Biofuels for transport: The challenge of properly assessing their environmental impact

Biocarburants liquides pour le transport : le défi d'une correcte évaluation du bilan environnemental

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Abstract

The objective of this paper is to discuss technically some causes of uncertainties related to the environmental assessment of liquid biofuels for transport. This discussion of causes of uncertainties is mainly based on three topics taken as examples. The three topics selected due to their importance in order to illustrate the complexity of environmental assessment of biofuels and the difficulty to reduce uncertainties are: 1) agro-environmental impact of bioethanol (from sugar cane) in Brazil and biodiesel (from palm oil) in Malaysia. These two tropical countries were selected due to their role of leader at world level and their strong export potential to the European Union, 2) N_2O (Nitrous oxide) emissions related to biofuels cultivation and land use change and 3) Life cycle assessment (LCA) of biodiesel from palm oil in Malaysia. These three topics are discussed and recommendations are formulated in order to reduce scientific uncertainty, for example through the development of internationally agreed sustainability certification systems with corresponding verification measures or further research on emissions and indirect land use change.

Keywords

Biofuels for transport. N_2O emissions. Life cycle assessment. Biodiesel from palm oil. Agro-environmental assessment.

Résumé

L'objectif de cet article est de fournir une discussion technique et scientifique de certaines causes d'incertitude liées à l'évaluation du bilan environnemental des biocarburants liquides pour le transport. Cette discussion porte essentiellement sur trois sujets choisis comme exemples. Les trois sujets choisis pour illustrer la complexité de l'évaluation du bilan environnemental des biocarburants et la difficulté à réduire les incertitudes de diagnostic sont : 1) suivi agro-environnemental du bio-éthanol (de canne à sucre) au Brésil et du bio-diesel (à partir de l'huile de palme) en Malaisie. Ces deux pays tropicaux ont été choisis du fait de leur rôle de leader au niveau mondial et leur fort potentiel d'exportation vers l'Union européenne ; 2) émissions de N_2O (protoxyde d'azote) dues aux cultures pour biocarburants et au changement éventuel d'utilisation du sol avant la mise en culture ; 3) analyse de cycle de vie (ACV) du biodiesel produit à partir de l'huile de palme en Malaisie. Après discussion de ces trois sujets, des recommandations sont formulées afin de réduire les incertitudes scientifiques, par exemple à travers la mise en place de systèmes de certification/développement durable, objets d'un accord international et associés à des mécanismes de contrôle ou à travers un développement de la recherche sur les effets indirects possibles des biocarburants.

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1. Introduction

The objective of this paper is to discuss technically some causes of uncertainties related to the environmental assessment of biofuels for transport. This discussion of causes of uncertainties is mainly based on three topics taken as examples. The three topics selected due to their importance in order to illustrate the complexity of biofuels assessment and the difficulties to reduce uncertainties are: 1) agro-environmental impact of bioethanol in Brazil and biodiesel in Malaysia (Countries with major export potential), 2) N_2O (Nitrous oxide) emissions and 3) Life cycle assessment (LCA) of biodiesel from palm oil.

In the context of this paper on bioenergy, biomass is considered to be the organic fraction of agricultural products (including vegetal and animal substances), from silviculture and related industries, as well as the organic part of industrial and municipal waste. This includes for example wood, straw, energy crops, agricultural waste, agro-industrial waste, plants and animal waste.

Agriculture contributes to various extents to provide food but also fibre, fodder and fuel (the 4Fs) and started more recently to provide feedstock for green chemistry or bio-based materials. If the production of biofuel or biogas by agriculture is often presented as a new option valid for the future, it should be mentioned that in the 19th century in Europe, about 20% of agricultural land was used to grow crops for non-food purposes and fodder. The land used for energy purposes disappeared with the development of mechanization and now again the issue of bioenergy is being discussed in some cases for its utility at farm level but most of the time in order to produce energy for the entire society.

Bioenergy is the production of energy from biomass for uses in transport, heat or electricity. In the specific field of liquid biofuels for transport (called biofuels in this document), important programs have been launched in Brazil since the end of the seventies but also more recently in the United States. In the United States mainly corn is used for ethanol production. In Europe, rapeseed, sugar beet, wheat and rye areused for 1st generation biofuels. Second generation biofuels from ligno-cellulosic material might become operational in a few years.

1.1. A few points regarding the production, use of liquid biofuels for transport in the European Union and imports

The European Union (EU) has ongoing plans based on the implementation of European, national, regional and local activities in the field of bioenergy. The targets defined at EU level in the 2003 Biofuels Directive (EC/2003/30) are 5.75% of biofuels on the transportation fuel market in 2010. In addition the European Commission called for 10% in 2020 (EC Proposal, Draft Directive 23/1/2008). This proposal has been approved but the 10% biofuels target has been transformed into 10% from renewable sources, i.e. including for example hydrogen or electric cars.

Unlike the other biofuel key players, the EU produces more biodiesel than bioethanol. 54.6% of transport fuels consumed in the EU are diesel versus 45.4% for gasoline. This proportion is not reflected in the production of biofuels: biodiesel accounts for more than 80% of EU total biofuels production. In 2007, the EU major producers of biodiesel were Germany (50.6%), France (15.3%), and Italy (6.35%) [EBB 2008]. The main feedstock for the production of biodiesel is rapeseed oil which corresponds approximately to 90% of the EU biodiesel production. In the EU, the expansion of biodiesel production has put pressure on the rapeseed market. The areas dedicated to the cultivation of rapeseed and sunflower seeds for energy use have increased from 780,000 ha in 2004 to 1,634,000 ha in 2006, corresponding to 22.5% of the total area dedicated to both crops. This expansion is taking place in areas traditionally dedicated to food crops. Currently, the EU is using about 40% of its rapeseed production and about 62% of its rapeseed oil production for the manufacturing of biodiesel. The pressure on rapeseed areas is mainly due to the relatively low productivity of this feedstock in terms of litres of biodiesel per hectare. As a consequence, between 2002-2003 and 2006-2007, rapeseed oil prices have increased by 63%.

Biofuels incorporation rates show great variations per EU Member State but the total for EU 25 was around 1% in 2005, thus behind the targets set towards the 2010 objective. Moreover there has been a biodiesel yearly growth of only 16.8% in 2007 compared to 54% in 2006 and 65% in 2005 (production increased from 4.9 Million tonnes in 2006 to 5.7 Million in 2007) [European Biodiesel Board 2008, EuRObserv' Er 2006]. According to Jank *et al.*, [2007], if the EU decides to limit the oilseed area dedicated to biodiesel feedstock to 50% of the total oilseed area, the EU will need to import 4.16 Million tonnes of vegetable oil or biodiesel.

In Brazil, sugar cane and more recently soybean are the main crops used for energy purposes. Although in 2007 the EU is the world's third largest producer of ethanol (2.1 Billion litres, it is far behind the United States (24.6 BL) and Brazil (19 BL). EU ethanol production increased only by 11% compared to 2006 (in 2006 the increase was 71 % compared to 2005, with 1.5 BL). The main producers of ethanol in the EU were (2007) France (32.6%), Germany (22.2%), Spain (19.6%) and Poland (8%) [eBio 2008]. Contrary to the situation in Brazil with the development of biofuels so far mainly based on a single crop i.e. sugar cane, ethanol in the EU is produced from a large variety of feedstock (wheat, sugar beet, barley, rye…) which account for the major part of the production, followed by sugar beet. Sugar beet is the most efficient crop for bioethanol in Europe, with production estimates around 7,250 litres of ethanol per hectare (3,125 for cereals). Presently Germany and France are the main producers of ethanol fromsugar beet and there is a potential for expansion. Since EU ethanol production is much smaller than biodiesel production and since it is based on the utilisation of various feedstocks of which the EU is a net exporter of some, ethanol has so far had no significant impact on agricultural land availability and commodity prices. On the contrary it provides a new option to sugar beet producers after the reform of the sugar Common Market Organisation adopted in February 2006 that reduced the sugar beet price by almost 40% and limited the sugar export opportunities to the World Trade Organization WTO quota. In 2007, Brazil exported more than 887 Million liters of ethanol to the EU (50% of 2007 EU production) [eBio 2008, MAPA 2008]. In the EU, the main countries of destination were Netherlands, United Kingdom, Sweden and Finland. In addition, ethanol production might also be considered in the future in some ACP countries.

1.2. Production and use of liquid biofuels for transport: a controversial issue

In order to reach European targets, there is a consensus on the need for Europe to complement at short and mid term biofuels produced from European feedstock with imports of biofuels from tropical countries. The amount of biofuels imports needed by the EU depends on the scenarios chosen and the sustainability benefits expected or taken into account by various groups. It is clear that in relation to imports from non EU countries, there is a wide variability of dominant trends between the attitudes on this issue of European countries as different as Sweden (low agricultural potential, presently importing ethanol from Brazil), Netherlands (strongly involved in international trade of biomass and biofuels, limited land availability, competitive harbour infrastructure), France, Germany or Poland (all countries with strong agricultural sectors), Cyprus and Malta (limited land availability).

Biofuels for transport are often considered as a tool to mitigate greenhouse gas emissions, reduce climate change, increase energy supply diversity and security of supply, as well as a new opportunity for agriculture and rural development. Biofuels have a domestic economic appeal because locally produced fuel creates jobs and keeps part of the energy bill within the country, the financial fluxes being totally different in the case of bioenergy or use of fossil oil. In the context of planned increase of oil prices at long term, the interest in biofuels has escalated sharply and the competitiveness of biofuels is expected to improve.

On the other hand, biofuels are often criticized at various levels, for reports on their low environmental performance, their negative consequences on tropical deforestation and the diversion of land use (“fuel against food”). Biofuels policies, especially in Europe and the United States are not a field of consensus, with very different scientific conclusions presented by different research groups as technical support for decision-making. In addition, the debate on biofuels is often made confusing by a lack of agreement or even preliminary discussion of the policy drivers and their respective weights: climate change, environment, security of supply, employment, transport, agriculture, rural development, international cooperation… The criticism towards biofuels development comes to a large extent from the scientific community [Crutzen *et al.,* 2007, Searchinger *et al.,* 2008], from some international organizations [OECD 2007] and to a large extent from part of the media [Holt-Gimenez, 2007]. Some NGOs are calling for a moratorium on biofuels. Another aggravating factor is the lack of consensus on Life cycle assessment results in relation to biofuels (see for example [Farrell *et al.,* 2006] on the issue of US corn ethanol and [Connor *et al.,* 2006]).

It should be noted that biofuels policies are not only a case of disagreement on technical results or governance but also a field of international dispute: Brazil initiated a WTO case against US ethanol and farm subsidies. US presently protects its own ethanol producers by a 0.54$ per gallon tariff [Licht 2007]. This takes place in an ethanol global market still at a very preliminary stage but characterized by strong differences in production costs according to the feedstock and the geographic origin. Jank *et al.,* [2007] reports production costs (expressed in US$ cents/litre) of 22 for sugar cane ethanol from Brazil (2005), 40 for US corn ethanol, 50-75 for EU ethanol and 44-81 EU biodiesel (2004).

2. Biofuels in Brazil and Malaysia and associated agro-environmental impact

2.1. Bioethanol (Brazil)

The Pro Alcool Programme started in Brazil in 1975, after the first oil crisis and mainly for security of supply concerns. Initially benefiting from public-support mechanisms, the activities were liberalized at the end of the 1990’s, even if there are still some differential taxation schemes at State level. Information on the use of biomass for bioenergy in Brazil can be found in Focus on Brazil [IEA 2006]. From 1983 to 1988, 90% of the 800 000 new cars sold each year on average were using ethanol. Due to the strong increase in consumption, a severe shortage of ethanol happened at the end of 1989, provoking a loss of consumer trust in the security of ethanol supply and Pro Alcool Programme. Due to these problems, by the end of the 1990s, the sales of ethanol fuelled cars amounted to less than 15% of total car sales. In 2003, car manufacturers introduced "flex fuel" vehicles and it is estimated that "flex fuel" vehicles correspond now to more than 3 quarters of new car sales in Brazil. Pure gasoline is no longer sold. The share of biofuels in road-transport fuel was estimated at 14% in 2004.

Most of the reduction in the cost of production of ethanol in recent years came from the agricultural part of ethanol production. It is estimated that around 60% to 70% of the final cost of ethanol corresponds to the cost of the sugar cane. Agricultural yield has therefore a strong impact on the final cost of ethanol. Average productivity in Brazil is around 65 t/ha but it can reach 100 to 110 t/ha in Sao Paulo State which is the main ethanol producing region. Since the beginning of Pro Alcool Programme, yields have improved of 33% in Sao Paulo due to the introduction of new varieties and the improvement of agricultural practices. There has also been a development of mechanization. In the period 2001-2006, in the mid-west, southeast and southern regions, about 35% of the area planted with sugar cane has been harvested mechanically and the mechanized harvesting rate can reach 90% in some regions.

It should also be noted that there has been an historical evolution of Pro Alcool with a progressive change of technological priorities. This is especially to be taken into account when comparing respective advantages/disadvantages of EU local production or imports. Initially the main focus of the Pro Alcool was put on the increase of equipment productivity. The size of Brazilian mills also increased. The focus then shifted to the improvement of conversion efficiencies. Over the past 15 years, special attention has been paid to a better management of the processing units. As a consequence, presently, almost all sugar-cane distilleries in Brazil use bagasse-fired steam turbine systems to provide steam and electricity to cover their site needs. For example, most biomass cogeneration takes place in Sao Paulo State with 40 sugar mills selling 1.3 GW of surplus power to the electrical grid. Bagasse-based co-generation is developed in order to reduce the country's traditional reliance on hydropower and in addition this improves the competitiveness of Brazilian ethanol.

Regarding processing conditions, on average 5 m^3 of water are used for each ton of sugar cane processed, even if values range from 0.7 m^3/t to 20 m^3/t. According to [Macedo 2005], the levels of water withdrawal and release for industrial use have substantially decreased over the past years from around 5 m^3/t sugar cane collected in 1990 and 1997 to 1.83 m^3/t sugar cane in 2004 (sampling in Sao Paulo State). In the conversion to ethanol, the reduction of water consumption was mainly due to reuses and recycling, process improvements and substitution of wet cane washing with dry cane washing; in the higher values of water use (5 m^3/t) sugar cane washing, evaporation and cooling in condensers and fermentation cooling accounted for 87% of the water use. It seems possible to decrease water collection to 1 m^3/t with no release, by optimizing both the reuse and use of waste water for irrigation [Moreira 2007].

In the past, direct discharge of vinasse (liquid residue from the distillation of ethanol, rich in potassium and organic matter) to water streams was a cause of significant environmental damage. For each litre of ethanol, 10 to 15 litres of vinasse are produced. Vinasse began to be recycled to the cane fields in 1978 when the first legislation governing the disposal of vinasse was passed. The current practice is full recycling of vinasse and industrial wastewaters. The application of vinasse is optimized for specific topographic, soil, and environmental conditions. Filter cake, another waste stream is also recycled as a fertiliser. Nutrient recycling in turn has reduced application of fertilisers. The highly intensive production systems for ethanol have been in the past a cause of environmental damage mainly due to the use of fertilisers and pesticides. Sugar cane cropping is also a source of air pollution due to burning prior to manual harvesting. The phase-out of burning is taking place in Brazil with a deadline for complete phase out in 2022.

According to UNICA (Brazilian Sugar Cane Industry Association), sugar cane culture in Brazil is considered to have relatively small soil erosion loss compared to soybean and corn for example. This situation keeps improving as harvesting without burning expands and reduced preparation techniques are introduced, thereby reducing losses to very low rates that are comparable to those for direct planting in annual crops.

According to Dufey *et al.,* [2006], the main issues of concern related to sugar production development are: natural habitat conversion and species loss, water uptake and reduced water flow, soil erosion and loss of fertility, water pollution, pollution from burning cane fields, air pollution and solid waste from processing cane.

Average ethanol production yields have grown from 3,900 litres per hectare and year (L/(ha*year)) in the early 1980s to 5,600 L/(ha*year) in the late 1990s. In the most efficient units, yields are as high as 8,000 to 10,000 L/(ha*year). Sugar cane crops are virtually non-irrigated in Brazil except for some small areas (supplementary irrigation). The annual rainfall in Sao Paulo State is roughly 1,000-2,500 mm/year.

Regarding the area requirements, in 2007, 11.6% of the cultivated area was used for sugar cane, compared to 23.8% for corn and nearly 35% for soybeans [IBGE 2008] (out of presently about 59 Mha of arable land [IBGE 2008] and about 172 Mha of pasture land [IBGE 2006]).

World sugar production in 2007/2008 (October/September) is estimated by FAO to reach 169 Million tonnes (raw sugar equivalent), 2.7% more than in the previous year, and about 12 Million tonnes higher than the projected world sugar consumption of 157 Million tonnes. Virtually all of the growth in output would stem from developing countries. Brazil is set to produce 32.2 Million tonnes of sugar in 2007/08, relatively unchanged from 2006/07. This is despite a record level cane harvest, following relatively favourable weather conditions, which boosted yields. It is estimated that around 55% of Brazil's 2007/08 sugarcane harvest will be converted into ethanol rather than into sugar [FAOSTAT 2007].

It has been estimated [Earth Policy Institute] that expanding the sugarcane area from about 6.7 Million ha in 2007 to some 8 Million ha would allow Brazil to become self sufficient in automotive fuel within a few years while conserving its sugar production and exports.

According to projections from the sugar/ethanol sector in Brazil, increasing internal and export market demands for sugar and ethanol can easily be met. It is assumed that the industry should be able to produce 33.7 Million tonnes of sugar (12.8 Million tonnes for internal consumption and 20.9 Million tonnes for export) and 26.4 Million m^3 of ethanol (of which 4.4 Million m^3 for export) by the year 2015. This would mean an increase of about 230 Million tonnes of sugarcane in ten years – a doubling in the ethanol production and an increase of 44% in sugar production [WWI 2007].

On 21 July 2007, the Brazilian Government announced a new set of measures to eliminate part of the misunderstanding related to the country's sugar cane ethanol. Part of the new legislation will be largely symbolic. Brazil will now explicitly outlaw the growing of cane in both the Amazon and the Pantanal through the creation of a zoning system for sugar cane with a restrictive map. It should be noted that even before these new measures, sugar cane was not grown anywhere in the Amazon due to agro-technical reasons. The Centre-South region of Brazil has not only good climatic and soil conditions: in addition, it has a good infrastructure, a functioning capital market and a sugar industry structure that allows cooperation between different players in the supply chain to achieve high efficiency and low costs. The agro-industry involved has reached a high level of control of biomass planting, harvesting and logistics, with species diversification thus improving adaptation to climate variability (more than 500 commercial varieties of sugar cane for different microclimates and local conditions) and finally ensuring stability in volumes and prices against variability of production conditions. The combination of all these factors exist in the South of the country, not in the Amazonian region, so the reports on direct negative impact of ethanol production on Amazonian deforestation are not based on facts. Nevertheless, the main criticism on biofuels development coming for example from environmental NGOs and part of the scientific community is not on direct impact but on indirect impact and displacement effects which are far more complex to address.

2.2. Biodiesel (Malaysia)

The Malaysia National Biofuels Policy [Malaysia Energy Centre 2005] was launched in August 2005. The Government is promoting among other the use of biodiesel in public fleets. The blend is not compulsory yet but it will be in the next phase of the implementation plan.

For oil palm, the oil extraction rate is 20% and the palm oil yield about 4 t/(ha*year). It should be noted that the best fields can produce 7-8 tonnes annually. The planting density ranges from 136-160 palms per hectare. The economic lifespan is 20-30 years. An oil palm usually bears fruits from 30 months after planting. Malaysia humid tropical climate with a temperature range of 24°C to 32°C throughout the year, an annual rainfall of about 2000 mm evenly distributed is very adapted to the cultivation of oil palm. According to Oil Worlds 2007, average oil yield is 3.74 t/(ha*year) for oil palm (mesocarp) against 0.38 for soybean, 0.48 for sunflower and 0.67 for rapeseed. The average oil yield from rape-seed in Europe is 1.3 t/ha*year.

Malaysia produced 200 Million litres of biodiesel in 2006, consuming 1% of the 15.88 Mt of palm oil produced. In 2007, the production remained almost unchanged and has already totalled 5.3 Mt in January-April 2008 (of the 17 Mt expected for this year) [Department of Statistics Malaysia 2008]. In 2006 less than 13 per cent, or 4.17 Mha (from 54,000 ha in 1960) [Basiron 2007] of Malaysia's land is planted with oil palm (the bulk of oil palm estates was previously planted with rubber, coconut and cocoa) (with 7.9 Mha of land used for agriculture [FAOSTAT 2005]).

Malaysia is the world's largest exporter of palm oil selling around 13.5 Million tonnes with a relatively low domestic consumption. Malaysia share of global oils and fats trade was 27.9% in 2006 [Oil World]. According to MPOB 2008, the EU was, after China, the second destination for Malaysian palm oil in 2007 with 2 Million tonnes (about 18% against less than 1% to the USA), almost half of the total palm oil imported in EU in 2007.

For the future, improved planting materials and better management techniques are foreseen. Domestic consumption is relatively low and Malaysia exports most of its palm oil and kernel oil.

MPOB is reporting costs of oil production (US $ per ton), of 228 for Malaysia, 400 for soybean (USA), 648 for rapeseed (Canada), 900 for rapeseed (Europe). The palm sector in Malaysia corresponds to the employment of 860,000 persons with 100,000 small holders with 650.000 ha.

The issue of sustainable resource development, specifically of palm tree cultivation, has been discussed for example in Brussels in June 2007 during the Meeting on Sustainable Resources Development organized by the Malaysian Palm Oil Council [MPOC]).

In 2006, the palm oil plantations had the following distribution in Malaysia: 2.34 Million ha (56%) in Peninsular Malaysia, 0.59 Million ha (14%) in Sarawak and 1.24 Million ha (30%) in Sabah. If there are clear advantages of oil palm in relation to other options, concern has been expressed especially by NGOs about the impact of oil palm plantations development on tropical deforestation. According to FAO 2007, based on country reporting, the total forest area in Malaysia (in thousands of ha) was 20,890, i.e. 63.6% of the land and the forest plantation area 1,573. The annual change (in thousands of ha) 1990-2000 was – 78 (– 0.4%) and – 140 (– 0.7%) for the period 2000-2005.

According to Stibig *et al.,* [2007], based on TREES (Tropical Ecosystem Environment Observations by Satellites) Project activities, *"since the mid 1990's Malaysia's and Indonesia's oil palm plantation area has grown from 2.4 to 4 Million ha and from 1.7 to 6 Million ha respectively: in Indonesia almost 3 times as much has been cleared for expansion and further huge expansion is foreseen. However, it should be stressed that not all oil palm development on Borneo or Sumatra will lead to forest conversion. In Peninsular Malaysia the conversion of state land forest to oil palm plantation is of limited extent".*

Concerning the use of peatlands in South East Asia, an assessment of CO_2 emissions from drained peatlands in SE Asia has been performed in the PEAT-CO_2 project [Hooijer *et al.,* 2006]. In this study, present and future emissions from drained peatlands were quantified using available data on peat extent and depth, present and projected land use and water management practices, decomposition rates and fire emissions. This study estimated that current likely CO_2 emissions caused by decomposition of drained peatlands amount to 632 Mt/y (between 355 and 874 Mt/y). For comparison, the agricultural sector for EU27 is estimated to emit about 430 Mt CO_2-eq [EEA, 2008]. The authors consider that these emissions will increase in coming decades unless land management practices and peatland development are changed. In addition, over 1997-2006 an estimated average of 1400 Mt/y in CO_2 emissions was caused by peatland fires also associated with drainage and degradation. The current total peatland CO_2 emission of 2000 Mt/y equals almost 8% of global emissions from fossil fuel burning. These emissions have been rapidly increasing since 1985. Over 90% of this emission originates from Indonesia. Nevertheless, there is a large variability of CO_2 emissions assessments due to the type of peat soil considered, the drainage depth and the land use [Hooijer *et al.,* 2006, Melling *et al.,* 2005a/b, Melling *et al.,* 2007, Verwer *et al.,* 2008] and the authors discuss the following uncertainties sources: Input data (Peat thickness, extent and distribution of peat lands, carbon content of SE Asian peat, carbon storage, land use/land cover, % of peatland drained, drainage depth). Other uncertainties sources mentioned by the authors were emission relations (relation between drainage depth and CO_2 emissions, CH_4 emissions, peat fires) and uncertainties in the projections (deforestation trend assessment, drainage trend assessment, land use projections).

Further, is should be mentioned that Hooijer *et al.,* [2006] estimated that 25% of plantations in Malaysia and Indonesia are on peat (present + future plantations), while MPOB reports only 6% of existing plantations in Malaysia on peatland.

From the analysis of the present situation regarding bioethanol from sugar cane in Brazil and biodiesel from palm oil in Malaysia it appears that the major causes of uncertainties are related to:

- difficulty to quantify the indirect impact of biofuels policies on land use, especially on tropical deforestation,
- difficulty to quantify the acreage of peat soils or peat swamp forests converted to agriculture for bioenergy reasons and the corresponding GHG emissions,
- difficulty to access detailed digital cartography of soils and land use.

At this stage, regarding the assessment of future indirect impact of biofuels policies at global level, the coupling of economic-trade/land use change/GHG emissions models is still a research topic associated to high uncertainties due to the various assumptions related to each modelling component.

3. N_2O emissions and related uncertainties

3.1. N_2O emissions and the uncertainty of biofuel-GHG budgets

For the GHG balance of biofuels, the emissions occurring during cultivation are an important element and they are one of the most important sources of uncertainty [see e. g., Adler *et al.,* 2007; Porder *et al.,* 2009; Scharlemann and Laurance, 2008; Smeets *et al.,* 2009]. If only the direct land use effects are considered, it is fair to say that most of this uncertainty derives from the difficulty to accurately estimate the emissions of N_2O that go ahead with all soil cultivations. The reason is that nitrogen, once it enters the system as "reactive" nitrogen (all form of N with the exception of the inert molecular nitrogen, N_2), undergoes several steps of transformations until it is eventually transformed back as N_2 [this is referred to as the "nitrogen cascade", Galloway *et al.,* 2003]. Particular importance have the processes of nitrification (converting ammonia to nitrate) and denitrification (converting nitrate back to molecular nitrogen), which both release traces of N_2O in varying quantities.

The resulting high variability of N_2O fluxes in space and in time, and the equally high variability in indirect emissions pathways is one of the largest sources of uncertainty for estimating N_2O emissions from agricultural soils. In field studies for direct N_2O fluxes, coefficients of variation up to 200% have been observed and the part of the variability in fluxes can be explained with the major soil parameters, such as soil organic carbon, pH, and soil drainage texture determining soil moisture and redox-potential [e.g., Dobbie and Smith, 2001; Granli and Bøckman, 1994; Yanai *et al.,* 2003]. Further soil compaction influencing [e.g. Sitaula *et al.,* 2000; van Groenigen *et al.,* 2005], and tillage methods [Skiba and Smith, 2000] are both influencing water- and oxygen status in the soil and thus determine whether the aerobic process of nitrification or the anaerobic process of denitrification, both potential sources for N_2O, can take place.

Also year-to-year variability is very high and is mainly driven by the weather [e.g., Baggs *et al.,* 2003]. Within a year, high N_2O emissions are frequently observed following the application of fertilizer nitrogen, but can also be related to springtime freezing/thawing events [e.g. Flessa *et al.,* 1995; Maljanen *et al.,* 2004]. These emissions are typically very large and can represent about half of the annual total emissions [Regina *et al.,* 2004]. They are mainly explained by the increased availability of organic material due to the death of microorganisms combined with anaerobic conditions. A similar effect is given for cycles of wetting and drying [e.g. Davidson, 1991; Zheng *et al.,* 2004].

The resulting variability is overlaid with effects that are active at a larger scale, such as climatic differences, management systems, variations in soils type and landscape morphology at a medium to large scale. So far, however, it was not possible to explain large-scale variations by large-scale drivers and most assessments rely on the up-scaling of small-scale estimates. The difficulty here is to assure that these are effectively representing the larger scale [see e.g., Leip, 2009].

3.2. Approaches to estimate N_2O fluxes vary from very simple to very complex

There are various options to estimate N_2O fluxes associated with the cultivation of crops. These methodologies are differing by the complexity, of the calculation method and number of variables that are taken into account – from single-input global values [Crutzen *et al.,* 2008] to data-hungry methods that are applicable at a high resolution [Leip *et al.,* 2008].

Even though there is little doubt about the high degree of variability in measured data, the most widely used method is the IPCC emission factor of 1.25% of N-input [IPCC, 2001] or the – recently updated – factor of 1.0% [IPCC, 2006]. Both factors have thus as only parameter the input of nitrogen (as fertilizer, organic nitrogen, or crop residue). Next to the factor to estimate direct N_2O emissions occurring on the field, the IPCC provides also a method to estimate the so-called indirect N_2O emissions, which occurs further down in the "Nitrogen cascade" [Galloway *et al.,* 2003]. Even though many experts are aware that the average N_2O emissions in their country might be different from the IPCC estimates, the default factors are nevertheless used in most national GHG inventories because robust data to estimate country-specific factors are not available [Leip *et al.,* 2005].

A compilation of all studies giving annual estimates of N_2O fluxes and sufficient ancillary information is provided by [Stehfest and Bouwman, 2006], improving on earlier work of [Bouwman *et al.,* 2002]. The authors develop a statistical method on the basis of these data including N application rate and type, crop type, soil and climate information and the length of the experiment in the analysis. Crop type, fertilizer type and N application rate are significant management-related factors for N_2O emissions. Applying this model globally, [Stehfest and Bouwman, 2006] find an average fertilizer-induced emission factor of 0.9% of the N-input, but obviously regional differences are high. However, using this method to assess the contribution of N_2O to the GHG balance of first-generation biofuels, [Smeets *et al.,* 2009] conclude that the statistical model remains to be among the largest contributors of uncertainty changing the overall GHG saving by potentially more than 100% points. More detailed statistical analyses become possible for smaller regions. Particularly in Europe, the density of N_2O measurements is relatively high so that the application of a method based on ecosystemic stratification might become possible [Jungkunst and Freibauer, 2005] which can be seen as a further development of regression-approach developed by [Freibauer, 2003]. Still, even in Europe, the number of measurements is scarce and process-based models are seen as the only possibility to extrapolate into "unexplored" conditions and thus give a truly complete picture of larger regions [see for example, Adler *et al.,* 2007; Leip *et al.,* 2008; Werner *et al.,* 2007].

Example one: global approach by Crutzen et al., (2008)

[Crutzen *et al.,* 2008] propose a global emission factor for N_2O emissions of 3-5% of nitrogen needed to grow (biofuel) crops. This emission factor stems from a global analysis of the increase of atmospheric N_2O concentration and the anthropogenic generation of "new" nitrogen. This approach is very attractive as it comprehensively includes both direct and indirect emissions of N_2O, without the need to "track the fate of nitrogen" as this is done in the IPCC methodology. Accordingly, the Crutzen-emission factor can be regarded as much more robust than any of the emission factors contained in the IPCC guidelines. On the other hand, there is the risk of double counting in the case that a significant part of the nitrogen taken up by the crops is not "new" (i.e. obtained through the input of synthetic fertilizer, biological nitrogen fixation or also by draining the nitrogen pool in soils) but stems from the application of manure or from atmospheric deposition [Leip, 2007]. Being robust at the global level, however, implies also that the emission factor cannot be used to estimate local or even regional N_2O fluxes. As soon as a "subsample" of global N generation is evaluated, the Crutzen emission factor becomes much more uncertain and should be corroborated (or substituted) by a more flexible approach.

Example two: detailed approach by Leip et al. (2008)

Through the combination of an economic model, a downscaling procedure of the most important anthropogenic drivers (geo-referencing of land use activities and quantification of farm input) and a mechanistic biogeochemistry model, [Leip *et al.,* 2008] established a framework that allows the evaluation of GHG fluxes from agricultural soils using a state-of-the art mechanistic model [Li, 2000]. This is embedded into a realistic setting including the most likely environmental conditions of cultivation of crops and regionally estimated farm input consistent with the economic environment (for example livestock number, feed and fertilizer import etc.). The possibility simulate a large number of spatial units allows the assessment of the spatial variability. A disadvantage of this methodology is that it has been set-up for Europe and can not easily be implemented in other parts of the world. Where the validity of the mechanistic model can be shown on the basis of experimental data (which are scarce in large areas of the world [see Stehfest and Bouwman, 2006], appropriate environmental datasets in combination with estimates of farm management might become a significant hurdle. A first application of the method to rapeseed cultivation in Europe, combining simulated emission fluxes by the biogeochemistry model with literature data (including also CO_2 fluxes occurring during farming-energy consumption), for example, that this occurs on soils characterized by relatively high N_2O flux rates offsetting a large part of the GHG savings when using the crops as feedstock for biofuels [Erisman *et al.,* 2009]. Using sugar beet leads to a better overall GHG balance due to the lower N-input needed.

3.3 Discussion

- The evaluation of the best method is not only a scientific problem, but must be seen in the framework of the policy framework.
- For example, if thresholds for minimum GHG savings are set, then the decision to use a global approach excludes N_2O emissions from distinctions of biofuel feedstock with respect to their origin, however the selection of the emissions factor could well influence the "ranking" of biofuel crops.
- In the case that GHG certificates are issued for crops cultivated for the production of biofuels only and not for those that enter the feed, food or fibre industry, the use of a detailed methodology might lead of a shift between land used for biofuel production or other uses, without any real impact on total GHG fluxes.
- Consequently, detailed methodologies can pay only off if thresholds for GHG emissions from the field are applied to the whole production of a crop in a country. This could be the average CO_2-eq emissions over the country, or to assure environmental integrity, the demand that a minimum share of the production be sustainable with regard to the threshold. Only in that case, the application of detailed models, which take into considerations the local environmental conditions (soil, climate etc.) in combination with a realistic estimate of the spatial distribution of the cultivated areas would be important. In case that such a model could be properly be set-up, which requires high quality environmental datasets and realistic estimates for farm-input, the use of aggregated emission indicators would also lead to a minimum uncertainty in the estimate of the GHG balance of the biofuel.
- Even if under such assumption it could be possible to reduce the uncertainty associated with N_2O emissions from field on which crops for the production of biofuels are grown, it is important to note that both emissions and uncertainty of emissions from *indirect land use effects* are at least of the same order of magnitude but have the potential to be much more important than direct land emissions [Fargione *et al.,* 2008; Searchinger *et al.,* 2008]. Today, it is of utmost importance to improve our knowledge of these indirect land use emissions and our capabilities to accurately predict the GHG impact of biofuel (targets) comprehensively [Porder *et al.,* 2009].

4. Life cycle assessment of biofuels

Biofuels for transport are generally considered to be environmentally friendly since they save non-renewable energy resources, are biodegradable and – at least at first glance – CO_2 neutral. The latter is of course only true for the direct combustion of biofuels which releases the same amount of CO_2 into the atmosphere that earlier has been taken up by the plants. However, when looking at the entire life cycle of biofuels – from biomass cultivation (including the input of fertilizers, pesticides etc.) through conversion into biofuels and their energetic use – substantial amounts of (non-renewable) energy resources are used which in turn cause greenhouse gas (GHG) emissions. Thus, biofuels are not CO_2 neutral from a life cycle point of view. The same holds true for other potential environmental impacts: the use of biofuels is not implicitly environmentally friendly simply because the feedstock – biomass – is a renewable resource.

In the 1990s, a method has been developed which addresses the environmental aspects and potential environmental impacts (e.g. use of resources and the environmental consequences of releases) throughout a product's life cycle: life cycle assessment (LCA). The method is internationally standardized (ISO standards 14040 and 14044) and considers the input and output flows (raw and other materials, energy and wastes, wastewater, emissions, etc.) and potential environmental impacts (e.g. greenhouse effect, acidification etc.) of the considered product system (product or service) along its entire life cycle ("cradle-to-grave", from raw material acquisition through production and final disposal).

LCAs usually address a number of environmental impact categories, such as use of resources, greenhouse effect, acidification, eutrophication, stratospheric ozone depletion, summer smog (photo-oxidant formation), human toxicity and ecotoxicity. In recent years, however, the scope of many studies was restricted to two of them: the use of non-renewable energy resources and greenhouse effect. In this case, the LCA methodology is used to obtain energy and greenhouse gas balances, which are not to be confused with a full LCA.

Despite all standardization, the results of LCAs or energy and GHG balances may vary quite substantially. This can be due to a) differences in accounting for co-products (substitution versus allocation), b) differences in system boundaries (e.g. exclusion of land use changes) or c) differences in basic data (e.g. N_2O emission factors). In the following chapters, the results of LCAs as well as energy and GHG balances of biofuels will be presented. Using the example of palm oil biodiesel, the main reasons for variations in the results will be identified.

4.1. Greenhouse gas balances of biofuels

One of the most commonly cited collections of energy and greenhouse gas (GHG) balances of biofuels is the JEC Well-to-Wheels study [JEC 2007]. It gives ranges of GHG emissions in grams CO_2 equivalents per kilometre (g CO_{2eq}/km), e.g. for bioethanol used as neat fuel: 58 to 130 for bioethanol from sugar beet, 32 to 209 for wheat, 19 to 22 for wheat straw and 21 to 25 for sugar cane. As for biodiesel, the values are 83 to 100 for rapeseed, 46 to 60 for sunflower and 14 to 17 for farmed wood. These ranges are due to the different pathways studied in the report (for example concerning the use of by-products). As an order of magnitude the following GHG savings could be found in the same report [JEC 2007] in terms of kg CO_2eq/(ha*year): 660 for wheat and 4,429 for sugar beet for ethanol production; 1,505 for rapeseed, 1,545 for sunflower and 4,806 for farmed wood for biodiesel production.

IFEU [2004] also gives ranges of GHG savings in terms of kg CO_2eq/(ha*a): 800 to 3,700 for bioethanol from wheat, 3,000 to 11,000 for bioethanol from sugar beet, 500 to 2,800 for biodiesel from rapeseed, and 1,500 to 4,000 for biodiesel from sunflower.

According to Jank *et al.,* [2007], the greenhouse gas balance of bioethanol in tonnes of CO_2 equivalents per tonne of oil equivalent (t CO_2eq/toe) is 2.17 for sugar beet, 1.85 for wheat, 0.41 for sugar cane and 0.33 for straw. In the case of biodiesel production, it is 2.6 for soy, 1.73 for palm and 0.27 for wood.

According to the Worldwatch Institute [2007], *"the vast majority of studies have found that, even when all fossil fuels throughout the life cycle are accounted for, producing and using biofuels made from current feedstocks result in substantial reductions in GHG emissions relative to petrol fuels".* Several studies have assessed the net emissions reductions resulting from sugar cane ethanol in Brazil, and all have concluded that the benefits far exceed those from grain-based ethanol produced in Europe and the US. The lower life cycle climate impacts of Brazilian sugar cane ethanol are related to two main factors: high cane yields and use of bagasse for energy or co-generation.

All the above mentioned studies, however, do not take into account GHG emissions from land cover and/or land use changes. One of the first studies to cover this issue was published by WWF [2007] showing that GHG balances could even turn out negative, i.e. that the use of biofuels could cause higher GHG emissions than the use of conventional fuels.

Another way to assess the benefits of biofuels is to quantify how many years it takes for the biofuel carbon savings from avoided fossil fuel combustion to offset the losses in ecosystem carbon from clearing land to grow new feedstocks (or Ecosystem "Carbon Payback Time", ECPT). Fargione *et al.,* [2008] estimated that converting rainforests, peatlands, savannas or grasslands to produce food-crop based biofuels in Brazil, Southeast Asia and the United States creates a "biofuel carbon debt" by releasing 17 to 420 times more CO_2 than the annual greenhouse gas reductions that these biofuels would provide by displacing fossil fuels. On the other hand, it was considered that biofuels made from waste biomass or from biomass grown on degraded or abandoned agricultural lands planted with perennials are associated to a small or no carbon debt and can provide immediate GHG advantages.

Similarly, the indirect or "leakage" land use impacts of US corn ethanol have been estimated by Searchinger *et al.,* [2008] to double the greenhouse gas emissions per fuel mile compared to conventional gasoline over 30 years. Gibbs *et al.,* [2008] also concluded that clearing tropical forests and grasslands to produce biofuels leads to long-term carbon debt while only converting degraded lands will provide carbon savings (even if the highest yielding biofuel crops from clearing forests are taken into account). It should be noted that growing crops on these marginal lands may require significantly more land area than other regions due to relatively lower yields, and will likely require more energy-intensive management such as fertilizer application or irrigation to remain productive that should change the obtained ECPT value.

The content of the Fargione *et al.,* and Searchinger *et al.,* papers has been questioned by a letter to Science of Kline and Dale [2008] suggesting that an improved understanding of the forces behind land-use change leads to more favourable conclusions regarding the potential for biofuels to reduce greenhouse gas emissions.

4.2. Greenhouse gas balance of palm oil biodiesel

Quite a number of greenhouse gas balances for palm oil and downstream products such as palm oil biodiesel (palm oil methyl ester, PME) can be found in literature, e.g. Germer & Sauerborn [2008], Reijnders & Huijbregts [2008], Reinhardt *et al.,* [2007], Schmidt [2007], Wicke *et al.,* [2007], Wicke *et al.,* [2008], WWF [2007] and Yusoff & Hansen [2007]. The results of these greenhouse gas (GHG) balances vary quite substantially, mainly depending on whether and how direct land use changes are considered and to a lesser degree depending on differences in basic data.

4.2.1. GHG emissions related to direct land use changes

Direct land use changes (LUC), i.e. the conversion of natural ecosystems (e.g. forest land) into agricultural land (e.g. an oil palm plantation), induce changes in site quality, e.g. in terms of biodiversity and carbon stocks. So far, it is not possible to quantify the impacts of LUC on biological diversity by means of life cycle assessment. Changes in above-ground and below-ground carbon stock, however, can lead to very significant GHG emissions, which have to be included in the GHG balance. These emissions can either result from singular processes (e.g. clear-cutting) – which require an annualization – or from continuous processes (e.g. peat subsidence) that prevail for many years after land conversion. If fire is used to clear the site (slash-and-burn), emissions of methane and nitrous oxide have to be considered in the GHG balance. A detailed analysis by Reinhardt *et al.,* [2007] has shown that the two most important influencing factors are:

- Magnitude of carbon stock change: Depending on the previous land use, the amount of carbon stored in both the above-ground and below-ground vegetation as well as in the soil differs considerably. Most authors that include LUC use basic data from IPCC [2006] for carbon stocks of vegetation and mineral soils but not for organic soils due to poor documentation of the latter. GHG emissions from vegetation fires are only included in Germer & Sauerborn [2008] and Rettenmaier *et al.,* [2008], the latter also covering peat fires.
- Annualization: GHG emissions resulting from singular events such as clear-cutting of natural forests have to be evenly distributed over a certain period of time (i.e. annualized). As the length of this period is not specified by LCA standards, it is up to the user to define an adequate time span. Many opt for 100 years, others for 25 years which equals one plantation cycle (economic life span of oil palms) whereas IPCC stipulates an annualization over 20 years. This dispute cannot be solved scientifically, so there is a strong need for a political consensus which time span should be used as a general rule.

The qualitative results of GHG balances, i.e. whether the life cycle GHG emissions are higher or lower than those of conventional fuel, are heavily dependent on these two factors. For example, if natural forest on mineral soil is cleared and the resulting GHG emissions are annualized over 100 years, the result of GHG balance is positive. However, if annualization over 25 years is chosen, the result is negative. In other cases, the results are more uniform: clear-cutting of peat (swamp) forest always leads to negative GHG balances, irrespective of annualization, whereas establishment of oil palm plantations on degraded land always induces positive GHG balances.

Regarding basic data for continuous processes such as CO_2 emissions due to peat subsidence and N_2O volatilization due to fertilization of organic soils, IPCC [2006] unfortunately doesn't give clear guidance. For example, if drained peat soils are classified as 'drained organic soils in managed forests', CO_2 emissions are as low as 1.36 t C/(ha*a). However, if they are classified as 'cultivated organic soils' the figure is considerably higher: 20 t C/(ha*a). In the above mentioned GHG studies, values from 8.6 t C/(ha*a) [Germer & Sauerborn 2008] up to 25 t C/(ha*a) [IFEU 2007] are used, the latter based on an equation by Hooijer *et al.,* [2007] and a drainage depth of 1 metre. Melling *et al.,* [2005a] criticize Hooijer's figures but derive their own ones from disturbed ecosystems [Verwer *et al.,* 2008].

Similar problems arise regarding N_2O emissions from organic soils. In addition to the N_2O emission factor (1% of N input), which was already discussed in chapter 3, IPCC [2006] gives a second emission factor (EF_2) for N application on organic soils. This fixed sum has to be added to the input-dependent term. Again, it is the question whether drained peat soils are classified as "tropical organic forest soils" or as "tropical organic crop and grassland soils", leading to emissions of 8 kg N_2O-N/(ha*a) and 16 kg N_2O-N/(ha*a), respectively. In the above mentioned GHG studies, values from 4.1 kg N_2O-N/(ha*a) [Germer & Sauerborn 2008] to 16 kg N_2O-N/(ha*a) [IFEU 2007, Schmidt 2007] are used.

Fortunately, the choice of emission factors doesn't influence the qualitative results, but of course the quantitative ones. In order to obtain more accurate results for the GHG balances, further research is needed, especially regarding GHG emissions from tropical organic soils.

Of course, the establishment of an oil palm plantation does not necessarily lead to a direct land use change, i.e. the conversion of natural ecosystems into agricultural land. In the 1990s, oil palm plantations were often replacing other plantations like rubber, coconut or cocoa. Assuming a constant demand for the products earlier produced on these plantations, their production was displaced to other areas. The existing agricultural production in those areas might have been displaced to a third area and so on. As agricultural land is a finite entity, new land must be reclaimed elsewhere, again leading to direct land use changes. These so-called indirect effects are not yet incorporated into LCAs as the underlying mechanisms are not fully understood yet. Several suggestions like Fritsche's "risk adder" [2007] are currently discussed in science and politics, but more efforts are required to address this issue.

4.2.2. GHG emissions related to palm oil production

Next to land use change, cultivation and conversion are two critical stages along the life cycle of palm oil biodiesel, which can be optimised considerably [Helms *et al.,* 2006, Reinhardt *et al.,* 2008]. In order to cover the variability of palm oil production, IFEU [2007] developed two scenarios: "typical practice" and "good practice". The following parameters differ from each other (for details see [Reinhardt *et al.,* 2007, Rettenmaier *et al.,* 2007]):

- Cultivation: By applying good agricultural practices, the yield of an oil palm plantation can be increased from typically 3.5 tonnes palm oil/(ha*a) to 4.0 tonnes palm oil/(ha*a). "Good practice" includes improved planting material, tailored fertilisation and just-in-time harvesting.

- Conversion: Great optimisation potentials emerge from the energetic utilisation of the entire amount of fibres and shells (50 % are required for the internal power supply of the oil mill) and biogas, which is produced during the anaerobic digestion of the "palm oil mill effluent" (POME). The surplus power could be fed into the public grid.

The increase in yield, the energetic utilisation of co-products as well as the retention and utilisation of the biogas from POME treatment improves the greenhouse gas balance of palm oil biodiesel: the disadvantage (i.e. net GHG emission) accounts for "as little as" 5.9 tonnes of CO_2 equivalents/(ha*a) instead of 9.7 tonnes of CO_2 equivalents/(ha*a) without optimisation.

A comparison of input data for palm oil production [Rettenmaier *et al.* 2008] showed much less variability as compared to the input data for land use changes. All of them point at a significant potential to optimize both oil palm cultivation and palm oil extraction.

4.3. Other environmental impacts of palm oil biodiesel

As mentioned before, LCAs address more environmental impact categories than just the use of non-renewable energy resources and greenhouse effect. However, if acidification, eutrophication, stratospheric ozone depletion, summer smog (photo-oxidant formation), human toxicity and ecotoxicity are to be assessed, much more input data regarding emissions of air pollutants are required. The following example shows, how they can be obtained.

Example: Emissions of air pollutants related to the use of biodiesel

Generally, neat vegetable oils show properties related to melting point, viscosity and flash point, which hamper their use in regular diesel engines. Therefore, neat vegetable oils are usually converted into fuels with properties similar to those of conventional diesel fuel, either by transesterification into fatty acid methyl esters (FAME) – also called biodiesel – or by hydrotreatment into synthetic biofuels. In this way, both biodiesel and hydrotreated vegetable oil (HVO) can be used in regular diesel engines and replace conventional diesel fuel.

Diesel engine emissions contain mutagenic and carcinogenic polycyclic aromatic hydrocarbons (PAH). Their formation depends on the type of engine, the engine load, the fuel properties, and the effectiveness of the exhaust gas aftertreatment [Krahl *et al.,* 2007]. A comparison of the mutagenic effects of diesel engine emissions from rapeseed biodiesel (rapeseed oil methyl ester, RME) and a reference diesel fuel revealed similarly low emissions of mutagenic compounds for both fuels. Additionally, the regulated emissions of total hydrocarbons (HC), carbon monoxide (CO), nitrogen oxides (NO_x), and particulate matter (PM) were determined. Regarding the regulated emissions, the results for RME remained below the limiting values except for NO_x which showed an increase of up to 15% [Krahl *et al.,* 2007]. Unfortunately, similar analyses for palm oil biodiesel (palm oil methyl ester, PME) are currently still under progress [Krahl 2009].

In a follow-up study, the Krahl and his colleagues showed a non-linear behaviour of the results for mutagenicity, i.e. the non-regulated emissions. Diesel fuel (B0), RME (B100) and various blends thereof (B5, B10, B20, B30, B40, and B50) were investigated. An increase of mutagenicity with increasing content of RME was observed for B10 and B20 followed by a decrease for B30 and B40 [Krahl *et al.,* 2008]. The authors hypothesize that this is due to oligomerisation as biodiesel oligomers may have a higher boiling point than biodiesel or may even boil under decomposition like neat vegetable oil.

The results for the regulated emissions mentioned above are used as input data for the environmental impact categories acidification, eutrophication, summer smog (photo-oxidant formation), human toxicity and ecotoxicity.

4.4. Conclusions

Life cycle assessment is a very suitable tool to assess the environmental impacts of biofuels. It could be shown that the varying results are due to differences in both system boundaries and basic data, of which the former are more important. As far as greenhouse gas balances are concerned, the largest influencing factor is GHG emissions from land use changes. If LUC – as it is common scientific consensus – are included in the balance, the qualitative results (positive or negative) are rather similar. The quantitative results, however, are differing due to varying basic data. Here, more efforts are needed to harmonize the underlying basic data.

5. Biofuels environmental certification

In Europe, the main ongoing initiatives related to biofuels certification are performed at national level, especially in Netherlands [Cramer 2007], United Kingdom and Germany [Fehrenbach *et al.,* 2008]. Other EU Member States are also starting the preparation of certification schemes within the framework of the preparation of the National Biomass Plans. A detailed synthesis overview of recent developments in sustainable biomass certification has been prepared by Van Dam J. *et al.,* [2007] and Scarlat [2008]. A new recent international initiative is the Round Table on Sustainable Biofuels coordinated by the *« École Polytechnique Fédérale de Lausanne »,* Switzerland (see [RSB]).

On the specific issue of tropical feedstock used for biofuels production, the Roundtable on Sustainable

Palm Oil [RSPO], the Better Sugar Initiative [BSI] and the Round Table on Responsible Soy [RTRS] are important since they are the basis for the definition of sustainability schemes based on the combination of NGOs activities (mainly WWF) with those of industrial partners, exporters and other stakeholders.

At this stage, the RSPO proposed certification system does not include a GHG emissions saving criterium but its participants consider this is indirectly taken into account through other criteria. A critical assessment of sustainability schemes for biofuels has been published by Friends of the Earth Europe [April 2008]. This document questions the expected effectiveness of applying sustainability criteria to agrofuel and animal feedstock production in the Mercosur Region.

Regarding the biofuels certification schemes in preparation or discussion in the European Union and its links to tropical countries exports, the authors of this paper wish to stress:

- Usefulness of a certification system not only for biofuels (or biomass or bioenergy) but for agriculture independently of the final use. This would avoid the risk of a double standard policy between fuel and food ("green" sugar in car tanks and non certified sugar for food or green palm oil in car tanks and non certified palm oil for cooking?).
- Need of a sustainability certification system covering not only environmental issues but also social ones and thus ensuring the active participation of farmers.
- Need to clarify the relationship and compatibility between national and international systems.

Other open questions are for example:

- Should only certified biofuels or also non-certified biofuels have access to markets? Should there be a performance rewarding system (green biofuels for example more supported than lesser green biofuels) and through which mechanisms?
- How will the sustainability certification schemes treated during WTO negotiations and how can we ensure that these certification schemes will not become an instrument preventing international trade or the growth of a biofuels sector?

6. Conclusion and recommendations

In order to reduce uncertainties related to the environmental assessment of biofuels and provide options to lower the controversy, especially in Europe and the United States, about the advantages/disadvantages of biofuels and bioenergy, the following points can be stated:

- The success of the Brazilian experience with ethanol from sugar cane is based on the achievements of a programme started more than 30 years ago, initially with public support, then progressively liberalized. If the complex issue of indirect land use change is not quantified, the environmental record of the Brazilian Programme has been improving. In our view, the economic or environmental comparisons between oil derived fuels on one hand, European, US and tropical biofuels on the other hand are only valid if they take into account externalities, financial fluxes and the difference in maturity between several technology options.
- Biofuels certification is an opportunity both for exporters from tropical countries and for importers, for example from the European Union. Extreme care must be taken in order to make sure that biofuels certification will provide a fair treatment both to European and tropical biofuels feedstock productions, and will be acceptable for WTO standards. Biofuels certification should allow the development of international trade, not make it difficult or impossible by being based on criteria so detailed their verification will be too complex or costly to check. The implementation of sustainability certification systems and the corresponding verification mechanisms, for example through remote sensing, will allow to reduce some uncertainties.
- More research is needed on GHG emissions quantification in relation to biofuels, especially considering N_2O emissions, the contribution of peat soils to emissions in case of land use change, indirect effects on tropical deforestation, the price interactions between food and biofuel prices. Life cycle assessment of biofuels is a useful tool of analysis only if it is transparent but the results are associated to a high level of uncertainty, often due to different methodological choices. The indirect effect (displacement, leakage…) of EU and US policies on land use/land cover in tropical countries is a complex issue which requires more research using among others global macro-economic and land use/land cover models.
- Crops must be grown in a sustainable way whatever their final use. Do we need green sugar in car tanks and any sugar in coffee cups? All crops have advantages/disadvantages and it is our responsibility that the biofuels development based on tropical feedstock takes into account How (i.e. farming practices) and not What (i.e. this crop is "good" and this one is "bad").
- The final decision for a country or group of countries to implement biofuel policies should be based on the combination of policies such as: transport, environment, energy, climate change, agriculture, rural development, employment, security of supply, development and aid… part of the confusion presently observed in the biofuels debate is in our view linked to scientific uncertainty mixed with policy drivers confusion and commercial interests.

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