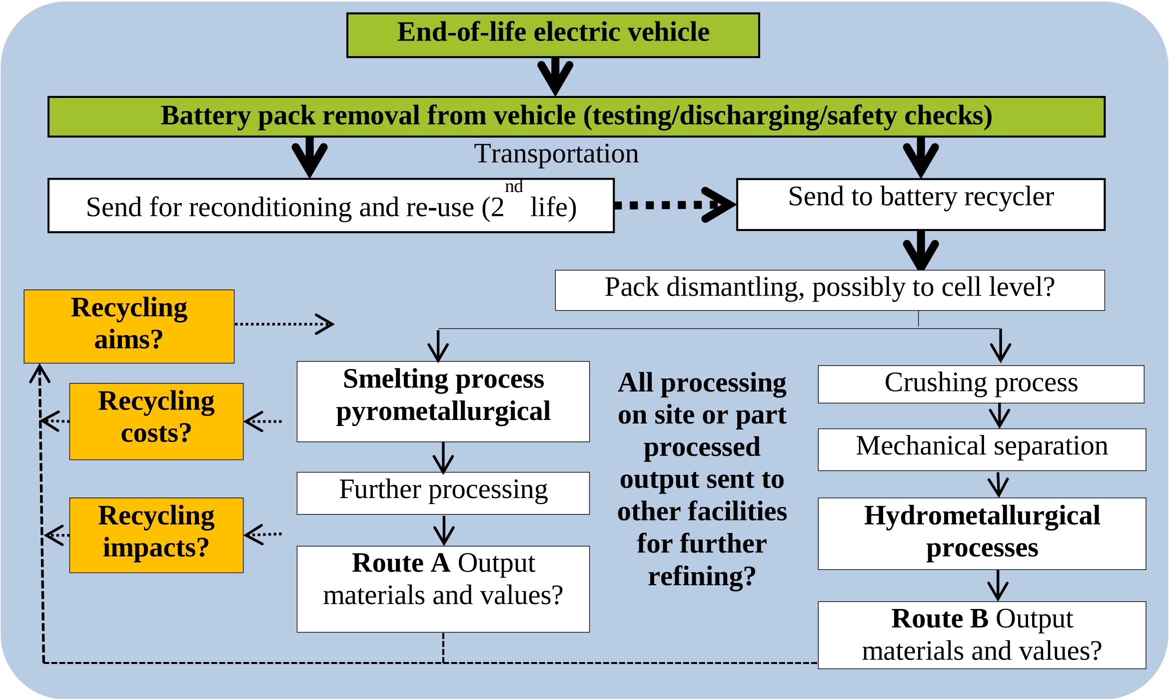

Figure 2 Potential recycling routes

The first technique is a pyrometallurgical process where high temperatures are used to separate the valuable metallic fractions. The second is a hydrometallurgical process where metals are separated via liquid processes such as leaching and precipitation, following mechanical shredding of the cells [27].

The manufacturing processes used to produce the final battery packs from raw materials (e.g. those involved in production of the electronics and electrode materials) are significant contributors to the impacts. These impacts will not be mitigated through recycling, unless components are reused or materials (e.g. cathode powders) can be reclaimed in forms suitable for reuse in batteries without the need for significant additional processing.

2.3 Secondary effects of batteries

The selected battery also has repercussions on the vehicle’s in-use impacts. This necessitates multiple trade-offs to be made, e.g. production impacts verses lifetime and energy efficiency. A few assessments have begun to help quantify the influences of some of these factors [20] [28] but the complexity involved, rapidly developing technologies, data limitations and considerations given above, have resulted in a general lack of data in the current literature surrounding these interconnections.

2.4 Non-powertrain factors

Other parameters can alter the impacts of vehicles, e.g. vehicle mass, auxiliary power draws, and the coefficients of drag and rolling resistance [13]. Vehicles with alternative powertrains, which are designed to provide optimal efficiency, typically have other non-powertrain parameters far more optimised than conventional vehicles. This often creates problems in comparisons because it is not clear what gains are due to the powertrain and what result from the other parameters. For example, the energy required to propel a vehicle with a coefficient of drag and frontal area equal to the 2010 Toyota Prius, over the NEDC, is approximately 8% lower than for a vehicle modelled using the parameters for a Volkswagen Golf MK6 of the same year. This assumes no powertrain losses, a mass of 1300kg and all other parameters are identical.

2.5 Economic considerations

BEVs can offer in-use cost benefits. However, much of the in-use fuel cost seen by consumers for conventional fuel arises from taxation in some countries. Fuel taxation is an important source of revenue for many nations and if a significant number of vehicles were to use an alternative fuel, with lower taxation, some way of recouping this loss will be necessary. Figure 3 shows the level of fuel taxation that would need to be applied to alternative fuels, so that the UK revenue received per km would be equal to that of an efficient petrol vehicle.

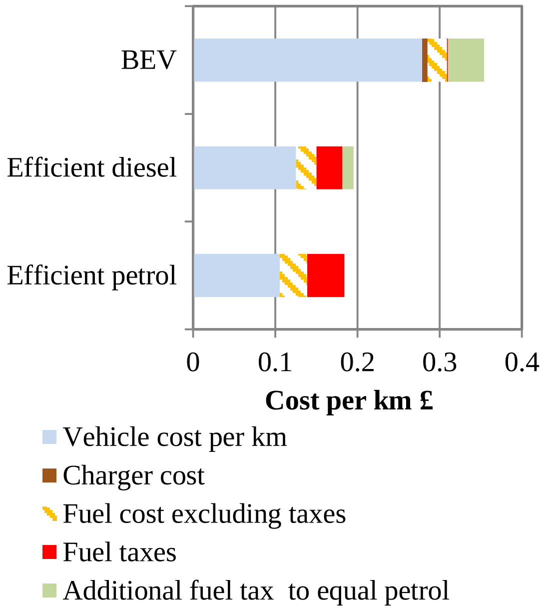

Figure 3: Effects of UK taxation on vehicle fuel costs in pounds sterling (£) (adapted from [1])

Different results would be achieved depending upon the country assessed, due to substantial differences in taxation levels and fuel costs.

Figure 3 also shows the effects of the currently higher BEV costs compared to conventional vehicles. The majority of these higher costs result from their battery packs. These are anticipated to drop substantially with mass production and technology refinements, but will remain a large contributor to the overall vehicle cost [29].

External costs add a further layer of complexity to the selection of vehicles. These are the economic and social costs arising from transportation. For example, damage to human health, materials and crops due to vehicle emissions. Quantifying the cost of these factors is difficult due to factors such as variations with population density and overall pollution levels [2] [30]. However these factors can be major considerations. For example, anthropogenic particulate matter emissions, for which transportation is a major contributor, have been estimated to reduce the life expectancy of UK residents by an average of six months and cost £15 billion annually [12]. Mitigation of these emissions in urban areas, through the use of alternative vehicles such as BEVs which can eradicate tailpipe emissions, could therefore offer significant overall benefits.

2.5.1 Recycling economics

Between 2010 and 2050 it is estimated that over 7 billion tonnes of materials will be consumed in the production of vehicles [1]. This material also needs to be dealt with when the vehicles reach their end-of-life. This phase is becoming increasingly regulated, for example the EU Directive 2000/53/EC on end-of life vehicles. This Directive stipulates minimum recycling rates and requires any negative costs arising from the treatment of waste vehicles to be borne by the automotive manufacturers.

Presently end-of-life treatment facilities are able to generate a profit from waste vehicles, meaning automotive manufacturers do not incur any costs [31]. With changes to vehicle architecture, e.g. substitution of steel (currently a main revenue source for vehicle recyclers) by lightweight materials such as carbon fibre and the inclusion of large new components e.g. batteries and hydrogen tanks, waste vehicles could become unprofitable. This is due to the currently limited recycling of materials such as carbon fibre and the processing needed to reclaim those that are highly mixed in components e.g. batteries, see Figure 2 [32].

A further aspect to recycling is the strategic importance it could have in mitigating import dependence in many countries. Several materials utilised in current or alternative vehicles are reliant on supplies from a limited number of countries. For example, 97% of rare earth metals are produced in China and most platinum group metals are sourced from South Africa or Russia. This leads to the supply and cost of these materials being very sensitive to factors such as export taxes, production quotas and environmental policies in the supplying countries [6]. Recycling could therefore provide a valuable source for some materials, as well as potential environmental benefits [32].

3 Summary

We have highlighted some of the factors that can affect the assessments of vehicles and the complexity these add. Major factors such as the source of electricity used to charge BEVs and the driving cycle used to model the in-use phase are documented in many existing assessments. However other factors and their interactions that should be included in future frameworks, such as the production and recycling impacts of alternative powertrain components, are less well understood.

For example, the production impacts of batteries for electric vehicles were indicated to potentially be a significant contributor to the overall vehicle's lifetime. However, further quantification of their impacts is required. Some of the economic considerations of alternative vehicles were also highlighted. These showed that, along with direct vehicle costs, taxation revenues, external costs and recycling profitability may also be affected.

All the factors discussed can influence the whole life impacts of vehicles. The potential variability in many of them can introduce significant discrepancies in the overall results of vehicle LCAs and this should therefore be appreciated when reviewing their results.

In order to improve the credibility of whole life sustainability assessments, more research is required to better quantify, incorporate and appreciate many of these factors and the effects of various trade-offs.

Abbreviations

BEV	Battery Electric vehicle
FCV	Fuel Cell Vehicle
GHG	Greenhouse gas
LCA	Life Cycle Assessment
NEDC	New European Driving Cycle

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