Time and Scale Aspects in Life Cycle Assessment of Emerging Technologies

Case Study on Alternative Transport Fuels

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Abstract

Life cycle assessments (LCAs, including well-to-wheel studies) that are to support decisions that strive to change large technical systems need to consider time- and scale-related factors that are given little attention in standard LCA procedures. We suggest that it is important to look beyond the current situation and study many possible future states, what we call “stylised states”, to explore general technology differences. We choose to address three issues in this report. Our case study deals with alternative fuels for transportation, and relates to a recent well-to-wheel study performed by CONCAWE, EUCAR and JRC. The methodological results, though, could be of equal importance when studying other major technologies.

First, shifting time frame gives room for technical development that should affect not only the choice of performance data, but perhaps also the functional unit and the selection of technologies under study.

Second, background systems such as heat and power production change over time, and we exemplify by using three different systems, mainly based on coal, natural gas and short rotation forestry, respectively. Increased production volumes may for some technologies also change the background system, which is of particular importance for technologies that are used in their own production processes. We show that for biofuels changes in background systems have consequences not only for greenhouse gas (GHG) emissions and agricultural land use for each fuel chain, but also for the ranking order of e.g. wheat ethanol and RME, in terms of GHG emissions. We use what we call a “net output approach”, which implies that a fraction of the produced biofuel is used for its own production. Accordingly, the functional unit used in this study is 1 MJ fuel available for other purposes than producing fuel.

Finally, different types of feedstock are available in different quantities and different by-product markets vary in size. Allocation of environmental impact between product and by-products is here made through system expansion, and we study some possible markets for by-products. To give an example of by-product effects, current key markets for ethanol by-products in EU-15 correspond to an ethanol production that covers about 2 % of demand, and for RME about 3 %, that is, well below the 5.75 % EU biofuel target for 2010. Therefore, the GHG emissions and agricultural land use allocated to the fuels differ between a low and a high market penetration.

Combining the results, we show that time and scale are important factors for the ranking of wheat ethanol, RME and wood methanol in terms of GHG emissions and agricultural land use, as the results are dependent on assumptions regarding background system and by-product markets. We indicate that agricultural land use results can be weighted in GHG terms in several ways, e.g. by using short rotation forestry or solar panels as a reference, an approach that would require further research.

Preface

This CPM report is a result of the ongoing project Sustainable Technology Paths (STEP) carried out at the Department of Environmental Systems Analysis (ESA), Chalmers. We greatly acknowledge CPM for the financial support, and Göteborg Energy Ltd. Research Foundation for the complementary funding of the project.

We would also like to thank:

- our reference group from the CPM member companies, Linköping University, the Swedish Energy Agency and the Committee on Transport and Communications of the Swedish Parliament;

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Göteborg, September 2004

Karl Jonasson
Björn Sandén

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Appendix

1 Introduction

While economic growth will increase the demand for energy services, the carbon dioxide (CO_2) emissions inherently linked to current fossil fuel energy technologies need to be reduced substantially over the century to "prevent dangerous anthropogenic interference with the climate system" (UNFCCC 1992). As a consequence, there will be a need for development and large-scale diffusion of a range of new technologies based on renewable energy for conversion, storage, transport and efficient use of energy.

Road transportation is currently responsible for 20-25 % of world CO_2-emissions and is to more than 99 % dependent on fossil oil (IEA 2000). To reduce oil dependence and greenhouse gas (GHG) emissions, different policies are now being implemented to increase the share of motor fuels based on renewable energy. In the short term, the EU directive "on the promotion of the use of biofuels or other renewable fuels for transport" force the member states to set targets for the minimum use of renewable fuels (EU 2003b). The reference value is 2 % by the end of 2005, and 5.75 % by the end of 2010, based on energy content (lower calorific value), of all petrol and diesel used for transport purposes. For the medium term, the European Commission Green Paper "Towards a European strategy for the security of energy supply" set the goal that 20 % of road transport fuel should be alternative fuels by 2020 (EU 2001). There are also other important objectives of switching to biofuels, such as reducing the emissions of urban air pollutants, and promoting the agricultural sector of the EU (Calzoni et al. 2000).[1]

Several different fuels are proposed as alternatives to petrol and diesel, e.g. ethanol, FAME, methanol, DME, Fischer-Tropsch diesel, natural gas, biogas and hydrogen. The visions of the future importance of different fuels vary greatly, especially in the medium term (2020) (ViewLS 2004). The fuels differ regarding resource base, energy efficiency, environmental impact, technical maturity and requirements for system change. The introduction (or opposition to introduction) of a fuel is often motivated with

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[1] This is the case for corn ethanol in the US, as well (Shapouri et al. 2002).

support from some kind of environmental systems analysis, and different analyses give different recommendations.[2]

The type of environmental assessment that is most commonly used is life-cycle assessments (LCA), in this application often termed well-to-wheel (WTW) studies (for an overview, see MacLean and Lave (2003)). They usually consider energy use and emissions during fuel production and vehicle use, while not including the production of vehicles and production facilities. Well-to-tank (WTT) studies, like this one, do not include vehicle use either.

Standard LCA methodology is developed to answer questions about environmental impacts of the current (or historical) production and use of one unit of a product or of minor product or process changes. When this methodology is used to provide answers to questions about strategic technological choices, i.e. not decisions that aim at optimising a process in an existing technological environment but with the long-term goal of changing large-scale technological systems, the result is of little value and in the worst case interpretations of the result may be grossly misleading. This observation is of particular importance for assessments of GHG emissions resulting from emerging energy and transport technologies. In this paper we explore some time and scale aspects that could improve the usefulness of LCA or WTW studies as a support for strategic technology choice. Some of these aspects have previously been discussed in general terms in e.g. Frischknecht (1997) and Weidema et al. (2004).

There are also other tools available for environmental and economic evaluation of transport fuels. Energy systems models aim at finding minimal cost options for allocation of resources and choice of conversion technologies for many end-use sectors under emission constraints, and cost-benefit analysis consider the complete fuel costs, including externalities.[3] They all point out different aspects of transportation, and they all have their strengths and weaknesses. LCA is suitable for evaluating similarly functioning products and product systems, and we have chosen to focus on fuels suitable for vehicles similar to those used today (see section 3.1 for a critical discussion). This does not mean that there is not a need for

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[2] See for example the debate on net energy output from corn ethanol in the USA (Shapouri et al. 2002; Pimentel 2003).

[3] See e.g. Azar et al. (2003) or Gielen et al. (2003) on energy systems models.

considering other aspects of transportation, like a decreased use of transport or completely new transport solutions, implying e.g. social changes.

2 *Prospective state-oriented technology LCA*

LCAs can be categorised in different ways. First, some studies are *retrospective*, looking back at historic environmental impact, while others are *prospective*, looking forward and considering effects of different decisions (Tillman 2000). Retrospective studies are almost always what we here would like to term *state-oriented*, i.e. they use plant-specific or average data to illustrate the life-cycle impact of a product. Prospective studies are often *change-oriented* and use marginal data. Change-oriented studies explore the consequences of an action.[4] But prospective studies could also be state-oriented comparing not changes on the margin but future states. We suggest that an additional distinction can me made, that between *product LCA* and *technology LCA,* where the former seeks to investigate the impact of a specific production process, plant or product, while the latter is an assessment of a more general technology.

Our aim is to develop methodology for technology LCA. In change-oriented technology LCA a key methodological problem is to select which cause-effect chains that should be included and how to quantify the effects (Karlström and Sandén 2004). In the following we will instead focus on prospective state-oriented technology LCA. Then the key methodological problem is to analyse a relevant state.

The relevant state depends on the character of the problem and technology status. In the case of global warming, local emissions of GHG add to a global stock: there is no correlation between the localisation of GHG emissions and climatic effects, and GHG emissions do not primarily lead to an instant problem, but builds up a problem of climatic change over time. The cumulative emissions over several decades, and not the current emissions, are the main cause of concern. In addition, the penetration of all alternative fuels is currently small and most technologies are immature. For these reasons the current state is not very relevant. The environmental impact in a future state with larger scale of adoption of the studied alternative fuel is more relevant. But the future is uncertain. Hence, there is a need to analyse many possible future states and stylised states. By a “stylised state” we denote an extreme state (e.g. a state where all electricity

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[4] Ekvall et al. (2004) use the terms attributional and consequential LCA for state-oriented and change-oriented LCA, respectively.

and heat is produced from coal) that is unlikely to materialise but that could illustrate important technology differences in a clear way.[5]

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[5] This is similar to the cornerstone scenarios discussed in Weidema et al. (2004), but their approach implies scenarios more plausible than those in our approach.

3 Time and scale aspects in LCA

The methodological implications of introducing time and scale considerations in LCA are here divided into three parts, where the first has to do with choice of data and functional unit, and connections to technical development. Secondly, we discuss the influence of the choice of background system, and finally different markets for by-products are studied, and related to feedstock utilisation.

3.1 Technical change and choice of data, scope and functional unit

The first and most obvious observation is that technology performance will change over time. Crops, components such as engines, and processes such as farming and fuel processing become better over time as more knowledge is gained. This factor is not only dependent on time, but also on scale. More production implies more learning and larger scale of production gives room for increased efficiency.[6] An increased scale also generates incentives for system optimisation. Plants dedicated to produce fuel instead of food or paper, engines optimised for a new fuel and vehicles optimised for a new propulsion system will create more efficient systems than those relying on current practices.

But, the further into the future we look, flexibility increases. First of all, the use of agricultural products for fuel instead of food production puts other demands on the farming processes. The hygiene demands on fertilisers could possibly be decreased (as discussed in Bernesson et al. (2004)), and higher-yield varieties can be used, perhaps in new areas, as conventional product quality can be ignored (Venturi and Venturi 2003). This implies, however, that by-products most likely cannot be used for animal feed purposes, and that the farmers cannot easily choose between the markets for food and for fuels.

The implication of this is that data on current performance should be used with care. Assumptions on future performance at different scales of

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[6] See Moreira and Goldemberg (1999) for an account of technology development in sugar cane ethanol production in Brazil.

adoption could be enhanced by calculated physical limits, expert estimates, trend extrapolation, and experience curves.[7,8] We choose to use the selection of data published in a recent WTW-study performed by Edwards et al. (2003), discussed in section 4, as employing these tools are beyond the scope of this report.

Not only data, but also the relevant choice of alternative well-to-wheel chains changes over time. In the short term, say within five or even ten years, a lot of things in the well-to-wheel chain are fixed. The only real alternatives are fuels that fit into the existing fuel-vehicle system and that are already produced, albeit for other purposes. The only fuels that use a renewable energy resource that have these properties are ethanol and FAME (fatty acid methyl esters, e.g. RME, rapeseed methyl ester) from crops that are already produced in large quantities. Ethanol can be used as blends in petrol or in petrol engines with some modifications, or in diesel engines equipped with sparking plugs or with a fuel additive (Baky et al. 2002).[9] FAME can be used as blends in diesel or in slightly adjusted diesel engines.

Over time, farmers can switch to short rotation forestry (SRF), and engines and infrastructure could be adapted to e.g. DME or methanol. Over time, solar hydrogen production and fuel cells could offer a realistic way to provide transport, using a fraction of the land required to produce energy crops. Over time, also new transport modes could open up for new ways of supplying mobility which introduce a vast range of possible and perhaps realistic alternatives to provide a functional unit such as person kilometre or tonne kilometre. In short, many options are not relevant for comparison in the short term, but if the short term is irrelevant, even seemingly farfetched alternatives could be of interest.

Similarly, technical change open up for radically new functions and combinations. A vehicle with a fuel cell and an electrical engine could for

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[7] See e.g. Weidema et al. (2004) on applicability of different future-studies methods in LCA.

[8] Experience curves normally relate unit cost to the volume of cumulative production, but could probably also be used to estimate relationships between cumulative production and performance data.

[9] In USA ethanol is increasingly used as a petrol additive to substitute for MTBE (methyl tertiary butyl ether), which has recently been banned in a number of states since encountered in drinking water.

example be constructed in a very different way, providing a range of alternative and improved services. It could also be plugged into the electricity grid to supply peak power. How can such new options be taken into account in a comparative study? This indicates the difficulties of specifying relevant and comparable alternatives and functional units.

Using the current technical and economic feasibility “in the middle of the resource-transport service chain” as selection criteria becomes less appropriate the further into the future we look. Instead it probably becomes more fruitful to look at resource efficiency at one end-point, and the potential to fulfil needs and solve social problems at the other end-point of the resource-transport service chain (see figure 1).

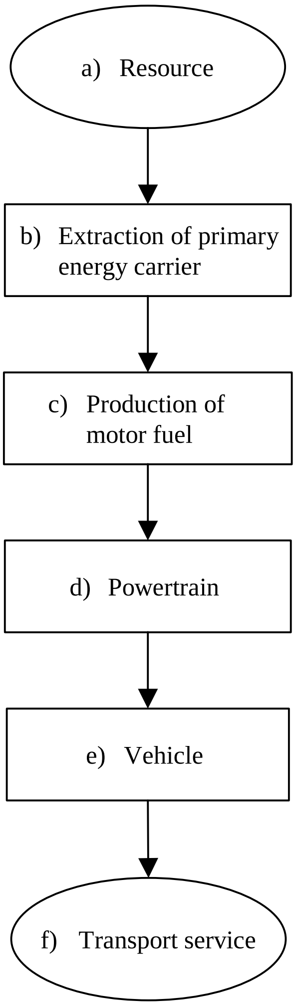

Figure 1: Over time, each stage a-f in the resource-transport service chain is subject to development, which in some cases may result in new and additional functions of a stage. Moreover, the way between stages a and f may be fulfilled in a totally different manner in a future solution.

3.2 Background and foreground systems

For convenience, the production processes in prospective LCA could be divided into a foreground and a background system, where the foreground system consists of those processes directly affected by decisions based on the study. The background system consists of all other processes included in the study, which only indirectly are affected by measures taken in the foreground system (Tillman 2000). In this WTT study, we assume the decision-maker to have an influence on policies governing the prerequisites for the production of biofuels (foreground system), without explicitly changing the electricity system or the supply of fossil fuels (background system).

Changes in background systems could be divided into those related to time (not affected by the foreground system), e.g. new technology used for heat and electricity production and transition to bio-based input materials, and those related to the scale of penetration of the studied product (indirectly affected by the foreground system), typically that a fuel is used for its own production and distribution.[10]

To illustrate the impact of such differences, we use the state studied in Edwards et al. (2003) as a base case and compare it with five stylised states with different background systems. The inputs in the base case are coal, heavy fuel oil, natural gas, the current electricity mix in the EU, input materials (such as methanol and hexane) of fossil origin and diesel. We assume that all coal, heavy fuel oil and natural gas are used for process heat, and can be replaced by coal in our coal cases and wood from short rotation forestry in our wood cases. Correspondingly, the electricity mix is replaced by electricity from coal in the coal cases and wood in the wood cases. In half of the cases diesel is replaced by the biofuel produced.

Edwards et al. (2003) assume that all inputs of transport fuel that are required to produce biofuels are diesel (figure 2, case I*). This assumption could possibly be relevant if the task is to estimate a minor change of the current state, but not if the assessment aims at comparing different fuels for strategic purposes. An increased scale of biofuel production implies that

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[10] Studying large technological changes, the background system eventually becomes a part of the foreground system.

more biofuels also are used as inputs. Biofuel use for biofuel production can be taken into account in two different ways (figure 2, case I and II).[11]

First, an increased scale of biofuel production can be assumed to lead to a change of the transport background system. Consequently, transport input can be modelled to reflect different biofuel shares (x) and scenarios of increasing shares (figure 2, case I). For low penetration levels (x close to zero) this general case degenerates to case I*. For 100 % penetration (x = 1) only biofuel is used as transport fuel input. In comparison to case I* this leads to an increased demand for other inputs per functional unit, such as heat and electricity. Case I can be termed a *gross output approach*.

When evaluating well-to-tank environmental impact of a fuel, the results are usually given in emissions and resource use per energy unit of the studied fuel. Consequently one will have to choose if the energy output the environmental impact relates to is the gross energy output of the fuel production system or the net output that is left when a part of the gross output is used in fuel production processes (see figure 2). In the *net output approach* (case II), the functional unit is 1 MJ of fuel available for other purposes than producing fuel. In comparison to case I this leads to an additional increase of other inputs per functional unit. The net output approach can be used for any scale of biofuel penetration. Here we choose to use the net output approach since it better reflects the inherent differences between fuels. This approach can be analogously used for other products that are used for their own production.

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[11] For this purpose, only the energy content of the fuel is regarded, i.e. potential efficiency differences in the vehicles are not included. The biofuel is assumed to directly replace diesel on an energy basis.

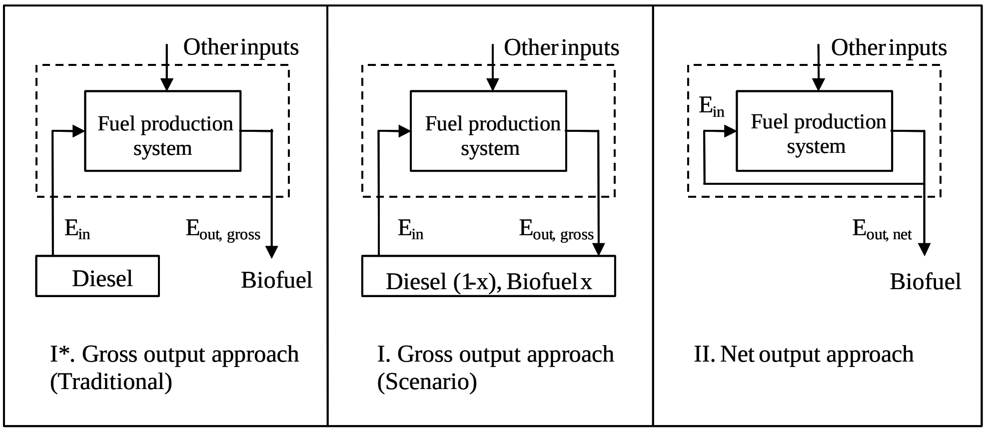

Figure 2: Biofuel use for biofuel production can be taken into account in two different ways. The environmental impact of a fuel can be related to $E_{out, gross}$ *(gross output approach) or* $E_{out, net}$ *(net output approach), where* $E_{out, gross}$ *is the energy content in the produced fuel,* E_{in} *is the energy content of the fuel used for production and distribution, and* $E_{out, net}$ *is the energy content of the fuel available for other purposes (*$E_{out, net} = E_{out, gross} - E_{in}$*).*

3.3 By-product credits

The life-cycle environmental impact of a product also depends on the type of feedstock used and the production of by-products. This introduces a different kind of scale dependency. Different types of feedstock (such as waste and farmland) are available in different quantities and different by-product markets vary in size.[12] Here we illustrate this by giving examples of how by-product credits for wheat ethanol and rapeseed methyl ester (RME) production in EU-15 could change with the level of market penetration.

As for many other products, the production of biofuels will result in one or more by-products that could be used for various purposes. In well-to-wheel

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[12] The quantity of biomass available for biofuel production is limited by the land area that can be used for fuel crop production (e.g. set-aside land), the selected crops and their respective yield, usable surplus from other sectors (e.g. from wheat, sugar and wine production), and usable residues from other sectors (e.g. corn stover, wheat straw, rape straw, forest residues and black liquor). For an overview of usable residues potential, see e.g. Kim and Dale (2004). No resource potential or land availability estimations have been included in this analysis, for such figures see e.g. Hall (1997), Hoogwijk et al. (2003) or IEA (2004).

studies, the environmental impact of fuel production then has to be allocated between the fuel and its by-products. There are several methods for doing this, and in the ISO standard for life cycle assessments it is recommended that when inputs and outputs cannot be directly connected to a product, the system should be expanded to include the additional functions related to the by-products. If this is not possible, the inputs and outputs should be allocated according to physical relationships or economic value (Ahlström 2002). In our example we use the system expansion method, so that credits are given to the fuels for avoided production of products that can be replaced by the by-products. This should reflect the actual environmental impact of producing the fuel, and the results can be compared with those for other fuels.[13]

Irrespective of allocation method, the use of by-products is usually decided by the price situation, and increased production may lead to price decreases and that by-products are sold on new markets. Using the system expansion approach, we do not consider this secondary scale effect, but use the present size of different markets. With economic allocation the uncertainty is higher since price changes need to be modelled. One major advantage with economic allocation, though, is that it reflects the underlying economic reasons for production. The problem with allocation based on physical properties, such as energy contents or mass, is that the different properties of products are not taken into account. When studying biofuels, for example, the energy contents does not reflect the variation in quality between solid biofuels, liquid biofuels and products used for chemical or food purposes.

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[13] An elegant example of how to deal with system expansion when studying corn ethanol is given in Kim and Dale (2002).

4 CONCAWE/EUCAR/JRC well-to-wheel study

Recently, a study of various alternative fuels considered for the future European market (2010 and beyond) was published. The project is a joint evaluation performed by CONCAWE (Oil Companies' European Association for Environment, Health and Safety), EUCAR (European Council for Automotive R&D) and JRC (the Joint Research Centre of the EU Commission). Some of the objectives of the study were to establish a consensual well-to-wheels energy use and GHG emissions assessment, and to have the outcome accepted as a reference by relevant stakeholders. Several fuels were included, all with an assumed market potential of 5-15 % in 2010-2020. The studied indicators were energy use, GHG emissions and costs. Data apply to EU-15 (Edwards et al. 2003).

We use this study to quantitatively explore how results could change when we change some factors that are sensitive to time and scale. For the quantitative assessment wheat ethanol and rapeseed methyl ester (RME) are selected for review, as they are gaining much attention in Europe (ViewLS 2004), and they entail interesting by-product and land-use aspects. Methanol from gasification of wood from short rotation forestry is also included in the study, as an example of a biofuel with higher production efficiency.[14,15] The environmental indicators studied are GHG emissions (carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), measured in CO_2-equivalents) and agricultural land use (area measure). The CO_2-equivalents are calculated from the IPCC factors with a 100-year time frame, published in Houghton et al. (2001). To calculate N_2O emissions from soils, the GREASE tool (GReenhouse Emissions from Agricultural Soils in Europe) developed at the JRC Institute of Environment and Sustainability was used (Edwards et al. 2003).

Edwards et al. (2003) assume that the effect of increased biofuel production in the EU will be reduced cereals export, which in turn would result in yield improvements and marginal intensification in other parts of the world. Therefore no reference crop is regarded. If the increase were assumed to come from e.g. grasslands converted to new arable area, the GHG emissions

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[14] Short rotation forestry can be e.g. poplar or willow grown on agricultural land (Edwards et al. 2003).

[15] Data for methanol production are based on pilot projects (Edwards et al. 2003).

for the biofuels would have been higher, due to a reduction in the soil carbon stock. This is consistent with the marginal approach used in Edwards et al. (2003). To be fully consequent, however, only the increase in environmental impact due to more efficient agriculture should be attributed to the fuels, but this is not done in Edwards et al. (2003). Here, all the GHG emissions and land use connected to the growing of crops used for fuel production are included in the analysis.

The choice of data in Edwards et al. (2003) is generally characterized by marginal thinking, but we choose to use present average data where applicable, i.e. for the electricity mix and for the supply of natural gas. This is appropriate for state-oriented LCA, even though estimated future averages would be preferable for our stylised future states. We adopt the assumption used in Edwards et al. (2003), that ethanol and RME are transported within the existing infrastructure for petrol and diesel. (Data choice is briefly discussed in section 3.1.)

Edwards et al. (2003) has been criticised for not using the wheat straw by-product (see section 5.2), and for some of the results not being fully explained by the underlying calculations (Bauen et al. 2004). We have also found some calculation mistakes in the first version of the appended spreadsheets (Jonasson 2004).

In most WTW studies, accordance with future vehicle emission standards, such as the US Environmental Protection Agency (EPA) Tier 2 (GM 2001) or Euro IV (Ahlvik and Brandberg 2001; GM 2002), is assumed. Edwards et al. (2003) use Euro III for 2002 vehicles and Euro IV for 2010 vehicles.[16]

The wood-to-electricity process is assumed to be the large-scale IGCC (Integrated Gasification and Combined Cycle) plant used in Edwards et al. (2003). We have assumed that methanol used for esterification of rapeseed oil is of fossil origin, except for in the wood cases, where it is produced through gasification of wood from short rotation forestry. (Distribution of this methanol is not included.)

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[16] This will have an indirect effect on the fuels studied, and possibly also on the production processes required.

Agricultural yields and lower heating values (LHV) of crops are given in table 1.

Table 1: Agricultural yields and lower heating values (LHV) of crops. The energy yields are calculated values.

	Mass yield (ton/ha)	LHV (GJ/ton)	Energy yield (GJ/ha)
Rapeseed[a]	3.0	23.8	72.1
SRF wood[b]	10	18.5	185
Wheat[c]	6.7	17	114

[a] 2002 data (EU 2003a); [b] (Edwards et al. 2003); [c] (EU 2003a)

5 Results

We start by looking at the effect of producing the fuels with different alternative background systems, without taking by-product credits into account (section 5.1). These are studied separately for varying by-product market sizes in section 5.2, and finally the results are combined to yield the overall results on time and scale aspects in technology LCA presented in section 5.3.

5.1 Stylised background systems

The stylised states analysed for each fuel are two coal cases (denoted by C), two mixed cases (M), and two wood cases (W), briefly described in section 3.2. For each category, the alternatives diesel (D) and the studied biofuel (B) are used as process energy and transport fuel. The results are presented in figure 3. The mixed cases with diesel (MD) are similar to those presented in Edwards et al. (2003), but with some modifications in data choice (see section 4) and without by-product credits. The value for diesel production is shown for comparison, and includes the non-renewable CO_2 resulting from diesel combustion.[17]

Clearly, the GHG emissions are highest in the coal cases (C), and lowest in the wood cases (W) for each fuel. There are still some GHG emissions in the wood case with biofuels (WB), because of input materials based on fossil resources, but mainly due to N_2O emissions from soils and production of fertiliser (more than 95 % of the GHG emissions in this case).[18] For high GHG emissions, the emissions increase when substituting biofuel for diesel used for process energy and distribution. The reason for this is that the emissions from biofuel production are higher than the emissions from diesel production and use in those cases, or that the net output approach gives a decreased output of biofuel.

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[17] The well-to-tank values for petrol and diesel are about the same (Edwards et al. 2003).

[18] Hexane is used for rapeseed oil extraction and various minor input chemicals are used for esterification of the oil.

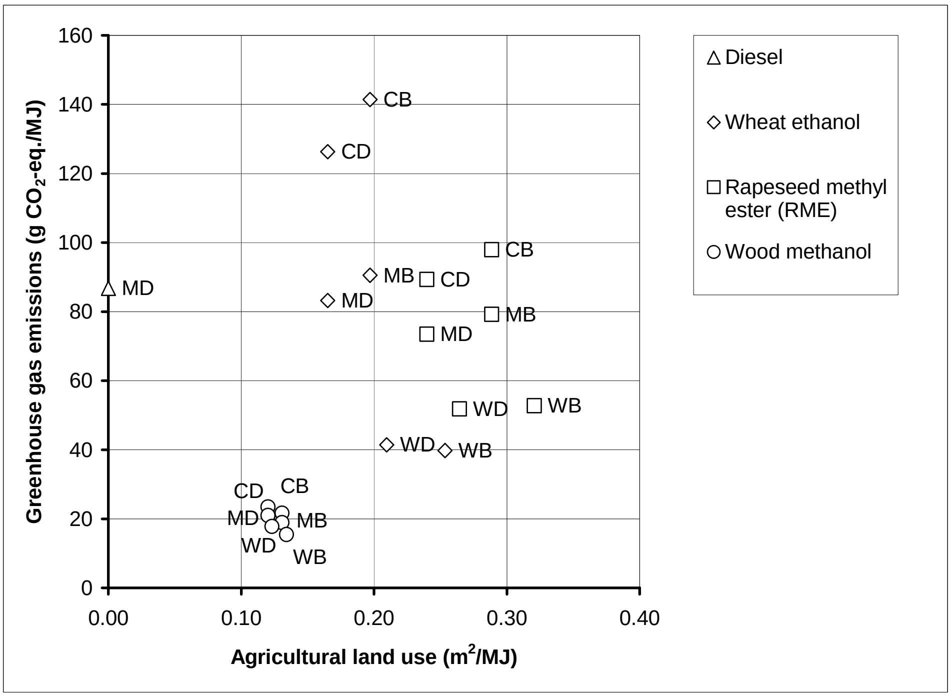

Figure 3: GHG emissions and agricultural land use from fuel production for the coal (C), mixed (M) and wood cases (W), in combination with diesel (D) and the studied biofuel (B). The value for diesel production is shown for comparison, and includes the non-renewable CO_2 resulting from diesel combustion. No by-product credits are included.

The GHG emissions are higher for wheat ethanol than for RME in the coal cases (C), the difference gets smaller in the mixed cases (M), and they are higher for RME in the wood cases (W). This is due to that RME has a larger share of its GHG emissions from soils and production of fertiliser. Methanol gives the lowest GHG emissions in all cases.

The differences in agricultural and processing yields give the difference in agricultural land use between the three fuels, where methanol has the lowest and RME has the highest use in each case. Agricultural land use is higher in the wood cases (W), due to the land needed for forestry to produce heat and electricity. It gets even higher when the studied biofuel is used for production and distribution (B).

The influence of using the net energy approach is about 9 % higher emissions and land use per functional unit for wheat ethanol and RME, and about 4 % higher for methanol, than if we had used the gross energy approach with 100 % biofuels in the background system. For diesel the effect of choosing the net energy approach is assumed to be small, and is not included in the calculations.

5.2 By-product market variations

By-product from production of ethanol is *DDGS (Distiller's Dried Grains with Solubles),* and from RME *rapeseed meal* and *glycerine,* while there are no marketed by-products from methanol production. We assume that the by-products are used for the functions proposed by Edwards et al. (2003). DDGS and rapeseed meal are here used as ingredients in animal feed products, replacing soya meal as a protein source. Glycerine is used as a chemical, replacing synthetic glycerine or propylene glycol of fossil origin. It can also be used as an ingredient in animal feed products, replacing wheat as an energy source. When these higher value markets have been saturated, we assume that DDGS, rapeseed meal and glycerine can be used as fuels for heat production. Increased supply of these products may though lead to the emergence of new fields of application, and hence new markets.[19]

In the EU, wheat straw is often (to 65 %) ploughed back into the soil to prevent soil degradation, as is generally done with rape straw (Edwards et al. 2003). We have not assumed any by-product credits for straw, but it can be argued that especially wheat straw should be counted as a by-product, as it can be collected and used for energy purposes, e.g. directly in ethanol production. From a GHG point-of-view this could make wheat ethanol more advantageous, as straw yield can be as much as 65 % of the grain output, based on mass (Edwards et al. 2003).[20]

The European market for glycerine is estimated at 325 kton/yr, and for propylene glycol 488 kton/yr (Tefac 2004). The market for animal feed is assumed to be equal to the feed production in Europe, which in 2001 was

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[19] Stillage and rapeseed meal could e.g. be digested to yield biogas, which also would avoid the need for drying of stillage to produce DDGS. We have no figures on this, but investigations on digestion of wheat stillage are going on at Agroetanol (2004). Gärtner and Reinhardt (2003) indicate that digestion of rapeseed meal is more favourable than using it as animal feed, but less favourable than direct combustion, when energy use and GHG emissions are concerned. This was when biogas was used for power production, replacing German electricity, and the same conclusions were drawn with or without utilisation of excess heat. See Edwards et al. (2003) for a similar example on sugar beet ethanol. Stillage could also be used as fertilizer.

[20] The same discussion applies to corn stover in the US (Sheehan et al. 2004).

117 Mton (FEFAC 2004). The maximum recommended contents of the by-products in animal feed products and the resulting total markets are given in table 2. These numbers may be a bit conservative, and can possibly be increased (Lantmännen 2004).

Table 2: The animal feed markets for the by-products and the maximum recommended contents in the respective feed products.

	Feed production (Mton/yr)[a]	Max. DDGS contents (Mton/yr)[b]	Max. rapeseed meal contents (Mton/yr)[b]	Max. glycerine contents (Mton/yr)[b]
Cattle	35.4	3.54 (10%)	6.02 (17%)	3.54 (10%)
Pigs	42.6	4.26 (10%)	5.11 (12%)	4.26 (10%)
Poultry	38.6	0 (0%)	3.86 (10%)	3.86 (10%)
Total market	117	7.80	15.0	11.7

[a] (FEFAC 2004); [b] (Lantmännen 2004)

The lower heating values (LHV) of by-products and replaced products are used when substituting fuels used for heat production. When replacing for animal feed purposes, digestible protein or energy contents are used, according to the qualities of the by-products (see table 3). Digestible energy for glycerine is assumed to be 95 % of LHV, the same fraction as for wheat (Edwards et al. 2003).

Table 3: Lower heating values (LHV) and digestible energy and protein contents of the by-products and the products they are assumed to replace (Edwards et al. 2003).

	LHV (MJ/kg)	Digestible energy (MJ/kg)	Digestible protein (% of dry matter)
DDGS	17.8	not used	38.5
Rapeseed meal	18[a]	not used	39.6
Glycerine	16	15	not used
Wheat	17	16	not used
Soya meal	not used	not used	49

[a] Same as for wood assumed, but the figure is depending on how much oil is left in the product. An interval of 15.29-20.06 MJ/kg is given in Bernesson et al. (2004).

We use the mixed background system with ethanol and RME used for their own production (MB-case, net output approach) for the assessment of by-product market variations. This means that an increased market penetration of the biofuel does not affect emissions per unit of production, when by-products are not considered. Similarly, the avoided use of transport fuel from the production of the replaced products is taken into account by increasing net biofuel output. The by-product heat mainly replaces heat from natural gas, both in biofuel production and in other applications, since natural gas is the dominating fuel for heat production in the mixed case. (Results for the CB- and WB-cases are presented in the appendix.)

Figure 4 and 5 show the by-product credits in terms of decreased GHG emissions and agricultural land use for wheat ethanol and RME, respectively. The credit is related to the respective biofuel share of petrol and diesel used for transport in EU-15 (the market penetration of the biofuel compared to the use of petrol and diesel in EU-15 today).[21] Both marginal credit steps and the resulting average credit curves are given.

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[21] The figures only reflect changes over scale. The expected increase of total fuel demand over time is not considered. Data is taken from (EU 2003c).

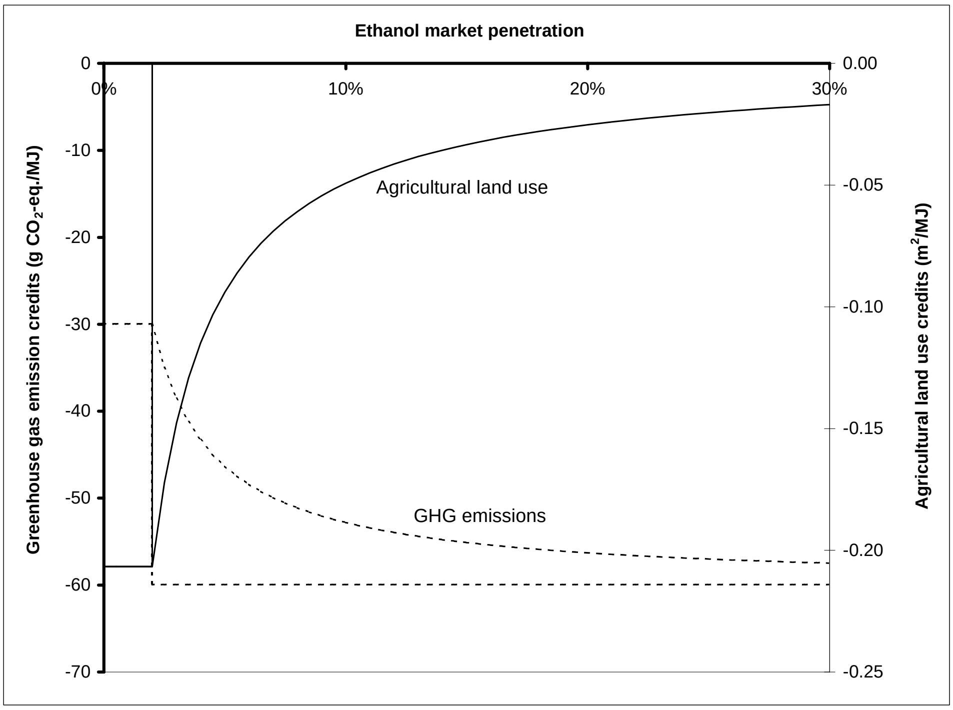

Figure 4: By-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for DDGS from ethanol production (case MB). The steps in the figure correspond to different by-product market potentials, measured in percent of petrol/diesel in EU-15 replaced by the ethanol produced. The smooth lines are the average credits for ethanol at a certain penetration rate. Negative values indicate that the amount should be subtracted from the ethanol results without allocation (from figure 3). No interesting effects occur above 30 % penetration.

The step in figure 4 (2.0 %) is limited by the market size for DDGS replacing soya meal in animal feed products. When this market is saturated the DDGS is used for heat production, replacing natural gas. The effect on the by-product credits of the step is that GHG emission credits increase, while the agricultural land use credits decrease above this level of penetration. No interesting effects occur above 30 % penetration.

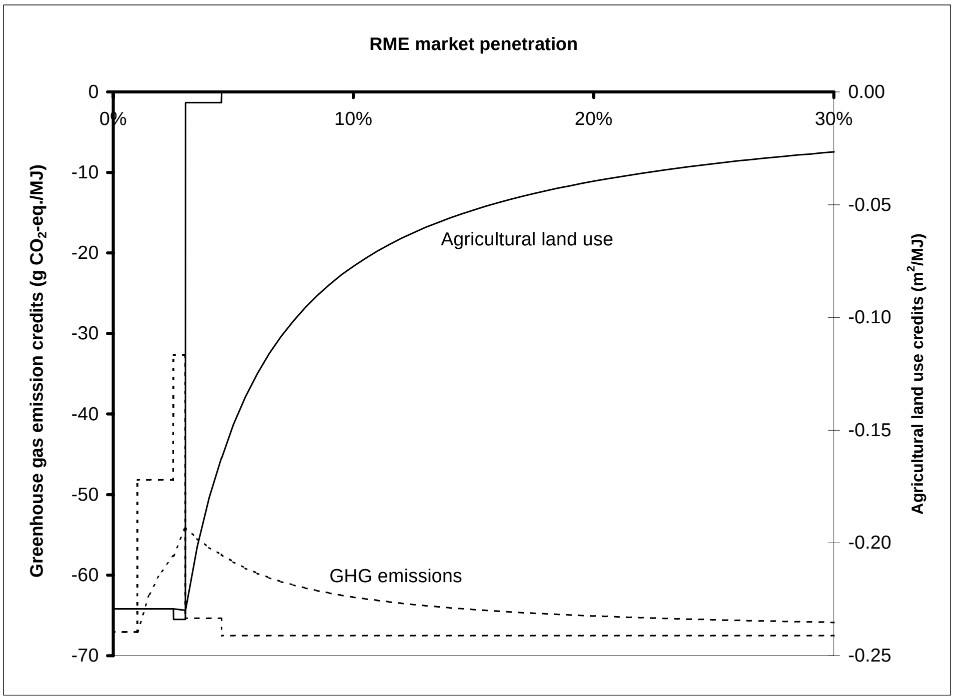

Figure 5: By-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for both rapeseed meal and glycerine from RME production (case MB). The steps in the figure correspond to different by-product market potentials, measured in percent of petrol/diesel in EU-15 replaced by the RME produced. The smooth lines are the average credits for RME at a certain penetration rate. Negative values indicate that the amount should be subtracted from the RME results without allocation (from figure 3). No interesting effects occur above 30 % penetration.

The first step in figure 5 (0.9 % penetration) is limited by the market for glycerine replacing synthetic glycerine, the second by glycerine replacing propylene glycol (2.3 %) and the third by rapeseed meal replacing soya meal in animal feed products (3.1 %). Up to this point, the GHG emission and land use credits decrease or remain constant with each step. However, when rapeseed meal and later glycerine (at 4.3 %) start to be used for heat production and substitute natural gas, the GHG emission credit rapidly increases, while the marginal land use credit goes to zero.

As scale increases, a larger amount of by-products are used for heat production, and as long as the market is not saturated, they can be assumed to replace the heating fuel of the background system. This is consistent with the net output approach, which proposes that the by-products are used directly in the production of the biofuels.

The by-product credits are added to the GHG emissions and agricultural land use without allocation from the previous section, to yield the results shown i figures 6 and 7.

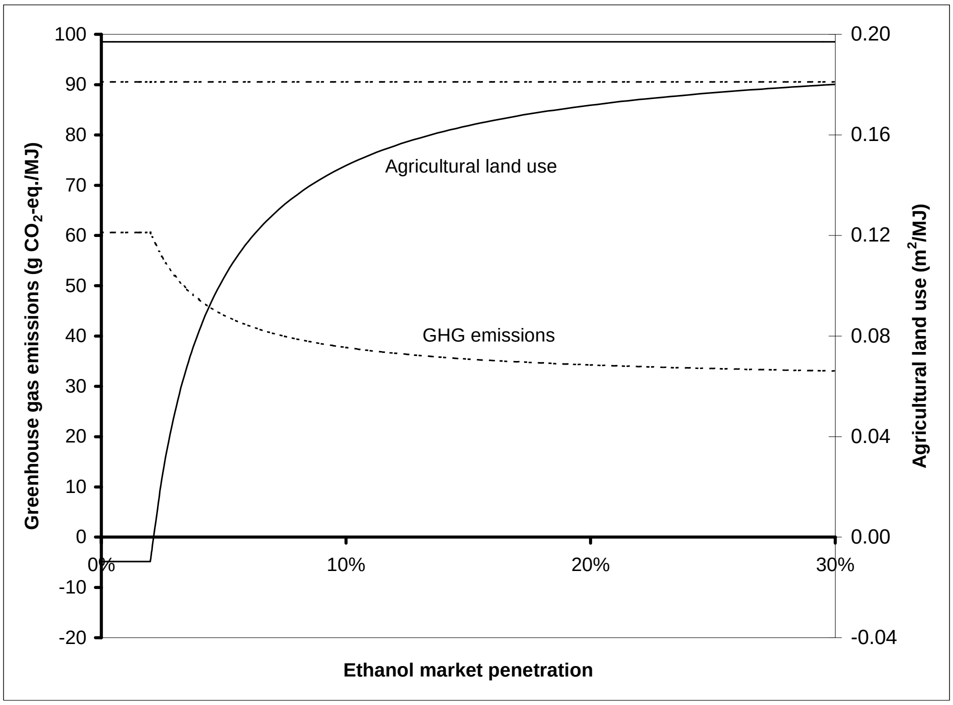

Figure 6: GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for ethanol at different penetration rates, with credits for different uses of the by-product considered. The horizontal lines at 91 g CO_2-eq./MJ and 0.20 m^2/MJ illustrate the GHG emissions and land use if no by-product credits are included. Below zero emissions is a result of the system expansion method.

Apart from what has already been noticed, figure 6 shows that for a low market penetration of ethanol, the agricultural land use is negative. This indicates that the by-product credit is larger than the land use for ethanol production without allocation.

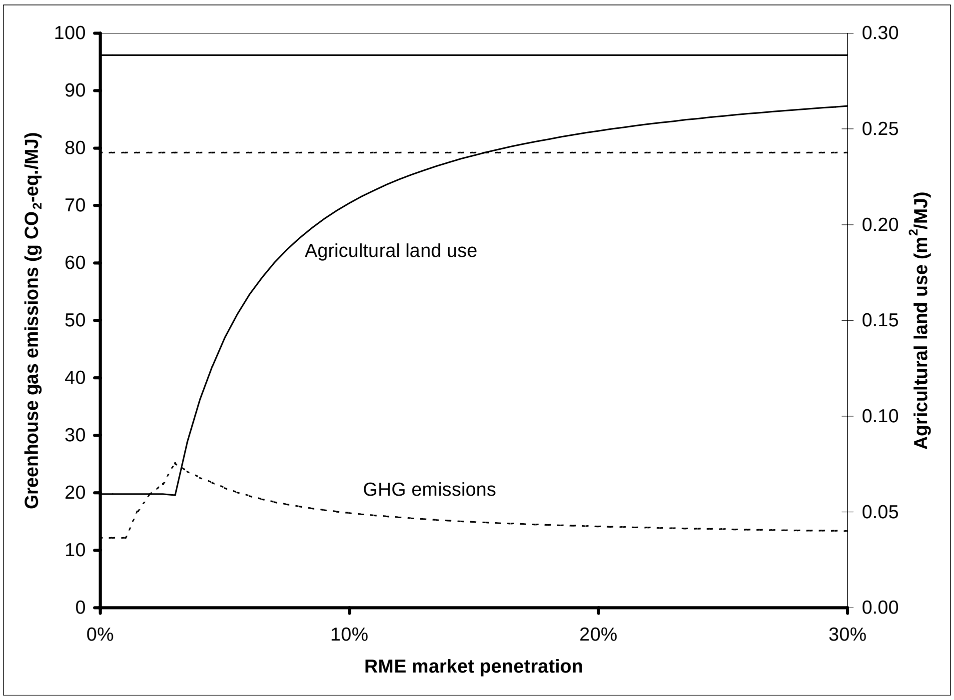

Figure 7: GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for RME at different penetration rates, with credits for different uses of the by-products considered. The horizontal lines at 79 g CO_2-eq./MJ and 0.29 m^2/MJ illustrate the GHG emissions and land use if no by-product credits are included.

In the here illustrated MB-cases, GHG emissions are minimized if the by-products are used for heat production replacing natural gas and not for anything else. This would not be the case with a bioenergy background system (WB) in which heat from by-products would replace heat from wood (see figures in the appendix). The situation would also change if saved bioenergy feedstock or land is given a GHG credit and it is assumed that residual land is used for harnessing of energy that can replace fossil fuels in another sector, or replace more transport fuel. This can be done by continued cultivation of present crops, or by switching to more area efficient energy production. Producing short rotation energy forest on the saved land would give even higher GHG credits, and using the land for direct solar energy conversion would lead to an increased area efficiency of at least a factor of ten above what could be reached for bioenergy plantations (Johansson and Burnham 1993).

On the other hand, if an equal GHG value were given to the land used to grow the wheat or rapeseed used for biofuel production, total GHG emission would rise well above the diesel GHG emissions. This reflects the fact that using a given land area to substitute heat from bioenergy for heat from natural gas gives a much larger GHG reduction than using the same area to substitute ethanol or RME for petrol and diesel.

5.3 Stylised states with varying by-product markets

We can now combine the results for varying by-product markets with the stylised background systems used in section 5.1. For this purpose we assume the extreme conditions set by a low market penetration (<1 %), as well as a high market penetration of biofuel (~100 %).[22] To begin with, we analyse the variations in GHG emissions and agricultural land use for each fuel, with all combinations of background systems and by-product markets (figure 8 and 9). The three value points for each case mark out the possible span in emissions and agricultural land use of the different background systems and by-product markets used in the assessment. The well-to-tank GHG emissions given in Edwards et al. (2003) are 7.1 g CO_2-eq./MJ for wood methanol, 74.1 g CO_2-eq./MJ for wheat ethanol, and 48.9-53.9 g CO_2-eq./MJ for RME, depending on glycerine use. Agricultural land use is not considered.

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[22] A low market penetration is implicitly assumed in the allocation process of most LCAs of transport fuels, and is a reasonable approximation when modelling the present situation for biofuels in Europe. For future states, if implying large technological changes, this assumption is not valid.

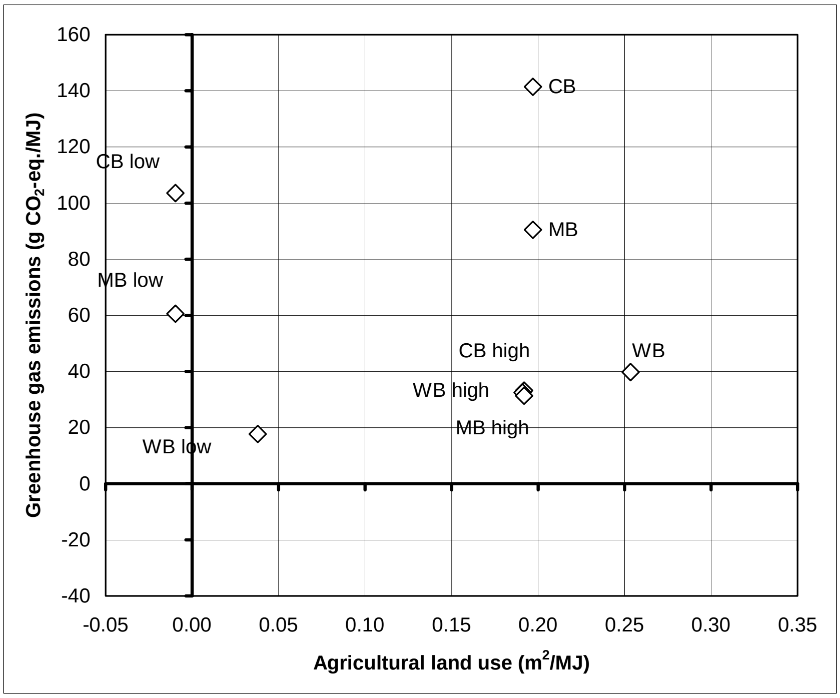

Figure 8: GHG emissions and agricultural land use from wheat ethanol production for the coal (C), mixed (M) and wood cases (W), in combination with the biofuel (B). The results are given with by-product credits for a low market penetration (low) and a high market penetration (high) of ethanol, and without by-product credits.

For ethanol, the effect of including by-product credits with a low market penetration of the fuel is a decrease of GHG emissions and agricultural land use, where the latter is eliminated (CB low, MB low), or almost eliminated (WB low), because of the avoided production of soya meal.

With a high market penetration there is a very large decrease in GHG emissions in the coal case (CB high), a large decrease in the mixed case (MB high), and a small decrease in the wood case (WB high), as the by-product is used for energy purposes. There is also a slight decrease in land use due to avoided use of transport fuel when replacing coal, natural gas or wood, which gives a larger net output of ethanol. The absolute result is about the same for all background systems, which is a consequence of the relation between the results without by-product credits, and the amount and energy contents of the by-product.

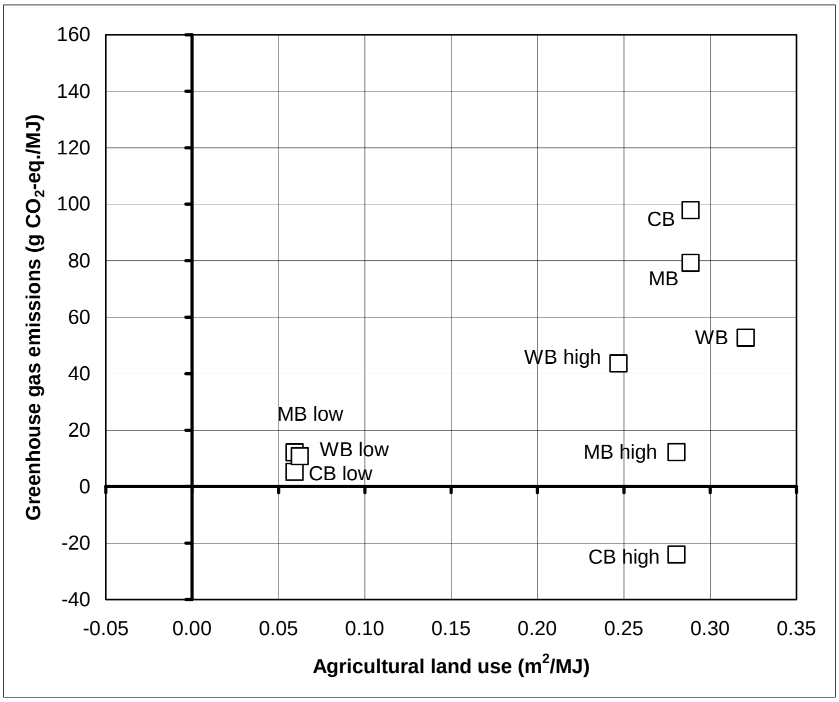

Figure 9: GHG emissions and agricultural land use from RME production for the coal (C), mixed (M) and wood cases (W), in combination with the biofuel (B). The results are given with by-product credits for a low market penetration (low) and a high market penetration (high) of RME, and without by-product credits.

For RME, the effect of including by-product credits with a low market penetration of the fuel is a very large to large decrease of GHG emissions for the coal (CB low), mixed (MB low) and wood cases (WB low), respectively. The absolute result is about the same for all background systems.

With a high market penetration there is a very large decrease in GHG emissions for the coal case (CB high), a large decrease for the mixed case (MB high) and a small decrease for the wood case (WB high). There is also a small reduction in land use due to a larger net output of RME.

Finally, we can use these results to compare the different fuels when they are produced in systems with a low and a high market penetration of the biofuels, respectively (figures 10 and 11).

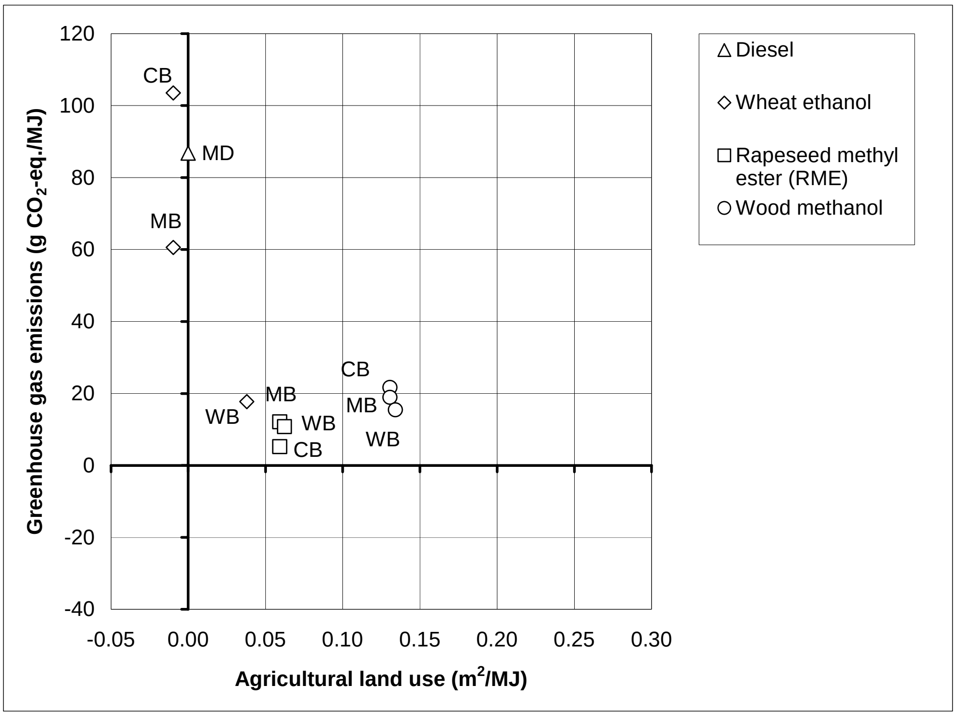

Figure 10: GHG emissions and agricultural land use from fuel production for the coal (C), mixed (M) and wood cases (W), in combination with the studied biofuel (B). The results are given for a low market penetration of the biofuels. The value for diesel production is shown for comparison, and includes the non-renewable CO_2 resulting from diesel combustion. The values for methanol are not affected by the market penetration of the fuel.

We can see from figure 10 that in systems with a low penetration of biofuels, both GHG emissions and agricultural land use are lower for RME than for methanol in all cases. For ethanol, the GHG emissions are largely dependent on background system, while the land use is lower than for both RME and methanol, and even below zero in the MB- and CB-cases. The values for methanol are not affected by the market penetration of the fuel.

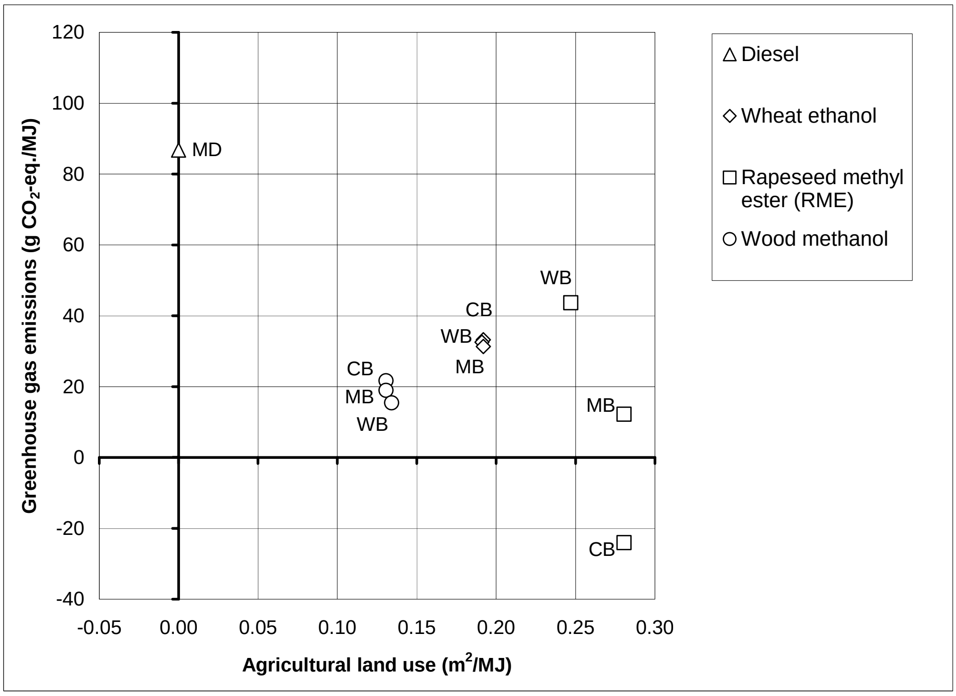

Figure 11: GHG emissions and agricultural land use from fuel production for the coal (C), mixed (M) and wood cases (W), in combination with the studied biofuel (B). The results are given for a high market penetration of each biofuel. The value for diesel production is shown for comparison, and includes the non-renewable CO_2 resulting from diesel combustion. The values for methanol are not affected by the market penetration of the fuel.

In systems with a high penetration of biofuels (figure 11) the results are quite different! Here, both ethanol and RME have a higher land use than methanol, with RME having the highest. For ethanol, GHG emissions are slightly higher than for methanol in all cases, while for RME, they vary from the highest (case WB) to the lowest value (case CB), which is even negative.

The fact that RME looks good from a GHG point-of-view in the coal case (CB) should not be interpreted that using coal for heat and power is to prefer, but *if* such a system is present, this value for RME should be used in comparisons with other fuels. A more reasonable conclusion is, that if we are aiming at a system based on short rotation forestry and biofuels (case WB), RME should not be chosen before ethanol and methanol.

6 *Conclusions*

Standard LCA methodology (including well-to-wheel studies) is developed to answer questions about environmental impacts of the current (or historical) production and use of one unit of a product, or of minor product or process changes. When this methodology is used to provide answers to questions about strategic technological choices, with the long-term goal of changing large-scale technological systems, the result is of little value and in the worst case interpretations of the result may be grossly misleading. For such choices, it is important to include time- and scale-related factors, and to look beyond the current situation. We suggest to study many possible future states, what we call “stylised states”, to explore general technology differences. This case study deals with alternative fuels for transportation, but the methodological results could be of equal importance when studying other major technologies in society. We choose to address three issues in this report.

First, shifting time frame gives room for technical development that should affect not only the choice of performance data, but perhaps also the functional unit and the selection of technologies under study.

Second, background systems such as heat and power production change over time, and we exemplify by using three different systems, mainly based on coal, natural gas and short rotation forestry, respectively. Increased production volumes may for some technologies also change the background system, which is of particular importance for technologies that are used in their own production processes. We show that for biofuels changes in background systems have consequences not only for greenhouse gas (GHG) emissions and agricultural land use for each fuel chain, but also for the ranking order of e.g. wheat ethanol and RME, in terms of GHG emissions. We use what we call a “net output approach”, which implies that a fraction of the produced biofuel is used for its own production. Accordingly, the functional unit used in this study is 1 MJ fuel available for other purposes than producing fuel.

Finally, different types of feedstock are available in different quantities and different by-product markets vary in size. Allocation of environmental impact between product and by-products is here made through system expansion, and we study some possible markets for by-products. To give an example of by-product effects, current key markets for ethanol by-products in EU-15 correspond to an ethanol production that covers about 2 % of demand, and for RME about 3 %, that is, well below the 5.75 % EU biofuel target for 2010. Therefore, the GHG emissions and agricultural land use allocated to the fuels differ between a low and a high market penetration.

Combining the results, we show that time and scale are important factors for the ranking of wheat ethanol, RME and wood methanol in terms of GHG emissions and agricultural land use, as the results are dependent on assumptions regarding background system and by-product markets. We indicate that agricultural land use results can be weighted in GHG terms in several ways, e.g. by using short rotation forestry or solar panels as a reference, an approach that would require further research.

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APPENDIX

By-product market variations

Wheat ethanol

In figures A, C and E, by-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for DDGS from ethanol production are shown (cases CB, MB and WB). The steps in the figure correspond to different by-product market potentials, measured in percent of petrol/diesel in EU-15 replaced by the ethanol produced. The smooth lines are the average credits for ethanol at a certain penetration rate. Negative values indicate that the amount should be subtracted from the ethanol results without allocation. No interesting effects occur above 30 % penetration.

In figures B, D and F, GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for ethanol are shown (cases CB, MB and WB), at different penetration rates, with credits for different uses of the by-product considered. The horizontal lines illustrate the GHG emissions and land use if no by-product credits are included. Below zero emissions is a result of the system expansion method.

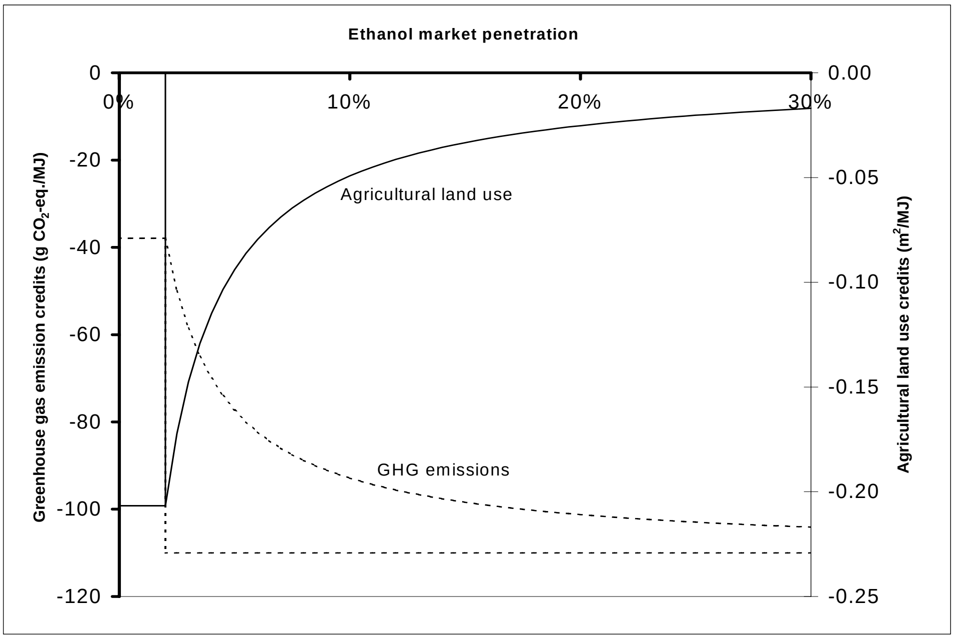

Figure A: By-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for DDGS from ethanol production, case CB.

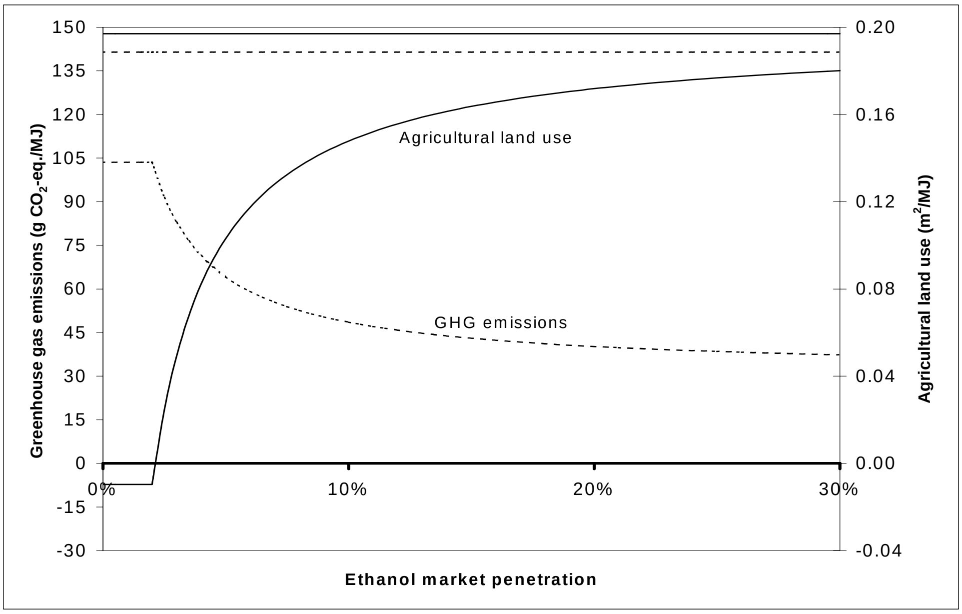

Figure B: GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for ethanol, case CB.

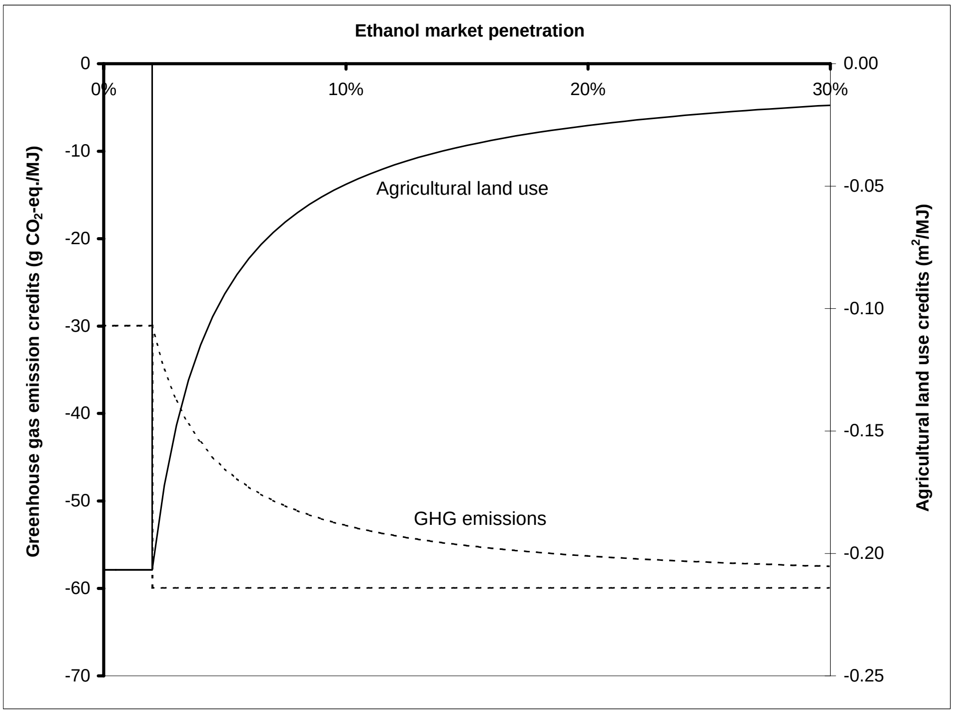

Figure C: By-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for DDGS from ethanol production, case MB.

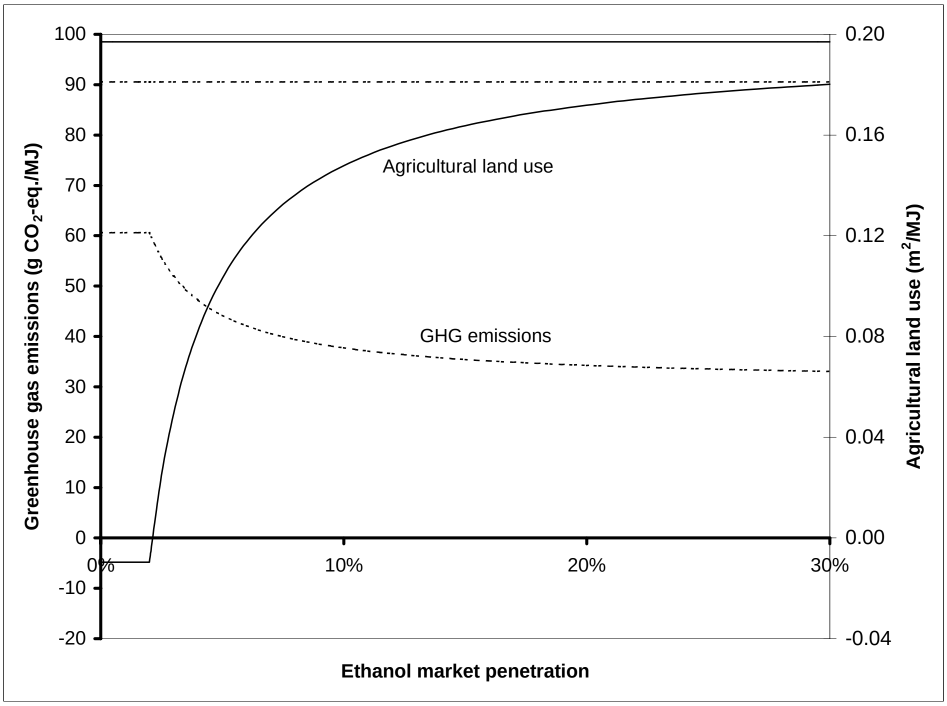

Figure D: GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for ethanol, case MB.

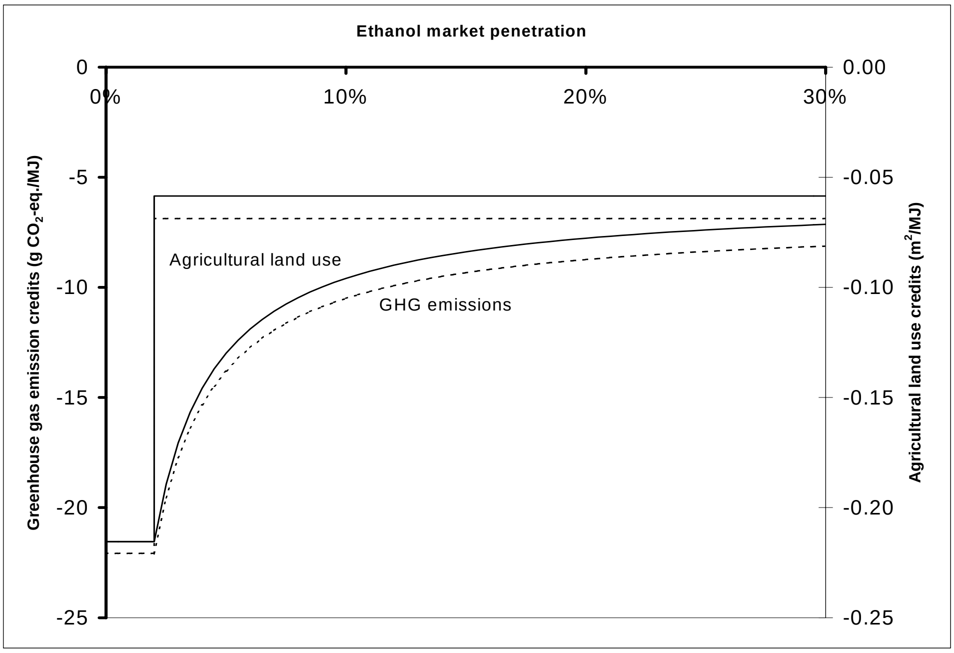

Figure E: By-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for DDGS from ethanol production, case WB.

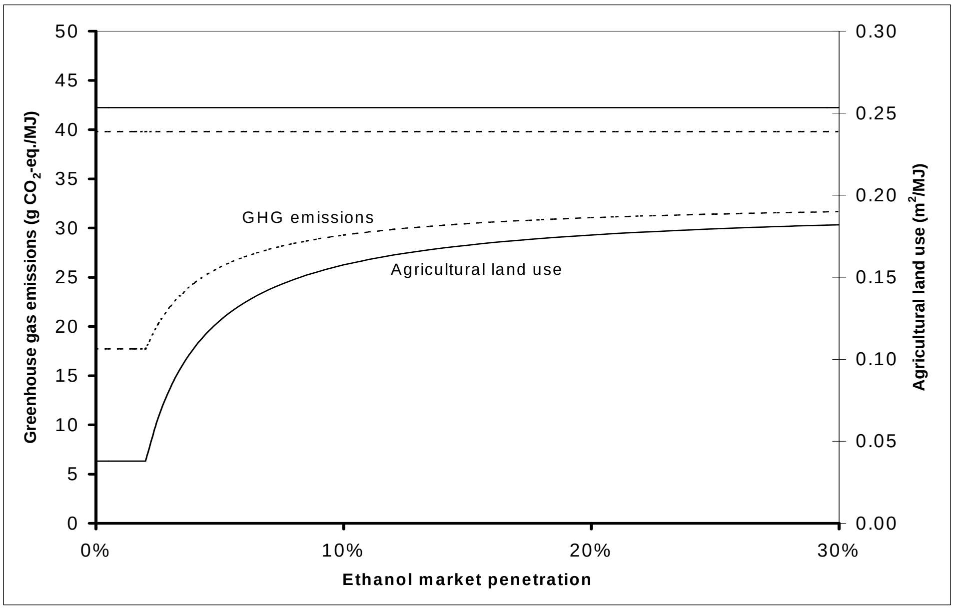

Figure F: GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for ethanol, case WB.

RME (rapeseed methyl ester)

In figures G, I and K, by-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for both rapeseed meal and glycerine from RME production are shown (cases CB, MB and WB). The steps in the figure correspond to different by-product market potentials, measured in percent of petrol/diesel in EU-15 replaced by the RME produced. The smooth lines are the average credits for RME at a certain penetration rate. Negative values indicate that the amount should be subtracted from the RME results without allocation. No interesting effects occur above 30 % penetration.

In figures H, J and L, GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for RME are shown (cases CB, MB and WB), at different penetration rates, with credits for different uses of the by-products considered. The horizontal lines illustrate the GHG emissions and land use if no by-product credits are included. Below zero emissions is a result of the system expansion method.

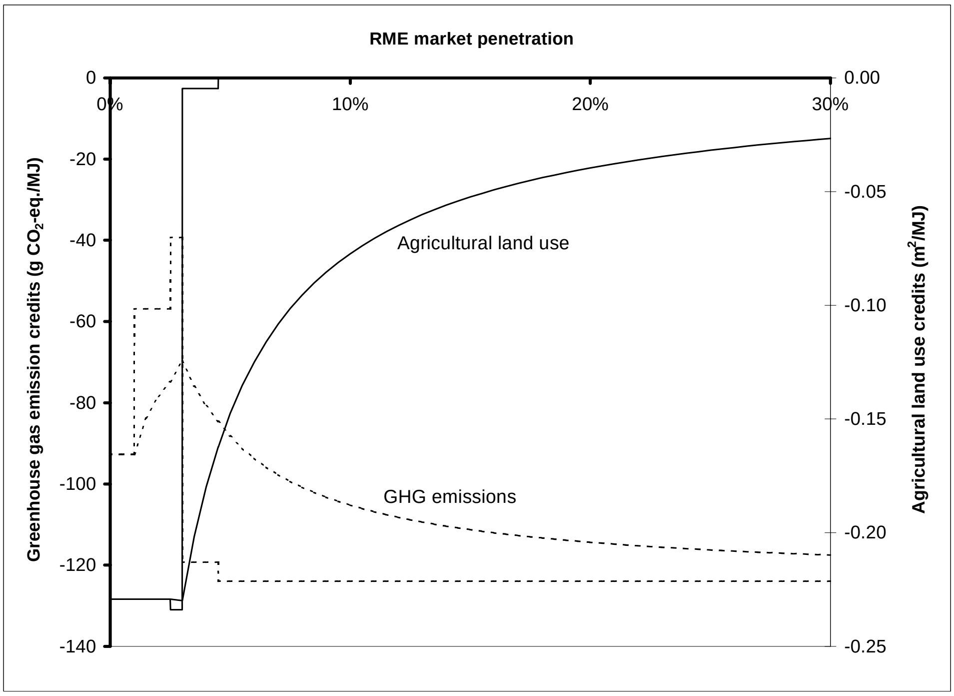

Figure G: By-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for both rapeseed meal and glycerine from RME production, case CB.

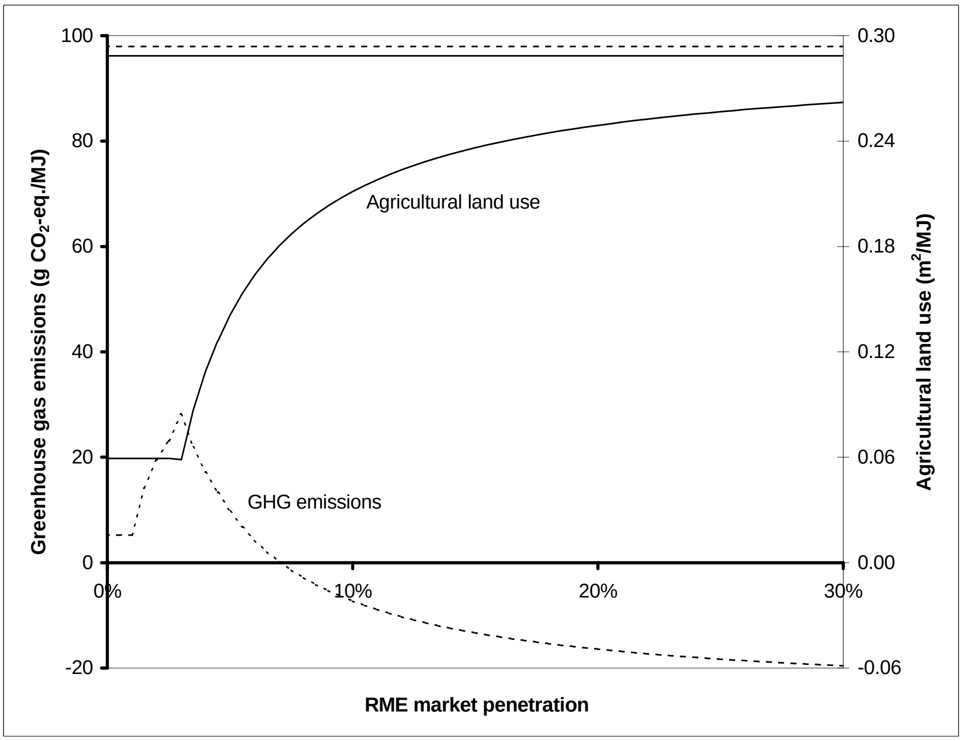

Figure H: GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for RME, case CB.

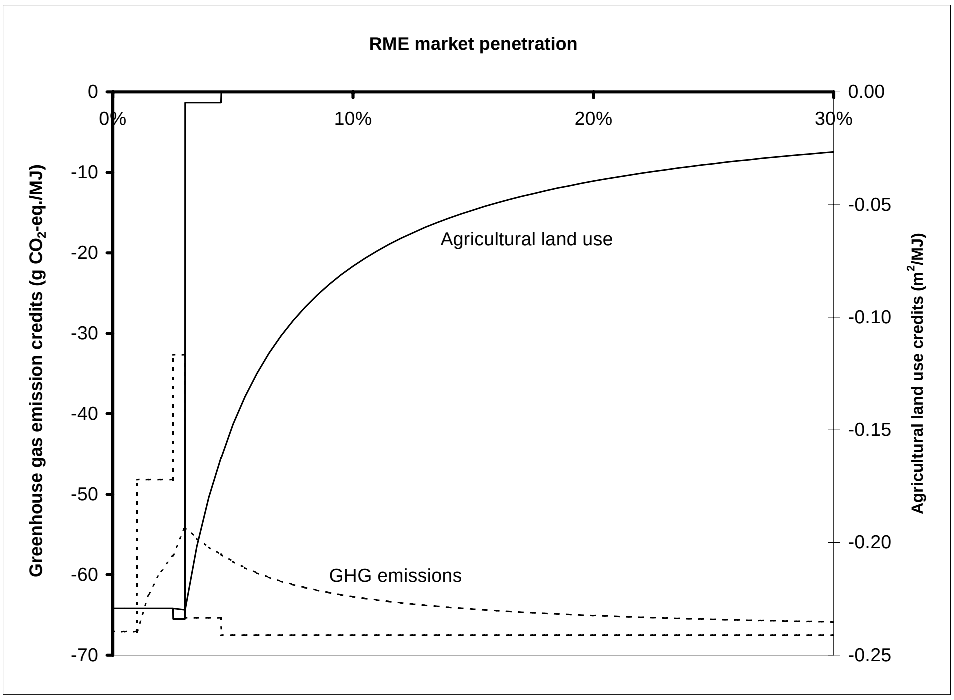

Figure I: By-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for both rapeseed meal and glycerine from RME production, case MB.

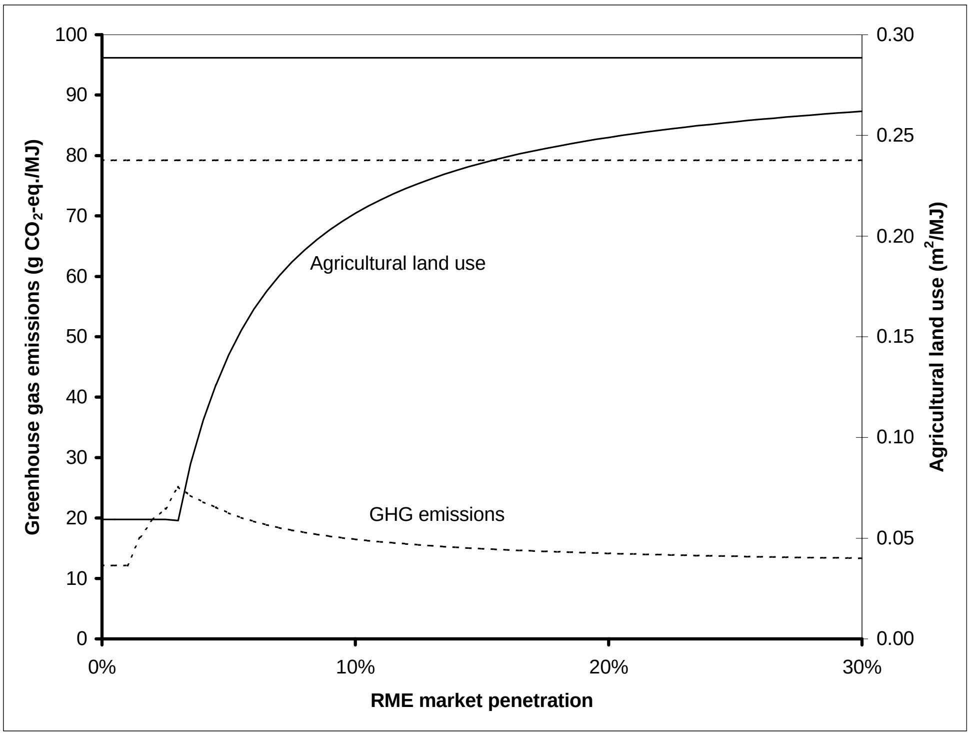

Figure J: GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for RME, case MB.

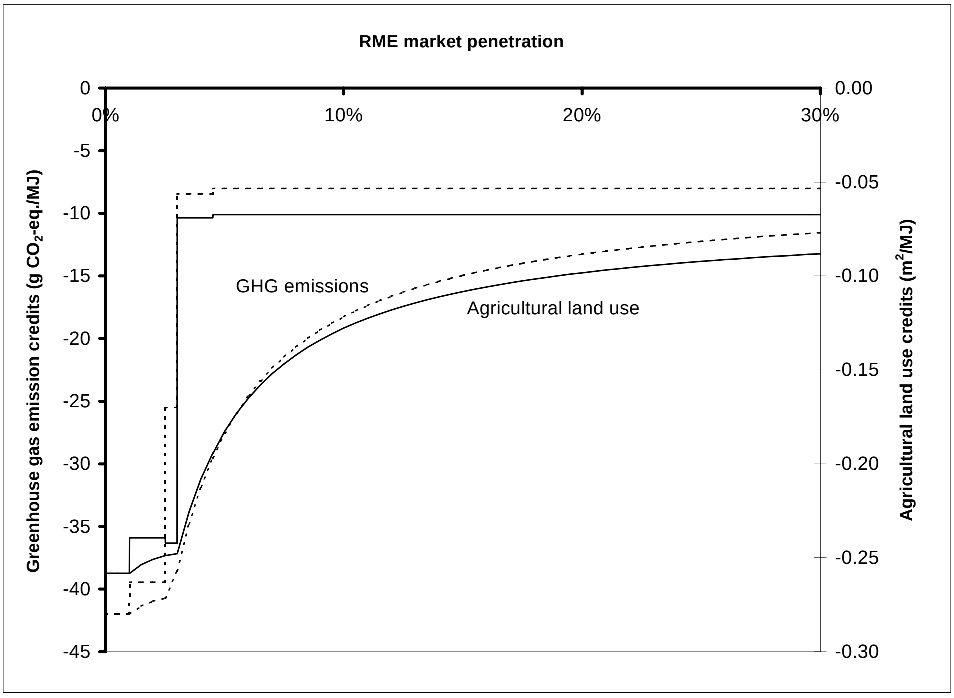

Figure K: By-product GHG emission credits (dotted line, left axis) and agricultural land use credits (black line, right axis) for both rapeseed meal and glycerine from RME production, case WB.

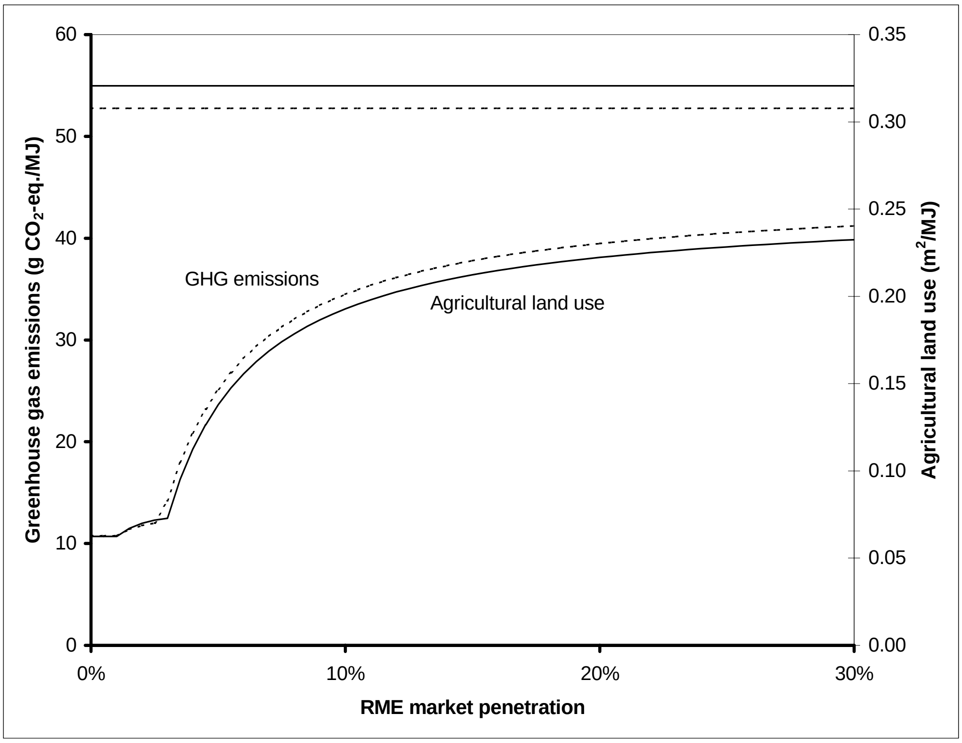

Figure L: GHG emissions (dotted line, left axis) and agricultural land use (black line, right axis) for RME, case WB.