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INTEGRATED ASSESSMENT OF VEHICLE FUELS WITH LIFECYCLE SUSTAINABILITY ASSESSMENT – TESTED FOR TWO FOSSIL FUEL AND TWO BIOFUEL VALUE CHAINS

Report from a project within the collaborative research program *Renewable transportation fuels and systems*

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THE SWEDISH KNOWLEDGE CENTRE FOR RENEWABLE TRANSPORTATION FUELS

PREFACE

This project has been carried out within the collaborative research program *Renewable transportation fuels and systems* (Förnybara drivmedel och system), Project no. 39120-1. The project has been financed by the Swedish Energy Agency and f3 – Swedish Knowledge Centre for Renewable Transportation Fuels.

f3 Swedish Knowledge Centre for Renewable Transportation Fuels is a networking organization which focuses on development of environmentally, economically and socially sustainable renewable fuels, and

- Provides a broad, scientifically based and trustworthy source of knowledge for industry, governments and public authorities
- Carries through system oriented research related to the entire renewable fuels value chain
- Acts as national platform stimulating interaction nationally and internationally.

f3 partners include Sweden's most active universities and research institutes within the field, as well as a broad range of industry companies with high relevance. f3 has no political agenda and does not conduct lobbying activities for specific fuels or systems, nor for the f3 partners' respective areas of interest.

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EXECUTIVE SUMMARY

The production and use of vehicle fuels results in both environmental and socio-economic impacts. In the Renewable Energy Directive (RED) the European Union (EU) implemented mandatory sustainability criteria for biofuels for transport and liquid biofuels. These include demand for reductions in greenhouse gas (GHG) emissions and restrictions related to land with high biodiversity value. This directive and the vast majority of the available studies enfolding vehicle fuels, focus on environmental impacts, and in many cases primarily on GHG emissions. To move towards sustainable development, a broader scope of sustainability issues needs to be taken into account in future assessment efforts and policy.

In order to address a broad range of sustainability aspects a method labelled Life Cycle Sustainability Assessment (LCSA) can be employed. It combines three different lifecycle methods, corresponding to the three pillars of sustainable development; environmental-LCA (E-LCA), socialLCA (S-LCA) and life cycle cost (LCC).

In recognition of these knowledge gaps, the overall aim of this project is to examine the use of LCSA to assess the sustainability performance of transportation fuels. This is achieved by applying it to four selected fossil and renewable vehicle fuel value chains. The principal aim of this work is to develop the methodology of LCSA with focus on a full integration step in the assessment. The integration of different sustainability perspectives is a challenge, as it is inevitably based on value judgements. In this analysis we apply the Multi Criteria Decision Analysis (MCDA) methodology using different stakeholder profiles for the integration. This approach has the advantage that it increases transparency on these value judgements. Further, as a part of this work, the policy relevance of LCSA results is discussed briefly.

The analysis considers four vehicle fuel value chains: Petrol based on crude oil from Nigeria ; petrol based on crud from Russia; Ethanol based on sugarcane grown in Brazil, and ethanol based on corn (maize) grown in the USA. Both biofuels represent first generation biofuels. These vehicle fuels were selected so as to build on an earlier study where an S-LCA was conducted for nine vehicle fuel chains.[1] They were also attractive as they have relatively high data availability. These four fuels were also found to have relatively high potential risks of negative social impacts in the previous study.

The LCSA conducted in this study is done by integrating S-LCA results with results from E-LCA and LCC. In addition to the compilation of comparable E-LCA and LCC results we seek to detail the S-LCA results in the previous study as well as complementing them with positive social impacts in order to provide a more detailed analysis.

The main contribution of this project is related to the steps taken towards aggregating the different sustainability perspectives into one holistic outcome for sustainability. This is done using three different stakeholder profiles. These represent different worldviews and value judgments when prioritizing between the different sustainability perspectives. The result shows that the ranking order of the different vehicle fuels chains are quite different for the different stakeholder profiles. This

[1] Ekener-Petersen, E., J. Höglund and G. Finnveden (2014). "Screening potential social impacts of fossil fuels and biofuels for vehicles." Energy Policy 73: 416-426.

shows that there is not always one single answer for the most sustainable choice between different alternatives. Rather this is dependent on different priorities held by different stakeholders, or the population they represent.

All three underlying lifecycle methods – E-LCA, S-LCA and LCC - have different methodological limitations. Further, they are to various extents relatively new and still under development. One issue identified for all three methods is the lack of robust and updated databases for data collection. This causes problems as the data requirements for assessments are considerable. Thus the importance of data quality is emphasized. The MCDA method offers, however, a possibility to address uncertainties based on variable data quality. In general, the MCDA methodology seems to offer many useful features to ameliorate the effects of a number of data-related complications. As such, it seems to offer a good tool for the aggregation step in LCSA. This stated, the lack of robust and updated databases imply that the actual LCSA-results for the included vehicle fuels may not be representative of the current situation regarding sustainability performance.

In this project, positive social impacts were handled and integrated separately. By considering the positive social impacts separately, the influence of the positive impacts on the end result of an S-LCA becomes visible. Although this was done in a limited way in this analysis, it is important to include positive impacts separately in future S-LCA efforts, to be able to distinguish the contribution from positive impacts to the total social impact. This may inform future action to enhance these positive contributions. Yet, the lack of data makes this a difficult task, needing further work.

Another important contribution, we believe, is the attempt to assess both fossil and renewable vehicle fuel chains with the same assessment tool. In the future, all vehicle fuels should be evaluated on their total sustainability performance at the same level of detail.

Finally, we believe that the methodology approach examined in this work may be useful for efforts to leave the ‘silo’-thinking that can be found in sustainability discourse behind. Instead of this, actors can be motivated to focus on broad, comprehensive sustainability implications of various product life cycles. Once the underlying data and methodology-related limitations have been improved, we believe that LCSA in combination with MCDA has true potential to provide a useful tool for sustainability assessment in a life cycle perspective.

LCSA could be used as an information tool to guide the formulation of policy, and as an assessment tool providing information to assess overall success (or failure) of policy interventions. In conclusion however, we stress that it is important that communication with stakeholders and decision makers should be clear in terms of data quality and of the assumptions and complex assessments required for this assessment method. This is vital if it is to be useful in policy-making and development of specific policy instruments.

SAMMANFATTNING

Produktion och användning av fordonsbränslen bidrar till miljöpåverkan samt har sociala och socioekonomiska effekter. Europeiska unionens direktiv om förnybar energi (RED) innehåller hållbarhetskriterier för biodrivmedel och flytande biobränslen för andra energiändamål som omfattar minskade utsläpp av växthusgaser och begränsningar kopplat till områden med hög biodiversitet. Dessa hållbarhetskriterier, i likhet med de flesta studier om hållbara drivmedel, fokuserar på miljöpåverkan, och i många fall främst på utsläppen av växthusgaser. För att gå mot en hållbar utveckling behövs dock en bredare ansats, där fler hållbarhetsfrågor beaktas i analyser såväl som i styrmedel.

För att analysera en mängd hållbarhetsaspekter kopplade till en produkt i ett livscykelperspektiv kan en metod som kallas Life Cycle Sustainability Assessment (LCSA) användas. Denna metod sammanför tre olika livscykelmetoder - motsvarande de tre perspektiven i hållbar utveckling – nämligen miljö-LCA, social-LCA och livscykelkostnadsanalys (LCC). Det övergripande syftet med denna studie är att undersöka möjligheterna att använda LCSA för att bedöma en produkts hållbarhetsprestanda genom att tillämpa LCSA på ett urval fossila och förnybara fordonsbränslen. Tyngdpunkten ligger på att vidareutveckla LCSA metoden, och inkludera en komplett integration av de tre separata bedömningarna i slutet av analysen. Att integrera olika hållbarhetsperspektiven är en utmaning, eftersom det oundvikligen baseras på värderingar. Vi använder metoden för multikriterieanalys (MCDA) och olika aktörsprofiler vid integreringen, vilket möjliggör transparens angående dessa värderingar och en insikt i hur olika värderingar påverkar utfallet. Även policyrelevansen för LCSA-metoden diskuteras översiktligt.

Följande fyra bränslen ingår i analysen: bensin baserad på råolja från Nigeria respektive Ryssland samt etanol baserad på sockerrör odlade i Brasilien och baserad på majs som odlas i USA (i båda fallen första generationens biodrivmedel). Vi valde dessa fordonsbränslen i första hand eftersom de ingick i en tidigare studie där en S-LCA genomfördes för nio olika drivmedelskedjor.[2] Det finns vidare relativt god tillgång till data för dessa kedjor. Därtill pekade den tidigare studien på relativt höga potentiella risker för negativ social påverkan för dessa fyra drivmedel.

I denna studie utför vi alltså en LCSA genom att integrera S-LCA resultat med resultat från E-LCA och LCC. Utöver att sammanställa E-LCA och LCC-resultat som är möjliga att jämföra försöker vi öka detaljeringsgraden av S-LCA resultaten i den tidigare studien samt komplettera dem med möjlig positiv social påverkan, i syfte att nå en mer detaljerad och heltäckande analys.

Ett av de viktigaste resultaten av detta projekt är kopplat till försöket att aggregera olika hållbarhetsperspektiv till ett helhetsresultat för hållbarhet. Detta görs med hjälp av tre olika aktörsprofiler som representerar tre olika "världsbilder" med olika värderingar kring prioriteringen mellan de olika hållbarhetsperspektiven. Resultatet visar att rangordningen av de olika drivmedelskedjorna blir olika beroende på de olika profilernas prioriteringar. Detta visar att det inte alltid finns ett enda svar på vad som är det mest hållbara valet mellan olika alternativ, utan att detta snarare beror av prioriteringar hos beslutsfattare, eller hos befolkningen de representerar.

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[2] Ekener-Petersen, E., J. Höglund and G. Finnveden (2014). "Screening potential social impacts of fossil fuels and biofuels for vehicles." Energy Policy 73: 416-426.

Alla de tre underliggande livscykelmetoderna – E-LCA, S-LCA and LCC – har olika begränsningar och är i olika utsträckning under utveckling. Ett gemensamt problemområde som har identifierats för alla tre metoderna är bristen på robusta och uppdaterade databaser för datainsamling. Detta orsakar problem då stora mängder data behöver samlas in till dessa bedömningar; vikten av hög datakvalitet måste alltså betonas. MCDA-metoden ger emellertid en möjlighet att hantera osäkerheter baserat på bristande datakvalitet. En allmän slutsats är att MCDA som metod erbjuder många användbara funktioner för att hantera olika svårigheter relaterade till tillgång och kvalitet på data samt möjligheter att synliggöra dessa. Bristen på uppdaterade databaser innebär emellertid att de faktiska LCSA-resultaten för de inkluderade drivmedlen inte med säkerhet är representativa för den faktiska hållbarhetsprestandan hos dessa bränslen i dagsläget.

I detta projekt lades positiv social påverkan till den negativa och hanterades och integrerades separat. Genom att beakta positiv social påverkan för sig, kan man tydligare se vilken inverkan den har på slutresultatet från en S-LCA. Även om detta gjordes på ett begränsat sätt i denna analys visar det på vikten av att inkludera positiva påverkan separat i framtida S-LCA. Dock gör bristen på data detta till en svår uppgift och ytterligare arbete behövs för att tillgängliggöra positiv social data.

Ett annat viktigt bidrag från vårt arbete är vår ansats att utvärdera både fossila och förnybara kedjor med samma verktyg. I framtiden bör alla drivmedel utvärderas utifrån sin totala påverkan på hållbarhet på ett likartat sätt.

LCSA kan användas som ett informationsverktyg för att guida utvecklingen av styrmedel och andra policies och som ett analysverktyg för att ge information som möjliggör en utvärdering av effekterna av olika styrmedel. Det är emellertid viktigt att i kommunikationen med intressenter och beslutsfattare samt när resultaten presenteras vara tydlig vad gäller datakvaliteten samt de antaganden och komplexa bedömningar som krävs för att denna bedömningsmetod ska vara användbar i såväl politiskt beslutsfattande som vid utvecklandet av specifika styrmedel.

Slutligen anser vi att den undersökta metoden LCSA, trots sina begränsningar, är användbar i den strävan som finns efter att lämna struprörstänkandet inom hållbarhet och istället fokusera på bredare, mer holistiska hållbarhetskonsekvenser från olika produktlivscykler. När de underliggande metoderna förbättrats och tillhörande datamässiga begränsningar har minskat, bedömer vi att LCSA i kombination med MCDA kan utgöra ett användbart verktyg för hållbarhetsbedömning i ett livscykelperspektiv.

CONTENTS

1 INTRODUCTION

The production and use of vehicle fuels can lead to environmental as well as social and socio-economic impacts. In recent years, there has been a debate on the sustainability of biofuels for transport. Linked to this, in the Renewable Energy Directive (RED) the European Union (EU) implemented mandatory sustainability criteria for both biofuels for transport and for liquid biofuels for other energy purposes. These includes demands for reductions in greenhouse gas (GHG) emissions and restrictions related to areas of high biodiversity value (European Parliament, 2009a). Thus, the sustainability criteria in the RED focus mainly on environmental aspects but the life cycle perspective is an integral part.

The vast majority of the available studies on sustainable vehicle fuels also focus on the environment, and in many cases primarily on GHG emissions (Lazarevic & Martin, 2016). To move towards sustainable development, more sustainability issues of vehicle fuels needs to be taken into account. There is a great need among market players, researchers and decision-makers, for a more holistic understanding of more sustainability consequences of all vehicle fuels. Not least as the transport sector and the renewable fuels that are now covered by sustainability criteria (Renewable Energy) and these criteria will certainly be developed further in future.

In order to address a range of sustainability aspects, we find the development of Life Cycle Sustainability Assessment (LCSA) an interesting track. The basic idea is to bring together three different lifecycle methods - that correspond to the three pillars of sustainable development – in one method. Klöpffer (2008) suggested, therefore, that the LCSA is a merger of environmentally-LCA, social-LCA and life cycle cost (LCC) of the formula LCSA = LCA + SLCA + LCC (Cinelli, 2013). Since then, various alternative approaches have been developed for how a LCSA can be built up.

In a previous f3 project (Social and socioeconomic aspects of vehicle fuels) conducted by researchers at KTH and IVL Swedish Environmental Research Institute, potential social life cycle impacts of the production and distribution of vehicle fuels, including both fossil fuels and biofuels, was studied (Ekener Petersen et al., 2013). The screening of potential social impacts with social LCA demonstrates the potential risks of negative social impacts in all analyzed fuel chains (Ekener Petersen et al., 2014). In summary, the conclusions of the project were that the methodology of social LCA (S-LCA) will enable decision-makers to identify hotspots where the negative social consequences arise in the life cycle of vehicles fuels and that the country of origin are expected to be decisive for the actual social impact (Ekener Petersen et al., 2014). However, to get more reliable results at a more detailed level, a more detailed study needed be carried out.

The overall aim in this project is to examine the potential for assessing the sustainability performance of products using an integrated life cycle assessment approach, namely LCSA, by applying it on selected fossil and renewable vehicle fuels. The focus is on developing the methodology of LCSA in this field, as LCSA applications has so far generally not included a full integration step in the end of the assessment. We conduct a LCSA by integrating S-LCA results with results from environmental LCA (E-LCA) and Life Cycle Costing (LCC). Besides compiling comparable E-LCA and LCC results we try to detail the previous S-LCA results (Ekener Petersen et al., 2014) as well as complementing them with positive social impacts. The integration of different sustainability perspectives is a challenge, as it is inevitably based on value judgments. We use the Multi Criteria Decision Analysis (MCDA) methodology for the integration, as an attempt to achieve a higher degree of transparency on these values. Finally, the policy relevance of LCSA results is briefly discussed.

The analysis considers the following four different vehicle fuels, including fossil fuel based and biomass based fuels:

- petrol based on crude oil from Nigeria;
- petrol based on crude oil from Russia;
- ethanol based on sugarcane grown in Brazil (1st generation technology);
- ethanol based on corn grown in the USA (1st generation technology).

We selected these vehicle fuels because they were included in the preceding study (Ekener-Petersen et al., 2013) and have relatively high data availability. The preceding study also indicated that they have relatively high potential risks of negative social impacts (Ekener-Petersen et al., 2013). This makes them interesting objects to study for this extended assessment.

The preliminary findings in the report were discussed with stakeholders at a workshop organised within the project (see Appendix I). At this workshop the issue of how to present the results, and how to ensure sufficient understanding of the approach was addressed. Due to the simple fact that this project is a method development exercise, that there remain significant uncertainties in the underlying data and its representativeness, combined with the risk of results being presented out of context (mainly in figure format), the authors of this report have decided not to present the E-LCA results and the final LCSA results in too much detail. This is especially so with regards to the specific vehicle fuels chains included for method testing. Additional outreach of the project is presented in Appendix II.

2 METHODOLOGY

The overall approach is described in this section. More specific assumptions and limitations are described for each sustainability aspect in the following sections and throughout the text. Main responsibilities are as follows, other parts performed jointly. E-LCA, LCC and data for positive social impacts: IVL; S-LCA and world views: KTH; documentation of workshop and discussion of policy implications: LU; and MCDA analysis supported by Aron Larsson at Stockholm University/Mid Sweden University.

2.1 LCSA – OVERVIEW OF EXISTING METHODS ABD APPROACHES

The development of Life Cycle Sustainability Assessment (LCSA) took an step forward in 2008 when Klöpffer (2008), laid out the approach as a combination of the three existing life cycle approaches stated like LCSA=E-LCA + S-LCA + LCC. Since then, there has been an ongoing discussion and further development in the area, with important contributions from papers such as Finkbeiner et al (2010) and UNEP-SETAC (2011). There have also been different parallel methodologies proposed, such as CALCAS (Guinée et al., 2011) and PROSUITE (Blok et al., 2013). Sala et al. (2012a; 2012b) presented an overview of the development in the area, and and Guinée (2016) recently published a review of the concept.

The report of the UNEP/SETAC Life Cycle Initiative (UNEP/SETAC, 2011) is in alignment with Klöppfer (2008) with the three included methodologies (E-LCA, S-LCA and LCC) conducted in parallel, based on the standardized process for E-LCA within ISO 14040, in an iterative process. Thus, in the first phase of the work, definition of goal and scope including i.e. goal, functional unit, system boundaries, allocation principles and a coherent set of impact categories is commonly performed jointly for all three lifecycle approaches within the LCSA. Secondly, a common data collection effort is conducted, taking into consideration the need for qualitative as well as quantitative data, and allowing for generic and sector /national level data, all due to different needs and availability for the different approaches.

Also the two final steps in an E-LCA following the ISO 14040 standard, impact assessment and interpretation, should be conducted in the light of the existence of the three approaches within LCSA, although LCC does not require an impact assessment step as the impact results expressed as costs are directly measurable. To interpret the results, in alignment with the goal and scope formulation some sort of combination is needed. Within UNEP/SETAC (2011) it is proposed that this should be achieved either by a table with the three outcomes shown side by side, or some ‘traffic light’ design, displaying good and bad performance on a color scale from green to red over yellow for each of the included methodologies. In these approaches, the three perspectives are viewed separately, i.e. the reader will have to make the holistic sustainability consideration her-/himself. In this work the aim is to try to test the aggregation of the three sustainable development perspectives into one LCSA outcome. In this respect, the MCDA methodology offers an interesting possibility for this. Not least as it offers degree of transparency regarding the process to achieve the separate results and via more explicit presentation of values upon which the prioritization between the different sustainability perspectives is based. Using MCDA in the final step of an LCSA has been proposed by several authors (Cinelli et al., 2014) and a number of LCSA-studies have also employed MCDA methodology in this step (e.g. Ren et al., 2015 and Valente et al., 2013).

2.2 POINT OF DEPARTURE IN THIS STUDY

In this work, as already explained, we wish to conduct a LCSA on a selection of vehicle fuels. We aim to follow the LCSA definition LCSA=E-LCA + S-LCA + LCC by including as extensive LCA for the different sustainability aspects as possible. We are to some extent building on earlier work published in the area for the three separate life cycle assessments (see Section 2.1.1).

In the case of E-LCA we seek to include results of a full scale E-LCA. In the E-LCA for vehicle fuels identified in literature, the focus was to a large extent limited to CO_2 emissions (Lazarevic & Martin, 2016), however some studies exist that cover more aspects. Yet, to ensure that the E-LCA results cover a range of environmental impacts, and is comparable when using similar system boundaries and assumptions, specific value chains for the included transportation fuels were assessed using the GaBi software based on lifecycle inventory (LCI) data from Ecoinvent, (2015). We decided to not base our analysis on existing studies as to ensure that the results would be comparable using similar system boundaries and approaches. This E-LCA covers well to bunker station in Sweden i.e., well-to-tank and the refining is assumed to take place in Europe. It should be noted however, that some of the data in these databases are rather old, and in some cases the results are likely therefore not representative of the current situation.

The use phase; i.e., tank-to-wheel was assessed separately and the Network for Transport Measures (NTM) tool called NTM calc (NTM, 2016) was used. The results were weighted using standard methods, and weighting factors for EPS and Ecovalue were used (see Section 3.1.4). The method Stepwise was also used as an additional test. Stepwise yielded notably higher impacts for Brazilian ethanol, this being due to emissions of non-carcinogenic toxic compounds. However, it was not possible to trace the motivation behind these results within the scope of this project and thus the Stepwise results were not included in the following analyses.

The E-LCA is an attributional LCA (which is also the case for the LCC and S-LCA), which means that it accounts for emissions from the activities within the product life cycle only. This is the LCA approach generally used in certification systems and labelling (but it should also be compared to consequential LCA, which includes broader consequences in the system as a result of the product). The attributional LCA perspective chosen imply that indirect land use changes (ILUC) is not considered in the assessment as they represent potential consequences in life cycles of other products than the vehicle fuels.

Our LCC is based on input data from secondary sources documented in a literature review (see section 3.3). The costs do not include subsidies or support and represent an estimate of the different costs associated with the production and transportation of the fuels from well-to-bunker station in Sweden, i.e. well-to-tank. The LCC results give an approximation of the real life cycle costs for the different vehicle fuels. The potential impact of LCC on the LCSA results will be assessed in the sensitivity analysis.

The S-LCA is based on Ekener et al. (2014) but some amendments have been made compared to the approach in that study. Firstly, we wished to add a more elaborated approach for assessment of positive impacts in S-LCA. To date, S-LCA assessments conducted have mainly focused on the negative social impacts, and the positive impacts have been treated as inverted negative ones, or been aggregated with the (often overshadowing, in these approaches) negative ones and thus concealed in the overall outcome. In particular, when assessing social impacts with the Social Hotpots

Database, positive impacts are not at all included in the assessment (Ekener et al., 2014; Ekener et al., 2016). Building on the work in Ekener et al. (2016) we identified a number of positive impacts from the selected vehicle fuels (e.g. the number of employment opportunities was the item added to the S-LCA in Ekener et al., 2014). Secondly, we sought to detail some of the data in the earlier study to make it more relevant. This included classifying the risks into two levels of severity.

The LCA results will be aggregated using MCDA (described in the next section). The LCSA approach used in this report is illustrated in Figure 1.

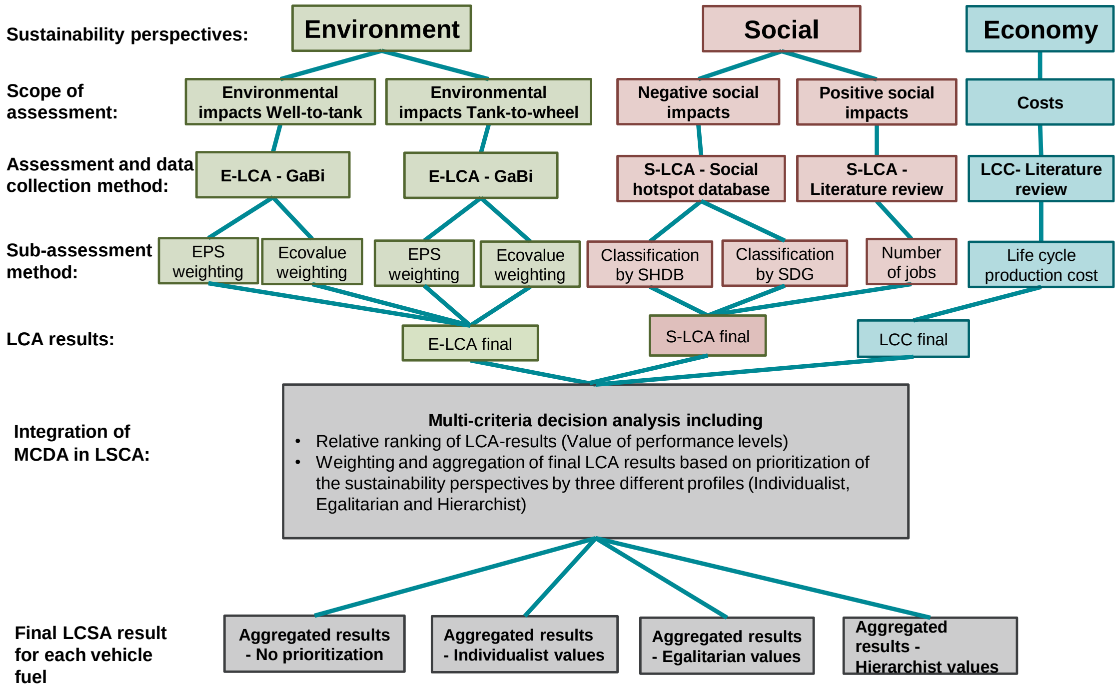

Figure 1. Illustration of LCSA approach used in this report. The number of jobs is used to represent potential positive social impacts.

It should be clarified that there are different system boundaries in the different assessments. The E-LCA includes the entire vehicle fuel value chain up to and including the use phase. However, well-to-tank and tank-to-wheel results are presented separately since the former represents pure ethanol and petrol and the latter ethanol E85 (85% ethanol, 15% petrol) and petrol E05 (5% ethanol, 95% petrol). An estimate of the total potential environmental impact can be represented by a summarized index of the two results which then imply that the production of petrol in E85 is approximated with the production of ethanol in E85.

The LCC represents well-to-tank (Göteborg) for pure ethanol and petrol respectively. The S-LCA does not include the use phase, as this phase is not considered to differ between the various vehicle fuels as much as other phases in terms of social impacts.

The S-LCA, based on the previous study, includes cultivation or extraction, refining/processing and transport. The additional work on the S-LCA conducted in this project, which focus on assessing potential positive social impacts, does not include the last phase in the original S-LCA, i.e. sea transport. Petrol and ethanol refers to the unblended fuels, while in those cases where blended ethanol and petrol is referred to E85 or E05 is used.

2.3 MULTI-CRITERIA DECISION ANALYSIS (MCDA)

Multi-Criteria Decision Analysis (MCDA) is a term describing a family of methods used to support decision making when there are conflicting objectives. The aim is to compare and rank decision alternatives based upon their performance in combination with the preferences of a decision maker. It is a widely recognized approach to manage assessments characterized by multiple attributes (cf. Steele et al., 2009; Cinelli et al., 2014; Niekamp et al., 2015). An MCDA method can for instance be used to combine environmental assessments such as Life Cycle Assessment (LCA) and Life Cycle Cost (LCC) in order to provide a holistic overview and find possible optimal solutions (Niekamp et al., 2015).

The Multi-Attribute Value Theory (MAVT) technique, used in this study, formally maps and transforms different perspectives into a value (utility) function, where the criteria adopts the same, dimensionless value scale. The measures can be used to evaluate or rank alternatives. It is a well-established methodology and the approach assigns a utility value to each option, where the utility is a real number representing how well the option is preferred in comparison to other options. The number is the sum of the marginal utilities from each criterion to the considered option. The values, which after the transformation are in utility units, make comparison between them possible, and also provide information about the relative importance between the criteria (cf., e.g., Linkov and Moberg, 2011).

The idea in MCDA is to be able to aggregate different kind of input, regardless of its format, and rank them based on decision-makers priorities. When data comes in different forms, maximum and minimum values are identified and they are all converted into a comparable scale (normalized), for example into a scale from 0 to 1. In the MCDA techniques, the graphical illustration of these scales is used to identify intervals and/or distances with the aim to establish rankings of the different alternatives, in combination with decision-maker priorities. One important function is often to identify the positions on these scales leading to a shift in the ranking. When the data is uncertain, one may use intervals instead of precise data, to see where in the different intervals a shift in ranking takes place. The result of a MCDA can be stipulated as numerical value, but can also be a verbal description, or other less quantitative way of indicating the impact of a decision option. MCDA can help interpret a mix of quantitative criteria expressed by indicators, qualitative criteria expressed by descriptors, and intermediate criteria expressed by scores (e.g., a scale 0-10). For a more specific and scientific description of the MCDA method used in this study, see Appendix III.

2.3.1 Weighting the three perspectives of sustainability in the LCSA

Different decision-makers, or rather the stakeholders they represent, might have different priorities between the three sustainability perspectives. To consider this potential difference in prioritization in in our LCSA, we used three different stakeholder profiles identified in Cultural Theory (CT) (Hofstetter et al., 2000). These profiles, characterized by their different world views, are used in some E-E-LCA studies as a base for prioritizing between different environmental issues (Basson and Petrie, 2007). They are identified in CT based on the identification of four potential stakeholder profiles on a on a scale of 'grid' in one dimension, and 'group' in the other. The grid dimension represents here structures in society, such as legislation, rules and regulation. The group dimension represents the emphasis put on relations with others. In practice, only three of the stakeholder profiles identified are are used, labelled Individualist, Egalitarian and Hierarchist. A common way to depict them on the two two scales group and grid is displayed in

Figure 2.

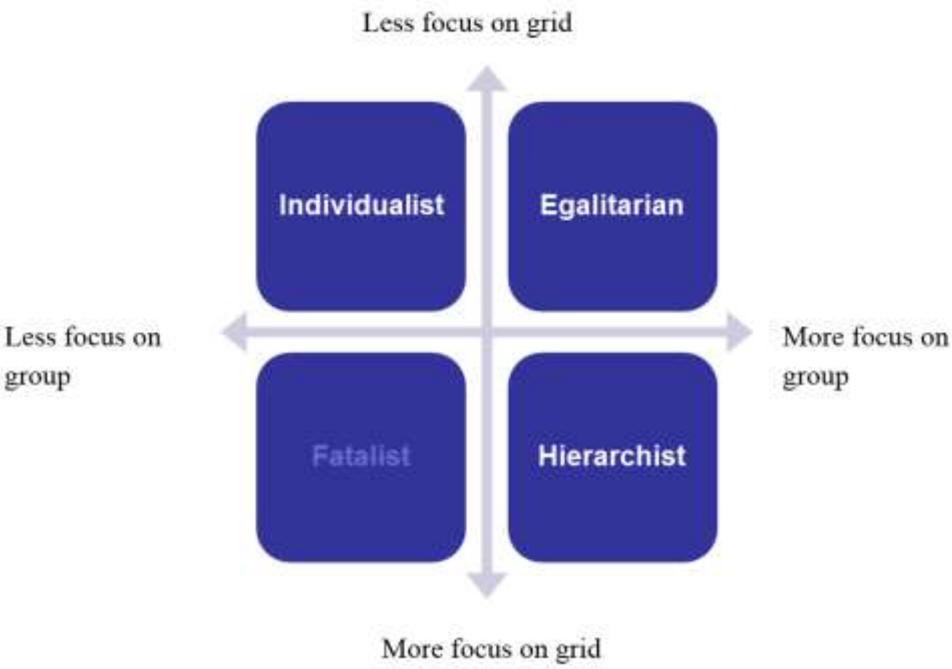

Figure 2. Mapping of different stakeholder profiles on dimensions group and grid, according to Cultural Theory (Conner et al., 2015).

The assumed perspectives and values of these stakeholder profiles, as identified in literature (De Schryver et al, 2011), are displayed in the upper part of Table 1. These values form the basis for our assumptions on the ranking between the three sustainability perspectives for the different stakeholder profiles (lower part of Table 1). This ranking is used in the MCDA assessment.

Table 1. Externally assumed perspectives of the different stakeholder profiles (De Schryver et al, 2011) (upper part, normal text), and own assumptions on priorities between the sustainability perspectives made within this project (lower part, bold text).

	Individualist	Egalitarian	Hierarchist
Group	Weak	Strong	Strong
Grid	Weak	Weak	Strong
Nature view (Nature is…)	…stable and able to recover	…fragile and unstable	…in equilibrium
Priority 1	*Economic*	*Social*	*Environmental*
Priority 2	*Environmental*	*Environmental*	*Economic*
Priority 3	*Social*	*Economic*	*Social*

3 CALCULATIONS AND RESULTS FOR THE SEPARATE SUSTAINABILITY PERSPECTIVES

As specified in the introduction four different vehicle fuel chains are studied; two fossil fuels and two biomass-based fuels:

- petrol refined from crude oil from Nigeria and Russia respectively and
- ethanol derived from sugarcane grown in Brazil and corn (maize) grown in the USA.

The life cycles considered for well-to-tank are displayed in figures 3 and 4, and tank-to-wheel assumptions are specified in section 3.1.3.

3.1 ENVIRONMENTAL LIFECYCLE ASSESSMENT (E-LCA)

There are many environmental LCA studies of biofuels and fossil fuel (e.g., Dones et al. 2007; Jungbluth et al. 2007; ADEME 2010; Börjesson et al. 2010; Eriksson and Ahlgren 2013). A short overview of our review of existing life cycle assessments is presented in Appendix IV. The focus in the E-LCA related work in this study has been to establish a comparative set of data for the value chains of the four different vehicle fuels considered in this study.

3.1.1 Comparable lifecycle inventory (LCI) data for the selected transport fuels

The environmental impacts of the selected transport fuels chains from well-to-tank are assessed using the GaBi software (Thinkstep, 2015), based on LCI data from Ecoinvent. The datasets used in order to extract the results are described in the section below. All datasets used are listed in Appendix V. The LCIA (life cycle impact assessment) results for the impact categories global warming, water consumption and non-renewable primary energy use are presented in Table 2. These impact categories represent the most well-represented impact categories in the existing life cycle assessments identified in the literature review in this project. The E-LCA result presented in Table 2 is compared to result from studies included in the literature review in Table 3 and Table 4. The results for more impact categories are presented in Appendix VI.

Table 2. LCIA results for selected impact categories for the studied transport fuel chains from well to tank.

Fuel	Category	Value	Unit
Petrol – Nigerian oil	Global Warming[1)]	0.029	kg CO2 eq./MJ fuel[4)]
	Water Consumption[2)]	0.034	kg water/MJ fuel
	Non-Renew. Prim. Energy Consumption[3)]	1.35	MJ/MJ fuel
Petrol – Russian oil	Global Warming	0.025	kg CO2 eq./MJ fuel
	Water Consumption	0.073	kg water/MJ fuel
	Non-Renew. Prim. Energy Consumption	1.40	MJ/MJ fuel
Ethanol – Brazilian sugar cane	Global Warming	0.02	kg CO2 eq./MJ fuel
	Water Consumption	0.67	kg water/MJ fuel
	Non-Renew. Prim. Energy Consumption	0.20	MJ/MJ fuel
Ethanol – US corn	Global Warming	0.080	kg CO2 eq./MJ fuel
	Water Consumption	0.81	kg water/MJ fuel
	Non-Renew. Prim. Energy Consumption	0.89	MJ/MJ fuel

[1)] IPCC global warming, excluding biogenic carbon
[2)] Total fresh water use (Thinkstep, 2015)
[3)] Primary energy from non-renewable resources (net calorific value) (Thinkstep, 2015)
[4)] Original results "per kg fuel" for both petrol and ethanol. Lower heating value petrol: 42.5 MJ/kg. Lower heating value ethanol: 26.8 MJ/kg.

Petrol based on Nigerian or Russian crude oil

Ecoinvent provides several life cycle inventories for oil-derived products in Switzerland and Europe, basing all datasets on Jungbluth et al. (2007). The year considered is 2000 and the modelled chain includes oil field exploration, crude oil production, long-distance transportation, oil refining and regional distribution. Moreover relevant production facilities and infrastructure, as well as transport services needed to supply energy and materials, and treatment processes needed for the production wastes are also considered (Dones et al., 2013). The phases included in the LCA modelling are illustrated in Figure 3.

Figure 3. Life cycle phases included in petrol production and distribution.

Country-specific data is used whenever available for crude oil production related activities. Furthermore the allocation of energy use and emissions between crude oil and natural gas under combined production is based on the lower heating values of both. Long distance transportation is based on national and international statistics on imports and exports, and tankers and pipelines are the considered means of transportation from each region producing crude oil to Europe. The refining process is assumed to take place in Europe and given that this activity delivers several intermediate products, allocation by mass is applied to each intermediate whenever possible, since no economic information about intermediate products is available and heating values are quite similar (Jungbluth et al., 2007; Eriksson and Ahlgren, 2013). The regional distribution accounts for transport of the fuel to storage tanks as well as to customers (filling stations, households and companies). Emissions during this phase are modelled on product-specific basis (Dones et al., 2007).

In order to model the production of petrol based on only Nigerian or Russian crude oil it was necessary to perform a modification in one Ecoinvent dataset, chosen to represent the mentioned fuel. The dataset in question is named "RER: petrol, unleaded, at refinery", and it is a cradle-to-gate life cycle inventory of petrol refined in Europe. As explained before, crude oil used in Europe is a mix of oils extracted in several countries. Therefore, in this Ecoinvent dataset, crude oil input is comprised of different shares of oils extracted in different countries, according to statistics of the International Energy Agency (Jungbluth, 2007). In order to represent the hypothetical situation where only Nigerian or Russian oil is used as input, the mentioned dataset was modified accordingly, and the only oil input assumed was instead Nigerian or Russian oil.

The dataset "RER: petrol, unleaded, at refinery", is a LCI of production of unleaded, high-sulphur content petrol, at the refinery; it does not contain any emissions, energy and resource use related to any of the subsequent phases, such as reducing the sulphur content in petrol, regional storage and transport to final consumer. This being said, the modified dataset "RER: petrol, unleaded, at refinery" having Nigerian or Russian oil as the only oil input, was linked to the dataset "RER: petrol, low-sulphur, at regional storage" which contains the emissions, energy and resource use of the aforementioned subsequent phases. Potential double counting was avoided.

Sugarcane based ethanol from Brazil

The Ecoinvent database has several life cycle inventories of biofuels for transport based on different sources and origin. Ethanol from sugar cane produced in Brazil is included specifically and therefore selected for this chain in this study. The LCI in question was compiled by Jungbluth et al. (2007) and the reference year is 2000. The dataset chosen to represent the process, "CH: ethanol, 99.7% in H2O, from biomass, production BR, at service station", accounts for the cultivation of sugar cane in Brazil, its transport to the mill, fermentation and dehydration processes, transport to Europe, regional storage and transport to service station (see Figure 4).

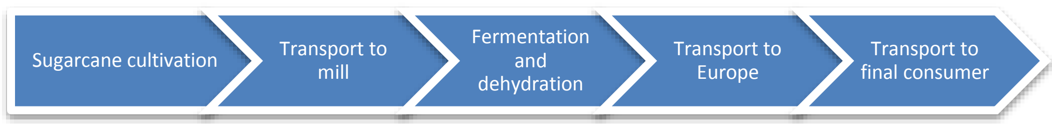

Figure 4: Life cycle phases included in Brazilian ethanol production and distribution

In all stages consumption of raw material, energy, infrastructure and land use as well as emissions to air and water are included. The cultivation is based on average values of studies conducted in different areas of Brazil (including the São Paulo state) and burning field emissions are taken into account. It is assumed that during ethanol production, bagasse is burned to produce electricity consumed during the process. A small share of surplus electricity is supposed to be sold to the grid. In order to tackle this multi-output situation, economic allocation between ethanol and electricity is applied. In order to reach Europe, ethanol is transported by truck, pipeline and rail to the Brazilian coast where it is loaded to an oversea tanker. Once in Europe (Rotterdam), barge, truck and rail transports are considered before reaching final destination in a regional storage (Switzerland, in this case). No modifications were made to this dataset.

Corn based ethanol from the USA

The LCI data for corn based ethanol from the USA in the Ecoinvent database are also based on Jungbluth (2007) and the reference year is 2000. The dataset accounts for the cultivation of corn in the USA, transport to the distillery, pretreatment, saccharification, fermentation, distillation, dehydration and stillage treatment processes, drying of co-products (including DDGS), transport to

Europe, regional storage and transport to service station (see Figure 5). Consumption of raw material, energy, infrastructure and land use as well as emissions to air and water is included in all stages.

Figure 5. Life cycle phases included in US corn ethanol production and distribution

Data for cultivation is based on statistics and are representative for 91% of the area cultivated with corn in the USA. Drying of grains is taken into account. The ethanol production process is based on dry-milling technology and the dehydration process is assumed performed by means of molecular sieves. Economic allocation is used between ethanol and DDGS. In order to reach Europe, ethanol is transported within the USA from the mid-west, by rail and road, to the east coast where it is loaded to an oversea tanker. Once in Europe (Rotterdam), barge, truck and rail transports are considered before reaching final destination in a regional storage (Switzerland, in this case). No modifications are made to this dataset.

3.1.2 Limitations linked to E-LCA well-to-tank results

A general limitation placed on the applicability of E-LCA results for this project, is the somewhat dated input data in the databases utilized (these are mainly from year 2000). While these data are sufficient for method-development work (the central aim of this work) it limits the 'real world' validity of the actual numbers yielded by the LCA tools.

For example, the climate impact of corn-based ethanol, in particular, has been indicated to have declined. This is partly due to changes in regulations and other policies, to a large extent driven by climate change considerations, but it is also driven by large-scale changes in energy markets. Biomass or biogas displaces to some extent natural gas for in-plant heat and power production in ethanol production plants in the US. In addition, natural gas has markedly displaced coal as the primary input for production of grid electricity in corn production regions of the USA, which reduces the climate impacts associated with purchased electricity (an important but subsidiary input to the production plants).

Another example is related to changes linked to the harvesting of sugarcane and ethanol production in Brazil. There have been restrictions in some areas (Sao Paulo) aiming at a phase out of the practice of burning sugarcane fields before harvest (i.e., increased use of mechanical harvesting). In addition, sugarcane production in Brazil has expanded considerably since 2000 and new mills are located, mainly, in vicinity to areas where mechanical harvesting is suitable.

Differences between LCA results for the same transport fuels from different studies can also arise from factors such as differences in the natural and techno-economic system modelled (e.g., origin of oil/feedstock, refining/milling technology, oil/fuel/by-products price and supply and demand) and methodological factors (e.g., data quality, by-products allocation, and system boundaries).

From Table 3, where the results are presented in "per MJ of fuel" using the low heating value for petrol of 42.5 MJ/kg (Jungbluth et al., 2007) it is indicated that the Ecoinvent results for petrol are somewhat higher than the corresponding results obtained by ADEME (2010) and the interval compiled in Eriksson and Ahlgren (2013) that are described in Appendix IV. However, the result can still be considered as in line with other studies, and is deemed adequate for this study.

Table 3. Comparison of results in the case of petrol based on Nigerian and Russian crude oil.

Fuel	Category	Value	Unit
Petrol – Nigerian oil (Ecoinvent, 2015)	Global Warming	29	g CO2 eq./MJ fuel
	Non-Renew. Prim. Energy Consumption	1.35	MJ/MJ fuel
Petrol – Russian oil (Ecoinvent, 2015)	Global Warming	25.2	g CO2 eq./MJ fuel
	Non-Renew. Prim. Energy Consumption	1.40	MJ/MJ fuel
Petrol (ADEME, 2010)	Global Warming	15.5	g CO2 eq./MJ fuel
	Non-Renew. Prim. Energy Consumption	1.22	MJ/MJ fuel
Petrol (Eriksson and Ahlgren, 2013)	Global Warming	6.7 - 27	g CO2 eq./MJ fuel
	Non-Renew. Prim. Energy Consumption	1.04 – 1.3	MJ/MJ fuel

For sugarcane based Brazilian ethanol the Ecoinvent data based results are in line with the result from the study by Ademe (2010) (see Appendix IV) for both global warming and non-renewable energy consumption (see Table 4). On the other hand, the global warming result for US corn based ethanol in the Ecoinvent case is higher than the corresponding result in Kim and Dale (2008), presented more in detail in Appendix IV. Ecoinvent uses an economic allocation approach to deal with the co-product DDGS, while Kim and Dale (2008) applies the system expansion approach, which leads to lower net total results. We have made are brief literature search trying to find more recent data for US corn ethanol in order to see whether the impacts may have changed since, but the picture from that was mixed.

Table 4. Comparison of results in the case of ethanol based on Brazilian sugarcane and USA corn.

Fuel	Category	Value	Unit
Ethanol – Brazilian sugarcane (Ecoinvent, 2015)	Global Warming	20	g CO2 eq./MJ fuel
	Non-Renew. Prim. Energy Consumption	0.21	MJ/MJ fuel
Ethanol – Brazilian sugarcane (ADEME, 2010)	Global Warming	25.3	g CO2 eq./MJ fuel
	Non-Renew. Prim. Energy Consumption	0.18	MJ/MJ fuel
Ethanol – US corn (Ecoinvent, 2015)	Global Warming	81	g CO2 eq./MJ fuel
	Non-Renew. Prim. Energy Consumption	0.89	MJ/MJ fuel
Ethanol – US corn (Kim and Dale, 2008)	Global Warming	57.1	g CO2 eq./MJ fuel
	Non-Renew. Prim. Energy Consumption	0.75	MJ/MJ fuel

3.1.3 Environmental impacts from use of fuels in transport

The results presented in the previous sections concern the lifecycle from origin well or extraction point to a bunker site in Gothenburg, i.e., well-to-tank. The last phase of the life-cycle, the use of the fuels in transport, was however not included in that analysis. At the same time the impacts from the end use phase will have major impacts – including among other things the impacts from fossil carbon versus biomass based carbon. This last part is generally called tank-to-wheel and E-LCA data for this part of the chain is also included in this study. The associated assumptions are described below.

The fuel chains studied here have either an origin in fossil or biomass resources and as the end-use of the fuels in a vehicle is not included in the E-LCA above the impact of net carbon emissions to the atmosphere from the burning of the fuels are not included. Biomass based carbon can be considered to not add net carbon to the atmosphere as this is already part of the biosphere, while for fossil-based carbon there is a net addition of carbon. From a climate impact perspective the distinction between a fossil fuel based or biomass based fuel is large and is also a divider in policy discussions (see for example European Parliament 2009a; European Parliament 2009b). An indicator for these impacts is the tank-to-wheel phase of the lifecycle.

There is a notable difference in fuel found for the tank-to-wheel phase and the previously presented fuels. While for the previous sections presentations have been for pure ethanol and pure petrol the fuels purchased at a petrol station would typically be a blend. In Sweden there are no pure ethanol cars. Instead, ethanol-based vehicle fuel (E85) in Sweden is a blend of 85% ethanol and 15% petrol, whereas petrol has a low blend of ethanol of 5% (sometimes referred to as E5). These blends can vary from the suppliers depending on whether it is summer or winter. The results for comparative key indicators for potential associated impacts from tank-to-wheel of ethanol (E85) and petrol (E5) car based on data from NTM (2016) is presented in Table 5.

Table 5. Comparative key indicators for potential associated impacts from tank-to-wheel of ethanol (E85) and petrol (E5) car (NTM, 2016).

	CO2e [kg]	Energy [MJ]	Volume [l]
Petrol (E5), Car Euro 5, average road (2%), 1 km	0.1	2	0.07
Ethanol (E85), Car Euro 5, average road (2%), 1 km	0.04	2	0.1

The associated carbon dioxide equivalents emissions would be about 2.5 times higher for the petrol (E5) than for the ethanol (E85). Note that these are data for blended fuels and in similar vehicles, thus the carbon dioxide emission for the ethanol fuel is higher than would be the case for unblended ethanol and the associated emission from the petrol is lower than would be the case for unblended fuel. The tank-to-wheel data provides critical sustainability information as it is linked to impacts on climate, but also on local environment.

3.1.4 E-LCA Weighting

In LCA there are several approaches to create an aggregated result in the form of a single aggregated index for the whole lifecycle including the whole set of all indicated environmental impacts. Typically these approaches include weighting factors that are multiplied to the emissions (CO_2, CH_4, NO_X, etc.) or environmental impacts (climate change potential, eutrophication potential, etc.) of the life cycle (Finnveden 1997; Finnveden et al., 2006; Ahlroth et al., 2011; Johnsen and Løkke, 2012; Finnveden et al., 2013; Ahlroth 2014). The result of each multiplication would have the same unit and be possible to summarize. Weighting factors are based on different frameworks that aim at representing the environmental burdens or costs in different ways. For example the weighting system referred to as Ecotax uses the Swedish environmental tax system as a basis for calculating the weighting factor (Finnveden et al., 2006). With another framework the weighting factors and, hence, the weighting results can be very different.

The results from the E-LCA of the selected four fuel value chains in this study have been subject to a weighting process. Here we have used the monetary weighting methods Environmental Priority Strategies 2015d impact assessment method (EPS) (Steen 1999a; Steen 1999b; Steen 2015a; Steen 2015b) and the Ecovalue system (Ahlroth 2009; Ahlroth and Finnveden, 2011; Ahlroth et al., 2011; Finnveden et al., 2013). Ecovalue include three different weighting sets (low, average and high) and all have been applied in our analysis. The three weighting sets aim to capture the uncertainty range when assigning environmental costs to certain environmental impacts. The weighting sets named low and high are expected to cover the lowest and highest weighting factors that are reasonable within the Ecovalue framework. The weighting set named average is expected to capture the "best" estimate and can be referred to as the default.

The aggregated E-LCA results for well-to-tank and tank-to-wheel after weighting is presented in Table 6 and Table 7. Note that the results for tank-to-wheel will not depend on the origin of the oil or ethanol. Due to the somewhat old available data combined with the risk of results being presented out of their context, the result is not presented in figure format. Detailed E-LCA results after weighting for different impact categories and the different weighting methods are presented in Appendix VII. Since the E-LCA results for well-to-tank and tank-to-well does not represent the exact same fuel it is not correct to summarize these two. However, a combined index of the two results will represent an estimate of the magnitude of the total potential environmental impact, but the climate impact of ethanol will be somewhat exaggerated.

The well-to-tank results provide a mixed picture, with some renewables having a lower environmental impact than fossil fuel, but also the other way around. The latter depends partly on the cultivation of feedstock for some biofuels, associated with a larger land demand than in the case of fossil fuels. For the use phase (tank-to-wheel) the renewable vehicle fuels has, as expected, a lower environmental impact than the fossil vehicle fuels. The rough estimate of the total potential environmental impact indicates again a mixed picture, with some fossil fuels and some renewable fuels being on top of environmental performance (valid for both weighting methods).

The E-LCA results are somewhat surprising since one could expect the biofuels to result in the lowest total environmental impact. However, the order among the alternatives for the estimated total potential environmental impact is in line with some other studies. For example, Yang et al. (2012) finds particular E85 fuels associated with higher weighted environmental impact than petrol. However, these results for the biofuels could depend on the somewhat old data used, where the environmental impact found in this study might be higher than actual impacts from current production (as already discussed in Section 3.1.2). This might be true for corn based ethanol as discussed before, even though a brief literature review conducted was, as earlier mentioned, not able to confirm that. It might also be valid in the case if ethanol from Brazilian sugarcane, showing relatively high emissions of particles and dust likely from the assumed burning of sugarcane in the fields. More recent data, based on less or no burning of sugarcane in the field, could show an improved environmental performance for sugarcane based ethanol.

The relatively small difference in E-LCA results for ethanol from Brazilian sugarcane and the fossil vehicle fuels should be further investigated before any firm conclusions can be made. The use of other weighting methods for E-LCA than the one chosen here might also result in other relations between the included vehicle fuel options.

Table 6. Results from applying weighting methodology to the E-LCA results – well to tank.

Method	Unit	Petrol, Nigerian oil	Petrol, Russian oil	Ethanol, Sugar-cane, Brazil	Ethanol, corn USA
EPS	TOTAL (Euro/MJ)	0.0248	0.0287	0.0238	0.0356
Ecovalue (average)	TOTAL (Euro/MJ)	0.0161	0.0199	0.0204	0.0380
Ecovalue (low)	TOTAL (Euro/MJ)	0.0064	0.0097	0.0060	0.0159
Ecovalue (high)	TOTAL (Euro/MJ)	0.0256	0.0295	0.0325	0.0595

Table 7. Results from applying weighting methodology to the E-LCA results – tank to wheel.

Method	Unit	Petrol (E5)	Ethanol (E85)
EPS	TOTAL (Euro/MJ)	0.0065	0.0023
Ecovalue (average)	TOTAL (Euro/MJ)	0.0150	0.0054
Ecovalue (low)	TOTAL (Euro/MJ)	0.0008	0.0004
Ecovalue (high)	TOTAL (Euro/MJ)	0.0293	0.0103

3.2 SOCIAL LIFECYCLE ASSESSMENT (S-LCA)

3.2.1 Social and socioeconomic impacts – assessment of potential risks

In an earlier paper (Ekener-Petersen et al., 2014) potential social and socioeconomic impacts of various biofuels and fossil fuels were screened by applying S-LCA methodology. Data for the screening were taken from the Social Hotspots Database (SHDB) (Norris et al., 2011), where social data on country and/or sector level can be found. The data in SHDB is organized in five social categories (Labor rights and Decent Work; Health & Safety; Human Rights; Governance; Community Infrastructure) with 22 social themes linked to them. Each social theme is in turn measured with a number of indicators. For each indicators, the risk for a negative social impact to occur on that indicator for a given country/sector combination is given as. The risk level is assessed based on collected data on the indicators and is defined separately for each indicator, generally with a kind of normalization approach, and is expressed as low, medium, high or very high risk. More information on the risk assessment for each individual indicator can be found in the supplement documentation for SHDB (SHDB 2016).

In the previous study, only high and very high risks identified in the included life cycle phases for a fuel chain were considered in the assessment, to limit the amount of data. These high and very high risks were listed, and the results were displayed by counting the number of high and very high risks for each fuel chain. The outcome of this exercise for the four fuel chains selected in the current study is illustrated in Figure 6. For example, in the case of petrol based on oil from Russia, in total more than 180 high and very high risks for negative social impacts on people and societies in the considered lifecycle phases were found.

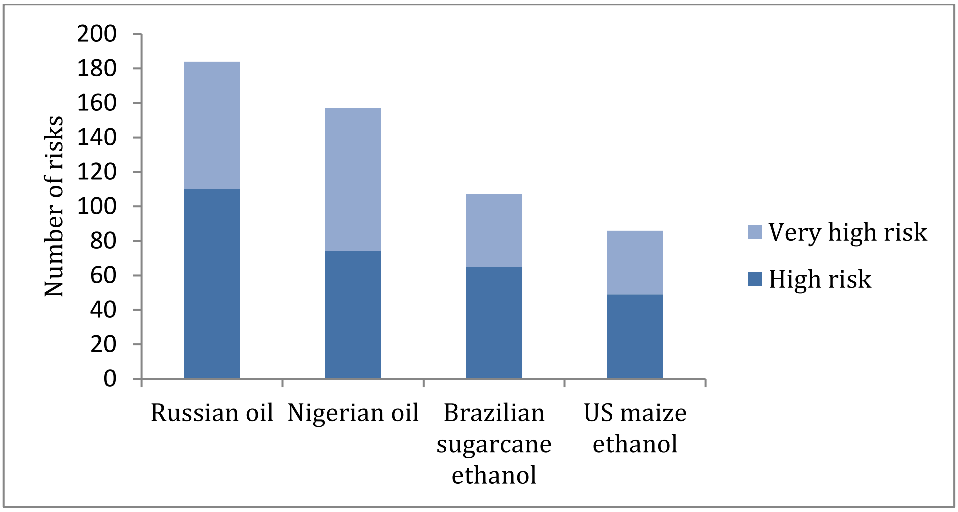

Figure 6. The number of very high and high social risks for the considered vehicle fuels, based on the results in an earlier project (Ekener-Petersen et al. 2014).

Some limitations to the results are that there are only 57 sectors included in the GTAP (Global Trade Analysis Project) database (GTAP 2016), on which the SHDB builds. As the GTAP database covers complete bilateral trade information and thereby represents the whole world economy, the sectors are highly aggregated. This means that each sector in GTAP includes several sub sectors, and the sector data collected might not be representative for the specific sub-sector in question. Another limitation is that, despite the ambition to include sector level data in the SHDB to as large extent possible, at present some data is only collected on country level, due to data deficiencies. Some data is by definition country level data, such as national legislation. Still, there is a possibility that some indicators identified as having a high of very high negative social impact on country level might in fact be non-existent in the sectors relevant for the actual life cycle examined.

3.2.2 Addressing the differentiation among the social risks

As elaborated on above, the results from the SHDB has some important limitations. One of these is the fact that all the risks assessed in the database are considered equal. As stated before, in the SHDB the risks of negative social impacts are structured in five social categories – Labor rights and Decent Work; Health & Safety; Human Rights; Governance; Community Infrastructure. To better distinguish between the social impacts in terms of severity, it would be useful to be able to rank them by marking the risks of social impacts that are perceived as more severe than others. In fact, not differentiating between the types of risks, as presently done, where very severe risks with devastating impacts on human well-being are counted equal to risks with more limited implications, may result in a skewed result considering the effect on human well-being.

However, any weighting exercise is inevitably based on value judgments. In the SHDB, a Social Hotspot Index is calculated, besides the risk assessment for individual indicators used in our assessment. In this index, some rough weighting has been performed by assigning a factor 1.5 to issues that are considered most important, while others count as 1. The selection of indicators considered more important than others is not motivated in any documentation of the SHDB database, and therefore it is unknown which values are underlying this selection. Yet, we have chosen to use this differentiation between risks as one input in our work.

The other approach chosen to differentiate between the risks in terms of severity is based on the Sustainable Development Goals (SDG's) adopted by the UN general Assembly in September 2015 (UN, 2015b). This is a recent, global and consensus based document, and represents a framework mirroring close-to globally agreed goals and aspirations linked to sustainable development. To create a differentiation between the risks based on SDGs, we assessed whether the risks were focused in the SDGs by being explicitly mentioned in the SDG targets. Our weighing approach is presented in Table 8.

Table 8. Differentiation of risks by their severity and corresponding weighting factors.

	Impact differentiation based on SHDB	Impact differentiation based on SDGs	Impact weighting factor
Severe impacts	Impacts assigned with the multiplying factor 1.5 in the Social Hotspot Index	Impacts directly addressed in any of the SDG goals or targets	1.5
Less severe impacts	Impacts without multiplying factor in the Social Hotspot Index	Impacts not directly addressed in the SDG goals or targets	1

The differentiation for the considered fuel chains by classifying the risks according to SHDB Index, as outlined above, resulted in the following distribution between high prioritized risk – weighted with 1.5 - and other risks (Figure 7).

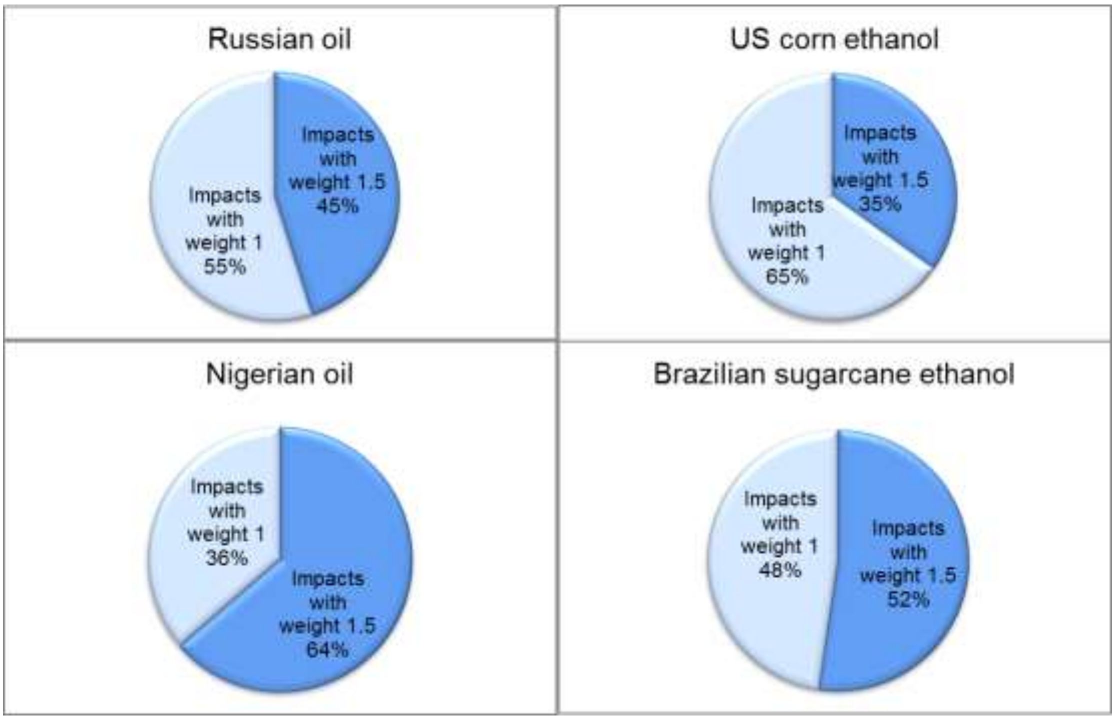

Figure 7. The share of high-prioritized social risks according to the SHDB Index for the fuel chains.

The differentiation for the fuel chains by classifying the risks according to the SDGs resulted in the following distribution between high prioritized risk and other risks (Figure 8).

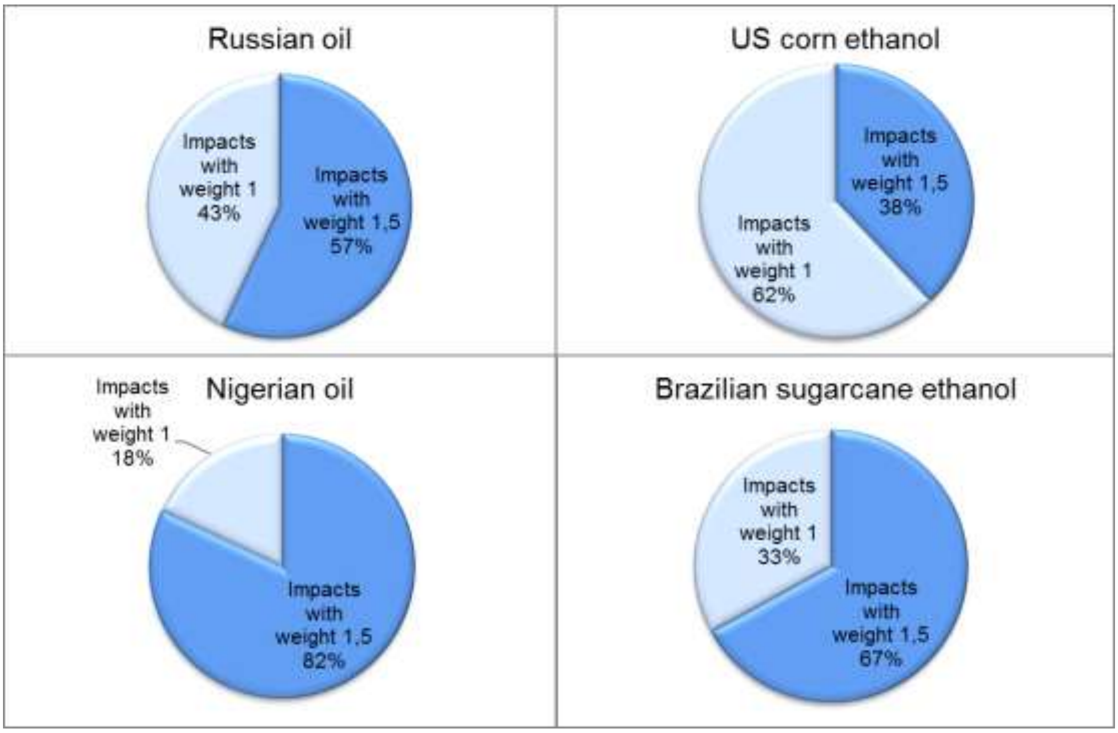

Figure 8. The share of high-prioritized social risks based on the SDGs for the fuel chains.

As can be seen, the high level risks are more present in the fuel chains of Nigerian oil as and the Brazil sugarcane ethanol in both classifications. The smallest percentage of high level risks is displayed for US corn ethanol. This differentiation is taken into account in the MCDA and is thus accounted for in the final results.

We also tried to differentiate based on the ratio between sector/country data for the different fuel chains, where a higher level of sector data would be considered more relevant and less uncertain. However, the result showed that the risks were based largely on country data in all chains (Table 9). It was concluded as a less useful base for differentiation and was thus not included in the further analysis.

Table 9. Percentage of identified risk with data on sector level, per fuel chain.

Fuel chain	% of identified risks with sector level data
Petrol based on Russian oil	4
Petrol based on Nigerian oil	10
Ethanol – Brazilian sugar cane	22
Ethanol – US corn	4

3.2.3 Positive aspects

Positive impacts are not assessed or represented within the SHDB. Thus introduction of such aspects will have to be done separately and in addition to the S-LCA results. There are a number of potential positive impacts, such as income opportunities, export opportunities, access to training and different forms of social safety and infrastructure networks that potentially can result from the value chains of the studied vehicle fuels. In the paper (Ekener et al., 2016) different approaches to better identify and take into account positive impacts in S-LCA was discussed. Further, the problems linked to aggregating positive and negative social impacts were also addressed in the article.

The positive aspects for vehicle fuels are in many cases local impacts such as consequence of processing biomass to bioenergy products, or from service companies linked to the oil sector. Individuals affected by activities in a specific sector may be grouped on two different levels. Firstly, those that are directly involved in the sector. These would typically be identified in for example labour statistics. Secondly, individuals involved in supporting organizations linked to that sector would be impacted; technical service of the equipment, transportation requirements and other professional services are found. As an example, in official statistics, 21 797 individuals work in the oil sector in Nigeria, while the figure including indirect jobs would be around 100 000 jobs (Alemu, 2015).

The total estimated number of direct and indirect jobs associated to the studied vehicle fuels is presented in Table 10. The national population in working age and the number of jobs per PJ associated with the studied vehicle fuels is presented in Table 11.

Table 10. Number of direct and indirect jobs associated to the different fuel (production country). References used are specified in the table.

Country	Production [million ton]	No of jobs	Source
Nigeria (oil) Russia (oil)	110.9 528.6	130 000 – 150 000* 610,000 – 730 000* 660 000**	*About 5-6 million people worldwide in the sector gives 1,16–1,39 jobs/ton produced. We estimated the number of jobs based on the national production times this factor. Volumes from (IEA, 2015), factors are based on various sources triangulated towards known numbers (e.g. USA). **Russia specific number (Pozharnitskaya and Tsibulnikova, 2014)
USA (ethanol)	44.2	357 407	Volume and number of jobs (RFA, 2016)
Brazil (ethanol)	21.2	503 000	Volume (RFA, 2016) and number of jobs from (IRENA, 2015)

Table 11. Population in working age and jobs associated with the fuels (UN, 2015).

	Nigeria	Russia	USA	Brazil
Population (15-64)	97,067,000	100,251,000	213,218,000	143,679,000
Production (PJ) *	4,993	23,786	1,110	532
Share of workforce in sector	0.14%	0.66%	0.17%	0.35%
No of jobs/PJ	28	28	322	946

* Ethanol 25.1 GJ/ton, Crude oil 45 GJ/ton

The highest share of a country's workforce involved in any of the vehicle fuel related sectors is in the oil sector in Russia, while the share in Nigerian oil sector is the lowest. Looking at the number of jobs associated per functional unit of the fuel in each country of origin the bioenergy stands out as much more labour intensive than fossil based fuels.

Both ethanol and oil products are typically, at least partly, exported and thus generate export income for the countries. The export income may provide means for development in the country. The export value and share of total exports for the vehicle fuels included in the study is presented in Table 12. As indicated, the fossil fuels represent large export products. For Nigeria the oil exports represents a very high share of the total export value. Also Russia is highly dependent on the oil exports. Export of ethanol represents for both USA and Brazil represent a marginal share of the total export. The level of dependency that both Nigeria and Russia are showing on oil exports would also indicate that their economies are exposed to changes in oil prices.

Table 12. Export value and share of total exports for crude and refined petroleum (OEC, 2016).

	Nigerian oil	Russian oil	USA corn ethanol	Brazil sugarcane ethanol
Export value (billion USD)*	78.2	191.9	1.96	0.99
Export share*	78.2%	43.0%	0.13%	0.43%

*(OEC, 2016)

The export market values are not included in the analysis due to the complexity of interpreting the final impact on social conditions from export, i.e. what would be a positive impact and what actually represents a risk. Thus, only employment generation (expressed as number of jobs per PJ fuel) is included as representative of the potential positive impacts in the forthcoming analyses. To quantify positive aspects might thus be a difficult task also in the cases where there are relatively large amount of statistics.

3.2.4 Handling positive and negative impacts

Ekener et al. (2016) discuss an ethical aspect of the handling of negative and positive impacts. If they are allowed to balance each other out, producing a neutral result, this can be a problem as the negative and positive impacts might not affect the same stakeholders. Even if they do, they cannot be assumed to outweigh each other without considering the views of the afflicted stakeholder. For example, suffering from occupational health impacts can most likely not be outweighed by improved take-back practices for consumers wanting to return purchased items. For a more detailed discussion on how positive social impacts can be taken into consideration along with negative impacts in SLCA in the case of vehicle fuels, see Ekener et al., 2016.

One way of handling this problem is the use of MCDA, as is done in this study. MCDA methods offer different ways of aggregation, providing transparency on the way it is done. The way positive and negative social aspects are aggregated in this study described in section 4.1.

3.3 LIFECYCLE COST (LCC)

The cost of fuels can be assessed in different ways. Several studies look at costs associated to production of petrol (Hackney and de Neufville, 2001; Goedecke et al., 2007; Restianti and Gheewala, 2012; IEA, 2015) and there are also studies looking at the costs linked to the production of ethanol (Hackney and de Neufville, 2001; Goedecke et al., 2007; Bai, 2009; Mulugetta, 2009; Bai et al., 2010; Edenhofer et al., 2011; IRENA, 2013).

The approach applied here to provide comparable indicative LCC data for the selected transport fuel chains is an assessment where the different parts of the life cycle of the fuel have been associated with a cost. The results include costs for raw material extraction/growing and processing of raw material to fuel (assumed to take place in the country of origin), including costs for capital goods, inputs, labour and transport within and between these parts and also a bulk sea transport from each country to Gothenburg, Sweden. This analysis was based on a literature study combining data, in order to get results representing equal and comparable systems. The LCC value gives an indication of the total cost for production of a certain unit of product. LCC provides an opportunity to make comparative analysis of different investment options and give details on where costs are generated along the value chain.

There is an expansion of the LCC approach that includes internalization of externalities in the assessment (Schau et al., 2011), but there are few studies carried out presenting environmental LCC results on biofuels and fossil fuels. However, the monetarization of external environmental effects is already included in the E-LCA and should not be included specifically in the LCC in order to avoid double counting (Suwelack and Wüst, 2015). As indicated earlier, the LCC results included in this study should be regarded as an indicator representing the economical aspect of sustainability in LCSA. But there is a risk that the LCC-results anyhow to some extent include internalized environmental costs as e.g., environmental taxes even if the ambition has been to avoid this.

3.3.1 Petrol based on Nigerian or Russian crude oil

The main data source for petrol (both Nigerian and Russian) came from Luo et al. (2009), who presented a general value for the cradle-to-gate costs of petrol production. Due to lack of available data we did not find any appropriate way to differentiate the LCC cost of petrol production between

Nigerian and Russian oil. Instead, this general value was used for both countries. The only difference in LCC calculations between petrol from Nigeria and Russia is the small cost for transport to Gothenburg (see Table 13 and Table 14).

Table 13. LCC of petrol with origin in Nigerian crude oil (Luo et al., 2009).

LCC petrol – general	0.59 USD/kg
Exchange rate	1.1 USD/€
Energy of petrol	43.1 MJ/kg
Cost for bulk transport to Göteborg	0.0008 €/MJ
LCC Nigerian crude oil	**0.0132 €/MJ**

Table 14. LCC of petrol with origin in Russian crude oil (Luo et al., 2009).

LCC petrol – general*	0.59 USD/kg
Exchange rate	1.1 USD/€
Energy of petrol**	43.1 MJ/kg
Cost for bulk transport to Göteborg	0.00014 €/MJ
LCC Russian crude oil	**0.0126 €/MJ**

3.3.2 Sugarcane based ethanol from Brazil

The ethanol from Brazil is produced using sugarcane as feedstock. Typically the ethanol factory also co-produces sugar and electricity. This means that the LCC will need to divide investment costs between these three products. The capital investment costs in the LCC data provided by Luo et al. (2009) have been divided, based on economic value, so that only the part belonging to the ethanol production is included here.

Data on costs for production of ethanol from Brazil was taken from Luo et al. (2009). Based on the data collected the life cycle cost for ethanol from sugarcane feedstock Brazil will be 0.0111 €/MJ (see Table 15).

Table 15. LCC of ethanol based on sugarcane Brazil (Luo et al., 2009).

LCC ethanol Brazil	0.30 USD/kg
Exchange rate	1.1 USD/€
Energy of ethanol	26.9 MJ/kg
Cost for bulk transport to Göteborg	0.0009 €/MJ
LCC Ethanol based on sugarcane Brazil	**0.0111 €/MJ**

3.3.3 Corn-based ethanol from the USA

The cost for the ethanol produced in the US using corn as a feedstock comes from Pimentel and Patzek (2005). Based on the data collected the life cycle cost for ethanol from corn feedstock USA will be 0.0203 €/MJ (Table 16).

Thus it should be noted that the main data for the ethanol from corn comes from a different source than the other fuel systems presented here. The results should be comparative but there is a certain risk that details in terms of what has been included or not does not correspond exactly.

The reason for the big difference in LCC costs for ethanol from corn and sugarcane can come from the agricultural part, but also the fuel process. Over 60%, of the cost of corn ethanol comes from the agriculture, i.e. raw material production, the rest come from the ethanol production process and bulk transport. The production process for corn ethanol also includes a hydrolysis step, which is not necessary when making ethanol from sugarcane (IRENA, 2013). Unfortunately no similar cost breakdown information was found for the sugarcane ethanol and for petrol.

What comparisons that can be made from the LCC values representing different sectors presented here (i.e., between the fossil and biomass based vehicle fuels) should be further discussed but are considered possible to use as representatives of LCC in this study. For example to what extent the production of the included vehicle fuels is driven by the presented production cost or the impact of support systems etc. remains unclear.

Table 16. LCC of ethanol based on corn feedstock USA (Pimentel and Patzek, 2005).

LCC ethanol USA *	0.45 USD/l
Exchange rate	1.1 USD/€
Energy of ethanol**	21.2 MJ/l
Cost for bulk transport to Göteborg	0.0009 €/MJ
LCC Ethanol based on corn USA	**0.0203 €/MJ**

4 LIFECYCLE SUSTAINABILITY ASSESSMENT (LCSA) – MERGING THE PARTS

4.1 AGGREGATION WITH MCDA

In an LCSA, the results for the environmental, social and economic assessments are thought to be combined into one sustainability assessment. As described earlier, in this study we chose to aggregate the perspectives by MCDA methodology, using the MAVT technique (see section 2.2 and Appendix III). In a MCDA the way criteria included are structured can be displayed in a criteria tree hierarchy, where data for low-level criteria are aggregated in higher level criteria. The criteria hierarchy for this particular study is shown in Figure A2 in appendix III (which can be compared to Figure 1).

The low-level criteria are in this study always weighted equally e.g., in the case of E-LCA the result from EPS and Ecovalue average are assumed to be equally important.

For the S-LCA, the differentiation of risks based on SHDB and SDGs respectively, are also weighted equally and is thus considered equally important. In the next step, the negative (risks) and positive (jobs created per PJ fuel) social impacts are likewise weighted equally. As all data is normalized, i.e. put on a common scale ranging from for example 0-100, before entering the MCDA. The impact from the one indicator showing positive social impacts is assumed equally important to the impact from the whole range of indicators behind the negative social impact. It is thus the average performance of all positive as well as negative indicators that counts, regardless of how many indicators there are on each ‘side’. This can be compared to calculating an arithmetic mean of a number of figures – the number of figures included in the calculation has no impact on the resulting mean. Such an approach for aggregating negative and positive social impacts can be motivated by the fact that the one indicator for positive impacts is supposed to represent a larger number of potential positive impacts not captured specifically in this study.

After the ‘internal’ aggregation, the ‘decision maker’ is asked to prioritize between the three life cycle assessments (E-LCA, S-LCA, and LCC) for the final weighting. This is done using three different perspectives of imagined stakeholders i.e., taking different world views into account (see section 2.2.1). This kind of approach is advocated for by e.g., Hansson et al. (2011).

4.1.1 Outcome from the MCDA

The outcome of the MCDA is visualized in Figure 9-Figure 12. We present the evaluations from the perspective of each stakeholder profile; the Egalitarian, the Hierarchist, the Individualist, along with a case showing the outcome for equal weighting of the three sustainability perspectives. Please note that in these graphs, a higher bar represents a better outcome on sustainability performance. The underlying performance of the fuel on a sustainability perspective is multiplied with the priority for that perspective. Thus, a high priority on a perspective gives that the fuel’s performance on that perspective is having a larger impact on the total sustainability outcome (the final height of the bar) then would its performance have had with a low priority. Graphically, the higher priority held by a stakeholder profile for a certain sustainability perspective; the more enlarged is the part of the bar representing this perspective, relative to the other parts of the bar, i. e. the other perspectives. Due to lack of certainty regarding current representativeness of some of the underlying data, and to avoid misuse of the final outcome, the actual vehicle fuels are not presented in Figures 9-12. For example the climate impact of some of the biofuels has been indicated to have declined in recent years (see more in Section 3.1.2 and 5.1.2).

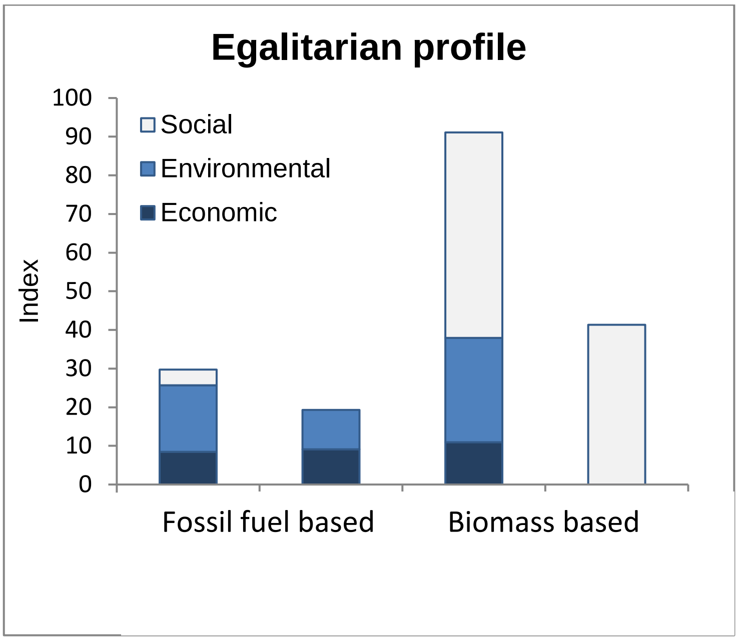

Figure 9. The Egalitarian's evaluation result, with prime priority on the social perspective.

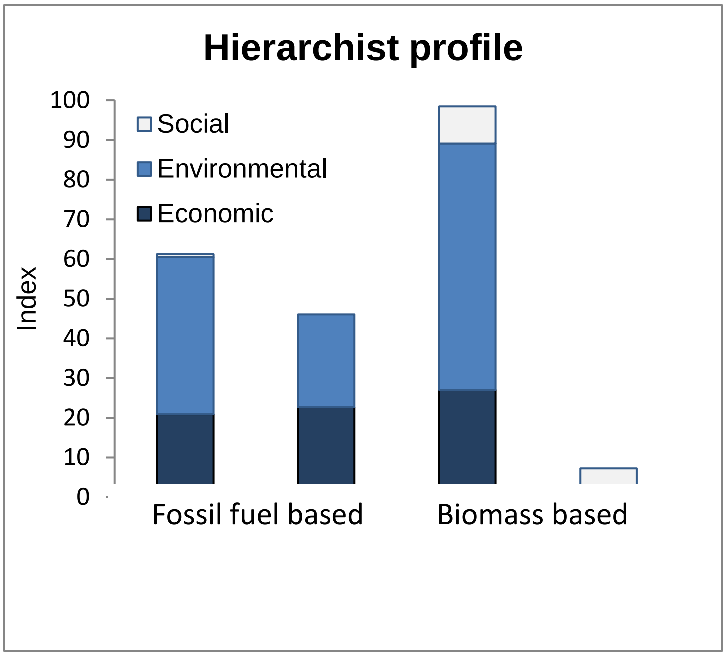

Figure 10. The Hierarchist's evaluation result, with prime priority on the environmental perspective.

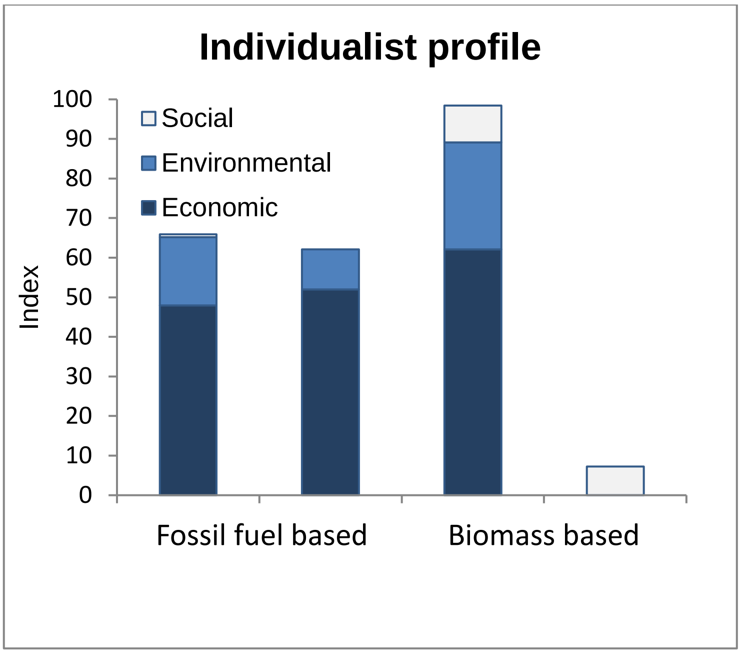

Figure 11. The Individualist's evaluation result, with prime priority on the economic perspective.

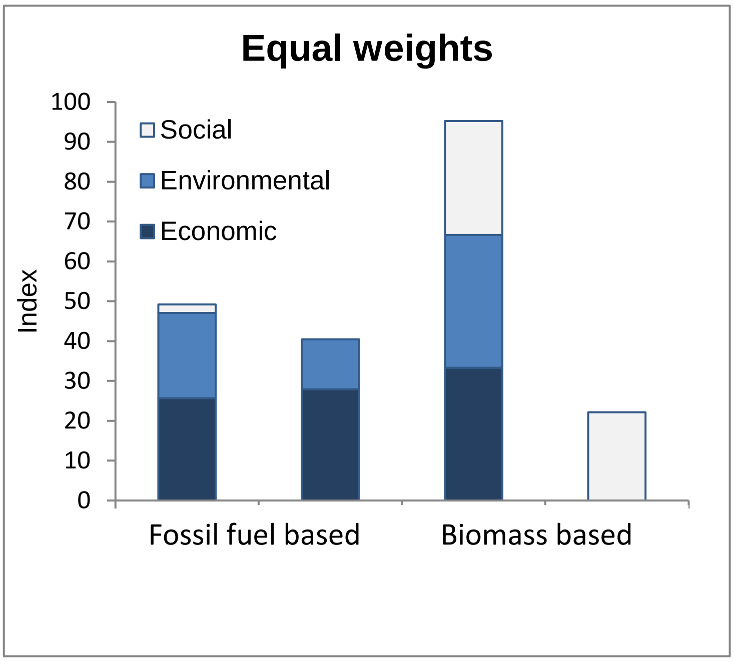

Figure 12. Evaluation result with equal weights for all three sustainability perspectives.

As illustrated, the ranking of the different fuels differs according to the three stakeholder profile considered or in equal weighting. One of the biomass based fuels has the best sustainability performance in all outcomes. When an Egalitarian perspective is taken, the second-best fuel from a sustainability perspective is also biomass based, while the second-best in the other three outcomes are all fossil fuels. Hence, it is clear that the differing prioritization of the sustainability aspects according to stakeholder perspectives markedly influences rank order of the assessed fuels and the outcome of the sustainability assessment.

4.2 SENSITIVITY ANALYSIS

As the approach used in this study is new, and the underlying data is linked to substantial limitations, we have made some sensitivity analyses. Our sensitivity analyses are generally designed by taking out some of the aspects, to see to what extent they are impacting the result.

4.2.1 With and without positive social aspects in S-LCA

In order to determine the impact of including positive social impacts we have taken out the positive aspects in the S-LCA assessment (represented by job creation). It should be noted that in this case, the outcome is not impacted by the (lower) number of positive impacts (we included only one) in relation to the large number of negative social risks. Rather, the strength of the normalized positive impacts is what contrasts the negatives ones. The sensitivity analysis shows that the inclusion of positive social impacts did not in this specific case change the rank order between the alternatives.

4.2.2 With and without production phase data in E-LCA

For biofuels, E-LCA studies generally focus on accounting for GHG emissions (that mainly occur from the use phase), which depends on the related policy development. In order to assess the impact of including full E-LCA results including a range of other environmental aspects or focusing solely on the main GHG emissions aspect we have taken out the results from the E-LCA linked to the production of the fuels. The sensitivity analysis shows that when only the environmental impacts from the use phase are accounted for in the E-LCA, the rank order of the alternatives switched in the case of the hierarchist profile and of equal weights of the sustainability perspectives, i.e. when no prioritizing was done between these perspectives. In these two cases, the two renewables were considered better that the two fossil vehicle fuel chains. When only use phase environmental impact were considered, and the prioritization was based on the egalitarian and individualist stakeholder profiles, the renewables improved their performance relative the fossils, but without causing a change in the rank order.

4.2.3 With and without LCC

We are, as mentioned before, not sure how to interpret the implication of LCC data in a sustainability assessment on a planetary level. Therefore, we chose to do the assessment for the environmental and social perspective only. As the LCC result were more or less the same for all fuel chains except one renewable fuel chain which had much higher costs, the only change was that the difference between this fuel and the other three diminished. The rank order between the fuels did however not change. This outcome was valid for all stakeholder profiles.

4.2.4 With uncertainty ranges for the S-LCA results

Since S-LCA methodology and databases are quite new and not so robust, is it is interesting to assess the impact of the S-LCA results for the LCSA outcome. Therefore, we chose to make intervals of the S-LCA data to address these uncertainties. The outcome of these sensitivity analysis did not change of rank order for any of the stakeholder profiles, including equal weights.

5 DISCUSSION

5.1 FORMING LCSA ANALYSIS

The development and use of LCSA to assess the broad sustainability implication of a product is driven by the wish to take a holistic perspective on sustainability, to avoid problem shifting between perspectives, and in order to shed light upon potential trade-offs between them. However, most LCSA efforts conducted thus far do not go all the way to integrate them to a common outcome. In order to contribute to a holistic view on sustainability in decision-making, we believe it is important to examine different ways of integrating the different sustainability perspectives, in as transparent ways as possible. In this work we attempted to do so and tried one possible approach. As expected, this approach has entailed several challenges, and highlighted new areas of uncertainty, but we hold that it has also contributed with some interesting finding.

We find it particularly interesting to display the different outcomes resulting from the three different worldviews captured in the stakeholder profiles used. This indicates that the ranking of the different fuel chains according to sustainability performance will very likely shift when different prioritizations is granted to differing sustainability perspectives. This underlines that subjectivity does play an important role, and that there is not one right answer to which product has the best sustainability performance. Rather, the ‘right’ choice can be very much dependent on the values of the decision-maker (or the stakeholders he/she represents). This reflects the reality of decision making – where different stakeholders absolutely do have different agendas and priorities. Further, it has the potential to help clarify the tradeoffs between differing choice pathways.

Moreover, this puts light on the issue of how to present the results and not to run into simplified conclusions on the outcome. LCSA results has for example been proposed to be presented in a sustainability dashboard (Finkbeiner et al., 2010), there are LCSA results that are presented in spider diagrams and with smilies. We have chosen to go one step further and actually aggregate the results in and display this aggregation graphically. It should be noted that in theses graphs, the underlying separate assessments are still visible. We have, as expanded on above, also chosen to present three different outcomes of the aggregation, in order to visualize the importance of the underlying value judgements for the outcome. The choice of ways to resent results is often influenced by the tension from the trade-off between accessibility to the result – i.e. the easiness of grasping the final outcome in for example a graph - and the preservation of the visibility of underlying uncertainties and imbedded values. We believe that there is not one right answer to this trade-off, and that the challenge is to achieve as much as possible of both these important features of the result. We find that the MCDA approach adopted here may be a tool that contributes to addressing this challenge by being relatively successful in addressing both these features.

As stated before, the actual outcome for the different fuel chains is not seen as the main result in this work. Rather, we wanted to illustrate the potential for making a holistic sustainability assessment with a life cycle perspective for some vehicle fuels. To be able to draw conclusions on specific fuels chains from this kind of assessment, it is clear that the underlying data needs to be improved. There remains a number of methodological issues that need to be resolved. We present the limitations we see in our results in sections 5.1.1 – 5.1.3 below.

5.1.1 The chosen approach

For E-LCA, the approach was limited by the somewhat dated input data in the databases utilized. This potentially affects the representation of the current situation negatively. In addition the chosen approach limited the possibility to directly consider indirect land use changes which is considered very relevant from a policy perspective. Other assumptions related to system boundaries and allocation might have a large impact on the results, for example related to modelling of the refineries and handling of by-products from the biofuel production as well as related to calculations of the climate impact.

For S-LCA, the inclusion of positive social impacts is judged to be an improvement compared to before. However, more positive impacts could have been included, such as economic development, capacity building, community engagement, infrastructure development and technological development (Ekener et al., 2016) - these not least as they are represented in the social risks database utilized. Also, the aggregation of positive and negative impacts, needs to be further discussed and developed. Here we have sought to examine that which was proposed in Ekener et al. (2016), i.e. to address the positive social impacts together with the negative ones in a MCDA approach.

Moreover, the issue of access to land and it’s potentially negative impact needs to be better covered than it presently is in the used database SHDB. A new database for social data, PSILCA (PSILCA 2016), has recently been launched, and seems promising on this issue as it includes far more granularity on sectors and a different set of criteria. Lastly, the modelled supply chain for the fuels contains rather broad, non-detailed phases, so the granularity of the data is quite coarse.

The lack of appropriate LCC studies and their limitations also represent a weakness for this study.

5.1.2 The data quality, coverage and comparability

For E-LCA, data coverage and comparability is in general quite good. However, lack of certainty regarding representativeness, and to some extent poor data quality is linked to the somewhat dated in-data in the databases utilized (these are mainly from year 2000). There has been some development related to in particular the included biofuel production value chains. Thus, in order to use the actual results presented for the included vehicle fuels and use it for comparison further sensitivity analysis are needed. For example, to what extent do the ranking of the different options and the actual difference between different options remain with new updated input data. The brief literature review conducted for US corn ethanol showed however a mixed picture and did not confirm and clear changes in impact.

In addition, the assessment of impacts of vehicle fuels on ecosystem services is not addressed in this study, implying that it does not cover all environmental aspects. The reason for this is that impacts on ecosystem services and biodiversity are, at present, only to a limited extent addressed in LCA approaches (Arbault et al., 2014; Bringezu et al., 2009; Zhang et al., 2010). More research on the integration of the concept of ecosystem services and LCA is needed before the possibility to include also these aspects in LCSA could be addressed. Ecosystem services linked to biofuels for transport are also assessed in another project (Biofuels and ecosystem services) within the same research program and the reader is encouraged to read the report from that project.

The data in SHDB is considered to be reliable in relation to the data sources from where it is withdrawn. However, the quality of data in these sources might be very varied. Some data is self-reported, and as social data might be politically sensitive, this should be taken into account when considering the results. Further, as discussed earlier, the data in SHDB is to a large extent country level data, which might be fine for some issues determined on a country level, such as national legislation, but might be misleading for issues that are determined on a sector, or even a company level, such as the presence of child labour.

One issue that has been challenging in the work is the interpretation of the economic perspective, the comparability of the presented LCC results between different sectors and to what extent the LCC results includes internalization of external environmental effects. The raised questions are, among others, if a low production cost is good or bad, and for whom. One might argue that a low production cost for delivering the same benefit is an indication of resource efficiency. However, that presupposes that all resources are correctly priced, which might not be the case. When including the economic perspective in an LCSA, a discussion is required on the actual implications of the shown result for sustainability.

5.1.3 Weighting

In the aggregation of E-LCA results on individual environmental indicators weighting is done in order to arrive at one single result for the environmental performance. In this case, it is important that the reasoning behind the weighting is clear and transparent. It is also advised to use different weighting approaches. We have done so, and let them impact the results on an equal share.

For S-LCA, the risks are weighted in a quite simplified way, just grading them on two levels of severity, based on two different sources (SDGs and SHDB). This weighting could be further developed and detailed in the future in the same way as weighting methods for E-LCA have been developed within the research community and made available to a larger audience. However, the most important issue is to make the weighting done transparent for the reader.

5.2 IMPLICATIONS FOR POLICY DEVELOPMENT

This study has been guided by the position that existing transportation fuel-related certification efforts addressing sustainability have significant limitations. Namely, that they have mainly been developed for biofuels at this stage; and while they to some extent address both environmental and social aspects, the latter are vaguely or weakly formulated, and most importantly, do not address these perspective simultaneously. Thus, existing certification schemes give inadequate guidance to market actors and stakeholders, and the sustainability performance of transport fuels risks being lower than anticipated.

Regarding the potential for LCSA to be used to improve policy making aiming at securing the sustainability if vehicle fuels, this study's results confirm that there is a need for insights into socio-economic, social, and environmental aspects linked to vehicle fuels. Further, it takes a view that the importance of including 'social aspects' has been strengthened by the fact that quite different 'relative rankings' between different fuels are yielded by the LCSA method. However, the results of the LCSA analysis, including the weighting process using a multi-criteria decision analysis (MCDA) tool (as they are at this point in time), might not be easy to follow for those not specialized in LCA techniques.

LCSA could be used as an information tool to guide the formulation of policy, and as an assessment tool providing information to assess overall success (or failure) of policy interventions. The LCSA approach applied in this study may be viewed as an 'indicator tool' generating both a 'composite indicator' (the whole) and a combination of a number of 'category indicators' e.g. GHG emissions, social risk, employment. Measures of performance in a number of areas (i.e. indicators of various types) are important to the application of a range of market-based policy instruments and hybrid regulatory systems increasingly applied to address transnational sustainability issues (Harnesk et al., 2015). An important role for assessment such as LCSA is to facilitate awareness of needs for moves to action (or inaction) – as indicators have a role to direct attention towards certain issues that are important. This is particularly true for social wellbeing and socio-economic issues that are not captured by existing metrics. Thus, a possible outcome of using LCSA is to use it to inform the further development of sustainability criteria and related certification systems. For example, negative environmental and social impacts identified by LCSA for any of the involved fuels could be used to set, or raise, the level of related criteria – or at least to indicate where such criteria might be needed. In addition, LCSA represents a tool that brings in several indicator measures and deliberately weighs them together and against each other so as to provide indications of the relative importance of different parameters to overall sustainability efforts.

The method tested has shown that it can highlight 'sustainability' differences between fuels that clearly differ from 'just' an environmental assessment. Differences clearly show up between fossil oils, between biofuels, and between fossil and renewable fuels. Moreover, many more aspects appear possible to capture than those covered by sustainability certifications systems. This indicates that such a method should indeed be able to contribute to development of sustainability requirements.

In policy implementation S-LCA could be used as one of the measures of 'threshold' sustainability in for example a certification system. Implementation of an S-LCA as a tool for 'sustainability' assessment linked to some form of performance threshold would conceivably increase the difficulty of sourcing vehicle fuels from any country with limited governance capacity. It seems feasible that environmental and social pressures to avoid fossil oil, and particularly oil from high-risk countries, could also be markedly enhanced.

One limitation with the LCSA approach is that stakeholders may have difficulties in understanding assumptions and methods used. The complexity of an LCSA effort increases the complexity of understanding the result, and puts pressure on the policy related communication. Robust data quality is important to manage or counteract alteration, or even perversion, of key LCSA messages by other actors. Stakeholders can also be increasingly involved in the prioritization and weighting of different sustainability aspects.

To gain better understanding by stakeholders for complex processes such as LCSA, stakeholder involvement and a continuous communications process might be very helpful (Suwelack and Wüst, 2015). The more complicated the indicators, the greater the need for relating to locally varying contexts and views of the indicator users (Chess et al., 2005). On this note, a need for greater stakeholder involvement in method development for LCSA is a key finding from the study. Thus, this work shows that further improvement of the LCSA methodology is likely to be required if it is to serve a meaningful role in policy formulation in areas affecting vehicle fuels. Further development of S-LCA methodologies would also seem best suited (in the short to medium term) for a focus on their role as an 'information tool'. By this, it is meant that S-LCA could be a very useful input for the scientific field and for the policy sphere when seeking to rationalize which policy directions to pursue regarding transportation fuels. However, the implications of further LCSA method development for future policy making related to vehicle fuels and, in particular the possibility to contribute to the development of sustainability requirements for transportation fuels need to be further assessed.

6 CONCLUSIONS

The main contribution of this project is related to the steps taken towards aggregating the different sustainability perspectives into one holistic outcome for sustainability. As this is done using three different 'worldviews' in prioritizing between the different perspectives, it is shown that the ranking order of the different vehicle fuels chains are quite different for the different stakeholder profiles. This shows that there is not always one single answer for the most sustainable choice between different alternatives. Rather this is dependent on different priorities held by different stakeholders, or the population they represent.

All three underlying lifecycle methods – E-LCA, S-LCA and LCC - have different methodological limitations. Further, they are to various extents relatively new and still under development. One issue identified for all three methods is the lack of robust and updated databases for data collection. This causes problems as the data requirements for assessments are considerable. Thus the importance of data quality is emphasized. The MCDA method offers, however, a possibility to address uncertainties based on variable data quality. In general, the MCDA methodology seems to offer many useful features to ameliorate the effects of a number of data-related complications. As such, it seems to offer a good tool for the aggregation step in LCSA. However, the lack of robust and updated databases implies that the actual LCSA-results for the included vehicle fuels may not be representative of the current situation regarding sustainability performance.

Positive social impacts were handled and integrated separately. By doing this, the influence of the positive impacts on the end result of an S-LCA becomes visible. Although this was done in a limited way in this analysis, it is important to include positive impacts separately in future S-LCA efforts, to be able to distinguish the contribution from positive impacts to the total social impact. This may inform future action to enhance these positive contributions. Yet, the lack of data makes this a difficult task, needing further work.

In the course of this work, the implication of adding the economic perspective of sustainability by using LCC came to be questioned. The interpretation of LCC in the light of the aim for sustainable development seems all but clear. This needs to be further examined.

An important contribution from this work, we believe, is the assessment of both fossil and renewable fuel chains in the same tool. In the current discourse, the renewable fuels are frequently addressed and assessed for their sustainability performance. We believe that in the future, all vehicle fuels should be evaluated on their total sustainability performance at the same level of detail.

Finally, the methodology approach examined in this work may be useful for efforts to leave the 'silo'-thinking that can be found in sustainability discourse behind. Instead of this actors can be motivated to focus on broad, comprehensive sustainability implications of various product life cycles. Once the underlying data and methodology-related limitations have been improved, we believe that LCSA in combination with MCDA has true potential to provide a useful tool for sustainability assessment in a life cycle perspective.

LCSA could be used as an information tool to guide the formulation of policy, and as an assessment tool providing information to assess overall success (or failure) of policy interventions. In conclusion however, we stress that it is important that communication with stakeholders and decision makers should be clear in terms of data quality and of the assumptions and complex assessments required for this assessment method. This is vital if it is to be useful in policy-making and development of specific policy instruments.

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APPENDIX I: SUMMARY REPORT OF WORKSHOP

INTEGRATED ASSESSMENT OF VEHICLE FUELS WITH SUSTAINABILITY LCA - SOCIAL AND ENVIRONMENTAL IMPACTS IN A LIFE CYCLE PERSPECTIVE

- A project financed by the Swedish Energy Agency, and f3 Swedish Knowledge Centre for Renewable Transportation Fuels within the collaborative research program Renewable transportation fuels and systems (samverkansprogrammet Förnybara drivmedel och system) and conducted by the Royal Institute of Technology KTH, IVL Swedish Environmental Research Institute and IIIEE, Lund University.

Workshop conducted: 09:30 to 12:00, Friday 9 September 2016.

PARTICIPANTS

Elisabeth Ekener, Julia Hansson, Philip Peck (Project team)

Stakeholders:

- Per Hanarp, Volvo
- Mattias Backmark, Preem
- Andreas Gundberg, Lantmännen
- Ebba Tamm, SPBI
- Nils Westling, 2050
- Lisbeth Dahlöf, Volvo/IVL
- Emmi Josza, Swedish Energy Agency (Energimyndigheten)
- Tomas Ekbom, Svebio
- Tomas Ekvall, IVL

BACKGROUND

The project is in its final stages of analysis and write up and this workshop was intended to help the project finalize their analysis and rationalize how to best present project results to the intended audience. In order to test the relevance of project outcomes to the intended audiences of the work, and so as to ensure that findings from the work could be packaged in a form suitable for the intended audiences, the workshop focused on presentation of the work and findings. This encompassed both work from a preceding study (underpinning this project), and the prime results yielded by this project.

CONDUCT OF WORKSHOP

The workshop involved an interactive presentation of project work, and substantive findings, with the audience. That is, the audience posed questions, and provided critique, to the project team throughout their extended presentation of the work. Discussion was conducted concurrently with the presentation and feedback. The closing 30 minutes of the 2.5 hour workshop involved a general discussion of the work and how it should (or could) be best presented to the Swedish transportation biofuels community.

The presentation and discussion addressed the following issues:

1. An introduction of the project presenting how the current situation (e.g. where the sustainability of transport biofuels is often simplified to only their ability to reduced greenhouse emissions as compared to fossil fuels. It was explained that the work within this project investigates the feasibility of a new method that offers potential to assess not only environmental impacts, but also captures a number of social and/or socio-economic consequences. This being labelled within the project as a Life Cycle Sustainability Assessment (LCSA) of transport fuels. In such an LCSA, a number of social, economic and environmental parameters are integrated. This work builds on and earlier f3 project: Social and socioeconomic aspects of vehicle fuels.

2. A presentation of work on fuel chains covered in the earlier project and how they led to, and underpin this project;

3. Delineation of potential pathways to integrate both positive and negative environmental impacts (rather than just negative as is often the case)

4. Explanation and presentation of how existing methods for LCSA have been added to within the project as a testing of how to extend such methodologies;

5. Presentation of initial results and explanation of areas of ’counter-intuitive outcomes

An important additional item within this was discussion of the implications of the work for policy development within the area – this being the last portion of the project work, which is still underway.

Key outcomes of the workshop

- When discussing background to the project, it was underlined by participants that ‘perceptions’ of fuel system viability, cost effectiveness, and environmental importance have proven to be leading ‘areas of difficulties’. Simply put, perceptions held in the market may NOT match the reality as seen by the f3 community. This topic of ‘perceptions’ – related to the manner in which LCSA results are communicated, or can be interpreted by stakeholders was taken up as an area of key concern throughout the workshop.

- In particular, there were concerns expressed by several of the participants that the LCSA results shown thus far may be interpreted in a manner damaging to the progress of the biofuels market in Sweden. It was stressed as a key issue that it be made very clear that this project is a METHOD DEVELOPMENT project. As such, a first attempt to examine fuel chains, both fossil and bio, in a new manner. The results cannot (yet) be seen as a proven method to provide a ranking of fuel chain merit based on LCSA.

- The concerns revolved around the issue that actors such as the media might quote the results shown in this study as true representations of ‘reality’ – a reality where a fossil chain can appear superior to a biofuel chain (e.g. Nigerian oil superior to US maize ethanol in this case), even where the project finds this result ‘notably counterintuitive’. The project members hold that such interpretation would be quite inappropriate. The apparently counterintuitive results obtain, rather indicate that the LCSA may be sensitive to a number of key assumptions or simplifications used a part of the method development.

- A need for refinement and testing of such key assumption areas is clearly an area where the project finds that substantive work will be required before this method can be applied as a comparison tool.

- Key actions arising from participant concerns discussed included: a) removing country names from fuels chains – thus presenting them as 'method test examples'; b) explicit reiteration throughout the final report that this project is a method project, not an 'assessment' project.

- Examples of key input and data assumption areas highlighted in the presentation and taken up in the discussion included:

  - Indicators of corruption at governmental and industry level

  - Indicators of workforce conditions etc. including OHS, child labour, freedom to organise, and more

  - Measures of environmental damage associated with production

  - The difficulties of comparing systems in an LCA tool when for example 'water' is compared to 'water'. Firstly, oil production does not use much water, secondly water is scarce in some areas and not in others. A risk is seen that a perverse situation can arise that oil can appear to be better than biofuels if a category such as water consumption is not managed carefully.

- A major issue with such raised by participants is the level of granularity for data – particularly from the risk database (SHDB) tested in the work. The differences can be extremely large between for example: continental US (OH&S & labour conditions) contra industrial maize ethanol production (OH&S conditions) in the mid-west EtOH production regions – and for example between Sao Paolo state sugarcane VERSUS other parts of country where labour conditions, technologies etc. can be markedly less developed.

CLOSING COMMENTS

The participants in the workshop were very engaged. They commented and critiqued throughout the entire workshop providing excellent contextual feedback to the work method and the results of the first trials of a method for LCSA. As indicated above, areas of particular interest were: a) assumptions and in-data for LCSA work (issues associated with a lack of precision and 'granularity' of such inputs; b) the manner in which stakeholders, particularly the media might use results of the LCSA work in its current status of development (issues regarding misuse of method development work).

As such, the feedback from participants has provided great value with regards to how the project is to be represented in the report, particularly with regards to explicit caveats regarding its use.

Pursuant to the conduct of the workshop, the project team judges the workshop to have been a successful and constructive event with regards to obtaining Stakeholder reactions and input important to report generation. However, the team also concludes that such an interaction may have had even greater value if a stakeholder involvement process of this type were held earlier in the process.

This work clearly examines knowledge gaps where there are strong vested interests (e.g. from the side of transportation biofuel actors in Sweden). In particular, earlier involvement of such stakeholders has the potential to both provide for broader inputs to the work and more immediate acceptance of the ideas such work contains.

In closing, it was agreed that there is a clear need to underline the key project output/role of method development in the report. Further, the final outputs must highlight the areas where such methods can be applied to solve issues of (apparently) higher relevance to some in audience.

Figure A1. Workshop participants and presenters.

APPENDIX II: OUTREACH OF PROJECT

Besides this report the project has resulted in the following publications:

Ekener, E., Hansson, J., Gustavsson, M., (2016). *Addressing positive impacts in social LCA – discussing current and new approaches exemplified by the case of vehicle fuels.* The International Journal of Life Cycle Assessment, 23 (3) p. 556–568.

Ekener, E., Hansson, J., Larsson, A., Peck, P., (2018). *Developing Life Cycle Sustainability Assessment methodology by applying values-based sustainability weighting - Tested on biomass based and fossil transportation fuels*. Journal of Cleaner Production 181 p.337-351.

The project has been presented at the following conferences/seminars/workshops:

- Fossil Free Fuels: Markets and Measures - a co-arrangement by f3, SICEC and Chalmers, January 2015, Göteborg, Sweden.
- 23rd European Biomass Conference and Exhibition (EUBCE) June, 2015, Vienna, Austria.
- Program conference for the Swedish Energy Agency and f3 collaborative research program Renewable transportation fuels and systems (samverkansprogrammet Förnybara drivmedel och system), February 2016, Göteborg, Sweden.
- 24th European Biomass Conference and Exhibition (EUBCE) June 2016, Amsterdam, the Netherlands.
- Workshop for biofuel actors to discuss the findings organized within the project, September 9, 2016, Stockholm, Sweden.
- International Advanced Biofuels Conference, May 19, 2017, Gothenburg.
- Program conference for the Swedish Energy Agency and f3 collaborative research program Renewable transportation fuels and systems (samverkansprogrammet Förnybara drivmedel och system), October 2017, Uppsala.

APPENDIX III: SCIENTIFIC DESCRIPTION OF THE MCDA METHOD USED IN THIS STUDY

The MAVT technique, used in this study, formally maps and transforms different perspectives into a value (utility) function, where the criteria adopts the same, dimensionless value scale. The measures can be used to evaluate or rank alternatives. The approach assigns a utility value to each option, where the utility is a real number representing how well the option is preferred in comparison to other options. The number is the sum of the marginal utilities from each criterion to the considered option.

In general, given M evaluation criteria, we can represent an alternative A_k with a vector of performance levels (x_{k1}, x_{k2}, …, x_{kM}). Conforming to the MCDA approach MAVT, the global value of a decision option A_k is given by the additive value function

$$V(A_k) = \sum_{i=1}^{M} w_i v_i(x_{ki}) \tag{1}$$

where M is the total number of criteria, w_i is the weight of criterion i and $v_i(x_{ki})$ is a value function representing the value of alternative A_k under criterion i. Alternative A_k is then preferred to alternative A_l if (and only if) $V(A_k) > V(A_l)$, see, e.g., (Eisenführ et al, 2010) for a comprehensive introduction to this preference model and (Santoyo-Castelazo and Azapagic, 2014) for its application in energy environmental appraisals in a previous case.

With respect to ranking of the options (in our case vehicle fuels) and assigning values to different options, the value of each option under each criterion from the perspective of the decision maker is captured in a so-called value function $v(x)$ such that $v : \boldsymbol{X} \rightarrow [0,1]$ where $\boldsymbol{X}$ is the range of the performance level indicator. Further, if we assume positive preference direction (the more the better) and let $x_{\min}$ denote the worst performance level of the available alternatives and $x_{\max}$ the best, and define $v(x_{\min}) = 0$ and $v(x_{\max}) = 1$. The most common way of doing this is to assign them proportionally such that $v(x) = (x - x_{\min}) / (x_{\max} - x_{\min})$ if the preference direction is positive, or $v(x) = (x_{\max} - x) / (x_{\max} - x_{\min})$ if the preference direction is negative (which is the case in E-LCA and LCC since the higher the cost the worse). This intuitive way to generate a value function (often labelled "proportional scores") was used in this study.

Since the performance level indicator for "Jobs created" is based upon four classes, LPI, MPI, HPI, and VPI it is however less feasible to apply proportional scores as value function for this criterion. Instead, we applied the "Cardinal Ranking" method for this criterion, see (Danielson and Ekenberg, 2016). In its simplest form, cardinal ranking use surrogate values derived from ranking statements. We use $>_{s(i)}$ to denote the strength of the rankings between alternatives, where $=_0$ means that they are ranked equally, and $a_i >_1 a_j$ means that option i is "better" than option j, $a_j >_2 a_k$ means that that option j is "much better" than option k and so forth. This can be represented such that each option is assigned a preference position $p(i) \in \{ 1,\ldots ,Q\}$ where lower position indicate stronger preference such that for two options whenever $a_i >_{s(i)} a_j$ then $s(i) = |p(i) - p(k)|$ and the surrogate value is simply given from

$$v_i = \frac{Q - p(i)}{Q - 1} \tag{2}$$

In our case, we then have settled with

- MPI is better than LPI
- HPI is much better than MPI
- VPI is much better than HPI

Since no option is deemed to have very high impact on jobs created (VPI), we have Q = 4 and preference positions p(HPI) = 1, p(MPI) = 3, and p(LPI) = 4 yielding $v_4(x_4) = (4\text{-}x_4) / 3$.

Table A1: Performance levels of fuels.

	E-LCA* (EURO/MJ)		S-LCA			
			Risks			
Fuel	**Ecovalue, x_1**	**EPS, x_2**	**SHDB, x_4**	**SDG, x_5**	**Jobs created, x_5**	**LCC, x_6**
Petrol – Nigerian oil	0,03116	0,03132	207	221	LPI	0,0143
Petrol – Russian oil	0,03494	0,03525	226	236	LPI	0,0124
Ethanol – Brazilian sugar cane	0,02575	0,02609	135	143	HPI	0,0111
Ethanol – US corn	0,04336	0,03791	101	103	MPI	0,0203

*Here the E-LCA for well-to-tank and tank-to-wheel have been combined to one indicator. This is not an LCA result for the supply chain as it concerns two different products, but it is an indicator representing this variable in the MCDA.

Table A2: Value functions and value of performance level.

	E-LCA		S-LCA			LCC
Fuel	**$v_1(x_1)$**	**$v_2(x_2)$**	**$v_3(x_3)$**	**$v_4(x_4)$**	**$v_5(x_5)$**	**$v_6(x_6)$**
Petrol – Nigerian oil	0,693	0,558	0,152	0,113	0	0,652
Petrol – Russian oil	0,478	0,225042	0	0	0	0,859
Ethanol – Brazilian sugar cane	1	1	0,728	0,699	1	1
Ethanol – US corn	0	0	1	1	0,333	0

From Table A2 it can be seen that all criteria besides ”Jobs created” have a negative preference direction, i.e. the lower the performance level the higher the value.

PUTTING WEIGHTS ON THE CRITERIA

With respect to criteria weights, they will now reflect how big is the difference between the worst and the best performance for each criterion i.e., “how big” each step from $v_i(x_{\min}) = 0$ and $v_i(x_{\max}) = 1$ is. Conforming to the so-called additive model we further have the following normalisation constraints on the weights, $0 \leq w_i \leq 1$ and $\Sigma w_i = 1$.

Now, there are essentially two different ways for combining the set of evaluation criteria in order to rank the decision options:

(i) Each criterion is given equal weight. The highest score on one criterion will off-set having the lowest score on another criterion. Or, alternatively, a threshold score in all criteria may be required for the option to be considered. In this latter approach, trade-offs between criteria is not addressed.

(ii) Unequally weighted criteria: The criteria weights express the importance of each criterion relative to other criteria. This approach has the advantage of better representing the importance of different criteria. Trade-offs are addressed.

The trade-off, or the marginal rate of substitution $t(i, j)$ between two criteria is then given by

$$t(i,j) = \frac{w_j\left[\max(x_i) - \min(x_i)\right]}{w_i\left[\max(x_j) - \min(x_j)\right]} \tag{3}$$

This trade-off is of concern for the case herein, since the trade-off between criteria assessed on the unit of measurement EUR/MJ is judged to be one, meaning that one EUR/MJ in Ecovalue should account for one EUR/MJ on EPS. Or put in other words, one Euro in EPS is assumed to be equal to on Euro in Ecovalue. This implies that the weight w_1 for EPS is $w_1 = 0.4$ and the weight w_2 for Ecovalue is $w_2 = 0.6$.

The use of cardinal rankings also apply to criteria weights. Given a slider with in total Q number of importance scale positions, on which each criterion G_i has the position $p(i) \in \{1 ,... ,Q\}$ on this importance scale where lower position indicate more importance, such that whenever $G_i >_{s(i)} G_j$, $s(i) =|p(i)- p(j)$. It has been shown that the obtaining the surrogate weight from Eq. (2) outperforms other approaches (Danielson and Ekenberg, 2016) and thus this is the selected approach herein and used to represent the different criteria rankings of the stakeholder profiles.

$$w_i = \frac{\frac{1}{p(i)} + \frac{Q+1-p(i)}{Q}}{\sum_{j=1}^{N}\left(\frac{1}{p(j)} + \frac{Q+1-p(j)}{Q}\right)} \tag{4}$$

The outcome of the MCDA is visualized as the bar chart of stacked focal point part-worth values for the direct sub-criteria of the selected criterion). The part-worth value φil for alternative Ai under criterion l is simply given by φil = wi·vij (xij) are the focal point weight for criterion i and the alternative value for the j:th fuel under criterion i. The height of each bar is then the sum φi1 + φi2 + … + φin where n is the number of direct sub-criteria.

It should be noted that it is not possible to compare the quantitative results of the MCDA from this study with a new fifth vehicle supply chain without redoing the MCDA. The reason is that this fifth supply chain might alter the end-points of the data sets (which are used to define the end values in the intervals in the MCDA).

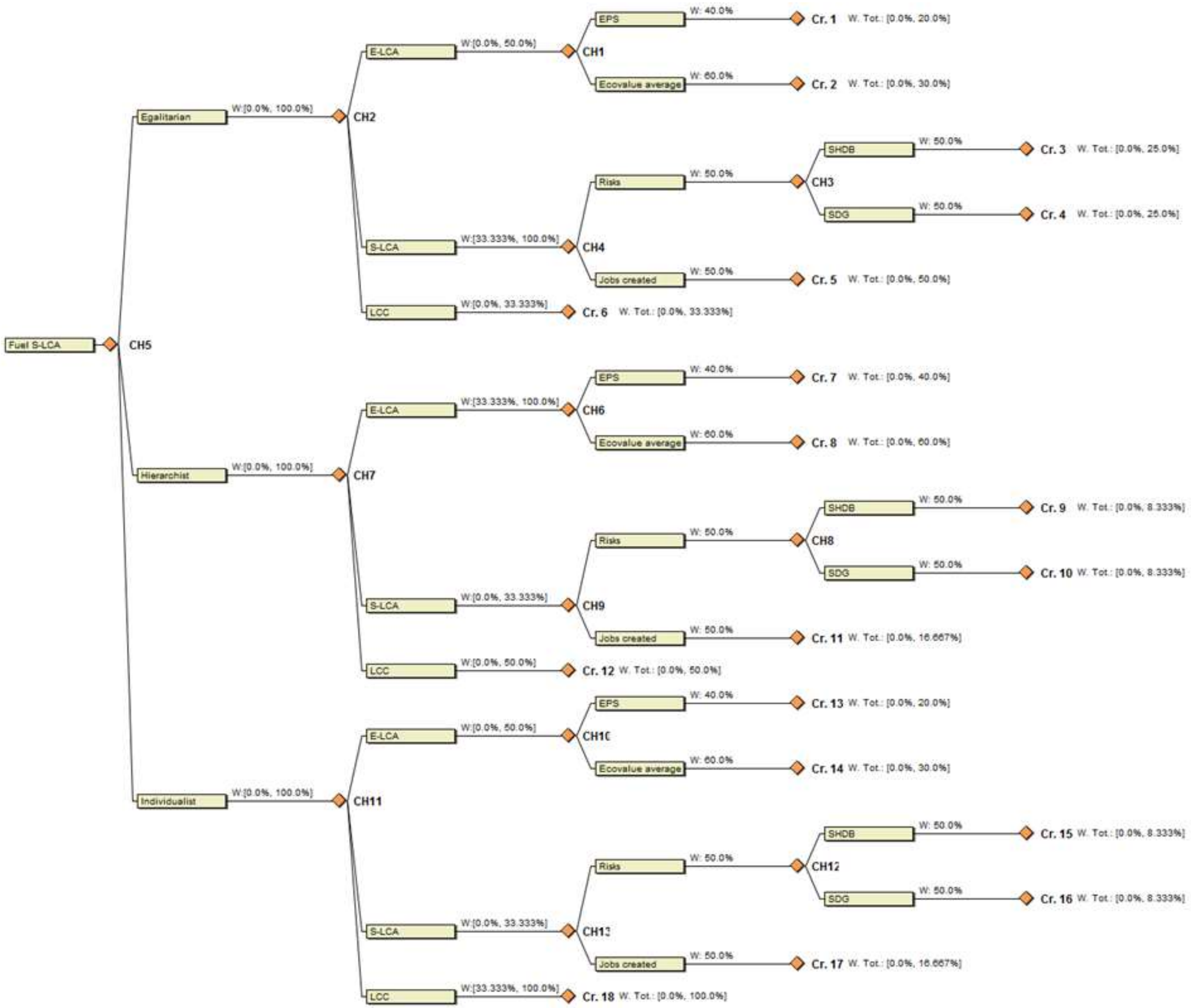

Figure A2. Complete criteria hierarchy for the MCDA.

APPENDIX IV: SHORT OVERVIEW OF REVIEW OF EXISTING LIFE CYCLE ASSESSMENTS

Jacobs Consultancy (2012) was the only LCA of the petrol production in Europe based on only Nigerian or Russian crude oil identified in our literature review. The main reason is most likely that crude oil for petrol production (as well as other petroleum products) in Europe originates from a number of countries and the petrol consists of a mixture of crude oil with different origin. Thus, most LCA:s of petrol production in Europe, model the crude oil extraction stage as a mix of the crude oils used in Europe, which make it impossible to define specific environmental impacts arising from petrol production with only Nigerian or Russian oil. For instance, in 2010, the former USSR countries were responsible for 38% of the crude oil supplied to Europe; Norway 14%; Libya 9%; Saudi Arabia and Iran 5% each, UK, Iraq and Nigeria 4% each and other countries together 17% (Jacobs Consultancy, 2012). However, even though some data is provided for Nigerian and Russian crude oil in Jacobs Consultancy (2012) specific detailed results are not disclosed publicly.

The French Environment and Energy Management Agency (ADEME) performed in 2010 a LCA of the first generation biofuels used in France, including also results for petrol and diesel produced from crude oil (ADEME 2010). The LCA focuses on five environmental parameters. The functional unit chosen is “km traveled” i.e., the internal fuel combustion in the vehicle is taken into account. Results in “MJ of produced fuel” are also presented but only for GHG emissions and energy use. The extraction phase is based on Ecoinvent datasets (Ecoinvent 2015), and the crude oil modelled is a mix of actual crude oil supplies to France. Refining data for energy use and CO_2 emissions is based on Edwards et al. (2007), where the allocation is incremental. The results are presented in Table A3.

Table A3: Energy use and GHG emissions for petrol (ADEME, 2010).

Category	Value	Unit
Non-renewable energy consumption[1)]	1.22	MJ/MJ fuel
GHG emissions (global warming)[2)]	0.0155	kg CO2 eq./MJ fuel

[1)] Including the “MJ” contained in the fuel

[2)] “Refining”, “raffinage” and “transport-distribution” phases (ADEME 2010, table 78)

Eriksson (2013) presents a literature review of LCA:s of petrol and diesel with different origin but with a European focus. The compiled results that focus on only energy use and GHG emissions (in CO_2-equivalents) were presented in both Well-to-Tank and Tank-to Wheel perspectives. In total, results from 9 studies were compiled, in a Well-to-Tank perspective varying from 0.04 to 0.3 MJ/MJ fuel for primary energy consumption and 6.7 to 27 g $CO_{2eq.}$/MJ fuel (with most results between 10-15 g $CO_{2eq.}$/MJ fuel) for GHG emissions.

As already mentioned ADEME performed in 2010 a LCA of the first generation biofuels used in France (Ademe, 2010). Ethanol based on Brazilian sugarcane as raw material is included and the LCA is based on Macedo et al (2004). As in the case with petrol, results in “MJ of produced fuel” are only presented for GHG emissions and energy use (see Table A4).

Table A4: Energy use and GHG emissions for the production of Brazilian ethanol (ADEME 2010).

Category	Value	Unit
Non-renewable energy consumption	0.183	MJ/MJ fuel
GHG emissions (global warming)	0.0253[1)]	kg CO2 eq./MJ fuel

1) "Cultivation", "processing" and "transport-distribution" phases (ADEME 2010, table 101)

In Kim (2008) a LCA of ethanol derived from corn grain grown in the USA is performed in order to investigate the environmental performance of fuel ethanol used in a compact vehicle fueled with E10 (90% petrol and 10% ethanol). The functional unit is 1 kg of ethanol and the system boundary includes corn cultivation in the US, transportation of corn grain, dry milling process, transportation and distribution of ethanol as well as vehicle operation. The co-product distilled dried grains with solubles (DDGS) is handled by the system expansion approach, where DDGS is assumed to replace corn grain and soybean meal. The potential impact categories analyzed are non-renewable energy consumption, GHG emissions, acidification, eutrophication and photochemical smog formation. Table A5 summarizes the results for non-renewable energy use and GHG emissions.

Table A5: Energy use and GHG emissions for the production of US corn based ethanol (Kim and Dale, 2008).

Category	Value	Unit
Non-renewable energy consumption	0.75[1)]	MJ/MJ[2)] fuel
GHG emissions (global warming)	0.0571[1)]	kg CO2 eq./MJ fuel

1) Results do not include "distribution of ethanol" nor "vehicle operation"

2) Original results "per kg fuel". LHV ethanol: 26.8 MJ/kg

APPENDIX V: DATASETS USED FOR E-LCA

Table A6: Datasets from the Ecoinvent Database (Ecoinvent, 2015) used in the GaBi modelling of the selected transport fuels.

Fuel	Dataset in Ecoinvent	Description
Petrol, Nigerian/Russian oil	RER: petrol, unleaded, at refinery	Description of all flows of materials and energy due to the throughput of 1kg crude oil in the refinery. The multi output-process 'crude oil, in refinery' delivers the co-products petrol, unleaded, bitumen, diesel, light fuel oil, heavy fuel oil, kerosene, naphtha, propane/ butane, refinery gas, secondary sulphur and electricity. The impacts of processing are allocated to the different products.
	RER: petrol, low-sulphur, at refinery	Estimation for the conversion of refinery production to low-sulphur petrol with a sulphur content < 50ppm (Today 150ppm). An additional energy use of 6% has been estimated. Data for additional emissions and additional infrastructure were not available.
	RER: petrol, low-sulphur, at regional storage	Inventory for the distribution of petroleum product to the final consumer (household, car, power plant, etc.) including all necessary transports.
Ethanol, sugarcane, Brazil	CH: ethanol, 99.7% in H2O, from biomass, production BR, at service station	The inventory for "ethanol, 99.7% in H2O, from biomass, production BR, at CH" is modelled with data of the regional distribution of petrol in Switzerland. The transports are modelled with the distance Brazil - Rotterdam for the transoceanic transport, the distance Rotterdam - Basel for the transport from the Netherlands to Switzerland, and standard distances for transports in Switzerland.
Ethanol, corn, US	CH: ethanol, 99.7% in H2O, from biomass, production US, at service station	Inventory refers to the distribution of 1 kg of anhydrous ethanol 99.7% in Switzerland. Ethanol is imported from US and produced from corn grains. Distribution to the final consumer (service station) including all necessary transports.

APPENDIX VI: DETAILED RESULTS FOR THE E-LCA WELL-TO-TANK

Table A7 presents the impact assessment results for the 12 impact categories comprised in the CML method (Guinee et al., 2002) for the modified Ecoinvent dataset “RER: petrol, low-sulphur, at regional storage”, with Nigerian crude oil being the only oil input in the dataset. The results refer to 1 MJ of low-sulphur petrol, at regional storage, assuming 42.5 MJ/kg fuel (lower heating value).

Table A7: Well-to-tank results for petrol based on Nigerian crude oil, per MJ of low-sulphur petrol, at regional storage (CML impact assessment method).

Category	Value	Unit
Abiotic Depletion (AD elements)	7.88E-09	kg Sb-Equiv./MJ fuel
Abiotic Depletion (AD fossil)	1.34	MJ/MJ fuel
Acidification	1.44E-04	kg SO2-Equiv./MJ fuel
Eutrophication	2.89E-05	kg Phosphate-Equiv./MJ fuel
Freshwater Aquatic Ecotoxicity	1.77E-03	kg DCB-Equiv./MJ fuel
Global Warming (GWP 100 years)	0.0289	kg CO2-Equiv./MJ fuel
Global Warming, excl. biogenic carbon (GWP 100 years)	0.0289	kg CO2-Equiv. /MJ fuel
Human Toxicity	6.48E-03	kg DCB-Equiv./MJ fuel
Marine Aquatic Ecotoxicity	5.26	kg DCB-Equiv./MJ fuel
Ozone Layer Depletion (steady state)	1.73E-08	kg R11-Equiv./MJ fuel
Photochem. Ozone Creation	6.65E-05	kg Ethene-Equiv./MJ fuel
Terrestric Ecotoxicity	8.53E-05	kg DCB-Equiv./MJ fuel

Table A8 presents the impact assessment results for the 12 impact categories recommended by the International Reference Life Cycle Data System (ILCD) (JRC, 2011), for the modified Ecoinvent dataset “RER: petrol, low-sulphur, at regional storage”, with Nigerian crude oil being the only oil input in the dataset. The results refer to 1 MJ of low-sulphur petrol, at regional storage, assuming 42.5 MJ/kg fuel (lower heating value).

Table A8: Well-to-tank results for petrol based on Nigerian crude oil, per MJ of low-sulphur petrol, at regional storage (ILCD recommended impact categories).

Category	Value	Unit
Acidification, accumulated exceedance	1.71E-04	Mole of H+ eq./MJ fuel
Ecotoxicity for aquatic fresh water, USEtox	2.58E-02	CTUe/MJ fuel
Freshwater eutrophication, EUTREND model, ReCiPe	1.91E-06	kg P eq/MJ fuel
Human toxicity cancer effects, USEtox	4.16E-10	CTUh/MJ fuel
Human toxicity non-canc. effects, USEtox	2.26E-09	CTUh/MJ fuel
Ionising radiation, human health effect model, ReCiPe	0.896	kg U235 eq./MJ fuel
IPCC global warming, excl biogenic carbon	0.0289	kg CO2-Equiv./MJ fuel
IPCC global warming, incl biogenic carbon	0.0289	kg CO2-Equiv./MJ fuel
Marine eutrophication, EUTREND model, ReCiPe	9.04E-07	kg N-Equiv./MJ fuel
Ozone depletion, WMO model, ReCiPe	1.73E-08	kg CFC-11 eq./MJ fuel
Particulate matter/Respiratory inorganics, RiskPoll	9.46E-06	kg PM2,5-Equiv./MJ fuel
Photoch. ozone form., LOTOS-EUROS model, ReCiPe	2.27E-04	kg NMVOC/MJ fuel

Table A9 presents the impact assessment results for the 12 impact categories comprised in the CML method (Guinee et al., 2002) for the modified Ecoinvent dataset “RER: petrol, low-sulphur, at regional storage”, with Russian crude oil being the only oil input in the dataset. The results refer to 1 MJ of low-sulphur petrol, at regional storage, assuming 42.5 MJ/kg fuel (lower heating value).

Table A9: Well-to-tank results for petrol based on Russian crude oil, per MJ of low-sulphur petrol, at regional storage (CML impact assessment method).

Category	Value	Unit
Abiotic Depletion (ADP elements)	2.69E-08	kg Sb-Equiv./MJ fuel
Abiotic Depletion (ADP fossil)	1.36	MJ/MJ fuel
Acidification	4.62E-04	kg SO2-Equiv./MJ fuel
Eutrophication	1.83E-04	kg Phosphate-Equiv./MJ fuel
Freshwater Aquatic Ecotoxicity	5.57E-03	kg DCB-Equiv./MJ fuel
Global Warming (GWP 100 years)	0.0251	kg CO2-Equiv./MJ fuel
Global Warming, excl. biogenic carbon (GWP 100 years)	0.0251	kg CO2-Equiv./MJ fuel
Human Toxicity	0.0144	kg DCB-Equiv./MJ fuel
Marine Aquatic Ecotoxicity	15.4	kg DCB-Equiv./MJ fuel
Ozone Layer Depletion (steady state)	1.78E-08	kg R11-Equiv./MJ fuel
Photochem. Ozone Creation	4.39E-05	kg Ethene-Equiv./MJ fuel
Terrestric Ecotoxicity	2.51E-04	kg DCB-Equiv./MJ fuel

Table A10 presents the impact assessment results for the 12 impact categories recommended by the ILCD (JRC, 2011), for the modified Ecoinvent dataset "RER: petrol, low-sulphur, at regional storage", with Russian crude oil being the only oil input in the dataset. The results refer to 1 MJ of low-sulphur petrol, at regional storage, assuming 42.5 MJ/kg fuel (lower heating value).

Table A10: Well-to-tank results for petrol based on Russian crude oil, per MJ of low-sulfur petrol, at regional storage (ILCD recommended impact categories).

Category	Value	Unit
Acidification, accumulated exceedance	5.21E-04	Mole of H+ eq./MJ fuel
Ecotoxicity for aquatic fresh water, USEtox	0.0788	CTUe/MJ fuel
Freshwater eutrophication, EUTREND model, ReCiPe	6.48E-06	kg P eq/MJ fuel
Human toxicity cancer effects, USEtox	1.81E-09	CTUh/MJ fuel
Human toxicity non-canc. effects, USEtox	6.5E-09	CTUh/MJ fuel
Ionising radiation, human health effect model, ReCiPe	2.99	kg U235 eq./MJ fuel
IPCC global warming, excl biogenic carbon	0.0251	kg CO2-Equiv./MJ fuel
IPCC global warming, incl biogenic carbon	0.0251	kg CO2-Equiv./MJ fuel
Marine eutrophication, EUTREND model, ReCiPe	2.18E-06	kg N-Equiv./MJ fuel
Ozone depletion, WMO model, ReCiPe	1.78E-08	kg CFC-11 eq./MJ fuel
Particulate matter/Respiratory inorganics, RiskPoll	3.05E-05	kg PM2,5-Equiv./MJ fuel
Photoch. ozone form., LOTOS-EUROS model, ReCiPe	1.66E-04	kg NMVOC/MJ fuel

Table A11 presents the impact assessment results for the 12 impact categories comprised in the CML method (Guinee, 2002) for the Ecoinvent dataset "CH: ethanol, 99.7% in H2O, from biomass, production BR, at service station". The results refer to 1 MJ of ethanol, assuming 26.8 MJ/kg fuel (lower heating value).

Table A11: Well-to-tank results for ethanol based on Brazilian sugarcane per MJ ethanol (CML impact assessment method).

Category	Value	Unit
Abiotic Depletion (ADP elements)	9.15E-08	kg Sb-Equiv./MJ fuel
Abiotic Depletion (ADP fossil)	0.184	MJ/MJ fuel
Acidification	2.65E-04	kg SO2-Equiv./MJ fuel
Eutrophication	9.43E-06	kg Phosphate-Equiv./MJ fuel
Freshwater Aquatic Ecotoxicity	0.0134	kg DCB-Equiv./MJ fuel
Global Warming (GWP 100 years)	-0.0757	kg CO2-Equiv./MJ fuel
Global Warming, excl. biogenic carbon (GWP 100 years)	0.02	kg CO2-Equiv./MJ fuel
Human Toxicity	0.154	kg DCB-Equiv./MJ fuel
Marine Aquatic Ecotoxicity	9.33	kg DCB-Equiv./MJ fuel
Ozone Layer Depletion (steady state)	2.20E-09	kg R11-Equiv./MJ fuel
Photochem. Ozone Creation	4.59E-04	kg Ethene-Equiv./MJ fuel
Terrestric Ecotoxicity	6.16E-03	kg DCB-Equiv./MJ fuel

Table A12 presents the impact assessment results for the 12 impact categories recommended by the ILCD (JRC, 2011), for the Ecoinvent dataset “CH: ethanol, 99.7% in H2O, from biomass, production BR, at service station”. The results refer to 1 MJ of ethanol, assuming 26.8 MJ/kg fuel (lower heating value).

Table A12: Well-to-tank results for ethanol based on Brazilian sugarcane per MJ ethanol (ILCD recommended impact categories).

Category	Value	Unit
Acidification, accumulated exceedance	3.89E-04	Mole of H+ eq./MJ fuel
Ecotoxicity for aquatic fresh water, USEtox	0.248	CTUe/MJ fuel
Freshwater eutrophication, EUTREND model, ReCiPe	6.97E-06	kg P eq./MJ fuel
Human toxicity cancer effects, USEtox	2.10E-09	CTUh/MJ fuel
Human toxicity non-canc. effects, USEtox	6.97E-08	CTUh/MJ fuel
Ionising radiation, human health effect model, ReCiPe	1.57	kg U235 eq./MJ fuel
IPCC global warming, excl biogenic carbon	0.02	kg CO2-Equiv./MJ fuel
IPCC global warming, incl biogenic carbon	-0.0757	kg CO2-Equiv./MJ fuel
Marine eutrophication, EUTREND model, ReCiPe	9.61E-06	kg N-Equiv./MJ fuel
Ozone depletion, WMO model, ReCiPe	2.22E-09	kg CFC-11 eq./MJ fuel
Particulate matter/Respiratory inorganics, RiskPoll	6.45E-05	kg PM2,5-Equiv./MJ fuel
Photoch. ozone form., LOTOS-EUROS model, ReCiPe	9.43E-04	kg NMVOC/MJ fuel

Table A13 presents the impact assessment results for the 12 impact categories comprised in the CML method (Guinee, 2002) for the Ecoinvent dataset “CH: ethanol, 99.7% in H2O, from biomass, production US, at service station”. The results refer to 1 MJ of ethanol, assuming 26.8 MJ/kg fuel (lower heating value).

Table A13: Well-to-tank results for US corn based ethanol per MJ ethanol (CML impact assessment method).

Category	Value	Unit
Abiotic Depletion (ADP elements)	1.43E-07	kg Sb-Equiv./MJ fuel
Abiotic Depletion (ADP fossil)	0.794	MJ/MJ fuel
Acidification	5.20E-04	kg SO2-Equiv./MJ fuel
Eutrophication	5.54E-04	kg Phosphate-Equiv./MJ fuel
Freshwater Aquatic Ecotoxicity	0.024	kg DCB-Equiv./MJ fuel
Global Warming (GWP 100 years)	9.25E-03	kg CO2-Equiv./MJ fuel
Global Warming, excl. biogenic carbon (GWP 100 years)	0.0808	kg CO2-Equiv./MJ fuel
Human Toxicity	0.0251	kg DCB-Equiv./MJ fuel
Marine Aquatic Ecotoxicity	27	kg DCB-Equiv./MJ fuel
Ozone Layer Depletion (steady state)	7.52E-09	kg R11-Equiv./MJ fuel
Photochem. Ozone Creation	3.61E-05	kg Ethene-Equiv./MJ fuel
Terrestric Ecotoxicity	9.67E-04	kg DCB-Equiv./MJ fuel

Table A14 presents the impact assessment results for the 12 impact categories recommended by the ILCD (JRC, 2011), for the Ecoinvent dataset "CH: ethanol, 99.7% in H2O, from biomass, production US, at service station". The results refer to 1 MJ of ethanol, assuming 26.8 MJ/kg fuel (lower heating value).

Table A14: Well-to-tank results for US corn based ethanol per MJ ethanol (ILCD recommended impact categories).

Category	Value	Unit
Acidification, accumulated exceedance	7.90E-04	Mole of H+ eq./MJ fuel
Ecotoxicity for aquatic fresh water, USEtox	4.65E-01	CTUe/MJ fuel
Freshwater eutrophication, EUTREND model, ReCiPe	2.25E-05	kg P eq/MJ fuel
Human toxicity cancer effects, USEtox	2.63E-09	CTUh/MJ fuel
Human toxicity non-canc. effects, USEtox	-1.23E-08	CTUh/MJ fuel
Ionising radiation, human health effect model, ReCiPe	6.24	kg U235 eq/MJ fuel
IPCC global warming, excl biogenic carbon	0.00808	kg CO2-Equiv./MJ fuel
IPCC global warming, incl biogenic carbon	9.25E-03	kg CO2-Equiv./MJ fuel
Marine eutrophication, EUTREND model, ReCiPe	7.53E-04	kg N-Equiv./MJ fuel
Ozone depletion, WMO model, ReCiPe	7.54E-09	kg CFC-11 eq./MJ fuel
Particulate matter/Respiratory inorganics, RiskPoll	3.35E-05	kg PM2,5-Equiv./MJ fuel
Photoch. ozone form., LOTOS-EUROS model, ReCiPe	2.74E-04	kg NMVOC/MJ fuel

APPENDIX VII: DETAILED E-LCA RESULTS AFTER WEIGHTING

E-LCA result after weighting for different subcategories and the different included weighting methods (EPS, Ecovalue average, low and high) are presented in this appendix.

EPS (ENVIRONMENTAL PRIORITY STRATEGIES)

Table A15: E-LCA result well-to-tank for different impact categories after weighting based on EPS.

Categories	Subcategories	Petrol based on Nigerian oil	Petrol based on Russian oil	Ethanol based on sugarcane molasse (Brazil)	Ethanol based on corn (US)
Resources	Energy resources	0.0123	0.0129	0.0008	0.0031
	Land use	0.0002	0.0014	0.0005	0.0102
	Material resources	0.0053	0.0087	0.0059	0.0116
Emissions to air	Heavy metals to air	0	0	0.00001	1.056E-05
	Inorganic emissions to air	0.0029	0.0031	0.0015	0.0082
	Organic emissions to air	0.0037	0.0012	0.0007	0.0007
	Particles to air	0.0006	0.0013	0.0143	0.0018
	Radioactive emissions to air	0	0	0.0000	1.10E-06
Emissions to fresh water	No major impact from sub-categories	0	0	0.0000	0
Emissions to fresh water	No major impact from sub-categories	0	0	0.0000	0
TOTAL (Euro/MJ)		**0.0248**	**0.0287**	**0.0238**	**0.0356**

Table A16: E-LCA result tank-to-wheel for different impact categories after weighting based on EPS.

Categories	Subcategory	Petrol (E05)	Ethanol (E85)
Resources	Energy resources	0	0
	Land use	0	0
	Material resources	0	0
Emissions to air	Heavy metals to air	0	0
	Inorganic emissions to air	0.0064	0.0022
	Organic emissions to air	5.809E-05	7.060E-05
	Particles to air	3.29E-05	2.84E-05
	Radioactive emissions to air	0	0
Emissions to fresh water	No major impact from subcategories	0	0
Emissions to fresh water	No major impact from subcategories	0	0
TOTAL (Euro/MJ)		**0.0065**	**0.0023**

ECOVALUE AVERAGE

Table A17: E-LCA result well-to-tank for different impact categories after weighting based on ECOVALUE AVERAGE.

Categories	Petrol based on Nigerian oil	Petrol based on Russian oil	Ethanol based on sugar-cane molasse (Brazil)	Ethanol based on corn (US)
Climate change [kg CO_2-Equiv.]	0.0088	0.0077	0.0045	0.0201
Freshwater eutrophication	0.0001	0.0005	0.0004	0.0013
Human toxicity	0.0009	0.0026	0.0091	0.0024
Marine ecotoxicity	0.0001	0.0002	0.0002	0.0004
Marine eutrophication	0.0000	0.0001	0.0001	0.0069
Particulate matter formation	0.0011	0.0030	0.0029	0.0033
Photochemical oxidant formation	0.0007	0.0005	0.0021	0.0005
Terrestrial acidification	0.0004	0.0013	0.0008	0.0017
Abiotic resources	0.0040	0.0042	0.0004	0.0013
TOTAL (Euro/MJ)	**0.0161**	**0.0199**	**0.0204**	**0.0380**

Table A18: E-LCA result tank-to-wheel for different impact categories after weighting based on ECOVALUE AVERAGE.

Categories	Petrol E05	Ethanol E85
Climate change [kg CO_2-Equiv.]	0.0147	0.0051
Freshwater eutrophication	0	0
Human toxicity	0	0
Marine ecotoxicity	0	0
Marine eutrophication	9.05E-06	7.82E-06
Particulate matter formation	0.0002	0.0001
Photochemical oxidant formation	9.86E-05	7.78E-05
Terrestrial acidification	4.49E-05	3.85E-05
Abiotic resources	0	0
TOTAL (Euro/MJ)	**0.0150**	**0.0054**

ECOVALUE LOW

Table A19: E-LCA result well-to-tank for different impact categories after weighting based on ECOVALUE LOW.

Categories	Petrol based on Nigerian oil	Petrol based on Russian oil	Ethanol based on sugar-cane molasse (Brazil)	Ethanol based on corn (US)
Climate change [kg CO_2-Equiv.]	0.0003	0.0003	0.0002	0.0007
Freshwater eutrophication	0.0001	0.0005	0.0004	0.0013
Human toxicity	6.06E-06	1.84E-05	6.51E-05	1.71E-05
Marine ecotoxicity	6.89E-05	0.0002	0.0002	0.0004
Marine eutrophication	3.67E-05	5.52E-05	0.0001	0.0069
Particulate matter formation	0.0011	0.0030	0.0029	0.0033
Photochemical oxidant formation	0.0003	0.0002	0.0011	0.0003
Terrestrial acidification	0.0004	0.0013	0.0008	0.0017
Abiotic resources	0.0040	0.0042	0.0004	0.0013
TOTAL (Euro/MJ)	**0.0064**	**0.0097**	**0.0060**	**0.0159**

Table A20: E-LCA result tank-to-wheel for different impact categories after weighting based on ECOVALUE LOW.

Categories	Petrol E05	Ethanol E85
Climate change [kg CO_2-Equiv.]	0.0005	0.0002
Freshwater eutrophication	0	0
Human toxicity	0	0
Marine ecotoxicity	0	0
Marine eutrophication	9.05E-06	7.82E-06
Particulate matter formation	0.0002	0.0001
Photochemical oxidant formation	5.11E-05	4.03E-05
Terrestrial acidification	4.49E-05	3.85E-05
Abiotic resources	0	0
TOTAL (Euro/MJ)	**0.0008**	**0.0004**

ECOVALUE HIGH

Table A21: E-LCA result well-to-tank for different impact categories after weighting based on ECOVALUE HIGH.

Categories	Petrol based on Nigerian oil	Petrol based on Russian oil	Ethanol based on sugar-cane molasse (Brazil)	Ethanol based on corn (US)
Climate change [kg CO_2-Equiv.]	0.0174	0.0151	0.0088	0.0396
Freshwater eutrophication	0.0001	0.0005	0.0004	0.0013
Human toxicity	0.0015	0.0045	0.0159	0.0042
Marine ecotoxicity	6.89E-05	0.0002	0.0002	0.0004
Marine eutrophication	3.67E-05	5.52E-05	0.0001	0.0069
Particulate matter formation	0.0011	0.0030	0.0029	0.0033
Photochemical oxidant formation	0.0010	0.0007	0.0031	0.0008
Terrestrial acidification	0.0004	0.0013	0.0008	0.0017
Abiotic resources	0.0040	0.0042	0.0004	0.0013
TOTAL (Euro/MJ)	**0.0256**	**0.0295**	**0.0325**	**0.0595**

Table A22: E-LCA result tank-to-wheel for different impact categories after weighting based on ECOVALUE HIGH.

Categories	Petrol E05	Ethanol E85
Climate change [kg CO_2-Equiv.]	0.0289	0.0100
Freshwater eutrophication	0	0
Human toxicity	0	0
Marine ecotoxicity	0	0
Marine eutrophication	9.05E-06	7.82E-06
Particulate matter formation	0.0002	0.0001
Photochemical oxidant formation	0.0001	0.0001
Terrestrial acidification	4.49E-05	3.85E-05
Abiotic resources	0	0
TOTAL (Euro/MJ)	**0.0293**	**0.0103**

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