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Review article

Biofuels, greenhouse gases and climate change. A review

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Abstract – Biofuels are fuels produced from biomass, mostly in liquid form, within a time frame sufficiently short to consider that their feedstock (biomass) can be renewed, contrarily to fossil fuels. This paper reviews the current and future biofuel technologies, and their development impacts (including on the climate) within given policy and economic frameworks. Current technologies make it possible to provide first generation biodiesel, ethanol or biogas to the transport sector to be blended with fossil fuels. Still under-development 2nd generation biofuels from lignocellulose should be available on the market by 2020. Research is active on the improvement of their conversion efficiency. A ten-fold increase compared with current cost-effective capacities would make them highly competitive. Within bioenergy policies, emphasis has been put on biofuels for transportation as this sector is fast-growing and represents a major source of anthropogenic greenhouse gas emissions. Compared with fossil fuels, biofuel combustion can emit less greenhouse gases throughout their life cycle, considering that part of the emitted CO_2 returns to the atmosphere where it was fixed from by photosynthesis in the first place. Life cycle assessment (LCA) is commonly used to assess the potential environmental impacts of biofuel chains, notably the impact on global warming. This tool, whose holistic nature is fundamental to avoid pollution trade-offs, is a standardised methodology that should make comparisons between biofuel and fossil fuel chains objective and thorough. However, it is a complex and time-consuming process, which requires lots of data, and whose methodology is still lacking harmonisation. Hence the life-cycle performances of biofuel chains vary widely in the literature. Furthermore, LCA is a site- and time-independent tool that cannot take into account the spatial and temporal dimensions of emissions, and can hardly serve as a decision-making tool either at local or regional levels. Focusing on greenhouse gases, emission factors used in LCAs give a rough estimate of the potential average emissions on a national level. However, they do not take into account the types of crop, soil or management practices, for instance. Modelling the impact of local factors on the determinism of greenhouse gas emissions can provide better estimates for LCA on the local level, which would be the relevant scale and degree of reliability for decision-making purposes. Nevertheless, a deeper understanding of the processes involved, most notably N_2O emissions, is still needed to definitely improve the accuracy of LCA. Perennial crops are a promising option for biofuels, due to their rapid and efficient use of nitrogen, and their limited farming operations. However, the main overall limiting factor to biofuel development will ultimately be land availability. Given the available land areas, population growth rate and consumption behaviours, it would be possible to reach by 2030 a global 10% biofuel share in the transport sector, contributing to lower global greenhouse gas emissions by up to 1 $GtCO_{2\,eq}.year^{-1}$ (IEA, 2006), provided that harmonised policies ensure that sustainability criteria for the production systems are respected worldwide. Furthermore, policies should also be more integrative across sectors, so that changes in energy efficiency, the automotive sector and global consumption patterns converge towards drastic reduction of the pressure on resources. Indeed, neither biofuels nor other energy source or carriers are likely to mitigate the impacts of anthropogenic pressure on resources in a range that would compensate for this pressure growth. Hence, the first step is to reduce this pressure by starting from the variable that drives it up, i.e. anthropic consumptions.

biofuels / energy crops / perennials / LCA / greenhouse gases / climate change / political and economic frameworks / bioenergy potential / land-use change / nitrous oxide / carbon dioxide / agricultural practices

Contents

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1. INTRODUCTION

Until the middle of the 19th century, American citizens lit their houses with whale-oil lamps. In 1892, the first Rudolf Diesel motor ran on peanut oil. Liquid fuels can be easily stored and transported and offer, for a given volume, a better exchange surface for combustion compared with solid fuels. Oils, in particular, can deliver a high energy amount by volume unit. No wonder then that biofuels were the first candidates to supply the newly developing automotive industry. However, they were almost immediately overtaken by petroleum products that appeared to be an energy godsend, remaining very cheap for more than a century. However, today the Black Gold Age is coming to an end.

In 2005, the world total primary energy supply approximated 11 430 Mtoe yr^{-1} (479 EJ yr^{-1} [1]), compared with 6130 Mtoe yr^{-1} (257 EJ yr^{-1}) in 1973 (IEA, 2007a). According to the FAO, the world population will grow from around 6.5 billion people today to 8.3 in 2030 (UN, 2006). World energy demand is expected to rise by some 60% by 2030. More than two-thirds of the growth in world energy use will come from the developing countries, where economic and population growths are highest (CEC, 2006a). Fossil fuels will continue to dominate energy supplies, meeting more than 80% of the projected increase in primary energy demand. Global oil reserves today exceed the cumulative projected production between now and 2030, but reserves will need to be "proved up" in order to avoid a peak in production before the end of the projection period. Effective exploitation capacity today is almost fully used, so growing demand for refined products can only be met with additional capacity (IEA, 2005). The exact cost of finding and exploiting new resources over the coming decades is uncertain, but will certainly be substantial. Financing the required investments in non-OECD countries is one of the biggest challenges posed by our energy-supply projections (IEA, 2005). As an example, Saudi Arabia, with 25% of the world's best proven reserves, is already investing US$50 billion to increase its production capacity by 2 million barrels per day (Mb/d); the global worldwide current production averaging 86 Mb/d (ASPO, 2008).

[1] Mtoe yr^{-1}: million ton oil equivalent: IEA conversion factor used throughout the article: 1 Mtoe yr^{-1} = 4.1868 × 10^4 TJ. Selected units for the article are Joules; however, conversions are indicated in brackets when quoted figures are given in other units.

According to some experts the peak oil will occur in 20 years, whereas others argue that the world is already at peak production. Although one might argue on the exact moment, it is generally accepted now that it will happen soon and that an energy transition is unavoidable (Van der Drift and Boerrigter, 2006). ASPO, The Association for the Study of Peak Oil, states that overall oil and gas production will be at their peak by 2010, and the conventional oil peak would already be overcome in most regions (ASPO, 2008). According to Andris Piebalgs, European Union Energy Commissioner, the oil crisis of the 1970s presented a discrepancy between oil supply and demand of only 5%, but in a post-peak oil scenario, the gap between supply capacity and demand could reach 20% within five years (as quoted by ASPO, 2008). In 2030, the European Union energy dependency on imports could account for 70% of its global energy needs. Today, this dependency is already around 50% and the energy demand has kept increasing by 1 to 2% each year since 1986 (EU DG-TREN, 2005). Cheap reserves will not be sufficient to fulfil the world's growing energy demand and fossil fuels have been shown to be the main anthropogenic cause of global warming. These increasing supply and environmental costs make petroleum no longer the only candidate as the universal energy source; other sources may now be competitive. However, there is no alternative energy godsend and, as industries have been relying on petroleum for too long, clean technologies are late. There is no other solution than diversifying the energy mix with a growing contribution of cleaner energy sources.

Current global energy supplies are dominated by fossil fuels (81% in 2005), with much smaller contributions from nuclear power (6.3%) and hydropower (2.2%). Bioenergy provides about 10% of the total energy supplies, making it by far the most important renewable energy source used; solar, wind and other renewable energy sources accounting for the last 0.5% (IEA, 2007a). On average, in the industrialised countries biomass contributes less than 10% to the total energy supplies, but in developing countries the proportion is as high as 20–30%. In a number of countries biomass supplies 50−90% of the total energy demand. A considerable part of this biomass use is, however, non-commercial, and relates to cooking and space heating, generally by the poorer part of the world's population (IEA Bioenergy, 2007). The contribution of bioenergy to the global supply mix has scarcely evolved since 1973, whereas other renewable energy sources have been consequently fostered and nuclear power widely developed (IEA, 2007a). Bioenergy could play a bigger role, especially in the industrial countries, which consume a lot of fossil energy and are therefore the main contributors to atmospheric pollution and global warming. According to the intergovernmental panel on climate change (IPCC), greenhouse gas emissions have already made the world 0.6 °C warmer during the last three decades. The EU-25 and the four other largest emitters, the United States, China, Russia and India, contribute all together approximately 61% of global emissions. Energy-related emissions represent 60% of global emissions in CO_2 equivalent (Baumert et al., 2005).

Transport is a major energy consumer [27.6% of total final energy consumption worldwide (IEA, 2007a), 31% in the EU-27 (EU DG-TREN, 2007); two-thirds of the projected increase in oil demand will come from transport (IEA World outlook 2005)] and a large greenhouse gas emitter. In 2004, the transport sector produced 6.3 $GtCO_2$, i.e. 23% of world energy-related CO_2 emissions (Ribeiro et al., 2007) or roughly 13.5% of global greenhouse gas emissions (Baumert et al., 2005). In the EU-27, this sector accounted for 22% of total greenhouse gas emissions in 2005 (EEA, 2008). Moreover, vehicle emissions are the single most rapidly growing source of CO_2 emissions. Achievement of a levelling off of vehicle emissions, given continuing growth in the number of vehicles on the road, requires both: (1) a substantial reduction in vehicle emissions during the next several years and (2) advances in technology in the longer term that fundamentally reduce CO_2 emissions, because energy will always be at a premium (Hansen, 2006). The automotive market is logically evolving towards electric motors, whose energy efficiency is roughly 7.5 times higher than that of internal combustion engines. The compacity and lightness of liquid fuels still enable fifty-fold higher energy storage than the best current batteries (Roby, 2006). Fuel cells may in the future replace these limited electro-chemical accumulators, but these are considered long-term technologies requiring significant research and development efforts. Their deployment also hinges on changes in the market and consumption behaviours. Lastly, electricity or H_2 are secondary energy carriers that need to be produced from primary energy sources, involving possibly high CO_2 emissions.

Biofuels can contribute to reducing the dependency on fossil fuels and lower greenhouse gas emissions from transport, provided that the savings of greenhouse gases through the use of bioenergy is not counteracted by an increase in the same emissions during the production and transformation of the biomass. Agriculture and land-use change already account for some 15% and 13% of global greenhouse gas emissions, respectively (Baumert et al., 2005; Smith et al., 2007; Houghton, 2008). Can biofuels finally be considered as an advantageous clean energy source? Here, we address this question by first reviewing the various biofuels, the state of the art of the technologies, and the current production and consumption rates. We then present the political and economic frameworks that aim at promoting the development of biofuels but still fail at convincing all stakeholders about biofuel sustainability. We finally address the issue of biofuel quality in terms of environmental impacts, with a special focus on greenhouse gas emissions and the potential of biofuels to contribute to climate change mitigation.

2. DEFINITIONS

This section provides definitions of the following key concepts: bioenergy, renewable, biofuels and biorefinery.

Bioenergy is the chemical energy contained in organic materials that can be converted into direct, useful energy sources via biological, mechanical or thermochemical processes. The most common and ancestral bioenergy source is firewood, which nowadays still represents 15% of global energy consumption (ADEME, 2006), some 90% of the world's

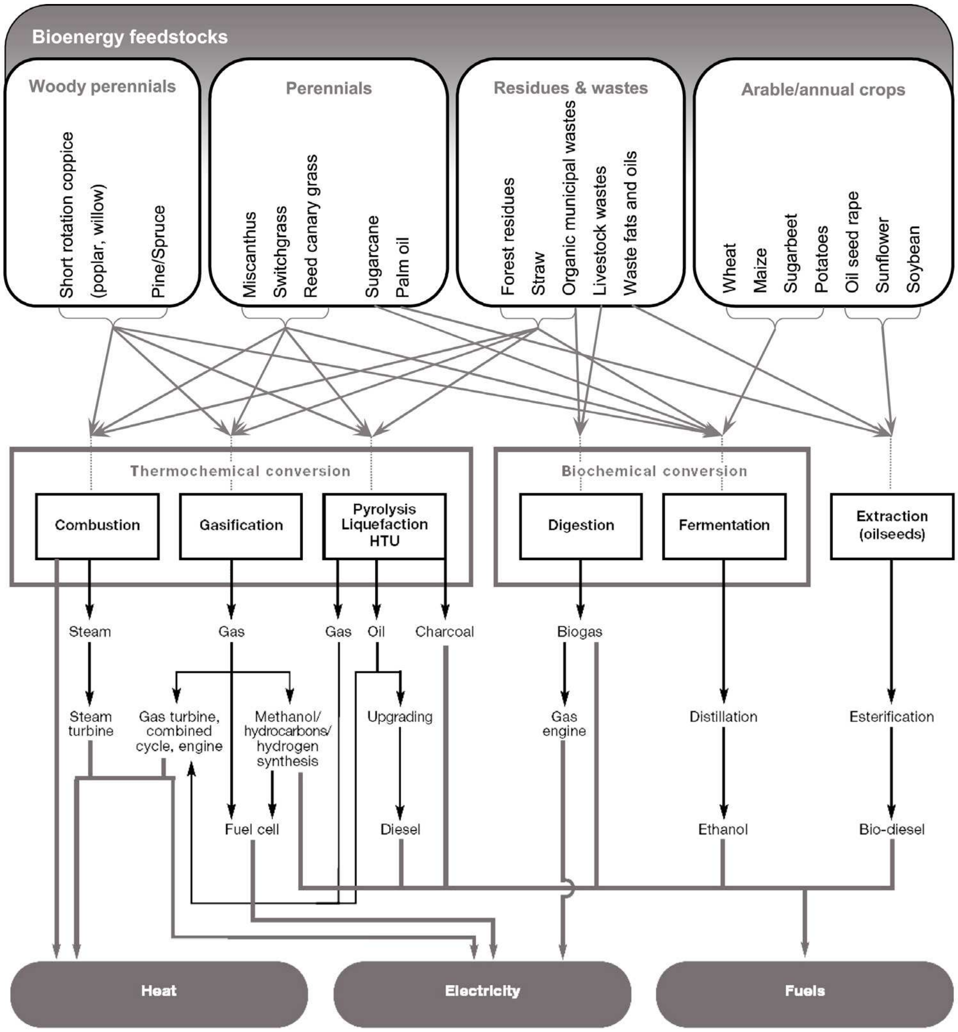

Figure 1. Bioenergy feedstock and conversion routes, adapted from Plassat, 2005 and UNDP, 2000. HTU: hydrothermal upgrading.

woodfuel being produced and consumed in the developing countries (Parikka, 2004). The prefix "bio", from the Greek "βίος" (meaning "life")[2], refers here to the origin of the energy converted through the metabolism of living organisms with, at the basis of the food chain, autotrophic organisms converting solar energy into chemical energy contained in the molecules they produce via photosynthesis[3]. The total sum of living organisms is called the biosphere in opposition to lithosphere, hydrosphere and atmosphere. It is also referred to as "biomass" or "biota" by biologists and ecologists. Biomass, in the energy sector, refers to biological material which can be used as fuel for transport, or an energy source to produce industrial or domestic heat and electricity (feedstock and conversion routes, Fig. 1). Bioenergy comes from biomass. In contrast, fossil energies are mineral resources, stocked in the lithosphere. The carbon fossil energy sources are the result of mineralisation transforming organic matter into mineral matter. This transformation takes millions of years, meaning that fossil resources are non-renewable on a human time scale.

"Renewable" does not mean "sustainable". Renewable resources consist of two main types of natural resources: flow resources and renewable stock resources. Flow resources, such as solar or wind energies, are non-limited resources despite intermittence. On the contrary, renewable stock resources, mainly biomass, are limited resources and their availability depends both on other primary natural resources (e.g. lands, water, ecosystems, etc.) and on natural regeneration/degeneration rates and/or anthropic production/consumption rates. The term "renewables" in the energy field encompasses all energies coming from renewable resources, e.g. photovoltaic energy, wind energy, bioenergy, etc. It is also referred to as RES, standing for renewable energy sources.

[2] Thus the prefix "bio" has nothing to do with the organic production label called "BIO" in France or Germany, for instance, which actually corresponds to specific management guidelines for agricultural production that aim at minimising the harmful impacts on the environment.

[3] Other autotrophic processes than photosynthesis exist when enlacing the whole biosphere, but they are less relevant in quantitative terms when focusing on bioenergy.

Considering biomass, "renewable" indicates that it will in theory stay available in an infinite time perspective as it can regenerate or be grown. However in practice, the renewal of biomass also depends on its management, which should ensure that primary resources are not overexploited or even depleted. If resource management is technically appropriate, environmentally non-degrading, socially favourable and economically viable, then the renewable resource will be exploited in a sustainable[4] way. The issue of sustainability being crucial in the field of bioenergy, the UN Executive Board for Clean Development Mechanisms released in December 2006 an official definition of "Renewable Biomass" including this sustainability dimension (UNFCCC, 2006). Among the five possible conditions where biomass can be defined as "renewable", the three that do not deal with residues or wastes have a first criterion mentioning that the land use shall not change except if land areas are reverted to forest. The second criterion implicitly linked to the first one states: "Sustainable management practices are undertaken on these land areas to ensure in particular that the level of carbon stocks on these land areas does not systematically decrease over time". This is a key element when comparing the CO_2 emissions from biofuels and fossil fuels. Indeed, the interesting fundamental carbon neutrality of combusted biomass is based on the fact that the emitted CO_2 from the plant originates from the atmosphere where it eventually goes back to. If land conversion to biomass production implies additional CO_2 emission through soil organic carbon losses, it may offset this carbon neutrality.

Highly dependent on the type of vegetation, the organic carbon stock is globally around 1.6 times higher in forest or permanent grassland soils than in crop soils (Antoni and Arrouays, 2007). Therefore, land-use change can lead to soil organic matter losses. Soil organic carbon content can also decrease in the long term as a consequence of the export of agricultural residues. Therefore, it is necessary to assess the net crop residue amount that would remain available for the bioenergy chain without degrading the soil quality in the long term (Saffih-Hdadi and Mary, 2008; Gabrielle and Gagnaire, 2008). Soil organic matter represents only a few percent of the total soil mass, but still constitutes a large organic carbon stock on a global scale, i.e. almost the same as the sum of the carbon stocks in the atmosphere and in the vegetation (Arrouays et al., 2002). Small but stable changes in this stock could critically impact the global carbon fluxes. Furthermore, soil organic matter plays a crucial role in soil quality. In an agricultural soil, whose main function is to provide nutrients and water to crops, soil organic matter permits the development of microorganisms decomposing organic matter into easy-absorbed mineral forms for the plant. It contributes to the soil cationic exchange capacity, which also influences the availability of essential minerals for the plant, and to the stability of the soil. Soil organic matter and forest biodiversity are among the precious resources whose conservation for future generations should not be jeopardised by land-use changes. For the producers, "Renewable Biomass" will no longer just mean "which can be grown" but also implies conditions for a sustainable production.

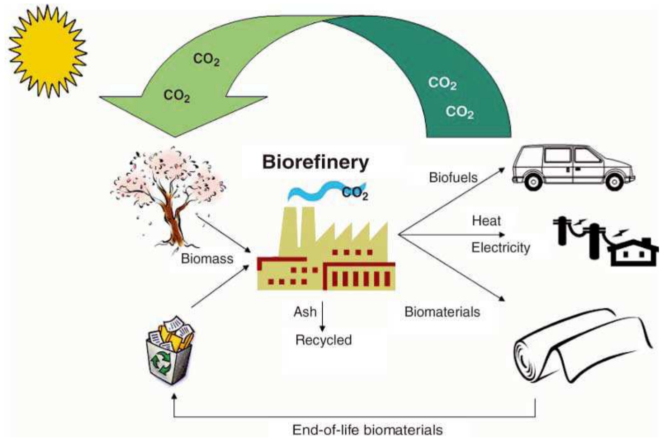

Figure 2. The fully integrated agro-biofuel-biomaterial-biopower cycle for sustainable technologies (Ragauskas et al., 2006).

Biofuels are biomass materials directly used as solid fuel or converted into liquid or gaseous fuels that can be stored, so that the harnessed energy can be released through combustion when needed. This chemical reaction permits the release of the binding energy that holds electrons to a nucleus in the organic molecules, in order to produce work and heat. In a narrower sense, biofuels may be only perceived as liquid or gas transportation fuels derived from biomass. Many different biomass raw materials can be used to produce biofuels including energy crops, agricultural residues or forest products, for example.

Biomass gives way to a whole product chain in which residues or co-products can largely contribute to the environmental and economic optimisation of the whole biomass value chain. Besides bioenergy, biomass can serve as a source of biomaterial (building materials, papers, etc.) and chemical compounds (solvents, pharmaceutics, cosmetics, biodegradable plastics, etc.). This last field of activities based on biomolecules is called "green chemistry". The overall integrated biorefinery that aims at using all the biomass compounds within one refinery complex is summarised in Figure 2. The power of the biorefinery concept is supported by economy of scale and by efficient use of all incoming bioresources. Using the petroleum industry as an illustrative example, ~5% of the total petroleum output from a conventional refinery goes to chemical products; the rest is used for transportation fuels and energy. Most visions for integrated biorefinery do not expect this ratio to change (Ragauskas et al., 2006).

To conclude, among renewables, biomass gives way to diverse bioenergy chains. In comparison with all the other renewables, bioenergy firstly presents the advantage that investments are generally lower. Furthermore, the diversity of raw materials and transformation processes offers a wide range of possibilities than can be adapted to different geographical locations, means and needs. Nevertheless, issues or challenges also arise when dealing with the development of bioenergy that will aim at finding the best cost/benefit equations,

[4] Based on the pillars of sustainability concept in the Brundland report, 1987.

Table I. Biofuel generations (Van der Drift and Boerrigter, 2006).

Biomass feedstock	1st generation biofuels	2nd generation biofuels	3rd generation biofuels
Vegetable oil	Pure Plant Oil (PPO, also called Virgin Plant Oil: VPO, or Straight Vegetable Oil:VGO) Fatty Acid Methyl Ester: FAME (e.g. Rape Seed Methyl Ester)		
Fermentable biomass	Biogas/Substitute Natural Gas		H_2
Starch/sugar	Ethanol/Ethyl Tertiary Butyl Ether (ETBE)		
Lignocellulose		Ethanol Fischer-Tropsch (FT) diesel* Dimethyl Ether (DME)* Methanol* Mixed Alcohols (MA)* Substitute Natural Gas: SNG	H_2*

Biofuels indicated with * are produced with synthesis gas (syngas, mainly H_2 and CO) as intermediate.

including externalities, depending on both the type and the amount of bioenergy produced. These equations appear to be especially difficult to solve in the case of transportation biofuels. Firstly, the reverse side of feedstock and technology multiplicity is that many options from worst to best biofuel chains exist, which finally creates an overall confusion and uncertainty about biofuels. This uncertainty is thereby hampering the development of production plants and new expensive technologies as it contributes to increasing investment risks. Secondly, whereas bioenergy chains, like the other renewables, were intrinsically thought to be advantageous first of all on a local scale, biofuels especially give way to international trade, that can raise the issue of externality displacement and imply the need for specific political and economic frameworks.

3. TRANSPORTATION BIOFUELS

Biofuels are nowadays commonly classed as 1st , 2nd or 3rd generation biofuels, as shown in Table I. First generation biofuels refer to those already considered as "traditional or conventional chains", whereas 2nd generation biofuels, requiring more complex and expensive processes, are not available yet on the market. The energy efficiency of a biofuel chain must be appraised considering two aspects, both dependent on feedstock type: the net energy yield per area unit and the energy cost for transformation processes. When considering plant biomass, the energy yield per hectare is a function of the type of plant, the climate, the soil properties and the crop management. C_4 plants, whose photosynthesis is more efficient, are especially energy cost-effective in humid tropical regions where water is not limiting, e.g. sugar cane in Brazil. On the other end of the spectrum, maize in the US necessitates considerable energy inputs.

There is among 1st generation biofuels no technological breakthrough that would lead to large differences in terms of energy efficiency. In temperate regions, oilseed crops typically generate lower yields per hectare than sugar or starch crops and are therefore more expensive to produce. But because oils seeds require less processing they still generally have positive global energy balances per unit of feedstock. Oilseed crops grown in tropical areas can thus be especially productive and competitive. Globally feedstock costs account for the majority of the eventual price of any 1st generation biofuel, while processing costs and a small proportion for transport represent most of the rest. For ethanol, feedstock comprises 50 to 70% of the production cost, while for biodiesel feedstock can be 60 to 80% of the production cost (Lang et al., 2001; Worldwatch Institute, 2007).

The split between 1st and 2nd generation biofuels lies in the fact that the latter are produced from lignocellulose, meaning that all types of vegetation and all parts of the plant are possible feedstock, whereas 1st generation biofuels only up-value specific parts of a few suitable plants. Hence, 2nd generation biofuels yield higher energy amounts per hectare than energy crops with a proportionally small specific organ of interest (such as seeds) as no part of the plant is left over. They also encompass a wider range of possible feedstock. Third generation biofuels are the follow-up of 2nd generation biofuels, from the same raw material up to H_2 production, whose energetic costs remain out of reach.

3.1. First generation biofuels

The production of 1st generation biofuels could rapidly be fostered as technologies ensue from the food industry. Pure plant oils or even cooking oils, also called yellow grease, can thus be directly used as fuel. However, complementary processes permit one to upgrade the biofuels in order to optimise their mixing with conventional fossil fuels without needing to adapt the motors.

3.1.1. Biodiesel

In a broad sense, biodiesel refers to pure and processed plant oils or animal fats. These oils and fats contain a mixture of triglycerides, free fatty acids, phospholipids, sterols, water, odorants and other impurities. Biodiesels are nowadays produced from a large range of oilseed crops, mainly rapeseed or canola, soybean and sunflower, palm oil and *Jatropha curcas* in tropical climates (see picture below). Other potential plant oil feedstock includes mustard seed, linseed, castor oil, peanut, cottonseed, coconut, *Lesquerella* spp. and micro-algae. There are as many different biodiesels as different oil compositions. Oilseed species vary considerably in their oil saturation and fatty acid content, characteristics that significantly affect the properties of the biodiesel produced.

Jatropha curcas fruit, Belize, February 2003
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The boiling and melting points of the fatty acids, methyl esters and glycerides increase with the number of carbon atoms in the carbon chain, but decrease with increasing numbers of double bonds (Ma and Hanna, 1999). Saturated fatty acids are more compactable, which enhances the oil energy density. But if they contain lots of saturated fatty acids, oils and fats are solid at room temperature and cannot be directly used as fuel in a diesel engine in their original form except in warm climates. The disadvantages of vegetable oils compared with petroleum diesel fuel are their higher viscosity, lower volatility and the reactivity of unsaturated hydrocarbon chains (Lang et al., 2001). Because of subsequent problems such as carbon deposits in the engine, engine durability and lubricating oil contamination, they must be chemically transformed to be compatible and used on a long term with existing engines (Ma and Hanna, 1999).

The most widespread biodiesels are methyl esters produced from plant oils combined with methanol through transesterification. The two other routes, microemulsion and pyrolysis, are not worth it; pyrolysis notably is expensive for modest throughputs and processing removes any environmental benefits of using a biofuel (Ma and Hanna, 1999). Transesterification is an alkali-catalysed reaction that requires 107.5 kg of methanol per ton of vegetable oil and results in the production of 1004.5 kg of methyl ester and 103 kg of glycerol (Graboski and McCormick, 1998). In this three-step reaction, triglycerides are converted into diglycerides, then monoglycerides and finally reduced to fatty acid esters, enhancing the viscosity of the final biodiesel. The viscosity of vegetable oils and that of their final esters is of the order of 10–20 times and twice that of diesel fuel, respectively (Lang et al., 2001). A pre-step and catalysis make it possible to deal with the impurities such as free fatty acids and water to improve the reaction kinetics (Ma and Hanna, 1999). Methanol is preferred over ethanol because of its physical and chemical properties as well as comparative low cost (Ma and Hanna, 1999; Lang et al., 2001), although it introduces a part of fossil fuel into the biodiesel. For different esters from the same vegetable oil, methyl esters also appeared to be the most volatile ones (Lang et al., 2001).

Biodiesel used as an additive to diesel fuel can improve its lubricity. This property is becoming increasingly valuable as recent legislation has mandated further regulation on the sulphur content of diesel fuels; these cleaner diesel fuels exhibiting reduced lubricity as compared with their high sulphur predecessors (Radich, 2004; Goodrum and Geller, 2005). Some fatty acids such as ricinoleic (castor oil) and lesquerolic acids (*Lesquerella* spp.) could be especially efficient in enhancing the lubricity of a diesel fuel to an acceptable level at concentrations as low as 0.25% (Goodrum and Geller, 2005). Blending biodiesel with diesel fuel can increase the neat cetane number. Cetane number increases with increasing length of both fatty acid chain and ester groups, while it is inversely related to the number of double bonds and as double bonds and carbonyl groups move toward the centre of the chain (Graboski and McCormick, 1998; Tripartite Task Force, 2007). Highly saturated oils, with a low number of double bonds, hence provide the fuel with superior oxidative stability and higher cetane number (Worldwatch Institute, 2007). The average cetane numbers are 50.9 for soy and 52.9 for rapeseed esters. For the other esters listed in Graboski and McCormick, 1998, including sunflower, cottonseed and palm oil, cetane numbers vary in the 48–60 range. In comparison, the cetane index for petroleum diesel ranges from 40 to 52 (Radich, 2004).

The co-products of the entire chain are the meal left in the seed after oil extraction, which is sold as animal feed, and the glycerine from glycerol recovery, used in cosmetics. However, the rapid expansion of biodiesel has already saturated the market of glycerine in Europe, undercutting its ability to reduce the biodiesel price, as it could offset 5% of the production cost (Worldwatch Institute, 2007).

3.1.2. Ethanol

Ethanol, on the contrary to biodiesel, is a single-compound biofuel whose final composition does not vary with the type of feedstock. The feedstock is sugar and starch crops, which are basically equally processed through pre-treatment, fermentation by yeasts and other microbes, and distillation. The main sugar crops are sugar cane and sugar beet. Sweet sorghum could also become an interesting ethanol feedstock as a multi-use crop, whose seeds are edible and whose stalk contains sugar. The main starch crops used nowadays are maize and wheat; also potatoes, cassava and sorghum grain to a lower extent. Sugar crops typically yield more ethanol per hectare with an overall better energy balance than starch crops because (1) sugar crops yield higher sugar amounts per hectare compared with starch crops; and (2) sugar can be directly fermented, whereas starch long polymers have to be hydrolysed before being fed to yeast for the ethylic fermentation.

"Wet-milling" and "dry-milling" are the two current common methods to treat the starchy crop parts at their entry in the process chain. In the wet-milling, grains are soaked and chemically sub-divided into rich starchy parts of primary interest (grain endosperms) and other parts that contain more protein and fibres and constitute diverse co-products (maize oil and syrup, gluten feed, germ meal, vitamins and amino acids). These co-products can contribute up to 25% of the process economy (Worldwatch Institute, 2007). The dry-milling method only consists of grinding the unprocessed heterogeneous seeds into granules. It is therefore less expensive but also leads to less diverse co-product production. The main co-product is the dried distillers grain (DDG) fed to animals that can digest high proportions of fibres, and contributes up to 20% of the process economy (Worldwatch Institute, 2007). In both "wet" and "dry" processes, the starch is finally hydrolysed into sugar typically using a high-temperature enzyme process (Fulton et al., 2004; Cardona and Sanchez, 2007). CO_2 from fermentation can also be sold as a co-product to beverage industries. Indeed, in conventional fermentation, approximately one-third of the carbon available in the sugar is lost as CO_2 (Strege, 2007).

The fermented ethanol must be distilled until enough water is removed to make the final anhydrous ethanol suitable for blending with gasoline (<1% of water in temperate climates). Indeed, water in ethanol blended with gasoline makes the fuel more sensitive to frost and increases the risk of phase separation in both storage and vehicle fuel tanks, which can cause serious operating problems for the engines (Tripartite Task Force, 2007; Balat et al., 2008). To improve the ethanol quality as a blend in low percentage, ethanol (47% on a mass basis) can be converted into Ethyl-Tertio-Butyl-Ether (ETBE) by reaction with isobutylene (53%). An ETBE blend of 15% corresponds to a blend in volume of 7% ethanol.

Since they both contain oxygen, ethanol and biodiesel are better combustibles than the substituted fossil oils, reducing the emission of pollutants such as CO, hydrocarbons (HC), sulphur oxide[5] and particulates by up to half of these emissions, depending on the biofuel and the blend mix (Shahid, 2007; Luneau and Fayet, 2007; Murugesan et al., 2008). Exhausts from blends with vegetable oils also depend on the engine load (Murugesan et al., 2008). Conclusions are not univocal concerning NOx emissions[6], but biofuels would tend to lead to slightly higher emissions (Graboski and McCormick, 1998; Radich, 2004), notably with blends of 20% of ethanol or biodiesel in a car driven in the city (Luneau and Fayet, 2007). In Murugesan et al. (2008), NOx emissions are reported to be in the range of ±10% as compared with diesel, depending on engine combustion characteristics (2008). Biofuels increase the octane level[7] (thanks to ethanol, Harijan et al., 2007; Balat et al., 2008) and cetane number (thanks to biodiesel, Radich, 2004). On the other hand, both ethanol and biodiesel may cause corrosion and are sensitive to cold weather.

The primary asset of biofuels is the convenience that they can be used as blends with conventional fuels in existing vehicles. However, unmarked blends are limited to certain extents according to fuel and vehicle specifications. As an example, in Europe, these blends on a volume basis are: 5% ethanol or 15% ETBE blends with gasoline, and currently 5% biodiesel in diesel fuel (Wiesenthal et al., 2007), up to 20–30% for specific fleets (Plassat, 2005). Beyond these limits, engines have to be adapted so that their performances will not be affected in the long run. This is, for instance, the case with the flexible-fuel vehicles that can run on low- and high-level ethanol blends up to 85–100% (also written as E85, E90, E100; biodiesel blends are noted B20, B30).

Divergences in biofuel technical specifications have been introduced worldwide due to feedstock variances, climatic conditions in each country and region, and the characteristics of the local markets (Tripartite Task Force, 2007). In consequence, automotive sector and biofuel blend-related strategies have also diverged. Diverse biofuel and blend standards were adopted worldwide by the various agencies, ABNT/ANP[8] in Brazil, ASTM International in the US and CEN[9] in Europe. The Tripartite Task Force has been working on implementing a road map to come up with international compatible biofuel-related standards to help increase the use of biofuels and avoid adverse trade implications in the global market.

The water content of ethanol shows how critical it may be to agree on standardised blends. Phase separation due to water occurs more readily at lower levels of ethanol in gasoline. In Brazil and the US, where ethanol blends reach up to 5.7–10% and 20–25%, respectively, phase separation is not an issue, whereas it becomes an issue in Europe with lower ethanol blends in gasoline. Additional dehydration increases production cost and can reduce productivity at the mill by up to 7% (Tripartite Task Force, 2007). Considering the additional environmental and economic costs related to biofuel upgrading, international harmonisation should urge a maximum level of blend flexibility correlated with a focus on minimum processing and optimum engine modifications.

3.1.3. Biogas

Biogas is produced through methanisation, i.e. the anaerobic digestion by bacteria of biodegradable matter such as municipal solid or agricultural waste, liquid slurry, solid manure, or maize silage, for instance. The more dry matter and fatty acids in the substrate contents, the greater the biogas yield (Moras, 2007). Apart from about 55 to 70% of methane (52–65% in examples in Tab. II), the actual fuel, biogas also

[5] Sulphur oxides (SOx) contribute to acid rain and can be carcinogenic.

[6] NOx are precursors to the formation of tropospheric ozone.

[7] Gasoline high octane value indicates a smaller likelihood that the fuel combusts too soon (low auto-ignite tendency), provoking engine knock problems. A high tendency to auto-ignite, or low octane rating, is undesirable in a spark ignition engine (gasoline) but desirable in a diesel engine (high cetane number).

[8] Brazilian Petroleum, Natural Gas and Biofuels Agency.

[9] Comité Européen de Normalisation.

Table II. Main feedstock, productivity per hectare, and co-products of 1st generation biofuels; FM: fresh matter.

Biomass feedstock	Countries regions	Yields t FM ha^{-1}	Biofuels L ha^{-1}	Fossil equivalent (on energy basis) L ha^{-1}	Total energy output /Fossil energy input	Co-products	References
SUGAR CROPS			ethanol (1L= 0.791 kg)	1L ethanol = 0.665 L gasoline*	Indicative values	Crop residues left in the field are often considered as fertilizing co-products and are important for sustaining soil C content	Fulton et al., 2004 FNR, 2007
SUGAR CANE	Brazil	85	7.080	4.708	~8 [2.1–8.3]	Yeast as cattle feed supplement; Bagasse as feedstock for feed. heat. Electricity, and cellulosic ethanol;	Brazilian Ministry in Xavier, 2007 Cardona and Sanchez, 2007
			6.500	4.323		Fructose as sweetener; invertase for food industry	Fulton et al., 2004
			6.800	4.522		23.8 t ha^{-1} bagasse (50% H_2O) = 7.854 kWh electricity	Moreira, 2000
		73.5	5.476	3.641		Energy from cane trash	Naylor et al.. 2007 in FAO, 2008a
	India	60.7	4.522	3.007			Naylor et al.. 2007 in FAO, 2008a
		66.5–71.3–	5.300	3.525		2.753–2.952 litres of cellulosic ethanol from 20.5–22 t ha^{-1} bagasse	Fulton et al., 2004 Kadam, 2000
	Thailand	55.7	–	–		104 kg sugar/t cane +45 kg molasses/t cane (+5 kg rice husk) =10.2 L ethanol [national availability~1Mt yr^{-1} molasses]	Nguyen and Gheewala, 2008 Prasertsri, 2006
	Global	65	4.550	3.026			Rajagopal et al.. 2007 in FAO, 2008a
SUGAR BEET	France	66.2	3.793 7.307	2.522 4.860	~2 [1.2–2.2]	3.7 t ha^{-1} of sugar. If only ethanol without sugar production 44 t green manure (10% dry matter)	ADEME/DIREM, 2002
		–	7.500	4.988		(Stabilized) molasses as fertilizer	France Betteraves (FB)/Passion Céréales, 2007
	Germany	55.5	5.439	3.617		Sugar beet pulp and dried slop as animal feed	FNR, 2007
	EU	– 68.9	5.500 –	3.658 –		Slop to biogas Heat	Fulton et al., 2004 JRC/EUCAR/CONCAWE, 2008
	Global	46	5.060	3.365			Rajagopal et al., 2007 in FAO, 2008a
SWEET SORGHUM	China	60 90	5.070 7.605	3.370 5.057	~1 [0.9–1.1]	3–4 t sorghum grains	USDA, 2007

Biomass feedstock	Countries regions	Yields t FM ha^{-1}	Biofuels L ha^{-1}	Fossil equivalent (on energy basis) L ha^{-1}	Total energy output /Fossil energy input	Co - products	References
STARCH CROPS							
MAIZE	Brazil	10	3.570	2.374	~1.5 [1.4–1.8]	Wet-milling process : fibre as substrate for enzymes production, CCDS as feed for non-ruminants, gluten meal (60% protein. high fat) and gluten feed (20% protein, low fat), corn oil, different chemicals and food-related products as vitamins and amino acids. Dry-milling process: DDGS as feed for ruminants (27–35% protein) 4.8 t ha^{-1} DDGS	Brazilian Ministry in Xavier, 2007
	USA	–	3.100	2.062			Fulton et al., 2004 Cardona and Sanchez, 2007
		8.7	3.471	2.308			Graboski, 2002
		9.4	3.751	2.494			Naylor et al., 2007 in FAO, 2008a
	Canada	7.2	3.384	2.250			Levelton, 1999
	China	5	2.028	1.349			USDA, 2007
		5	1.995	1.327			Naylor et al., 2007 in FAO, 2008a
	France	–	3.600	2394			FB/Passion Céréales, 2007
	Global	4.9	1.960	1.303			Rajagopal et al., 2007 in FAO, 2008a
WHEAT	France	7.5	2.600	1.729	~2 [1.2–4.2]	2.8 t ha^{-1} DDGS DDGS can be used as animal feed or for fuel Glycerol Straw for fertilizer or fuel	FB/Passion Céréales, 2007
		9	2.550	1.696			ADEME/DIREM, 2002
	Germany	–	2.560	1.702			FNR, 2007
		7.44	2.753	1.831			Öko-Institut, 2004
	EU	–	2.500	1.663			Fulton et al., 2004
		5.2	–	–			JRC/EUCAR/CONCAWE, 2008
	Global	2.8	952	633			Rajagopal et al., 2007 in FAO, 2008a
CASSAVA	Brazil	13.6	1.863	1.239	~2.85 2.1 in China		Naylor et al.. 2007 in FAO, 2008a
	Nigeria	10.8	1.480	984			
	Thailand	19	–	–			Nguyen et al., 2007
	Global	12	2.070	1.377			Rajagopal et al., 2007 in FAO, 2008a
RICE	Global	4.2	1.806	1.200	–		Rajagopal et al., 2007 in FAO, 2008a
BARLEY	Global	–	1.100	732	–	DDGS can be used as animal feed or for fuel Straw for fertilizer or fuel	Fulton et al., 2004
SORGHUM GRAIN	Global	1.3	494	329	–		Rajagopal et al., 2007 in FAO, 2008a
OIL PLANTS			Pure vegetable oils	rapeseed oil 1 L = 0.96l diesel fuel sunflower oil 1 L = 0.85l diesel fuel			ITDG, 2000 ADEME/DIREM, 2002 FNR, 2007
SUNFLOWER	France	3.34	1.522	1.461	4.7	Meal or cake as animal feed	ADEME/DIREM, 2002
RAPESEED		2.44	1.178	1.000	7.5		
	Germany	3.4	1.480	1.420	3–5		FNR, 2007

Biomass feedstock	Countries regions	Yields t FM ha^{-1}	Biofuels L ha^{-1}	Fossil equivalent (on energy basis) L ha^{-1}	Total energy output /Fossil energy input	Co - products	References
METHYL–ESTERS			Biodiesel (methyl-ester) 15 °C (1 L = 0.9 kg)	1 L biodiesel = 0.88–0.95 L diesel = 0.85 L diesel = 0.89 L diesel			Worldwatch Institute, 2007 ITDG, 2000 Lang et al., 2001
RAPESEED	Germany	3.4	1.550	1.380	~2.5 [1.3–3.7]	Meal or cake as animal feed or converted to biogas. steam and electricity Glycerol sold or converted to biogas then steam	FNR, 2007
	France	3.34	2.184	1.944			ADEME/DIREM, 2002
	EU	– 3.11	1.200 –	1.068 –			Fulton et al., 2004 JRC/EUCAR/CONCAWE, 2008
SUNFLOWER	France	2.44	1.690	1.504	3		ADEME/DIREM, 2002
	EU	– 2.44	1.000	890			Fulton et al., 2004 JRC/EUCAR/CONCAWE, 2008
SOYBEAN	USA	– 2.7	500 552	445 492	~3 [1.5–3.5]		Fulton et al., 2004 Naylor et al., 2007 in FAO, 2008a
	EU	–	700	623			Fulton et al., 2004
	Brazil	– 2.4 2.8	400 491 –	356 437 –			Fulton et al., 2004 Naylor et al., 2007 in FAO, 2008a JRC/EUCAR/CONCAWE, 2008
PALM OIL	Brazil	–	5.000	4.450	~9 [8.7–9.7]	Glycerol sold or converted to biogas then steam	Fulton et al., 2004
	Malaysia	–	6.000	5.340			Fulton et al., 2004
		20.6	4.736	4.215			Naylor et al., 2007 in FAO, 2008a
	Indonesia	17.8	4.092	3.642			Naylor et al., 2007 in FAO, 2008a
	Global	19	–	–			JRC/EUCAR/CONCAWE, 2008
JATROPHA	India	1.42	577	514	–	Husks and cakes as fertilizer or for fuel Detoxified meal or cakes as animal feed. Meal to produce biogas Shell as fuel Glycerol	Reinhardt et al., 2007
	Global	–	2.000	1.780			Fulton et al., 2004
WASTED VEGETABLE OIL	Global	–	–	–	[4.8–5.8]	Meal or cakes as animal feed	Rajagopal et al., 2007 in FAO, 2008a FAO, 2008a

Biomass feedstock	Countries regions	Yields m^3 t FM^{-1}	Biofuels m^3 ha^{-1}	Fossil equivalent (on energy basis) m^3 ha^{-1}	Total energy output /Fossil energy input	Co-products	References
DIGESTIBLE MATERIALS			Biogas	(% of CH_4) m^3 methane (LHV: 36 MJ m^{-3}			
MAIZE SILAGE	Germany	170–202	7.800–8.300	(52) 4.000–4.300	–		FNR, 2008 www.pleinchamp.com (03.03.08) Poiret, 1996 Moras, 2007
	France	200	6.660–8.500	(52) 3.500–4.450			
GRASS SILAGE	France	211	2.000–2.600	(54) 1.100–1.400			
	Germany	172	2.500–3.200	(54) 1.400–1.700			
SLURRY						CO_2 (25–45% of biogas) Digested biomass as fertilizer	Moras, 2007 FNR, 2008
Bovine	Germany	25 $m^3.t^{-1}$		(60) 15 $m^3.t^{-1}$	–		
	Global	26 $m^3.t^{-1}$		(60) 15.6 $m^3.t^{-1}$			
Pig	Germany	28 $m^3.t^{-1}$		(65) 18.2 $m^3.t^{-1}$			
	Global	36 $m^3.t^{-1}$		(65) 23.4 $m^3.t^{-1}$			
Chicken	Global	44 $m^3.t^{-1}$		(60) 26.4 $m^3.t^{-1}$			
MANURE							
Bovine	Germany	45 $m^3.t^{-1}$		(60) 27 $m^3.t^{-1}$	–		
	Global	25 $m^3.t^{-1}$		(60) 15 $m^3.t^{-1}$			
Pig	Germany	60 $m^3.t^{-1}$		(60) 36 $m^3.t^{-1}$			
Chicken	Germany	80 $m^3.t^{-1}$		(60) 48 $m^3.t^{-1}$			
BIOMASS WASTE							
Green waste	Germany	100 $m^3.t^{-1}$		(61) 61 $m^3.t^{-1}$	–		
	Global	60 $m^3.t^{-1}$ 151 $m^3.t^{-1}$		(61) 36.6 $m^3.t^{-1}$ (61) 92 $m^3.t^{-1}$			
Maize straw	Global	402 x 10^3 $L.t^{-1}$		–			
WASTED OILS	Global	800 x 10^3 $L.t^{-1}$		–	–		
MISCANTHUS (>3years)	Germany 2nd generation biofuels stand	15	6.081 L ha^{-1}	4.044 L ha^{-1} gasoline	20		Nguyen et al., 2007 FNR, 2007
CEREAL STRAW		6	2.390 L ha^{-1}	4.590 L ha^{-1} gasoline			
STRAW OR WOOD– BTL		–	4.030 L ha^{-1}	3.900 L ha^{-1} diesel.	–		

Notes: energy ratios are "approximate" values and range given in Worldwatch Institute (2007), except those in italics.
FM: fresh matter; CCDS, corn condensed distiller's solubles; DDGS, dried distiller's grains with solubles.
* In Nguyen et al. (2007), 1 L ethanol = 0.89 L gasoline based on fuel economy, i.e. taking into consideration that vehicle performance is enhanced due to ethanol's higher octane value.
Indicative provisional figures are given for 2nd generation biofuels in order to give a quick comparison point.

contains substantial amounts of CO_2, 30 to 45%, small quantities of hydrogen sulphide and other trace gases such as ammonia[10]. The separation of these components of biogas via a gas scrubber is an expensive prerequisite in order to use the biogas as fuel or to mix it with natural gas.

Biogas is less considered as transportation biofuel, because its target vehicle fleet remains marginal due notably to onboard gas storage constraints. The primary interest of biogas remains its local development as fuel for heat and power plants in rural areas. About 25 million households worldwide currently receive energy for lighting and cooking from biogas produced in household-scale digesters, including 20 million in China, 3.9 million in India and 150 thousand in Nepal (REN21, 2008). Hence, the two prevalent types of digesters are the Chinese "fixed dome" and the Indian "floating cover" that only differ by the gas collection method (ITDG, 2000). Biogas production in specifically designed digesters is the most widespread technology, although capturing methane from municipal waste landfill sites has been considerably developed. In the US, waste management including the recovering of methane produced by landfills has made possible to reduce these methane emissions by 50% over the years and has become one of the largest holders of greenhouse gas emission credits (Kram, 2007). Although the reaction takes several days to finally degrade just about 10 to 15% of the initial material, biogas permits one to take advantage of cheap feedstock and to diminish greenhouse gas emissions. Moreover, the solid residue of the process can be dried and used as fertiliser that has a high nutrient content and whose pathogenic germs have been killed during the digestion process, notably due to temperatures of 35 °C to more than 50 °C (ITDG, 2000; Baserga, 2000).

Biogas as transportation fuel could receive more attention in the coming decades, especially for use in city fleets and trucks, as has been the case in Sweden and Switzerland for a long time. Table II gives the productivities of common feedstock and biofuels worldwide.

3.2. Current 1st generation biofuel supply worldwide

Combustible renewables and waste are mostly consumed directly; only about 1.8% is consumed by the transport sector, about 17.6% by the industries and 80.6% by other sectors, notably households (IEA, 2007a). Production of heat and electricity dominate current bioenergy use with two key industrial sectors for application of state-of-the-art biomass combustion for power generation: the paper and pulp sector and cane-based sugar industry (IEA Bioenergy, 2007).

Global fuel ethanol production more than doubled between 2001 and 2006, while that of biodiesel expanded nearly sixfold. The US and Brazil currently dominate world ethanol production, which reached a record 52 billion litres in 2007 (FAO, 2008a). Close to half the world's fuel ethanol was produced in the US from 14% of its national maize production in 2006 (Möller et al., 2007), and more than two-fifths in Brazil from sugar cane, roughly providing 21%[11] of its transport fuel consumption (OECD, 2006). The remaining production comes from Spain, Sweden, France and Germany. China's ethanol from maize, wheat and sugar cane is mostly destined for industrial use.

In 2006, Europe accounted for 73% of all biodiesel[12] production worldwide, mainly from rapeseed and sunflower seeds, with Germany as the leading producer (40%), followed by the US, France and Italy generating most of the rest (Worldwatch Institute, 2007). In 2007, the EU still accounted for 60% of global biodiesel production, that amounted to 6.2 billion litres in 2007 (FAO, 2008a), but biodiesel production has increased in all producing countries; it has doubled in the USA and in most of the producing countries in Asia and Oceania, and more than tripled in Brazil. Figure 3 show the production shares of ethanol and biodiesel worldwide in 2007.

Despite the growth in biofuel consumption and a comparatively slower growth in oil consumption, biofuels still do not represent a significant share in worldwide liquid fuel supply; about 0.9% by volume, 0.6% by transport distance travelled (Worldwatch Institute, 2007). Within Europe, biofuels are essentially domestically produced and consumed, except in Sweden, where since 2004 all petrol has been blended with 5% ethanol, mainly originating from Brazil and wine production sites in Southern Europe; only 20% is produced nationally (Van der Drift and Boerrigter, 2006). International trade in biodiesel is minimal as yet: only 10% of biofuels produced in the world are sold internationally, with Brazil accounting for roughly half of those sales (IEA, 2006). However, trade is very likely to increase, notably due to the fact that only some regions of the world may be able to produce large biomass feedstock (see part 6).

3.3. Towards 2nd and 3rd generations of biofuels

Second generation biofuels are produced via biochemical (hydrolysis and fermentation) and thermochemical (pyrolysis or gasification) treatments. The biochemical or so-called "wet process" is very similar to the 1st generation ethanol except for the feedstock, which is not specific. Indeed, 2nd generation biofuels are all produced from lignocellulose, i.e. all kinds of vegetal biomass, as lignocellulose forms the basic structure of vegetal cell walls. Cell walls make up a substantial portion of the dried biomass: about 60–80% and 30–60% in the stems of woody and herbaceous plants, respectively, and about 15–30% in their leaves (Möller et al., 2007).

Lignocellulose consists of intricate assemblages of cellulose, hemicellulose and lignin, whose proportions and molecular organisation vary depending on the type of biomass. A typical range is 40 to 55% cellulose, 20 to 40% hemicellulose, and 10 to 25% lignin (Worldwatch Institute, 2007). The

[10] http://www.planete-energies.com (consulted on 10.03.2008).

[11] Various data: 13.2% in energy terms according to IEA 2006; 40% according to Xavier (2007).

[12] Biodiesel here does not take into consideration pure vegetable oils mostly directly consumed by farmers on the farm.

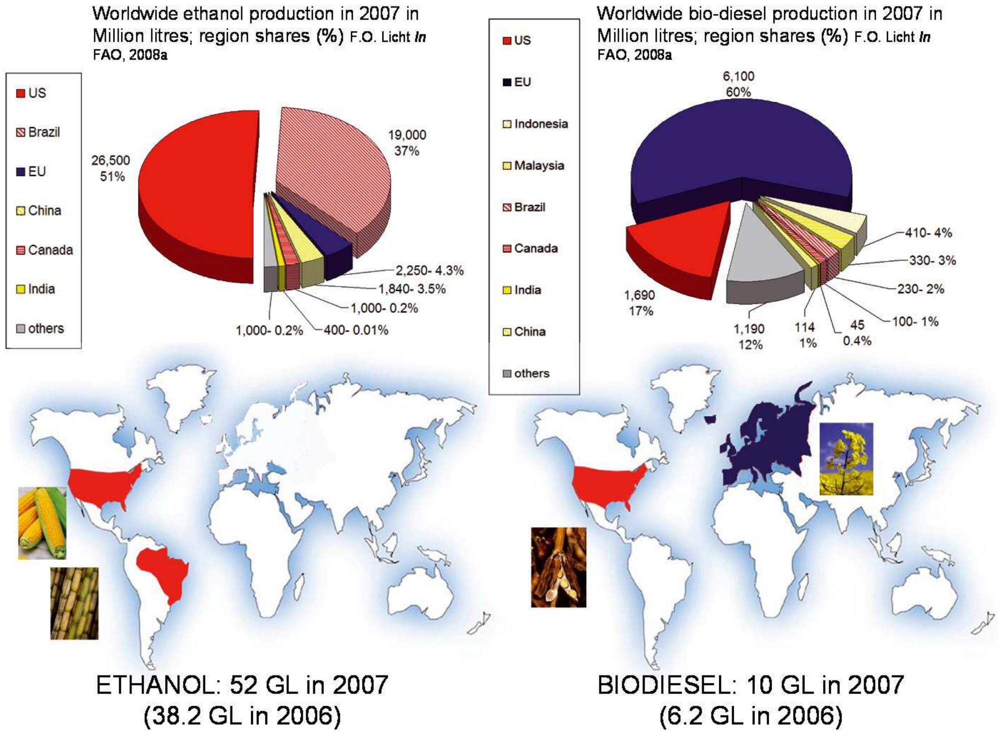

Figure 3. Worldwide ethanol and biodiesel productions in 2007 in Million litres; region shares (%). Drawn from data of F.O. Licht[13] in FAO, 2008a.

other minor components of cell walls are proteoglycans and pectins that glue together all the lignocellulosic compounds. The conformation of glucose residues in the crystalline cellulose core of cell-wall microfibrils forces the hydroxyl groups into radial orientation and the aliphatic hydrogen atoms into axial positions. It leads to the creation of strong interchain hydrogen bonds between adjacent chains in a cellulose sheet, which make cellulose resistant to enzymatic hydrolysis, and weaker hydrophobic interactions between cellulose sheets that contribute to the formation of a water layer near the hydrated cellulose surface, protecting cellulose from acid hydrolysis.
Furthermore, the microfibrils are embedded in the matrix of hemicelluloses and lignin, the latter also contributing to making cells walls hydrophobic and more resistant against enzymatic attack (Möller et al., 2006). Other molecules such as, for instance, waxes or inhibitors of fermentation, which naturally exist in the cell walls or are generated during conversion processes, also contribute to biomass recalcitrance (Himmel et al., 2007). This recalcitrance is the primary barrier to producing ethanol from lignocellulosic feedstock, commonly referred to as cellulosic ethanol. Indeed, expensive pre-treatments are necessary to break down this resistance, and reaching a cost-effective cell wall saccharification, i.e. the degradation of cell walls into monosaccharides, is the key that could really permit cellulosic ethanol to enter the market.

Pre-treatments include physical methods such as milling and grinding, high-pressure steaming and steam explosion, and biological (lignin- or/and cellulose-degrading organisms) or chemical methods (alkali or acid treatments, solvents) to solubilise parts of the hemicelluloses and the lignin. So far, methods such as ammonia fibre explosion (AFEX), wet oxidation and liquid hot water (LHW) treatment seem to be more successful for agricultural residues, whereas steam pre-treatment has resulted in high sugar yields for both forestry and agricultural residues (Hahn-Hägerdal et al., 2006). Monosaccharides from cellulose (glucoses) and hemicelluloses (pentose sugars) are then released through acid- or enzyme-catalysed hydrolysis, and finally fermented. Concentrated or dilute acid hydrolysis methods are more mature but very energy-intensive and present the disadvantage of potentially also degrading the monosaccharides. Enzymatic degradation, on the contrary, is more specific and perceived by many experts as a key to cost-effective saccharification, but none of these methods is currently cost-effective (Möller et al., 2006). As an example, hydrolysis of pre-treated lignocellulosic biomass requires one hundred-fold more enzymes than hydrolysis of starch (Tolan, 2006 in Möller et al., 2006).

Researchers have therefore been focusing, on the one hand, on improving the yields of pre-treatment and lowering their

[13] Christoph Berg is Managing Director at the commodity analysts F.O. Licht. F.O. Licht monitors the global soft commodity markets.

costs, and on the other hand, on developing integrated processes that make it possible to protect, separate and use the other compounds, such as the C5-sugars and the lignin, or the co-products such as furfural and fermentation gases. Traditional fermentation processes relied on yeasts and microbes that only convert C6-sugars (mainly glucose) into ethanol (Fulton et al., 2004). Researchers have already succeeded in producing several new yeast strains and bacteria, such as engineered *E. coli*, *K. oxytoca* and *Z. mobilis* (Balat et al., 2008), that exhibit varying degrees of ability to convert the full spectrum of available sugars into ethanol.

The introduction of simultaneous saccharification and fermentation (SSF) permitted a gain in efficiency of a 13% higher overall ethanol yield than separate hydrolysis and fermentation (SHF) (72.4% versus 59.1% of the theoretical maximum yield) (Öhgren et al., 2007). This gain is due to the fact that hydrolysed sugars are immediately fermented in the case of SSF, whereas their accumulation leads to enzyme inhibition in the case of SHF (Fulton et al., 2004). More recently, the simultaneous saccharification and co-fermentation of hexoses and pentoses (SSCF) has proved to be further advantageous as the hexose sugars continuously released by enzymatic hydrolysis increase the rate of glycolysis, so that pentose sugars are fermented faster and with higher yield (Hahn-Hägerdal et al., 2006). This makes it possible to lower the cost as both operations can be done in the same tank, added to the fact that enzyme manufacturers have recently reduced costs substantially thanks to biotechnology (Solomon et al., 2007; Balat et al., 2008). Nevertheless, further advances in discovering new hydrolases, new fermentation enzymes and organisms with process-tolerant traits such as tolerance to alcohol, pH and inhibitors, and advances in product recovery technology are required to reach commercial viability (US DOE, 2006a). Producing enzymes with combined tolerant traits is a real challenge, considering, for instance, that the majority of organisms cannot tolerate ethanol concentrations above 10–15% (w/v) (Balat et al., 2008). Moreover, optimal temperature and pH conditions vary depending on the enzymes and microorganisms involved in the different process stages, which can hamper the efficiency of the batch SSF or SSCF (Cardona and Sanchez, 2007; Öhgren et al., 2007). "Consolidated biomass processing" (CBP), the logical end point in the evolution of biomass conversion technology, would require a unique microbial community to produce all the enzymes for the saccharification and fermentation within a unique reactor vessel, but it has not been achieved yet (Fulton et al., 2004). For robust and complete conversion of polysacchaarides locked in biomass, the ultimate ethanologens will have to produce at least a dozen enzymes of different catalytic activities. Engineering such a yeast strain requires (1) screening thousands of combinations of biomass-degrading enzymes to identify the appropriate set of enzymes, then (2) managing to ensure that this strain is capable of simultaneously expressing the genes for all necessary enzymes (Hector et al., 2008).

Current biomass-conversion technologies were developed empirically and are based on limited understanding of the biological and chemical properties of biomass (Himmel et al., 2007). Therefore, all research efforts also rely in parallel on fundamental research to understand and characterise the cell walls of a very wide range of biomass feedstock better. This feedstock encompasses perennial grasses, short rotation coppices, cereal straws, and other biodegradable residues or waste. According to Möller et al. (2007), poplar, willow, miscanthus (see picture below) and wheat straw are the main relevant feedstock in Europe. In the US, attention is especially paid to maize stover, wood waste and switchgrass, whereas sugar cane producers are obviously more interested in converting the sugar cane bagasse. Research worldwide includes breeding programmes to develop new varieties with interesting phenotypes in terms of growth and resistance, but also in more specific biorefinery-related terms, i.e. regarding the cell wall composition. Research also includes genetic engineering. As an example, "Spartan maize" has been genetically modified within a research programme at the Michigan State University to express cellulase and hemicellulase in the plant's leaves and stover (Sticklen, 2007). Transgenic alfalfa has also demonstrated lower amounts of lignin, leading to drastic reduction of pre-treatment costs (Chapple et al., 2007). However, reduction in lignin content also leads to reduced biomass by up to 40%, which emphasises the need to determine whether cell wall manipulation may compromise the plant's structural integrity or susceptibility to pests and pathogens (Chapple et al., 2007).

Miscanthus at Estrees-Mons,
November 2008 bessou©INRA

Figure 4 shows the expected time frame for research advances and economically viable implementation of cellulosic ethanol over the next 5 to 15 years in the US. Considering the potential to sustainably harvest more than 1.3 billion metric tons of biomass from U.S. forest and agricultural lands by the mid-21st Century (Perlack et al., 2005), these projections illustrate the needed co-increase of technological performances and systems integration, and detail the research fields. Within 10 years, dedicated energy crops with composition and structure better suited for breakdown into sugars for fermentation,

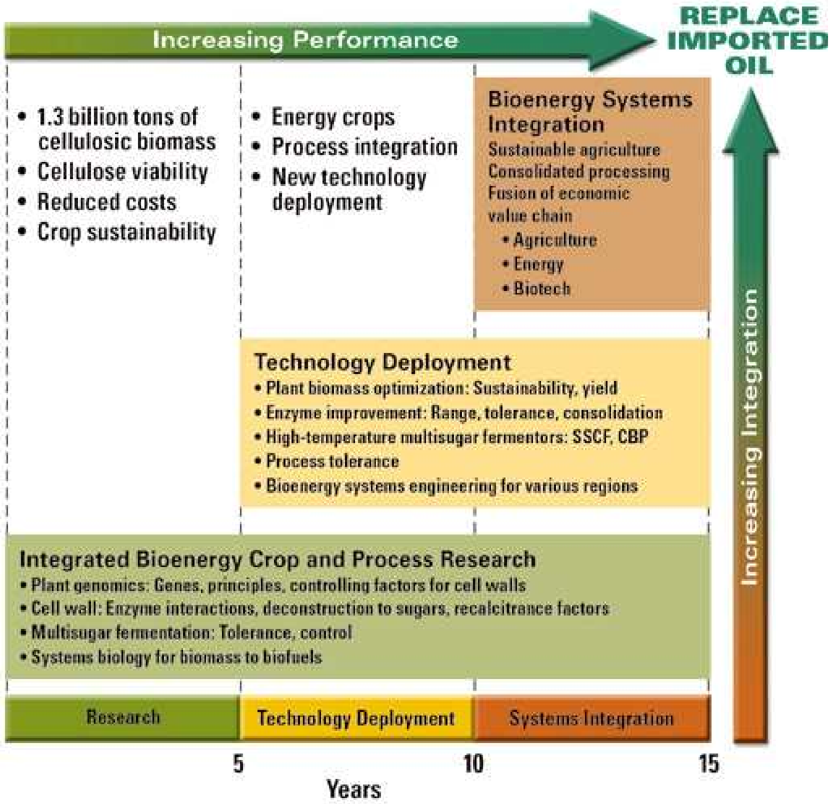

Figure 4. Phased development of bioenergy systems over the next 5 to 15 years (US DOE, 2006a). SSCF: simultaneous saccharification and co-fermentation; CBP: consolidated biomass processing.

high yield and robustness will be essential in contributing to achieving energy security.

While large deployment may not occur before ten years (US DOE, 2006a; BIOFRAC, 2006; FNR, 2006), or even fifteen years (Möller et al., 2007), several pilot and demonstration plants have already been built worldwide. Some twenty of such plants have been implemented since 1985 in the US, Canada, Brazil, Europe and Japan and about a dozen cellulosic ethanol commercial plants were being developed in 2007–2008, essentially in the US (Solomon et al., 2007) or under discussion in Canada and China. Steam pre-treatment with the addition of a catalyst for hydrolysis and improved enzymatic digestibility is the closest technology to commercialisation and has been widely tested in pilot-scale equipment (Hahn-Hägerdal et al., 2006). Considering the state-of-the-art technology in 2006, an estimated capital investment for a 220-million-litre cellulosic ethanol plant would approximate US$300 million, with the largest capital components for feedstock pre-treatment (17%) and simultaneous saccharification and fermentation (15%), and energy utilities (36%). The production cost could then approximate US$0.57 per litre with 40% related to the annualised capital charge and 46% to the feedstock and other raw materials (Solomon et al., 2007). In another estimate, production and transport of feedstock would represent about 21% and 26% of the total annual plant costs, respectively (Kaylen et al., 2000). In recent simulations, production costs, mostly based on the laboratory scale, range from 0.28 to 1.0 US$ per litre of cellulosic ethanol (Hahn-Hägerdal et al., 2006). In 2006, the production cost of dry mill ethanol from maize was US$0.44 per litre (Balat et al., 2008).

Scientists mostly argue that the technology is not mature yet for commercial production (Solomon et al., 2007; Cardona and Sanchez, 2007), whereas some industrials may be ready to take the chance. Still, all agree that tremendous increase in production volumes is the determinant techno-economic factor to reach commercial viability. About 86% of operating costs appeared to be proportional to the size of the plant (Kaylen et al., 2000). A drastic increase in production volumes and an "on-site" enzyme production, provided with governmental funds in the first development phase, appear to Murray Burke[14], Vice President and General Manager of the SunOpta BioProcess Group, the essential challenges to reach commercial production. As costs are highly linked to feedstock, whose price is volatile, diversification of feedstock, maximisation of ethanol yields and optimisation of the use or commercialisation of co-products must also be achieved. Current pilot plants can produce a few million litres a year, possibly integrated with an ethanol from-grain plant, which can be a near-term solution (Hahn-Hägerdal et al., 2006), but a ten-fold capacity increase

[14] Speech at the Platts Cellulosic Ethanol Conference in Chicago on October 31, 2006.

appears to be necessary for a complete switch of these plants to cellulosic ethanol plant.

Whereas lignin cannot easily be converted through biochemical processes, it can be burnt. Therefore, thermochemical processes are especially more effective in the case of plants with a high content of lignin, up to 30–35% of the biomass in some plants (Fulton et al., 2004; Möller et al., 2006). The main criteria for biochemical conversion of lignocellulosic feedstock are the quantity of sugars and the structure of the lignocellulose; in the case of thermochemical conversion, the main criteria are rather the biomass' bulk density, moisture and ash contents, and the calorific value (Möller et al., 2007). In a rough overview, agricultural residues and grasses with intrinsically higher sugar content and lower lignin content are generally more suitable for enzymatic conversion, whereas dense woody biomass with higher amounts of lignin and lower amounts of ash is comparatively more oriented toward thermochemical conversion. Ash can indeed lead to the slagging or fouling of heat-transfer surfaces during gasification. However, improvement of current technologies will notably permit one to reach efficient conversion ratios for a mix of the cheapest and most available feedstock within the supply area of an implemented technology (Worldwatch Institute, 2007).

The thermochemical pathway is referred to as Biomass to Liquid (BtL) as an analogy with the conventional fossil Gas to Liquid pathway (GtL). Nowadays, 8% of the syngas produced worldwide is converted into transportation fuels through GtL processes; the overall production of syngas corresponding to almost 2% of the total worldwide primary energy consumption. Thereby, thermochemical technologies are well developed but have to be adapted to biomass feedstock in qualitative terms as well as in terms of plants' scale, considering that biomass availability might appear to be a limiting factor (Van der Drift and Boerrigter, 2006). The core process is gasification, that involves using heat to break down the feedstock molecules and produce a synthetic gas or syngas (also called "bio-syngas" when biomass is the feedstock), and whose compound-mix of hydrogen, carbon monoxide and dioxide, water vapour, methane and nitrogen varies depending on the process (Fulton et al., 2004).

There are two major types of gasifier that were selected because of their high efficiency in producing H_2 and CO, although they still produce a different ratio of gases at different temperatures and with a differing level of cleanliness (ITDG, 2000; Van der Drift and Boerrigter, 2006). The fluidised bed gasifier, typically operated at 900 °C, has already been developed and demonstrated to produce heat and power from biomass. It requires, though, a catalytic reformer downstream to treat the produced gas so that it can fulfil the requirements to be converted into biofuels. On the contrary, the entrained flow gasifier, typically operated at 1300 °C, makes it possible to produce syngas without a catalytic reformer but needs energy-intensive pre-treatment of the biomass, such as torrefaction or pyrolysis, in order to reach a sufficient conversion rate. Torrefaction at 250–300 °C or flash/slow pyrolysis at 500 °C both turn solid biomass into a bio-coal or a bio-oil/char, respectively, that can be easily transported and fed to the gasifier. In all processes, syngas has to be further conditioned via gas cleaning and the H_2/CO ratio adjusted to be fed to a synthesis reactor and converted into final biofuels, such as Fischer-Tropsch diesel and naphta (basic gasoline), dimethyl ether (DME), methanol, mixed alcohols or hydrogen. Hydrothermal upgrading (HTU) is another process that makes it possible to transform biomass into a "biocrude" liquid by dissolving the cellulosic materials in water under high pressure but relatively low temperature. Bio-oils produced via pyrolysis or HTU can be subsequently upgraded into diverse hydrocarbon liquids and fuels (Fulton et al., 2004).

The diverse options are not incompatible; fluidised bed gasification can even occur as pre-treatment to feed an entrained flow gasifier. Choices are guided by the type of available biomass and the desired biofuels. Research projects are currently focusing on improving the pre-treatments, adapting the scale, and developing integration options such as polygeneration and mature biorefinery concepts to make the processes economically viable. Polygeneration refers here specifically to the up-value of all produced gases, and in particular of methane used as biogas to provide heat and power. Mature biorefinery (Fig. 5) is the combination of biochemical and thermo-chemical treatments that will permit one to produce as much biofuel (54% ethanol from sugars, 10% FT-diesel and 6% FT-gasoline from lignin and other residues) and heat/power (5% electricity also from lignin and other residues) as possible from lignocellulosic biomass (100%) with a minimum of energy input (21% captured for process energy or lost and <5% agricultural inputs: e.g. farming costs, feedstock transport) (US DOE, 2006a).

The frontier between 2nd and 3rd generation biofuels is conceptual and is not due to differences in biomass feedstock or radically new conversion processes. Still further technological breakthroughs will be needed to permit the economic viability of completely integrated biorefinery complexes, as well as technological revolutions in the transportation sector to introduce hydrogen as a competitive fuel for automotives. Hydrogen (H_2) is a fuel whose combustion produces only water. Although water vapour is the most significant greenhouse gas, its equilibrium in the atmosphere seems to be ensured by the natural water cycle. Hydrogen has been used by the aerospace industry since the 1960s and is nowadays especially used in the petrochemical industry to make ammonia fertilisers, to upgrade lower quality fractions in the refining of fossil fuels, and also to produce glass, lubricants, refined metals and processed foods (Zeman, 2007). According to Shell, the world market for distributed and centralised hydrogen is estimated at approximately 45 million tons per year. However, hydrogen is not to be found in nature under this diatom form and has to be produced from hydro-carbonates or water, requiring considerable energy inputs.

Hydrogen is designated as a 3rd generation biofuel, when it is produced from biomass via the thermo-chemical processes described above. However, this term would not be appropriate when talking about hydrogen coming from the conversion of fossil fuels, even if the processes ensured the storage of all emitted greenhouse gas during the conversion. 95% of today's hydrogen is produced from fossil combustible, most commonly methane (Demirbas, 2007), via steam reforming

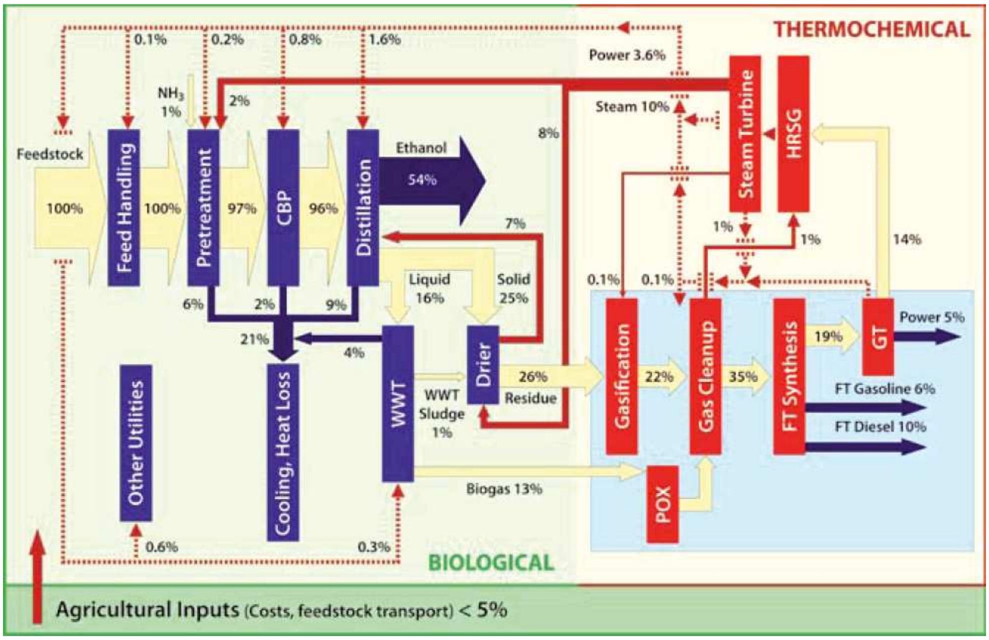

Figure 5. Mature biomass refining energy flows: example scenario in US DOE, 2006a. "Envisioning Mature Biomass Refineries," presented at First International Biorefinery Symposium, Washington, DC (July 20, 2005). Dotted arrows from above indicate energy inputs needed to run machinery.

that releases CO_2 into the atmosphere. Production via water electrolysis is three to four times more expensive and has a low energy yield (CEA, 2004). Hydrogen from renewables for fuel cell-driven vehicles might be a long-term solution, but its introduction needs breakthroughs in technology and cost and would require intermediate steps, to make a gradual growth of both fuel availability and number of vehicles possible. An effective intermediate step will be the use of hydrogen as a component in fuel production processes from biomass. This is applicable for today's fuel routes via synthesis gas, but will also be a serious option for future biorefineries (BIOFRAC, 2006). The development of a profitable hydrogen chain will take longer, especially considering the gas' inherent limit in terms of compression and storage on board. Although hydrogen contains three times as much energy as gasoline per unit weight, 4.6 litres of hydrogen compressed at 700 bars are needed to substitute 1 litre of gasoline (CEA, 2004). Moreover, as it is a flammable very small molecule, it requires specific hydrogen-proof material to be stored and transported. Currently hydrogen transportation is 50% more expensive than natural gas transportation, notably because one volume unit of hydrogen contains three times less energy than the same volume unit of natural gas (CEA, 2004).

Considering the risks and following costs implied in the development of new biofuel chains, industries' investments are significantly subordinated to the commercial perspectives that global policies underpin. These policies tend to respond to global issues and inevitably affect trade, as economic incentives often appear as efficient levers to reach targets.

4. POLITICAL AND ECONOMIC FRAMEWORKS

4.1. Climate change and greenhouse gas[15] emission trends

Lately biofuels have been fostered worldwide in a double context of energy insecurity and climate change. Except for a few exceptional cases, such as the Brazilian Pró-álcool Programme launched in 1975, it was not until the awareness of the risks associated with the depletion of fossil resources was drastically raised that biofuels and other renewables were widely given attention as real potential energy sources. Since the late 1980s, the more explicit the conclusions of the Intergovernmental Panel on Climate Change on the reality of climate change and the impact of anthropogenic greenhouse gas emissions have become, the more concrete the international policies and instruments to promote renewables have appeared. Needs for action and cooperation have been expressed within the framework of international agreements; such as the Framework Convention on Climate Change (UNFCCC) in 1992 and the Kyoto Protocol in 1997. Although they might not have federated enough stakeholders, notably the Kyoto Protocol which only entered into force in 2005 without some of the main CO_2 contributors, they gave way to the establishment of effective frameworks and national action plans.

The global average surface temperature on the Earth increased about 0.7 °C between the late 1800s and 2000, with a rate of about 0.2 °C per decade (IPCC, 2007) in the past three decades. However, taking into account the effects of

[15] By default examining greenhouse gas emissions includes "six" gases: CO_2, CH_4, N_2O, SF_6, PFCs and FCs.

orbital variations on climate, absent human influence, the natural trend would be toward a cooler climate, as peak warmth of the current interglacial period (Holocene) occurred 8–10 thousand years ago. Examination of prior interglacial periods reveals a strong correlation between the CO_2 and CH_4 concentrations in the atmosphere and temperature records. Nevertheless, in the past the temperature changes usually preceded the changes in gas concentrations. Today, anthropogenic greenhouse gas emissions are overwhelming and have reversed the order so that greenhouse gases are driving temperature increases. The climate system has not come to equilibrium with today's climate forcing and more warming is "in the pipeline". "Humans now control global climate, for better or worse" (Hansen, 2006). In other words, the IPCC stated in its last report: "Most of the observed increase in global average temperatures since the mid-20th century is very likely (probability >90%) due to the observed increase in anthropogenic greenhouse gas concentrations. Discernible human influences now extend to other aspects of climate, including ocean warming, continental-average temperatures, temperature extremes and wind patterns" (IPCC, 2007). Pre-industrial global atmospheric concentrations of CO_2, N_2O and CH_4 have increased markedly as a result of human activities since 1750[16] and now far exceed pre-industrial values determined from ice cores spanning many thousands of years.

Global increases in CO_2 concentration are due primarily to fossil fuel use and land-use change, while those of CH_4 and N_2O are primarily due to agriculture. If CO_2 emissions continue to increase by 1.5 to 2% per year, doubled CO_2 will be reached in approximately the year 2050. Encompassing the whole range of the six IPCC emission scenarios[17] from the lowest to the highest emissions, global warming could reach 1.8 to 4 °C by 2100 (IPCC, 2007). A global warming of 2 to 3 °C over the pre-industrial temperature would already "make the Earth a different planet" (Hansen, 2006). As a very critical issue, sea level rise illustrates how climate change can lead to exponential and irreversible impacts due to accumulation phenomena and positive feedbacks. IPCC scenarios give estimates of a sea level rise between 38 cm and 59 cm by the end of the 21st century relative to 1980–1999, due mostly to thermal expansion and excluding future rapid dynamical changes in ice flow. There is still no consensus on the long-term future of the ice sheet or its contribution to sea level rise. It is not possible to say how long it would take for sea level to change as feedbacks can lead to highly non-linear responses, nevertheless "it is almost inconceivable that under the business-as-usual scenario climate change would not yield a sea level change of the order of meters on the century timescale" (Hansen, 2007). Given the populations in 2000, a sea level rise of 6 m would displace 35 million inhabitants throughout the world and trouble is brewing for many species.

The distance that climate zones have moved so far is small, but the rate of movement of isotherms is now pole-ward at 50 km per decade and will double this century if we follow the business-as-usual scenario, surely causing the extinction of lots of species (Hansen, 2006). The IPCC stresses that: "Continued greenhouse gas emissions at or above current rates would cause further warming and imply many changes in the global climate system during the 21st century that would very likely be larger than those observed during the 20th century.[. . .] Sea ice is projected to shrink in both the Arctic and Antarctic under all scenarios[18]. It is very likely that hot extremes, heat waves and heavy precipitation events will continue to become more frequent. Even if the concentrations of all greenhouse gases and aerosols had been kept constant at year 2000 levels, a further warming of about 0.1 °C per decade would be expected" (IPCC, 2007). An alternative scenario aims at limiting the CO_2 peak at 475 ppm in 2100 before it should slowly decline thereafter and also requires a reduction of non-CO_2 forcing gases in order to hold warming to less than 1 °C. The 500 ppm scenario could make it possible to hold warming to less than 2 °C. From today's perspective, the 2 °C target is only achievable if global emissions are reduced below 10 $GtCO_{2eq}$ yr^{-1} in the longer term, meaning more than halving the 1990 level. In 2004, 51 $GtCO_{2eq}$ were added to the atmosphere, and the rise in 1990 emissions alone also produced an additional annual steady flow of 39 $GtCO_2$ due to climate time-lagged response to greenhouse gas emissions. If this development continues, it will be impossible to stay within the aforementioned limit for temperature increase (Fischedick et al., 2007); another decade of business-as-usual would eliminate the Alternative Scenario (Hansen, 2006).

Together, the 25 countries with the largest greenhouse gas emissions account for approximately 83% of global emissions. The largest emitter is the United States, with 21% of global emissions, followed by China with 15%. It follows that most of the remaining countries contribute little to the build-up of greenhouse gases in the atmosphere; 140 countries contribute only 10% of annual emissions (Baumert et al., 2005). The largest percentage increase since 1976 occurred in 2004, when more than 28 $GtCO_2$ were added to the atmosphere from fossil fuel combustion alone. Emission growth rates are highest among developing countries, where collectively CO_2 emissions increased by 47% over the 1990 to 2002 period. Among the major developing country emitters, growth was fastest in Indonesia (97%), South Korea (97%) and Iran (93%). During the same period, emissions also increased mainly in Canada (+20%) and Australia (+22%), whereas emissions in most developed countries did not change. During the 2003–2004 period, the CO_2 growth of 50% in China accounts for more than half of the worldwide CO_2 increase.

The Kaya Identity[19] model (Fig. 6) gives some clues to understanding the energy-related CO_2 emissions by using four

[16] Pre-industrial concentrations/in 2005: CO_2 (280/379 ppm); N_2O (270/319 ppb); CH_4 (715/1774 ppb), IPCC, 2007.

[17] SRES: special report on emission scenarios IPCC.

[18] Sea ice melting does not directly cause sea level rise like ice on continents; however, it can lead to the extinction of species that rely on these relatively scarce habitats. It also contributes to ocean thermal expansion.

[19] Model developed by the Japanese energy economist Yoichi Kaya in Environment, energy, and economy: strategies for sustainability, co-authored with Keiichi Yokobori as the output of the 1993 Tokyo Conference on Global Environment.

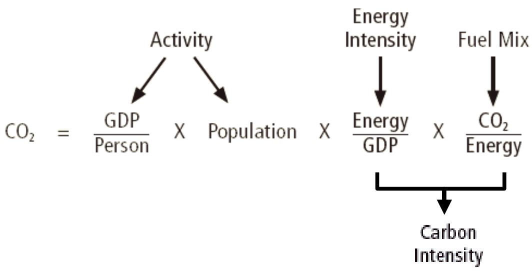

Figure 6. Kaya Identity model.

factors. Emission intensity or "carbon intensity" is a function of "energy intensity" and "fuel mix". Energy intensity reflects both the level of energy efficiency and the overall economic structure of a country. An economy based on heavy industrial production, for instance, is more likely to have higher carbon intensity than one where the service sector is dominant. However, energy-intensity levels are not well correlated with economic development. In several countries, it can be seen that declines in intensity were accompanied by significant increases in GDP, leading to increases in absolute CO_2 levels. The most notable case is China, where the effect of significant intensity declines, although China still heavily relies on the coal industry, was more than offset by substantial GDP growth. Likewise, the US decline in carbon intensity (17%) was offset by increases in population and GDP, giving a significant greenhouse gas emission growth in the US of 13% over the 1990 to 2002 period.

Whereas in agrarian economies with little heavy industry or energy production, land-use change, especially tropical deforestation, represents a larger share of CO_2 emissions, in a majority of countries, economic growth has, finally, the strongest influence on emission levels. This is the case in countries as diverse as the US, India, Australia and Iran (Baumert et al., 2005). Given the diversity of large emitting countries, it is simply not possible to adequately address the climate change problem without engaging both developed and developing countries, while adapting mitigation instruments to the specificity of influencing factors in the diverse countries. Fixed emission "caps" in particular may be impracticable in developing countries where economic growth is robust. Furthermore, differentiated per capita greenhouse gas emission targets rather than absolute emissions would reduce the effects of population growth on the commitments of Parties. Total emissions of CO_2 in 2004 show, for instance, China (3.7 tCO_2/capita) being far below the US (19.6 tCO_2/capita) and EU-27 (8.7 tCO_2/capita) (EEA, 2008). Also, projections of carbon intensity tend to exhibit less uncertainty than absolute emission forecasts (Baumert et al., 2005).

Between 1990 and 2004, European global greenhouse gas emissions had decreased from most sectors, particularly energy supply, agriculture and waste management; except from transport, which increased by nearly 26%. Had transport sector[20] emissions followed the same reduction trend as in society as a whole, total European Union-27 greenhouse gas emissions during the period 1990–2005 would have fallen by 14% instead of 7.9% (EEA, 2008). Only Germany, France and Portugal have managed to stabilise transport emissions in recent years. Nevertheless, EU-15 greenhouse gas emissions from transport are still expected to increase a further 35% above 1990 levels by 2010 if only existing policies and measures are used (EEA, 2006a). The transport sector represents the most significant climate policy challenge at two levels. First, transport contributes the lion's share of the emission increase of the European Union and in spite of the voluntary agreement to reduce the carbon content of travel for new vehicles, there does not appear to be in the near future a technological solution of a magnitude that could offset the effect of increased traffic and increased onboard equipment on CO_2 emissions. Second, further efforts to mitigate emissions in other sectors will be difficult to accept if governments do not undertake meaningful efforts in the transport sector (Barbier et al., 2004).

In the EU-25, despite an annual 1 t decrease in the average greenhouse gas emissions per capita of CO_{2e} between 1990 and 2004 and some successful decoupling of greenhouse gas emissions and economic growth, total EU-23 greenhouse gas emissions rose in 2004 by 0.3% compared with 2003 and were 5% below the 1990 level, the highest level since 1997. With existing policies and measures, and without additional ones, EU-23 greenhouse gas emissions are projected to keep on increasing and to be 2.1% below the 1990 level by 2010, meaning that the EU-15 Kyoto commitment of 8% emission reduction from this base-year level by 2008–2012 would not be reached, although the eight new member states had in 2004 emissions of only 76.8% of those in 1990 (EEA, 2006a).

At the UNFCCC meeting in Bali, December 2007, representatives of 180 countries agreed on a 'Bali roadmap' with the aim to achieve by the end of 2009 a global post-2012 climate change agreement to limit emissions, and address other issues such as adaptation to climate change, after the end of the Kyoto Protocol commitment period (2008–2012), whose targets will not be achieved. It should include both developed and developing countries, but with the largest emission reduction effort expected by the developed countries (indicatively in the range of 25 to 40% emission reductions by 2020 from 1990 levels). The European Council agreed in March 2007 on an integrated energy and climate change strategy. It endorses an EU objective of a 30% reduction in greenhouse gas emissions by 2020 compared with 1990 levels, provided that an international agreement can be reached with other industrialised countries. Without such an agreement, the EU would still pledge to a firm independent commitment to achieve at least a 20% reduction (EU, 2007). The EU Commission proposed to split the overall emissions reduction target into two: one for the sectors covered by the European Emissions Trading Scheme (ETS) and one for the non-trading sectors in which transport is included (EEA, 2008).

[20] The transport sector presented here consists of road transportation, domestic civil aviation, railways, national navigation and other transportation. It excludes emissions from international aviation and maritime transport (which are not covered by the Kyoto Protocol or current EU policies and measures). Road transport is by far the biggest transport emission source.

4.2. Biofuel-related policies

4.2.1. European policies

In Europe, several directives have been released; notably in 1997, the Energy White Paper and Action Plan, and the Green papers on Energy Supply (2000) and Energy Efficiency (2005). These policy instruments notably set indicative objectives in terms of consumption of renewables, e.g. on the share of green electricity (EU, 2001), the promotion of biofuels or emissions trading (EU, 2003), and the use of waste and disposal (2005). In parallel, other directives were released, which notably deal with energy efficiency improvement, economic incentives and eco-labels. However, none of the given targets were binding ones and the results have so far not been convincing. The White Paper (CEC, 1997) on the share of RES in total energy had proposed a common framework for action aiming to achieve the indicative objective of 12% for the contribution of renewables to the EU gross inland energy consumption by 2010, i.e. to double the share of renewables compared with 1997, including a tripling of biomass use. In 2003, the total amount of renewables used averaged only 6% of the EU gross inland energy consumption (EU DG-TREN, 2006a), about two-thirds of this contribution coming from biomass, i.e. 4% of EU total energy needs (CEC, 2005). Even if renewables consumption can widely differ between Member States, the challenge to reach the global objective remains entire; only 7% of the necessary growth of bioenergy production has been achieved globally (Fagernäs et al., 2006). In 2005, the indicative target[21] of 2% market share for biofuels stated in the EU Council Directive on "Biofuels" (EU, 2003) was not reached; biofuels apparently merely attained 1.4% of market share within the EU-25 (EU DG-TREN, 2006b). This share was better than in 2003 (0.6%) but if this trend continues, the 2010 target of 5.75% share will not be achieved: the forecast indicates a 4.2% share by 2010 (CEC, 2008).

Considering the need for a drastic reduction of greenhouse gases from transport and the still very low incorporation of biofuels, the European Union has decided to put into force a new directive that fixes mandatory targets, as was agreed during the European Union Summit in March 2007: 20% of the global energy consumption of the European Union has to be renewably sourced, including a minimum binding target of 10% within the transport sector (only consumption of gasoline and diesel are considered) for each Member State by 2020 (CEC, 2008). This directive proposal was published in January 2008 and should be followed up by concrete Member States' action plans by the end of March 2010. It will replace the former 2001/77/CE and 2003/30/EC directives after the 31st of December 2011. As a further incentive for investors, the Directive indicates that the share contribution by 2nd generation biofuels to the 10% binding target would count for double in comparison with the other biofuels (Article 18). For biofuels and other bioliquids, the directive sets up three conditional criteria of sustainability for their production, so that their consumption can be taken into account to fulfil the binding target and allow financial support (Article 15), although no sanction has been planned for non-compliance with these criteria or the non-fulfilment of both targets:

- a minimum saving of 35% of greenhouse gas emissions compared with the substitute fossil fuels,
- biomass feedstock must not be produced on soils within ecosystems considered to have a high value in terms of biodiversity: i.e. undisturbed forests, protected areas and specific permanent grasslands that shall be geographically identified by the Commission,
- biomass feedstock must not be produced on soils with high organic carbon contents: specific humid areas, notably virgin peat soils, and forests wider than 1 ha with canopy covers superior to 30%.

Concerning agricultural feedstock, supplementary environmental criteria of the European regulation related to good agricultural practices (CE 1782/2003 Article 5 and Annex III point A) remain applicable. Member States shall require economic operators to show that the environmental sustainability criteria set out have been fulfilled. In particular, the Directive stipulates the method to calculate the greenhouse gas emissions throughout the production chain and states that emission reduction due to the co-products shall be handled either by system extension in the case of co-generated electricity from agriculture, or by energy allocation[22] in all other cases. It also gives minima of emission reductions for each biofuel chain (Annex VII) that shall serve as reference.

However, the method does not specify how to take into account the N_2O emissions from the agricultural phase, and cannot at the Community level take into consideration the regional variability (see part 5). Furthermore, no independent certifying authority will be involved in the control of the respecting of the criteria, and biofuels are not included in the guarantee of origin system that will ensure the traceability of electricity, heating and cooling produced from renewable energy sources (Article 6–10). It is also mentioned that it would be technically and administratively unfeasible to apply EU environmental requirements for agriculture to biofuels and other bioliquids from third countries. In fact, since the proposal does not include any derogation for countries where the situations are constrasted with regard to continuous forest, peat soil or grassland resources as well as to certification access, establishing these biofuel sustainability criteria at a multilateral level could be perceived as a discriminatory measure according to the regulation of the World Trade Organization (Pons, 2008). Much remains to be done in order to establish multilateral agreements between the Community and third countries defining international standards of sustainability criteria, the greenhouse gas emission calculation method and certification control.

[21] Minimum indicative targets from the European Council Directive 2003/30/EC of 8 May 2003: 2% in 2005 and 5.75% in 2010 share of biofuels of all petrol and diesel for transport purposes placed on the market calculated on the basis of the energy content. (about 3% and 8.6% for ethanol; 2.2% and 6.4% for biodiesel when calculated on a volume basis).

[22] Based on the low calorific values.

4.2.2. US policies

For the last twenty years, the US government has also been putting into force several policies related to renewables in a view to reduce its dependency on imported oil. Starting with the Energy Tax Act of 1978, the US government has continuously maintained national tax incentives to encourage ethanol fuel production and use. Increases in ethanol excise tax exemption about every two years during the 80s–90s and loan guarantees to build up production facilities have notably fostered the growth of domestic maize-based ethanol (McDonald et al., 2004). In 2000, the Biomass Research and Development Act directed the departments of Energy and Agriculture to integrate their biomass Research and Development and established the Biomass Research and Development Technical Advisory Committee (BTAC), which advises the Secretary of Energy and the Secretary of Agriculture on strategic planning for biomass Research and Development. In 2002, this Committee set up a challenging goal requiring biomass to supply 5% of the nation's power and 25% of its chemicals, and biofuels to meet 20% (10%) of transportation fuel consumption by 2030 (2020). These contributions would represent all together 30% of the current petroleum consumption (Perlack et al., 2005). The 2002 Farm Bill also established new programmes and grants that support increased use of biofuels and biobased products as well as advanced biorefinery development (US DOE, 2006a). The need to substitute MTBE[23], which has been banned[24] in a growing numbers of US States due to its toxicity in high blends (formerly 15% in California, for instance), has also contributed to underpinning ethanol blends. However, biomass currently accounts for merely 4% of total energy consumption (BRDI, 2006), and biofuels, mostly maize ethanol, for around 2 to 3% of domestic transportation motor fuels (Kojima and Johnson, 2005; US DOE, 2006a).

With the growing energy consumption, US dependence on imported oil has reached severe levels. Between 1984 and 2005, crude oil imports increased 194%. In 2005, about 65% of crude oil and petroleum products were supplied by imports, representing 30% of the total US trade deficit. The overall demand for transportation fuels has increased 19% in the past ten years, with the vast majority of this growth reliant on imported petroleum (BRDI, 2006). During the last three years, the Government has hence especially insisted, with the Energy Policy Act of 2005, the American Competitiveness Initiative and the Advanced Energy Initiative (2006), on providing an aggressive strategy for tackling long-term energy challenges (US DOE, 2006b) and has shown ambitious goals in terms of energy efficiency and diversification, i.e. the increase in domestic production of conventional fuels as well as the development of new nuclear power generation, hydrogen and renewables.

In terms of bioenergy, the Energy Policy Act of 2005 (EPAct), a US$14-billion national energy plan, notably aims at fostering research programmes and partnerships between industries and academic institutions in order to develop advanced processes in bioproduct production. In this sense, it includes amendments to the Biomass Research and Development Act, focusing on four new technical areas for research and development activities: (1) develop crops and systems that improve feedstock production and processing, (2) convert recalcitrant cellulosic biomass into intermediates that can be used to produce biofuels and products, (3) develop technologies that yield a wide range of bioproducts that increase the feasibility of fuel production in a biorefinery, and (4) analyse biomass technologies for their impact on sustainability and environmental quality, security, and rural economic development.

The "Initiatives" strategies also particularly emphasise the role of technology development and innovations. The President's Advanced Energy Initiative provided a 22% increase in funding for clean-energy technology research at the Department of Energy in two vital areas: 1. "Changing the way we fuel our vehicles"; and 2. "Changing the way we power our homes and businesses". In 2007, a total budget of US$150 million was allocated to the DOE to fund biomass research and help to reduce the costs of producing advanced biofuels and ready technologies for their commercialisation.

Finally, biofuel production objectives are also underpinned by renewable content requirements for motor vehicle fuels. Called Renewable (or Alternative) Fuel Standard (RFS), the EPAct's provision requires gasoline sold in the US to be mixed with increasing amounts of renewable fuel on an annual average basis, up to at least 28 billion litres per year of biofuels by 2012 blended into the nation's fuel supply. In 2007, the US produced about 24.2 billion litres of ethanol and 1.7 billion litres of biodiesel, four times more ethanol than in 2000 and 80% more biodiesel than the previous year (US Government, 2008). In December 2007, President G.W. Bush signed the Energy Independence and Security Act, which notably responds to his "Twenty in Ten" challenge, a regulation to reduce gasoline consumption by 20% in ten years. This Act includes a new Renewable Fuel Standard, requiring fuel producers to supply at least 136 billion litres of renewable fuel in the year 2022, but also a Vehicle Fuel Economy Mandate, specifying a national mandatory fuel economy standard corresponding to a gain of 17 kilometres per litre by 2020. Several states in the US have adopted biofuel blend mandates; Louisiana, Montana, New Mexico, Oregon and Washington states, for instance, require ethanol (mostly 10%) in gasoline and/or biodiesel (2 to 5%) in highway diesel fuel with effective dates in the future. California is, moreover, developing a Low Carbon Fuel Standard for transportation fuels with a goal to reduce the carbon intensity by at least 10% by 2020. US policies undoubtedly boosted the biofuel supply and were massively followed by the member states as illustrated by the Governors' Ethanol Coalition, that includes 32 member states out of 50, as well as international representatives from Brazil, Canada, Mexico, Sweden and Thailand (US DOE, 2006a).

Although biofuels were initially thought to contribute to lowering US energy dependence on imports, the US still

[23] Methyl tertiary butyl ether: fossil oxigenate additive to gasoline.

[24] Contamination of groundwater by MTBE due to leaking tanks is especially severe in the US. Despite the decision to phase it out, the quantities of MTBE used in the US have not decreased due to its technical advantages that actually help to produce a cleaner burning gasoline (http://www.acfa.org).

imported about 1.7 billion litres of ethanol in 2007[25]. Even if all maize grain grown in the US were converted into ethanol, it would have satisfied just about 15% of the transportation needs (US DOE, 2006a). To reach the 30% vision of the BTAC, one billion dry tons of biomass feedstock would be needed annually, from the potential dedicated 1.3 billion dry tons from forestlands and agricultural lands, provided that large-scale bioenergy and biorefinery industries, including cellulosic ethanol plants, exist (Perlack et al., 2005). Therefore, the government especially focuses on the development of cellulosic ethanol (1 litre of non-grain-based ethanol is counted as 2.5 litres of grain-based ethanol to fulfil the RFS). This focus was confirmed when, on May 5, 2009, United States president Barack Obama signed a presidential directive on developing advanced biofuels, i.e. 2nd and 3rd generation biofuels. Cellulosic ethanol could enable greater greenhouse gas savings, which appears to be a crucial means for the US to drastically lower its greenhouse gas emissions, while it is besides not particularly willing to commit itself within the international emission reduction targets plan.

4.2.3. Chinese policies

As Chinese economy and consumption levels boom, Chinese energy policy is likely to significantly affect the worldwide energy market and is fatally expected to play a growing and major role in greenhouse gas emissions. In 1975, China became a net oil importer. Today, it depends on coal for around 70% of its primary energy and the main role of coal within the energy structure will remain unchanged for a long time to come. Nuclear power and renewables account for about 7% of primary energy consumption, the rest comes from fuel oil used in the transportation sector, whose consumption is growing rapidly (SCIO, 2007). Considering that China's energy efficiency is about 10% lower than that of the developed countries, and its per unit energy consumption of energy-intensive products is about 40% higher than the advanced international level, priority is given to the up-grading and widening of the domestic energy-grid by implementing more efficient and cleaner technologies. The 11th Five-Year Plan for National Economic and Social Development of the People's Republic of China outlines that the per-unit GDP energy consumption by 2010 will have decreased by 20% compared with 2005, and the total amount of major pollutants discharged will have been reduced by 10 percent (SCIO, 2007).

In terms of renewable energies, national targets are to reach contributions of 10% and a further 15% of total energy consumption by 2010 and 2020 (SCIO, 2007). The Renewable Energy Law of the People's Republic of China[26] (09/11/2005, Article 16) as well as China's Energy Conditions and Policies (28/12/2007, pages 17 and 37) directly but briefly endeavour the production of biofuels. The primary option for renewables is hydropower. This is notably illustrated by China's National Climate Change Programme, indicating that current measures are expected to lower greenhouse gas emissions by 2010 by 500 Mt CO_2 thanks to hydropower, 60 Mt CO_2 thanks to wind, solar, geothermal and tidal energy and only 30 Mt CO_2 thanks to bioenergy, essentially for heat and power (NDRC, 2007).

Over the past two decades, China's vehicle market has been the fastest growing in the world (+12% each year, Latner et al., 2007). China's consumption of crude oil totalled 323 million tons in 2005, including net crude-oil imports of 119 million tons. Consistent with new car use, the annual average growth rate for gasoline and diesel consumption during the period from 1990 to 2004 reached 6.8% and 10.1%, respectively. Thus, China views biofuel as a necessary strategic component to reach independence of imported oil (Latner et al., 2006). The development of biofuels, started in the late 80s, led to the first ethanol production in 2002. In 2004, the first recorded ethanol production was 300 000 tons and it increased more than four-fold within two years to reach 1.3 million tons in 2006, and an estimated 1.45 million tons in 2007. Ethanol is primarily converted from maize (>80%). Biodiesel, which was not introduced in the development programme until 2006, is mostly produced from animal fat or waste vegetable oils.

The 11th Five-Year Plan for biofuels that had suggested an implementation plan leading to a production of 5.2 million tons of biofuels by 2010 was not approved for food security concerns (Latner et al., 2007). China is already a net importer in all the major edible vegetable oils, the largest importer in the world, and a net sugar importer (Latner et al., 2006, 2007). As ethanol already accounts for 40% of the industrial maize use, higher demand for ethanol could turn China from a net exporter of maize into a net importer (Latner et al., 2006). Therefore, the government focuses more on the use of other crops such as cassava, sorghum, and the use of feedstock grown on non-arable lands, notably cellulosic feedstock for ethanol and *Jatropha* for biodiesel. It has already launched an E10 mandate in nine provinces that will be expanded to some of the other thirteen provinces. It is not clear today whether the ambitious government's target of a biofuel share of 15% of total transportation fuels by 2020 [about 12 million tons of biofuel (Latner et al., 2006)] remains on the agenda, since the Plan has been rejected. A realistic target would be 3 to 4 million tons of biofuels by 2010 (Latner et al., 2007), but the Chinese government needs to draw up new policies to ensure that its biofuel targets can be achieved efficiently and economically, said a researcher within the Chinese National Development and Reform Commission (Stanway, 2008).

The European Union, the United States of America and China are major emitters of greenhouse gases, and therefore show a growing political will to reduce their emissions, notably by developing cleaner energy sources. Nevertheless, many other countries, even some minor polluters, are implementing national strategies and policies to develop biofuels playing a more or less important part within renewables development plans. This is notably the case in some South American countries such as Colombia and Peru, but also in Asian countries such as India, Japan and Thailand. Industrialised countries may emphasise more their role in greenhouse gas reduction and energy diversification, while developing countries promote biofuels especially as an opportunity

[25] http://www.worldwatch.org (25/04/2008).

[26] Online: http://www.ccchina.gov.cn/en/Public_Right.asp?class=17.

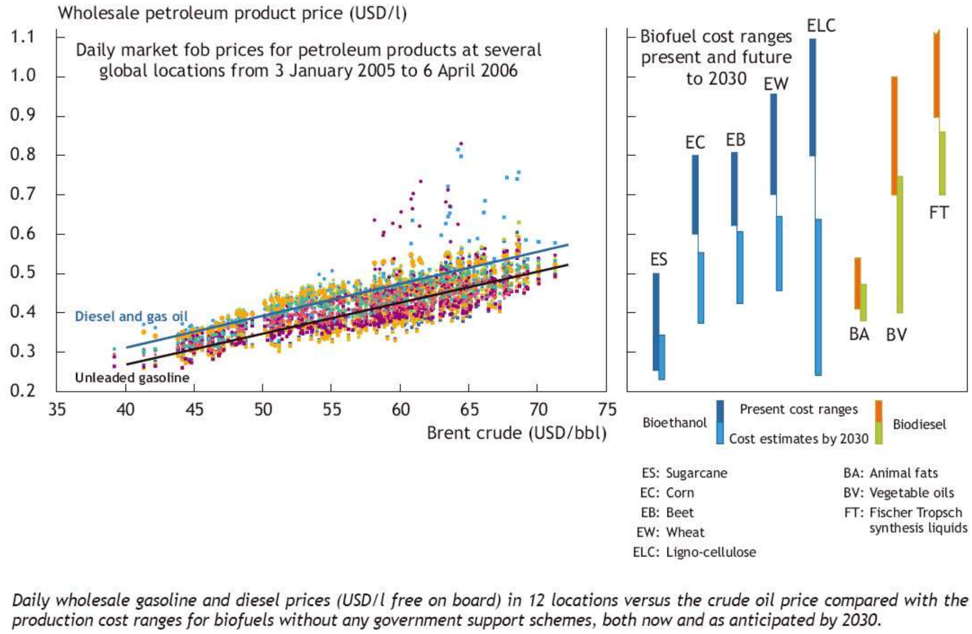

Figure 7. Biofuels current and prospected production costs without support schemes and petroleum products price evolution (IEA, 2007b).

to foster rural development and save foreign exchange. In all cases, biofuels are perceived as a means to contribute to energy security, when concerns are growing with the surge in oil price (Kojima and Johnson, 2005). However, the greatest barrier that has hampered biofuel large-scale commercialisation is their high cost of production compared with conventional fuels, two to four times higher (VIEWLS in pelkmans et al., 2006). Thereby, biofuel policies notably consist of implementing economic incentives to counterbalance high production costs and make biofuels competitive.

4.3. Economic incentives

Biofuels cannot be the panacea that petroleum has been for decades. The feedstock has to be produced (feedstock prices account for from two-thirds up to 90% of the total costs of 1st generation biofuels, Wiesenthal et al., 2007) and transformed, while petroleum just has to be looked for and exploited, which until resources started to become scarce, was largely cost-effective. Moreover, prices of agricultural commodities, especially those of crude materials, are highly volatile due to fluctuations in price inelastic supply and demand, meaning that a small shift in supply or demand results in a large price change. Supply may widely vary following climatic hazards and demand on such competitive markets, as agricultural commodities have only limited substitutes, and can be severely impacted by diverse factors, e.g. large purchase by governments (Clem, 1985). These fluctuations are further influenced by growing spill-over effects from one market to another, as global markets have become increasingly intertwined across all commodities and between commodities and the financial sector. Greater price uncertainty implies higher risk and growing speculations, which in turn can initiate a vicious cycle of even more enhancing price volatility. On the other hand, this uncertainty tends to limit opportunities to access credits and result in the adoption of low risk production technologies at the expense of innovation and entrepreneurship (FAO, 2007), which is notably critical in the development of second generation biofuels. Facing the high cost of biofuels, even Brazil's ethanol infrastructure model, that relies on an optimum combination of the very productive sugar cane and favourable climatic conditions, required huge taxpayer subsidies over decades before it could become viable (Xavier, 2007). Today, Brazil continues to maintain a significant tax differential between gasoline and ethanol (Kojima and Johnson, 2005).

Figure 7 gives an overview of the current and foreseen production costs of biofuels compared with petroleum products. Except for sugar cane-based ethanol and animal fat-based diesel, drastic cost reductions are still necessary for biofuels to become clearly competitive by 2030. Tax incentives, administered pricing, restrictive trade policies, credits and numerous other economic incentives are in force worldwide to underpin the production or consumption of biofuels, notably by making them artificially competitive.

Demand-side instruments, such as tax incentives and obligations (e.g. mandatory blends), are the most common mechanisms that have proven to be efficient in pushing biofuels onto the market (Wiesenthal et al., 2007). Tax incentives are tax provisions that grant special tax relief designed to encourage certain behaviours by taxpayers. Tax exemption on biofuels and higher excise taxes on fossil fuels permit one to compensate for the higher biofuel production costs (Tab. III) and create or enlarge a favourable price for biofuels relative to fossil

Table III. Production costs of ethanol and biodiesel and prices of petroleum-based fuel in major biofuel-producing countries, 2004. Data from OECD (2006).

Countries/Community	Ethanol	Gasoline	Biodiesel	Diesel
		(US$ per energy-equivalent litre*)		
US	0.82 (wheat) 0.45 (maize)	0.54 (with tax) 0.38 (without tax)	0.62 (vegetable oils)	0.57 (with tax) 0.37 (without tax)
Canada	0.85 0.50	0.68 0.40	0.50	0.68 0.39
EU-15	0.88 0.68	1.32 0.40	0.68	1.29 0.40
Poland	0.80 0.51	1.2 0.39	0.82	1.1 0.38
Brazil	0.35 (sugar cane)	0.84 0.39	0.65	0.49 0.38

Notes: * Biofuel prices accommodate differences in energy content. Ethanol is assumed to contain 0.66 the energy of 1 litre of gasoline, and biodiesel is assumed to contain 0.89 the energy of 1 litre of diesel. Note: when not specifically mentioned, the numbers follow the same order as for the US, i.e. 1st : wheat, 2nd : maize; and 1st : with tax, 2nd : without tax.

ones, providing a strong incentive for the consumer to prefer biofuels over fossil ones. Hence consumer acceptance was shown in a study of the United States General Accounting Office to be essential to the use of alternative fuels in the case studies of Brazil, Canada and New Zealand (US GAO, 2000). In the same study, it was also shown that the expected decline in ethanol use in the US, if tax exemption were eliminated, would be at least 50%.

In 2004, nine European Union member states had partly or completely detaxed biofuels: Austria, the Czech Republic, France, Germany, Italy, Lithuania, Spain, Sweden and the United Kingdom. All European Union Member States that achieved a high biofuel share had a full tax break in place and high fossil fuel tax levels. However, the reverse case does not seem true, which indicates that tax exemption is not a sufficient condition to reach a high share of biofuels (Pelkmans et al., 2006). Moreover, tax exemption implies budget losses for the governments, some 1140 million Euros in 2005 ($US 1419 million[27]) in Germany (Wiesenthal et al., 2007). These losses can be particularly critical in developing countries, where gasoline taxes are often a significant source of tax revenue more supported by high-income groups. Furthermore, tax expenditures aiming at favouring biofuels may fall under little scrutiny, while public expense on biofuels might need to be weighed against other social priorities (Kojima and Johnson, 2005).

Although tax incentives appear to be necessary to create and maintain a minimum biofuel demand, they are not sufficient to reach significant levels of biofuel consumption in most countries. In particular, ethanol tax incentives in the US, for instance, have failed in enhancing US energy security because they have not created enough usage to reduce petroleum imports and the likelihood of oil price shocks (US GAO, 2000). Thus, currently implemented schemes are mixed, i.e. some kind of tax incentives and/or obligations apply in parallel (Wiesenthal et al., 2007). In France, reductions of the interior consumption tax (Art. 265 bis A from the Duty Code 2006) by an average of 28 € ($US 35.2[24]) per hectolitre of biofuel blended with fossil fuels is combined with a supplementary tax (General Tax on Polluting Activities, TGAP, Art. 32 from the Finance Law 2005) on diesel and gasoline sales, which do not contain a minimum share of biofuels (Luneau and Fayet, 2007).

In Germany since 2007, a tax exemption system has been replaced by an obligation for fuel suppliers to provide a certain share of their total sales as biofuels. Obligations encompass mandatory blends, i.e. an obligation to add a certain % biofuels to fossil fuels, obligation to bring a certain quantity of biofuels on the market (e.g. the 10% share proposed in the European Union Directive CEC, 2008), and an obligation to bring a certain biofuel quantity on the market including a tradable renewable fuels certificate system (e.g. the Renewable Transport Fuels Obligation in the UK, the Green Power Certificates System in Flanders) (Pelkmans et al., 2006). In a broader sense, fuel standards, an authorised quota system for biofuel producers or filling station obligation (e.g. a mandate to fuel distributors to offer at least one renewable fuel in Sweden, Pelkmans et al., 2006) can also be perceived as obligation incentives, since they aim at introducing given quantities of biofuels on the market. Indeed, a major interest of obligation is the long-term visibility that they may offer to industries willing to produce biofuels, especially to those taking the risk to implement new technologies. This market prospect remains relative, though, as it depends on the governmental politics for the quota amounts (Pelkmans et al., 2006). Furthermore, in an obligation scheme, fuel suppliers will pay for the additional costs, meaning that they have an incentive to opt for the lowest cost biofuels, e.g. imported or low-blend biofuels. Fuel standards and low blends do not make any biofuel visibility possible, and obligations do

[27] Mean annual foreign exchange rates from the US Board of Governors of the Federal Reserve System annual databases.

not push industries to go beyond mandate targets (Pelkmans et al., 2006).

Finally, if high blends or certain technologies are to be promoted, neither obligations nor tax reductions are the appropriate instruments (Wiesenthal et al., 2007). It can be noticed that second generation biofuels will count for double compared with first generation ones, when considering US or European obligation systems, which in that case is a further incentive towards these biofuels. However, supply-side instruments may be more efficient in promoting specific biofuels. Moreover, while most of the initiatives in the biofuels field have been focusing on conversion and end-use sectors so far, there is a primordial need to support raw material producers to expand and secure feedstock supply in order to/and thereby be coherent with increasing biofuel share policies (European Commission, 2007). As an example, the E5 mandate in India was suspended in 2004, two years after its implementation, due to the lack of ethanol supply.

Supply-side instruments, mainly capital grants and feedstock support, have had limited success in pushing biofuels so far (Wiesenthal et al., 2007). Indeed, support for production facilities for 1st generation biofuels does not drastically impact the cost of biofuel due to the fact that capital does only represent a marginal part of the total cost in comparison with feedstock (0.01 € per litre with 10 million € support for a 15–20 million € investment to build a typical large biodiesel plant, Pelkmans et al., 2006). However, capital grants play an important role in fostering the development of the not yet mature second generation of biofuels, firstly as an incentive to build up demonstration plants, but also given that capital costs account for much more of the total cost when compared with 1st generation biofuels, some 60% and above (JRC/EUCAR/CONCAWE, 2006). The same pattern is observed with feedstock support. Although feedstock represents a large part of the total production costs, the energy crop premium indeed only lowers 1st generation biofuel costs by 0.01 to 0.04 € per litre. But as capital grants or loan programmes, feedstock subsidies appear to be an efficient instrument in order to support special types of biofuels (Wiesenthal et al., 2007).

In Europe, agricultural subsidies are defined by the Common Agricultural Policy (CAP). In 1992 following the Mac Sharry reform, liquid biofuels appeared as a way to compensate for the set-aside obligation that aimed at tackling the issue of overproduction and corresponded to 15% of the arable crops area in 1993–1994. In exchange, farmers were entitled to compensation payments (area and headage payments), and were allowed to grow non-food crops on set-aside lands (regulation n° 1765/92). In France, rapeseed methyl ester was favoured because it permitted the cultivation of the greatest area of set-aside lands for a given amount of public financial support considering its low yield per hectare (3.3 t/ha on average in 2005) (Sourie et al., 2005).

In 2003, the Fischler reform established "an income support for farmers", the single payment scheme (SPS), which replaces area and headage payments, cutting the link between subsidies and production. As a result, farmers can respond freely to increasing demand for energy crops (CEC, 2005). In particular, crops, which were eligible for direct payments only as non-food crops grown on set-aside lands, can from now on be grown on any kind of land. Moreover, in the past, only a limited range of energy crops could benefit from support, whereas this new reform has paved the way for farmers to grow more energy crops, including short rotation coppice and other perennial crops (CEC, 2005). Non-food crops including energy crops can be grown on set-aside lands under the condition that the use of the biomass is guaranteed by a contract between the farmer and the processing industry or by the farmer if the processing occurs on the farm (CEC, 2006a). In these cases, the 2003 reform also introduced a 45 € /ha premium for energy crops grown on non-set-aside land, the so-called Carbon Credit, with a budgetary ceiling of Maximum Guaranteed Area of 2 Million hectares subsidised in the EU (regulation1782/2003, amendment[28] in 2007). In 2006, this energy crop premium was already applied to some 1.2 to 1.3 Million hectares (Wiesenthal et al., 2007). In a further push to encourage the production of feedstock for renewable energy production, the Commission also proposed allowing the Member States to grant national aid of up to 50 percent of the costs of establishing multi-annual crops on areas on which an application for the energy crop aid has been made[29].

Finally, in November 2005, a major reform of the sugar regime was agreed. It aims at a progressive cut in price support of 36% over four years and the reduction of EU sugar subsidised exports from the current level of 7.6 Mt to the agreed Uruguay Agreement[30]'s limit of 1.4 Mt (OECD/FAO, 2007); meanwhile, "partial compensation" was introduced in the form of a direct decoupled payment. Sugar for the chemical and pharmaceutical industries and for ethanol production is excluded from production quotas. The lower sugar production quotas and the lower sugar price paid to EU farmers are likely to foster the production of sugar beet for ethanol, which is also eligible for the energy crop premium. The new US Farm Bill will also emphasise the role of agricultural subsidies to foster the development of biofuel with increased support to farmers. Despite the willingness of the former US President, G. Bush, to privilege direct payments upon coupled ones[31] or even the recent CAP reform, feedstock subsidies still imply market distortion.

As Mrs. Corre, Director General of the European Union of Ethanol Producers (UEPA), pointed out, a balanced trading framework is a pre-requisite to ensure the viability of the nascent industry and offer win-win opportunities, especially to developing countries. Also, biofuel producers need a self-running market with long-term visibility, and a 2–3-year perspective based on a yearly-tailored fiscal budget does not

[28] http://ec.europa.eu/agriculture/capreform/infosheets/energy_en.pdf.

[29] European Commission Press releases, IP/06/1243, Brussels, 22 September 2006.

[30] URAA, Uruguay Round Agreement on Agriculture in Marrakech, 1994.

[31] Communication of Charles E. Hanrahan, Senior Specialist at the Library of Congress in Washington, on 01/2008 at the Agroparistech.

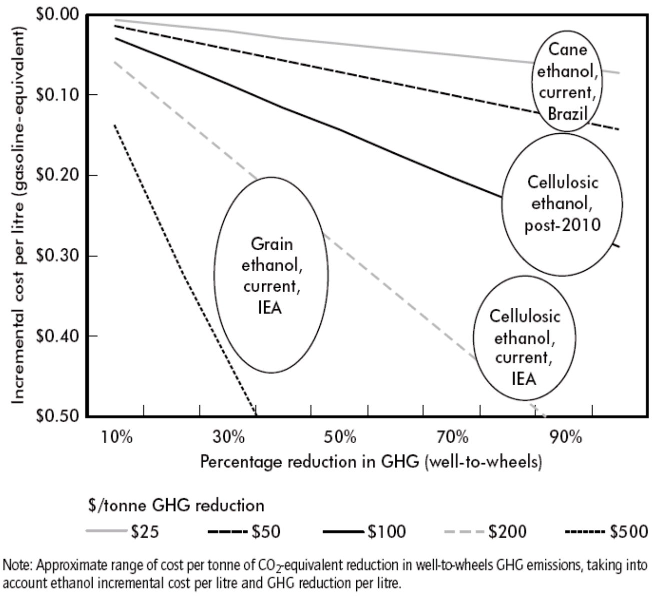

Figure 8. Biofuels incremental cost per ton of greenhouse gas reduction (Fulton et al., 2004).

offer that. Internalising the external benefits of fuel ethanol is the greatest help of all which could solve the problem (in EurObserv'ER, 2006). However, these externalities are not easy to quantify and to be given any value and the cost/benefit analysis also includes a lot of socio-economic factors that are in constant evolution, e.g. economic incentives, the cost of an oil barrel, etc. Taking the example of greenhouse gas emissions, which are given an economic value within the Emissions Trading Scheme (ETS), the assessment of CO_2 equivalent emissions throughout biofuel chains, first of all, still lacks precision (see part 5).

Transportation biofuels are not included in the current Emission Trading Scheme for CO_2 permits. The abatement cost of using biofuels to reduce greenhouse gas emissions, by substitution of fossil ones, is indeed very high due to their high production costs. Therefore, unless a given biofuel permits drastic greenhouse gas reductions with low costs (Fig. 8: Cane ethanol in Brazil, \$US/ton GHG reduction = \$US 25) the price of the ton CO_2 equivalent, i.e. of the tradable permit, would be too high within the current ETS "pricing engine" (Fig. 8: Grain ethanol in IEA countries, \$US/ton GHG reduction = \$US 200–500) and biofuels would not be a favoured solution for greenhouse gas reductions compared with green heat and power if permits were the only incentive (Fulton et al., 2004). This system makes emission reductions possible wherever abatement costs are the lowest, i.e. this trading scheme cannot lead to efficient greenhouse gas reductions where they are the highest as long as production costs are not significantly reduced.

Clean Development Mechanism (CDM) projects that involve an industrialised country buying carbon credit for a developing country, which uses the payment to produce biofuels or dedicated vehicles, etc., might be a better option to foster biofuels, depending on their greenhouse gas reduction potential. Also, reduction of tariff barriers within international trade for biofuels as environmental goods (under the classification of OECD, 2003 in Fulton et al., 2004) is an interesting instrument given the wide range of biofuels' potential and production costs worldwide. However, again, these opportunities are still hampered by the lack of data and agreement on real biofuels' environmental benefit.

Finally, the assessment of past biofuel success stories indicates that a portfolio of policy instruments, including supply- and demand-side instruments, is necessary to bring biofuels onto the market (Wiesenthal et al., 2007). This implies a need for interdisciplinary integration and harmonisation within ministries and governments at all levels. All over the world projected or implemented mandatory blends are on the agenda: the E10 mandate in Thailand and China, planned B2 and E10 mandates in Latin America, the possible ethanol blending mandate in Japan, the Brazilian mandatory B2 blend introduced in 2008 and to be increased to B5 in 2013, etc. However, these targets are far from being reached and illustrate a lack of policy harmonisation in terms of bioenergy strategy on a global level.

Furthermore, difference in blend standards can impact international trade as biofuel producers from different countries might not be supported by subsidies or taxes in the same way when considering various blend characteristics needed to get this help, as claimed by the European Biodiesel Board (EBB) in Bioenergy Business, 2007. This also stresses the need for harmonisation across several sectors and especially involving the automotive sector. Collaboration with car manufacturers has appeared crucial to ensure biofuel compatibility with engines, that was necessary to offer biofuel producers and consumers warranty (Wiesenthal et al., 2007). The introduction of policies on vehicle technology standards, so that all new vehicles would be compatible with a specific mixture of biodiesels, would permit one to lower the production costs of such vehicles and further incite biofuel consumption (Fulton et al., 2004). It would also pave the way to more flexibility and more coherence with other policies, which is lacking; as the example of the Spanish biofuel domestic surplus shows (Bioenergy Business, 2007).

Despite existing policies and measures, the major uncertainty factor, to decide which biofuel path should be fostered or not and to define better who should bear the additional costs, is due to diverging results on the energetic and environmental balances of biofuel chains. Calculations and results can drastically differ among studies and the lack of transparency behind hypotheses or the data quality also lead to some confusion. Therefore, growing doubts about the real ability of first generation biofuels to reduce overall greenhouse gas emissions and growing awareness of negative impacts of biofuel production on biodiversity, water and soil, point to the need for great caution in promoting biofuels further. Better knowledge of life-cycle greenhouse gas emissions from all energy uses of biomass, and strong sustainability criteria for biomass production, also addressing knock-on effects due to indirect land-use change, are needed to fully judge the benefits and limitations of biomass use (EEA, 2008).

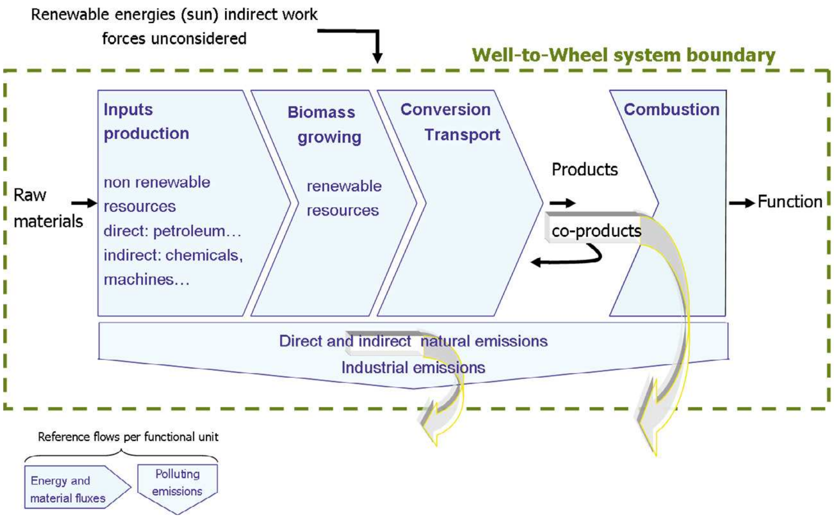

Figure 9. Biofuel LCA simplified well-to-wheel system.

5. BIOFUELS AND GREENHOUSE GASES

5.1. Assessing the environmental impacts of biofuels

5.1.1. Life cycle assessment of biofuel chains

5.1.1.1 The life cycle assessment tool

Life cycle assessment (LCA) is a holistic methodology developed in the 1980s[32], which assesses the potential environmental impacts of a product considering every step of the commodity chain from "the cradle to the grave". When comparing biofuels and fossil fuels, LCA appears to be an inescapable tool because the production of biofuels must be optimised, considering the environmental impacts throughout the whole commodity chain in order to avoid pollution trade-offs between ecological compartments or processing steps. This is, indeed, very important in "eco-design"; not to solve one environmental problem merely by shifting it to another stage in the product's life cycle (Guinée, 2002) and this is particularly crucial when it comes to decisions on national and international levels on global issues such as energy and climate change. The LCA methodology consists of 4 steps: "goal and scope definition", "inventory of extractions and emissions", "impact assessment" and "interpretation". An iterative approach should ensure that the system boundary and the inventory have been correctly adjusted, so that a comprehensive inventory of emissions makes it possible to correctly and completely characterise the selected impacts. A complementary sensitivity analysis then allows eliciting the weight of input data uncertainty and model assumptions on the final LCA results. Although the LCA tool has been standardised by the ISO norms 14 040 series (40/41, 43, 47–49, 1997/98, 2000/01[33]), there are several methods to apprehend the impacts and their characterisation, and no ready LCA adaptable to various cases. This firstly implies that the whole analysis has to be started from zero each time, but also that results might considerably vary between studies due to diverging background assumptions. LCA has been performed a lot during the last decade, notably in order to compare the environmental performances of bioenergy chains with one another or with fossil fuel chains. In this sense LCA can serve for decision-making but under some conditions on its construction that should lead to consensus on the LCA results.

Figure 9 shows a simplified scheme of a representative system boundary for a bioenergy chain from the extraction of raw materials to the combustion of the biofuel, the so-called Well-to-Wheel (WtW) system boundary. Delineating the system boundary is a decisive step of the "goal and scope definition" step of LCA, although it might evolve through iterative analyses of result sensitivity to assumptions. There is nowadays a global consensus on this WtW system boundary, although some divergences still appear, notably concerning the accounting of energy invested in farm machinery and infrastructure capital (Farrell et al., 2006). A distinction should be observed, though, between WtW assessments, that elicit the impacts of the fuel combustion, and Well-to-Tank (WtT) assessments, that assume total fuel combustion without further impact assessment. Renewable energies, upstream in the chain, are not included in the system boundary; essentially solar energy for photosynthesis, but also the so-called indirect energies such as human work, for example.

Within the system boundaries, from the first extraction to the last emission, all elementary flows are accounted throughout every step of the "tree of the product life cycle" in accordance with a quantified functional unit, i.e. the function that the studied product is to fulfil, e.g. to provide 1 MJ.

[32] By BUWAL, Bundesamt für Umwelt, Wald und Landschaft (Swiss federal office) and SETAC, Society of Environmental Toxicology and Chemistry (international scientific society).

[33] ISO 14040:2006 and ISO 14044:2006 replace the previous standards (ISO 14040:1997, ISO 14041:1999, ISO 14042:2000 and ISO 14043:2000). The new editions have been updated to improve the readability, while leaving the requirements and technical content unaffected, except for errors and inconsistencies.

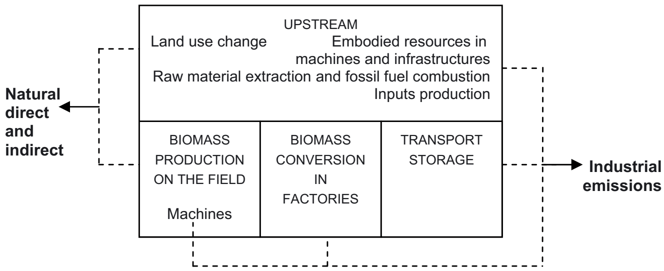

Figure 10. Natural and industrial emissions of a bioenergy chain (without combustion).

The "tree of the product life cycle" corresponds to the succession of the unit-processes. The blue boxes represent the main classes of processes within bioenergy chains, each one encompassing several unit-processes. For each unit-process, data are recorded on the inputs, the emissions, waste flows, and other environmental exchanges that are typically assumed to be linearly related to one of the product flows of the unit-process. All unit-process are linked through intermediate product flows, which makes the process system model linear with respect to the quantity of function it provides (Rebitzer et al., 2004). Inputs include raw materials and energy. Outputs are the products, the work or energy output and the polluting emissions. "Products" include the product of interest and all co-products; i.e. all the products, included biomaterials, waste or energy, that are concomitantly produced although all processes aim at optimising the production yield of the one product of interest.

A distinction can be made between co-produced materials that are directly generated from part of the feedstock such as straw or meal, and by-products such as glycerine or heat that are sub-products of other production processes (Malça and Freire, 2006). In this review all these "secondary" products are included in the "co-products". Direct emissions are emissions occurring during the production of the biomass due to natural biochemical and physical mechanisms within the soil-plant-atmosphere ecosystem. Indirect emissions can also occur on a wider space and time scale, following further reactions affecting the substances previously emitted, or upstream in the chain due to land-use changes. Indirect emissions also encompass methane emissions through fodder digestion by livestock. The term indirect emissions will be, in this review, reserved for these "natural second-order" emissions, whereas emissions through cultural operations, transport, conversion processes, storage, etc. will be referred to as industrial emissions (Fig. 10).

Energy ratios are a critical aspect of bioenergy chain assessments, since an important matter is to determine the fossil primary energy savings. Nevertheless, there is still a lack of consensus concerning the definition and designation of energy efficiency indicators to be used in a life-cycle perspective in particular to characterise renewable energy systems (Malça and Freire, 2006). The respective definition and use of among others "energy efficiency", "overall energy balance", "gross/net energy requirement" and "energy renewability efficiency" (Malça and Freire, 2006) will also be clarified when comparing bioenergy chain assessments.

5.1.1.2 Biofuels versus fossil fuels

Advantages of biofuels over fossil fuels depend on the environmental impacts that are considered. If e.g., savings of fossil resources as well as greenhouse gases are given the highest ecological importance, all biofuels compare favourably with their fossil counterparts if competition for other uses of the resource is not considered (Quirin et al., 2004), whereas taking into account the impacts such as acidification, eutrophication or ozone depletion reverses this trend. Indeed, these drawbacks are linked to intensive agricultural production that notably leads to nitrogen compound emissions responsible for those bad impacts on the environment. The relative advantages of biofuel chains between one another depend on the feedstock. Figure 11 illustrates the results of a comparative study of 109 biofuel chains by the IFEU, i.e. the ranges of greenhouse gas and primary fossil energy savings by substituting fossil fuels with the biofuels assessed in some 64 studies (Quirin et al., 2004).

Comparison on the basis of a unit MJ energy content of biofuels (Fig. 11a) makes it possible to compare all the chains, including biofuels from residues, whereas the comparison on a hectare basis (Fig. 11b) introduces a land-use perspective. Amongst 1st generation biofuels, bioethanol from sugar cane, molasses and sugar beet, as well as biogas from wastes (here compared with gasoline, but also advantageous compared with natural gas) show the best combined performances in terms of both greenhouse gas and primary fossil energy savings (Fig. 11a). Greenhouse gas emissions are partly linked to the combustion of fossil primary energy input throughout the chain; both savings are therefore connected and their ranges appear within similar orders of magnitude.

Due to the high energy content of oilseeds and less complex processes, relatively less primary energy input is necessary to produce a MJ of biodiesel or pure vegetable oil compared with a MJ of ethanol, whose distillation and dehydration are energy-intensive. In comparison, the primary energy input in refining gasoline or diesel is not significantly different between the two fuels. Therefore, diesel substitutes can save the most fossil energy. To a lesser extent, they also mostly emit less greenhouse gases than ethanol from starch crops and ETBE. Greenhouse gas savings vary a lot among biofuel chains and biodiesel roughly performs better than the worst ethanol chains but is worse than the best ethanol candidates. Ranges in energy and greenhouse gas savings of biodiesel chains are less connected than those of ethanol, since the conversion processes consume less energy. Greenhouse gas emissions depend therefore more on the feedstock types and the cropping systems. In the case of sugar crops, energy inputs are partly compensated for by higher yields per unit of agricultural input.

Further savings are also due to the co-products. This is notably the case with sugar cane, whose stalk or bagasse are burnt to produce energy input, or in the case of rapeseed

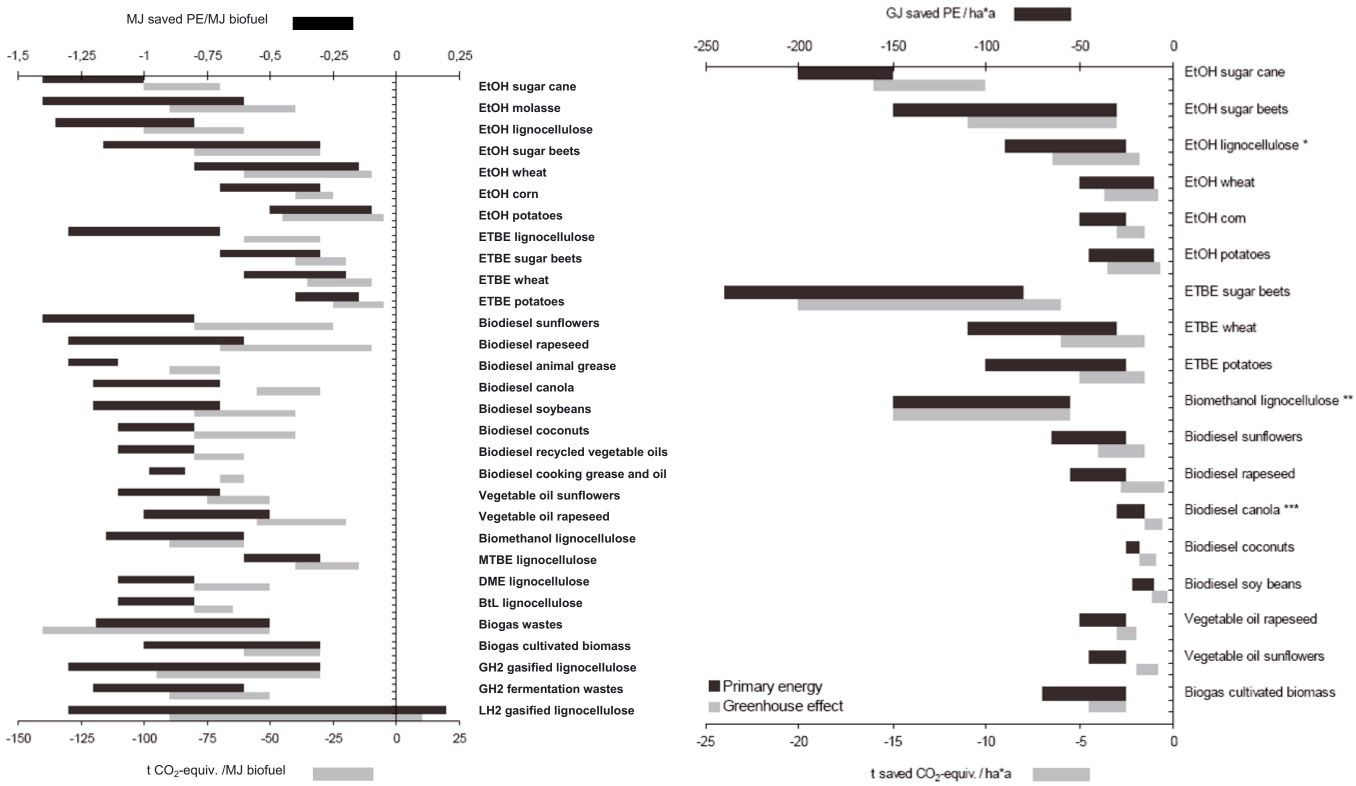

Figure 11. Results of the energy and greenhouse gas balances of the analysed biofuels as compared to their fossil counterparts (gasoline for ethanol, biomethanol, biogas and H_2; MTBE for ETBE; diesel for biodiesels, vegetable oils, DME and BtL) in MJ or GJ saved primary energy and saved metric tons CO_2 equivalent per MJ of biofuel (a) and per hectare a year (b). Negative values imply advantages for the biofuels; the zero mark means that the CO_2 equivalent emissions are balanced when the total life cycle (biofuel minus fossil fuel) is considered (Quirin et al., 2004).
* The spectrums for ethanol from lignocellulose are not unrestrictedly comparable with the others, since lignocellulose from biomass and that from organic residues are put together here.
** Only from cultivated biomass.
*** Canadian brand name of summer rapeseed.

biodiesel compared with pure rapeseed oil. The latter is not refined and would by itself lead to higher saved fossil energy and greenhouse gases. However, the production of biodiesel co-produces glycerine that substitutes fossil-based glycerine. This co-product can be used as a substitute for chemical glycerine or as animal feed. The savings from displacing this otherwise produced glycerine are included within the system of the biodiesel chain, and can be particularly significant in the case of substituted chemical glycerine, whose manufacturing is very energy-intensive (JRC/EUCAR/CONCAWE, 2006). Nevertheless, these savings are no longer true as soon as the glycerine market is satisfied. There are many diverse uses of glycerine, even as feedstock for energy (IEA, 2007b), but there are also many industrial processes that lead to glycerine co-production (Russi, 2008). Assuming that large-scale biodiesel production will cause a saturation of the glycerine market (Larson, 2006), some authors do not consider any savings from glycerine co-production (Russi, 2008), which equal using economic allocations for co-product handling.

From a hectare perspective (Fig. 11b), ethanol from sugar crops performs better than ethanol from starch crops, and much better than the bio-substitutes of diesel, due to higher crop yields. ETBE makes more significant savings possible than ethanol because of its high energy-related hectare yield that permits one to displace higher amounts of fossil fuel; for each MJ of ethanol that replaces gasoline, 3 MJ of ETBE[34] replace MTBE. However, on an energy-unit basis ethanol is slightly more interesting than ETBE, as ETBE requires a further processing step compared with ethanol. ETBE is, though, here compared with MTBE, which is produced with more energy input than gasoline but emits a little less greenhouse gas, as the bulk of the energy is provided by natural gas for

[34] 3 MJ of ETBE are produced from "1 MJ of ethanol and 2 MJ of isobutylene".

MTBE rather than heavier hydrocarbons in the case of gasoline (JRC/EUCAR/CONCAWE, 2006).

When comparing gasoline blended with MTBE, ETBE or ethanol (50% ex-wheat, 50% ex-sugar beet) combined with isooctane that compensates for the ethanol higher density, ETBE permits more CO_2 equivalent reduction per MJ of ethanol (Croezen et al., 2007; Higgins, 2007). Blending in ethanol or ETBE reduces the petroleum base fuel requirement for butanes and reformate octane numbers. Compared with a LCA in which ethanol and ETBE simply replace MTBE, these modifications in the refinery operations cause an additional 3% GHG savings in the case of ethanol, and about 20% in the case of ETBE. In the case of ethanol, its high vapour pressure requires a counterpart reduction of the petroleum base fuels' vapour pressure that partly offsets the savings due to the ethanol's higher octane numbers. In the case of ETBE, less volume of petroleum base fuels is needed for the same volume of ethanol (5%) with an even lower octane number requirement. Although this advantage for ETBE is to some extent undone by the higher GHG emissions related to its production and that of extra isobutylene, ETBE finally results in a higher net GHG reduction of 61 kg CO_2-eq/GJ ethanol converted into ETBE, against 37 kg CO_2-eq/GJ ethanol when 5 vol% ethanol is added in pure form (Croezen et al., 2007). Overall more greenhouse gas reduction is possible though, when engines are modified to run with higher blends of pure ethanol.

Results on 2nd generation biofuels are still theoretical and provisional, since their production is still at a pilot stage and technology keeps evolving, which is notably illustrated by the wide spectrum of lignocellulose or H_2 biofuel chains (Fig. 11a). Moreover, 2nd generation biofuels can proceed from a wider range of feedstock, diverse organic residues and theoretically all possible biomass. Regarding the spectrum of possible pathways, 2nd generation biofuel assessments are still scarce.

Mature 2nd generation biofuels are expected to have less impact on the environment. This firstly lies in the fact that the whole plant is transformed, increasing the energy yield for the same mechanical and chemical treatment, which implies that the environmental load is lower for the same energy unit. Furthermore, in the case of perennial grasses, the environmental advantages could be even more consistent. The biomass of perennial grasses has higher lignin and cellulose contents than the biomass of annual crops (Lewandowski et al., 2003). Perennial energy crops and short rotation forestry or coppice (SRF/SRC) generally have less impact on: soil erosion, nutrient inputs into the ground and surface water, pesticide pollution, and water abstraction. In contrast to annual crops, perennials require only one cultivation activity, i.e. preparation for planting, over a 10- to 20-year duration, and minimal nitrogen inputs (Heaton et al., 2004). Typical energy input/output ratios vary between 1/10 and 1/20 (IEA, 2006).

Perennials also have more extensive root systems present throughout the year, thus providing increased resistance to soil erosion and a more effective means of trapping nutrients and preventing nitrogen loss to drainage water. Production and turnover of belowground storage organs can furthermore add organic matter and carbon to the soil (Heaton et al., 2004). In addition, most nutrients also remain on the land under SRC or with perennial grasses, when the harvest takes place after the nutrient-rich leaves have dropped. As a result, soil carbon and quality tends to increase over time, especially when compared with conventional farming (IEA, 2006). Finally, annual crops on average need better quality land than perennials to achieve good productivities, whereas perennials can be grown on marginal lands, thereby achieving other potential benefits such as soil quality improvement or in some cases adding to biodiversity (Lewandowski et al., 2003; EEA, 2006b). However, perennial grasses and SRC, like other crops, may also contribute to ozone depletion, acidification and eutrophication if they are fertilised.

5.1.2. Limits of the LCA tool

5.1.2.1 Data quality

When compiling results from diverse biofuel LCAs, it appears that results widely differ between studies. This variation can be explained following two intertwined tracks: the intrinsic limits of LCA methodology and the lack of scientific background knowledge. LCA was first established for industrial production; so that differences between industrial and agricultural systems originate many methodological problems for agricultural LCA. The fact that industrial systems are mostly independent of their local environment has led to a site-independent methodology for LCA. However, the life cycle steps in close contact with the environment (such as agriculture or land filling) are site-dependent by nature (Kodera, 2007). Thus, in essence, LCA implies a site-independent model of a biofuel chain at a given period of time, which is merely compatible with the dual agricultural-industrial nature of biofuel chains. The first consequence is the difficulty of collecting data sets of representative quality; i.e. relevant, transparent, precise, complete and reproducible, while data collection and compilation are already often the most work- and time-consuming steps in LCA (Rebitzer et al., 2004). On the one hand, agricultural data sets are time- and site- (soil-climate) dependent, which implies uncertainty in extrapolation and modelling, and further variability in biofuel chain assessments. On the other hand, industrial data sets are not systematically accessible and transparent for outsider assessors.

Now, the attempt to make an exhaustive inventory should not be at the cost of data quality. According to Delucchi (2004), using literature-review estimates with an aggregated approach of processes instead of primary data from an appropriated input/output flow model for each process can amount to a percent or two of direct fuel-cycle GHG emissions. Three further percentage points in the fuel-cycle analysis would be related to data uncertainty with regard to estimates of the energy intensity of fuel production and the energy efficiency of motor vehicles (Delucchi, 2004). Comparing the life-cycle inventory for refinery products among several databases, Jimenez-Gonzalez and Overcash (2000) have shown that the variability in estimated emissions to the atmosphere, and waterborne and solid waste are approximately 50–150%, 1000%

and 30%, respectively (in Lo et al., 2005). Also, models based on international or national average values might not be useful or adaptable to more specific local production conditions for decision-making on regional levels. Moreover, a LCA steady-state model cannot take directly into account the variation in market demand or the technological advances, although they can be introduced through economic allocation or as prospective assumptions. Both of these factors are crucial for bioenergy; likewise social aspects, which are also not encompassed in LCA.

The varying quality of input data makes the comparison of diverse scenario outputs more difficult. The lack of transparency and homogeneity in background assumptions between different biofuel chain assessments may hide the fact that data might not always be reliable. To deal with data quality and uncertainties, tools exist and could, if systematically associated with the results, enlighten comparisons between assessments. A "pedigree matrix" permits one to establish data quality indicators (DQIs) that give scores to data sets (1 to 5) in function of their reliability, their completeness, and their temporal, geographical and further technological correlations linked to the goal and scope of the study (Weidema and Wesnaes, 1996). These scores make it possible to distinguish processes and flows for which input data quality is poor, and to focus on these inventory parts to compare their impacts on output data among different assessments. This qualitative approach is to be completed with statistical indicators, such as coefficient of variation, that highlight the data uncertainty: i.e. the basic uncertainty linked to typical measurement errors or normal fluctuation of the variables, and an "additional uncertainty" related to the data not being of the optimal quality as reflected by its pedigree scores. This additional uncertainty can be calculated or estimated. "Default uncertainty matrices" for different specific types of data or domains could also serve as references, when the uncertainties cannot be directly assessed (Weidema and Wesnaes, 1996). Such tools might often appear to be necessary in the case of agricultural systems for which data such as probabilistic distributions and the correlations of key parameters are particularly rarely available (Basset-Mens et al., 2006a).

5.1.2.2 Co-product handling

The complexity of bioenergy chains evidences limits of LCA. Indeed, LCA methodology leaves some degree of liberty, when it comes to dealing with the handling of co-products and the complexity of a wide range of environmental impacts. The LCA methodology stipulates that the product and co-products should be handled separately as long as possible, through sub-division of processes, in order to avoid problems of burden allocation. This is, however, impossible in the case of bioenergy chains, where product and co-products come from the same feedstock and are chemically linked. In this case, methods of co-product handling are suggested but none is mandatory.

The expansion of the system boundary implies that the process leading to the production of a co-product is taken into account as a co-function of the system, either by additive substitution or by subtractive substitution. In both cases, the flows corresponding to the production of the co-product are taken into account in accordance with energy and environmental loads of the co-product, when produced through a fossil chain; either as supplementary loads in the reference scenario (additive substitution), or as loads to be subtracted from the bioenergy chain (subtractive substitution).

The ISO 14 041 (1998) stipulates, where allocation cannot be avoided, that the inputs and outputs of the system should be partitioned between its different products or functions in a way which reflects the underlying physical relationships between them (physical allocation or apportioning). Physical allocation implies that environmental loads and energetic costs are partitioned between the product and the diverse co-products in accordance with defined mass or energy ratios. Although mass ratios are easily measurable and therefore more frequently reproducible across studies, they do not reflect a fair share in burdens. Indeed, since the biofuel justifies all the energy and material expenses, it should support the main share of the total burdens implied by its production, which is not the case, for instance, when a co-product weighs more than the biofuel itself. Energy allocation credits may be closer to the logics of the functional unit of biofuels; however, differing calculation on an energy content or energy consumed basis can also introduce further variations among studies. When physical relationships cannot be distinguished, then a financial allocation is the remaining option and consists of defining ratios in function of the market value of each product. This last option can be quite relevant in the sense that economics will still be the underlying driving factor of a biofuel chain development, while LCA does not in itself take into account economic factors. But it also may create a bias if market values are punctually considered. An economic analysis is necessary to speculate on the price evolution of the diverse products.

As explicitly recommended by the ISO norms, expansion of the system boundary is the first solution that should be examined in order to address the issue of co-product handling. The introduction of the substitutes within the system boundary permits one to elicit the real impact of the co-product production in a specific context. For instance, Prieur and Bouvart (2006) showed with the example of BtL from wood, that the relative greenhouse gas savings compared with the fossil chain could vary between around 68% and 104% when considering a substitute co-produced for electricity either from a French mix (mainly from nuclear power) or average European mix (more from coal), respectively. Variability in this case is even wider between the two types of substitute than between one of the two substitutions and mass or energy allocations.

Biofuel chains are not equally sensitive to co-product handling. Comparing the diverse co-product handling scenarios (three allocation ratios: mass, energy or market value; and expanding the system), the burden of primary energy that comes to ethanol varies from 89.7–95.6% of total primary energy in the case of sugar beet, whereas it varies from 42.7–91.2% in the case of wheat (Malça and Freire, 2006). Kim and Dale (2002) also found that burdens of corn ethanol widely vary following the co-product handling, within the comparable range of 48–80% for both dry and wet processes. In these studies, mass allocation tends to lower the biofuel burden, whereas system expansion maximises it. Methodological choices (e.g. system boundaries and allocation methods) have a large influence which may very well override many other types of uncertainty (Björklund, 2002 in Malça and Freire, 2006). Following a pragmatic approach, it may be very relevant to introduce substitute products that are representative for the specific local production at stake. However, system expansion requires that an alternative way of generating the exported function exists and that data are available and collected. Such ways may not be found, or on the contrary, some co-products can possibly substitute a wide range of products (Malça and Freire, 2006; Kodera, 2007).

The lack of a clear standard for choosing substitution products may lead to arbitrary choice and inconsistent calculations and results even if the methodology remains unchanged (Kodera, 2007). Ink has been spent on looking for a consensus on this methodological issue, but there is no single procedure to deal with the diverse co-products, not even for each biofuel chain. The ISO norm (14 041/1998) states that when an allocation issue arises, a sensitivity analysis on this allocation parameter should be done. Hence, each biofuel LCA should explore a range of possibilities. Scenarios of allocation ratios and substitution means should be elaborated and compared, taking into consideration the most relevant co-products substituting chains on the local scale.

5.1.2.3 Impact characterisation

A lot of published LCAs for biofuel are not in fact LCAs stricto sensu. Indeed, they may be assessments based on a life-cycle approach and guided by the LCA ISO norms, but rarely assess all the potential environmental impacts (Quirin et al., 2004; Blottnitz and Curran, 2007). By its holistic nature, LCA would require assessing all the potential impacts linked to the flows inventoried; nevertheless, the attempt at an exhaustive impact characterisation also faces methodological as well as scientific constraints. Life cycle impact assessment (LCIA) is the step that actually permits one to quantify the potential environmental impacts. It consists of classifying and aggregating the results of the inventory into category indicator results (Eq. (1)) that characterise how the environmental mechanisms for the chosen impact categories are modelled, and what the contributions are of the involved flows to these impacts.

$$\text{Impact Category Indicator}_i = \Sigma_j\,[\text{Characterisation Factors}\,(_{ij}) \times \text{Emission/Extraction Inventory}\,(_j)] \quad (1)$$

where "j" denotes the substances or resources (Brentrup et al., 2004).

Traditionally in LCIA, characterisation factors linearly express the contribution of a mass unit of an emitted substance to a given impact category. Depending on the scope and goal of the assessment, different impact assessment methods or approaches can be used, while the inventory remains the same. They actually differ in the choice of impact categories, the logic and approaches underpinning the characterisation (models and indicators). The main discussion here lies in the consideration of either midpoint impacts, or endpoint impacts, i.e. damage-oriented assessments.

Midpoints are considered to be points in the cause-effect chain (environmental mechanism), between stressors and endpoints (Bare et al., 2000 in UNEP, 2003). Midpoint impact categories are, for instance, climate change, stratospheric ozone depletion, human toxicity, acidification, eutrophication, etc. In damage-oriented assessment, impact pathways connect the inventory results across midpoints to one or more of the damage categories; i.e. classes of endpoints defined as areas of protection (AoP) that can be impacted both in their intrinsic and functional values (Jolliet et al., 2004). There exist different classifications of AoP that are actually connected; e.g. human health, the natural environment and manmade environment, human health, biodiversity (or ecosystem health), and natural resources (Heijungs et al., 2003). Life support functions (LSF), climate regulation, hydrological cycles, soil fertility and biogeochemical cycles, are classes of midpoints, just like areas of protection are classes of endpoints (Heijungs et al., 2003). Depending on the state-of-the-art knowledge, the representation of a pathway link may vary from a fully quantitative description, involving new contribution indicators such as disability-adjusted life years (DALYs), for instance, to a short qualitative description of the expected causal impact on subsequent pathway links (Jolliet et al., 2004).

Some LCA practitioners argue that endpoints are the elements of an environmental mechanism that are in themselves of value to society (Udo de Haes and Lindeijer, 2001 in UNEP, 2003). Others fear that uncertainties, due to a lack of sufficient data and robust models, may be extremely high beyond midpoints (UNEP, 2003). Hence midpoint approaches, such as CML (Heijungs et al., 1992), Eco-indicator 95 (Goedkoop, 1995) and EDIP (Wenzel et al., 1997), have been the main widespread approaches historically, whereas endpoint ones, such as EPS (Steen, 1999) and Eco-indicator 99 (Goedkoop and Spriensma, 2000) are more recently gaining interest, since they imply the analysis of trade-offs between and/or aggregation across impact categories. They also permit an integrative estimate of environmental externalities by monetary valuation of welfare losses due to impact on the AoP, which are fundamentally linked to societal values. Endpoint LCA results can be interpreted in light of marginal impact costs or distance-to-target performances, and can in this sense clearly serve to help design market-based internalisation instruments (e.g. taxes) (UNEP, 2003). Endpoint assessments are hence more directly understandable and useful for decision-makers; however, they lose in transparency when weighting is required to compare across categories, which does not systematically elicit links between midpoints and endpoints.

Therefore, both midpoint and endpoint approaches provide useful information with a trade-off between reliability and relevancy, and should be, to some practitioners' mind, conducted in parallel to determine how the results are affected (UNEP, 2003). In the case of biofuel LCAs, midpoint assessment appears to be relevant enough, though, given that impacts in terms of "climate change" and "fossil resources" are of central importance, and that uncertainties linked to data quality and co-products may already be a significant source of misinterpretation. Furthermore, a biofuel LCA is to be compared with a fossil fuel LCA, while weighting[35] across categories or other subjective scoring is not suitable when it comes to comparative assertions (ISO 14 042).

A more crucial issue is to improve impact characterisation, notably in order to account for the location and time of the emissions, waste generated, and resources depleted better, as well as the geographical zone and time period over which the contributions to different impacts should be considered (Pennington et al., 2004). Linear characterisation (Eq. (1)) aggregates the environmental loads at the time of the assessment, and neither takes into account the substance background concentration nor its temporal and geographical dependency on exposure and fate (UNEP, 2003). This implies that all impacts, irrespective of the moment and the place that they occur, are equally included (Udo de Haes et al., 1999). The fundament, referred to as the "less is better" principle, is that a pollutant remains a pollutant even if it is emitted in a place where it will not cause any harm; as such its emission shall be considered as contributing to potential impacts independently of site and time (Heijungs et al., 2003). It follows that impact categories are assumed to be independent of one another, and unless a precise scientific background permits one to justify the hypothesis to partition the contribution of the same substance to several impacts, the substance flow shall contribute in its entirety to each impact (Guinée, 2002). Firstly, these assumptions only lead to calculating potential impact scores, not actual damage (Khalifa, 1999; UNEP, 2003). Potential impacts thus represent the worst-case scenario, where some redundancy of a substance contribution is preferred to the risk of not considering its contribution to one impact. However, practitioners should be aware that double-counting may lead to poor decisions and that their models should try partitioning burdens as far as the state-of-the-art knowledge on the causality chain permits it (Reap et al., 2008).

Secondly, this dose-response modelling is not sufficient to describe complex environmental mechanisms, especially those where thresholds intervene (Khalifa, 1999; UNEP, 2003; Pennington et al., 2004). Simplifying assumptions and available scientific knowledge influences the accuracy of the indicators, which may vary among impact categories due to discrepancies between models and the corresponding environmental mechanisms. Hence the applicability of the characterisation factors depends on the accuracy, validity and characteristics of the models used (UNEP, 2003; Basset-Mens et al., 2006a). Lack of knowledge on the dose-response determinisms may jeopardise the reliability of the impact assessment. This is particularly true in the case of impacts on biodiversity or natural habitats, for example, for which the mechanisms are especially complex, and may also include other determinisms than chemical or physical ones. Khalifa (1999) emphasises that impact assessment within LCA is notably lacking in reliability for the impact categories of eutrophication, photochemical ozone, ecotoxicity, loss of habitats and biodiversity, as existing thresholds and non-linear dose-response notably are not considered. The reliability of the assessment is also in essence lower when it comes to local-range impacts that are more specifically dependent on the local ecosystems than global impacts. Truncations and assumptions about global homogeneity and steady-state conditions introduce the most severe errors in impact assessment. Indeed, first setting arbitrary time horizons skews results in favour of short- or long-term impacts, thus ignoring spatial variation, local uniqueness and environmental dynamics discounts the influence of environmental stress concentrations, leading to inaccurate estimates of potential damages (Reap et al., 2008). Spatial information in LCA is actually mandatory in order to contribute to solving the poor accordance between potential impact as calculated in LCA and the expected occurrence of actual impact (Khalifa, 1999; Heijungs et al., 2003; Potting and Hauschild, 2005).

Sophisticated LCIA have been developed in order to improve the level of detail, regarding in particular the temporal and spatial dimensions of the impacts. Nevertheless, only a few integrated approaches have been proposed so far (UNEP, 2003) and they were essentially developed within an endpoint characterisation, whereas regionally differentiated midpoints would also be better indicators (Heijungs et al., 2003). In particular, models have been used to determine regionalised fate and exposure factors, in order to account for background load and a priori tolerance of ecosystems to the emissions (Potting and Hauschild, 2005). For instance, Potting et al. (1998) used the RAINS model (IIASA) to produce acidification factors to be used within LCA in order to simulate acidification discrepancy better across 44 regions in Europe. This model integrates information on emission levels for each region with information on long-range atmospheric transport in order to estimate patterns of depositions and concentrations for comparison with critical loads and thresholds for acidification, eutrophication and photochemical ozone creation. These critical load functions (weighted for the size of the ecosystems) are used to construct so-called "protection isolines" for the grid element that consist of all combinations of S and N deposition for which a given fraction of ecosystems does not exceed critical loads, and thus in RAINS terminology is assumed to be protected against the adverse effects of acidification. Regional acidification factors were calculated by reducing one by one the emission levels of each separate region by 10%, and then relating the result to the reference situation (the initial emission level and area of unprotected ecosystems). Hence acidification factor ($AF_{s,i}$ in ha/tons) directly relates a change in emission of substance (s) in a region (i) to the change in unprotected ecosystems in its total deposition areas. A similar approach was notably used to calculate regional factors for terrestrial eutrophication (with NH_3, NO_x).

Impact characterisations are integrated over an infinite period of time, since the variation in emissions on the regional scale during the time period for integration is considered as marginal when compared with the total contributing emissions (Potting and Hauschild, 2005). Indeed, LCA steady-state assumption is founded on the "multiple sources-multiple

[35] Weighting and normalisation are two non-mandatory steps in LCA methodology.

receptors" character of present environmental problems, i.e. the temporal variation in the contribution from a single source emission is usually to a large extent cancelled out against the high background exposure from all sources together. Moreover, the large impact area of an emission and the overlapping with impact areas of neighbouring sources make the precise location of a source of less importance, which makes it possible to determine site-dependent factors on a regional scale (Potting and Hauschild, 2005). However, these approximations will not be true for local (exposures within the first kilometres from the source) or time-dependent impacts (such as very slow emissions or synergic impacts) (Potting and Hauschild, 2005). Modelling the combined impacts of agricultural inputs, the climate and the hydrological functioning of catchments, Basset-Mens et al. (2006a) determined N apparent fate factors that describe the part of leachable N that actually contributes to the annual stream nitrate flux. This study emphasises that, on the regional scale, these nitrate fate factors can imply large variations compared with the results of a standard LCA methodology. In the case of pig production, the eutrophication result was reduced by 5% to 32%, and the climate change impact varied between "no change" and an increase of 200% (Basset-Mens et al., 2006b).

Temporal and spatial dimensions are tightly intricate together. Especially meteorological conditions influence the determination of fate and exposure factors. Emission timing at different rates and locations defines site-specific emitting, fate and exposure conditions. For instance, the acidification factors calculated for the reference years 1990 and 2010 show that the difference between different calendar times can be notable (Potting and Hauschild, 2005). This would also be particularly relevant in the case of the region-specific fate factors for airborne nitrogen compounds causing aquatic eutrophication by Huijbregts and Seppälä (2000). These region-specific fate factors were modelled given European emission and meteorological data from 1985 to 1995. Regional NH_3 and NO_x fate factors express the fractions of these airborne emissions that actually end up in the aquatic environment, taking into account both direct deposition in the freshwater and marine environments and run-off from terrestrial systems into the aquatic environment. Now, for a short time frame, these regional fate factors may, in essence, vary depending on punctual variations in precipitation patterns compared with those used to model the fate factors. The characterisation factors used to assess the impact from a given process should also relate to the calendar time in which that process takes place (Potting and Hauschild, 2005).

However, time-dependent environmental processes may necessitate time horizons for impact integration that are not in accordance with an optimum for those site-dependent factors. Thus, time-dependent factors add to the continuing discussion within the LCA community on selecting integral limits and valuing impacts distributed in time (Reap et al., 2008). Furthermore, depending on the impact, the extraction/emission region might not be the same as the region where impact/damage occur. For both time and space, it would be necessary to distinguish factors at the extraction/emission point and impact/damage point (Heijungs et al., 2003). Thus, scale precision for spatial and temporal discrepancies might remain constrained by a geographical scale large enough to cover most of the impacts from an emission source. Site-dependent factors already mostly encompass characteristics that are relevant at a country-based level. Now, these factors appear to vary also on the regional scale. In the life cycle region-specific assessment method proposed by Yi et al. (2007), where "affected regions" are distinguished from "emitting regions", regional damage factors considerably vary within the 9 defined areas and are 0 to 3 times higher than the national average.

To conclude, LCA is a powerful tool but there remain theoretical and methodological open questions that can lead to diverging results and interpretations, in particular in the biofuel chain cases. While some practitioners work on sophisticating LCA (UNEP, 2003), some others look at ways to simplify it (Rebitzer et al., 2004). On the one hand, some limits (methodological, scope of the impacts: site independency, etc.) should be overcome to make LCA a better tool for decision-making purposes; on the other hand, it already is a challenging exercise (lots of data needed, various sources of uncertainty, crucial lack of scientific knowledge for impacts, etc.). A way to solve this puzzle may be to produce local specific LCAs, based on local inventories and a better knowledge of the local receptive environment. Although the requirement of additive site-dependent data is often put forward as an objection against spatial differentiation in LCA (UNEP, 2003; Potting and Hauschild, 2005), local data sets and fitting models with local receptor parameters can make it possible to diminish the uncertainty of the results.

Uncertainty affects all the assessment steps: from the input data uncertainty, through model uncertainty, up to uncertainty on the chosen uncertainty formalisms (e.g. determinations of probability distributions, etc.) and may, if completely represented, even entail a broad interval of imprecision that could finally make the results of comparative LCAs indistinguishable (Reap et al., 2008). As illustrated in a review by de Boer (2003), comparison between LCAs of diverse commodity chains across different case studies is hampered by the lack of international harmonisation on LCA methods among the studies, whereas a within-case-study comparison of diverse chains using the same LCA methodology appears suitable to track down the main differences in potential environmental impacts. Biofuel chains should be hence compared on a regional case-study-based scale, and further extrapolation across regions should be limited to LCAs that use harmonised methods.

5.2. Focus on greenhouse gas emissions from agriculture

"Climate change" is often presented as an example of well-established midpoint impact characterisation, notably because its global dimension particularly fits the "multiple sources-multiple receptors" background assumption. However, when assessing biofuel chains, it appears that greenhouse gas emissions in the field can show highly variable temporal and spatial patterns, and that these emissions contribute a significant share of the total greenhouse gas balance of the whole chain. Emissions during agricultural production contribute, for instance, 34%–44% to the greenhouse gas balance of corn ethanol in the US (Farrell et al., 2006) and up to more than 80% in the case of pure vegetable oils (ADEME/DIREM, 2002). Focusing on the agricultural phase only, many factors imply variations among biofuel chains. They are linked to the ecosystem characteristics and the cropping systems, both dependent on local conditions. After briefly describing the greenhouse gas emissions linked to agricultural activities, two aspects of the current limits in the assessment of biofuels' greenhouse gas balance will be detailed, i.e. the determinism of N_2O emissions and CO_2 emissions due to land-use change.

5.2.1. Overview of greenhouse gas emissions from agriculture

Agricultural activities contaminate the environment through three main impact pathways: land-use changes, the use of farm machines, and the use of inputs, e.g. fertilisers, that are sources of many diverse pollutants throughout their life cycle. Industrial emissions concerning the agricultural phase in a biofuel LCA encompass all the emissions linked to the production and use of machines for agricultural operations and of inputs such as fertilisers or pesticides. Briefly, the more intensive the use of machines and inputs, the higher the overall emissions of pollutants. In Farrell et al. (2006), agricultural practices across six compared corn with ethanol LCAs are responsible for 45% to 80% of all petroleum inputs and related emissions. This illustrates how the intensity of cropping systems influences the overall chain performance, but also that the lack of transparency in primary energy inputs adds to the global confusion when comparing LCAs. More attention is paid here to emissions that are the results of natural reactions within the coupled biogeochemical cycles of C and N. In general, CO_2, N_2O and CH_4 are by-products of the microbial activity, which is characterised by a transfer of electrons, hence depending on soil redox potential, dissolved organic carbon content, and the concentrations of the relevant electron acceptors (Li, 2007). These reduction-oxidation reactions are influenced by both the natural conditions and the agricultural activities, meaning that resulting emissions can vary widely and are therefore hard to predict.

Focusing on greenhouse gas emissions, in 2005 direct CH_4 and N_2O emissions from the agriculture sector worldwide accounted for about 5.1–6.1 $GtCO_{2eq}\ yr^{-1}$, equivalent to 10–12% of the total anthropogenic emissions of greenhouse gases (Smith et al., 2007). This includes 3.3 $GtCO_{2eq}\ yr^{-1}$ of CH_4 (50% of total CH_4 anthropogenic emissions[36]) essentially due to enteric fermentation from livestock (27% of agricultural greenhouse gases in Baumert et al., 2005 of a total of 6.2 $GtCO_{2eq}\ yr^{-1}$ in 2000), to rice cultivation on wetlands (10% of agricultural greenhouse gases in Baumert et al., 2005), and to manure management (7% of agricultural greenhouse gases in Baumert et al., 2005), and 2.8 $GtCO_{2eq}\ yr^{-1}$ of N_2O (60% of total N_2O anthropogenic emissions) (Smith et al., 2007) produced by microorganisms in the soils (40% of agricultural greenhouse gases in Baumert et al., 2005). Annual CO_2 emissions by agricultural lands are low compared with overall anthropogenic CO_2 emissions. The net flux between the atmosphere and the agricultural land, not considering energy-related emissions, is estimated to be approximately balanced at around 0.04 $GtCO_{2eq}\ yr^{-1}$ (Smith et al., 2007), while energy-related CO_2 emissions accounted for around 9% of global agricultural greenhouse gases[37] in 2000 (Baumert et al., 2005), although this share can be higher in industrial countries within intensive agricultural systems.

Moreover, land-use change and forestry (LUCF) are responsible for around 13% of global anthropogenic greenhouse gas emissions, i.e. some 5.4 $GtCO_{2eq}\ yr^{-1}$ on average during the period from 2000–2005, respectively (Houghton, 2008). The carbon flux includes emissions due to land clearing, emissions from forest products (including woodfuel) (80% in Duxbury and Mosier, 1993), and emissions from the oxidation of soil organic matter in the years following initial cultivation on former forest land (20% in Duxbury and Mosier, 1993). On the other hand, the carbon sinks accounted for are re-growing forest (vegetation and soils) after agricultural abandonment, reforestation, harvest and fire suppression. The assessed flux does not take into account the influence on carbon stocks of agricultural or silvicultural practices that do not imply changes in area, such as changes in species, no-till agriculture and thinning of forests, for instance. Finally, the assessment does not consider the indirect effects of fertilisation by N deposition or increased atmospheric concentration of CO_2, that could partly counterbalance the rise in CO_2 emissions (Houghton, 2003).

Agriculture is indirectly responsible for a large part of these emissions due to land clearing to convert lands into croplands or grasslands, primarily deforestation in developing countries driven by the conversion of forest to agricultural lands (Houghton, 2003; Baumert et al., 2005); 60% of released carbon due to land-use change between 1850 and 2000 came from the tropics and during the 1990s, the net carbon flux outside the tropics has actually turned into a net sink (Houghton, 2003). Nevertheless, anthropogenic CO_2 emissions from land-use change and the forestry sector are subject to *extraordinary uncertainties*[38], notably linked to the varying availability and quality of regional land-use data and to uncertainties in estimating forest growth rates and carbon stocks in ecosystems affected by various human management practices (Houghton, 2003). Thus, estimates of the carbon fluxes are uncertain in the order of ±150% for large fluxes, and ±50 $MtC\ yr^{-1}$ for estimates near zero (Houghton, 2003).

For the 1990s, IPCC estimates of CO_2 from land-use change ranged between 12% and 28% of world total CO_2 emissions (in Baumert et al., 2005). This sector also includes N_2O and CH_4 emissions, although no reliable global estimates make it possible to assess the share of these emissions that are linked to land-use change and forestry (Baumert et al., 2005).

[36] These shares are followed in the report by the mention "*medium agreement, medium evidence*", and the same for the balanced CO_2 net flux by agricultural soils "*low agreement, limited evidence*".

[37] The remaining 6% of agricultural greenhouse gases by subsector are undifferentiated sources of CH_4 and N_2O.

[38] As written by the authors Baumert et al., 2005, p. 91.

In order to assess the greenhouse gas emissions due to agricultural production better, a deeper understanding of the local determinism of CO_2, N_2O and CH_4 contributions is needed, which also takes into account the impacts of land-use change and agricultural practices on these emissions. We focus here, within the framework of biofuels from energy crops, on N_2O and CO_2 emissions.

5.2.2. N_2O *emissions*

5.2.2.1 Global budget

N losses from agricultural fields are a very critical issue for mainly two reasons. First, they represent a net loss of nutrient for the plant and a consequently supplementary cost in terms of fertilisation. Second, all the Nr[39] leaks outside of the soil-plant system are sources of pollution. The mechanisms of N losses are diverse, and so are their impacts on the environment. The determinism of N losses and the characterisation of their impacts are complex especially because of difficulties in considering changes in spatial and temporal scales between emission sources and final impacts, as reactive Nr is widely dispersed by hydrologic and atmospheric transport (Galloway et al., 2003). Furthermore, little is known about how to quantify synergic or antagonistic processes occurring between midpoint impact (e.g. acidification) and endpoint damage (e.g. water toxicity), inducing further uncertainty in indirect emissions that are linked to primary direct emissions.

A wide range of experiments and studies have been focusing on how to improve N fertilisation efficiency in order to firstly reduce the source of these losses as far as possible. In this sense, much progress has already been achieved during the last decades, notably by better adapting the type, doses and applications of fertilisers to the crop needs and pedo-climatic conditions. However, many questions still remain, especially concerning the determinisms of gaseous N losses on the field scale. The imbalance between total inputs and outputs of N in agricultural systems has puzzled scientists for more than 50 years (Wrage et al., 2001). ^{15}N balances show deficits in N fertiliser recovery that vary between 1% and 35% (Recous et al., 1988).

Special attention has increasingly been paid to nitrous oxide (N_2O), since it is an important agricultural greenhouse gas. Indeed, due to its long residence time in the atmosphere and its high relative absorption capacity per mass unit, its 100-year global warming potential is about 298[40] times that of CO_2 per mass unit (Forster et al., 2007). Considering its current concentration in the atmosphere, N_2O is the fourth largest single contributor to positive radiative forcing (after CO_2, CH_4 and tropospheric ozone[41]) (Denman et al., 2007). Its radiative forcing averaged 8.5% of total radiative forcing for the period 1750–2000, when CO_2 contributed to 85% of this total radiative forcing (Forster et al., 2007). N_2O is also the main source of stratospheric NO[42], that catalyses the photolysis of O_3 (Conrad, 1990).

Nitrous oxide is naturally produced in soils through the microbial processes of denitrification and nitrification. Nitrification is the aerobic oxidation of ammonium into nitrate, and denitrification is the anaerobic reduction of nitrate into nitrogen gas (N_2). N_2O is an obligate intermediate in the reaction sequence of denitrification and a by-product of nitrification (IPCC, 2006). Hence, the availability of inorganic Nr in the soil appears to be one of the main controlling factors of these reactions, and the intensification of agricultural activities leading to more use of N fertilisers enhances N_2O emissions by the soils (Mosier et al., 1998; IPCC, 2006).

Agriculture is the single biggest anthropogenic N_2O source (Denman et al., 2007), being the third overall most important source after soils under natural vegetation, especially land at tropical latitudes due to more rapid N cycling (Duxbury and Mosier, 1993), and N_2O release by oceans. Compared with CO_2 (Fig. 12), there is only one significant known sink of N_2O, i.e. its destruction in the stratosphere after an average residence time of 114 years in the atmosphere, and no robust evidence of soil N_2O sink strength. The amount of N_2O that is absorbed by soils, i.e. "consumed" by denitrification, is subject to extreme uncertainty. Net N_2O uptake by soils has been observed under different conditions, making it difficult to identify a set of conditions promoting N_2O uptake. However, factors opposing diffusion of N_2O in soil generally seem to increase its consumption, as well as low Nr and O_2 availability[43] (Chapuis-Lardy et al., 2007). N_2O uptake is often masked by larger N_2O production and may be indirectly accounted for in global budgets, provided that emission factors are based on all measured fluxes without discarding negative measurements.

More understanding of N_2O consumption by soils is needed to take into consideration its contribution to the global N_2O budget better; especially since the current global estimated sources and sinks of N_2O are not balanced (Chapuis-Lardy et al., 2007; Goldberg and Gebauer, 2008). Reported sources are larger than summed sinks and atmospheric increase.

[39] Nr means reactive nitrogen compounds, i.e. all inorganic and organic N compounds except N_2, that is a non-reactive N compound.

[40] The former GWP in the second IPCC assessment report was 310 eq CO_2 per kg, 298 includes the indirect negative radiative forcing due to the destruction of stratospheric ozone.

[41] Radiative forcing ($W.m^{-2}$), or global warming potential, refers to the change in the radiative balance on Earth's surface that is normally ensured by the natural greenhouse effect whose dominant contributing gases are water vapour (60–70% in Duxbury and Mosier, 1993), CO_2 (25% in Duxbury and Mosier, 1993) and O_3. A positive radiative forcing (warming) occurs when the concentration of greenhouse gases increases; a negative radiative forcing (cooling) when precursors that lead to the destruction of greenhouse gases are released into the atmosphere. Halocarbons are also main contributors to radiative forcing to an extent similar to that of tropospheric ozone (Forster et al., 2007). They are not mentioned amongst the first single contributors though, because they encompass several gas contributors.

[42] NO_x = NO + NO_2 which are in photochemical equilibrium. NO_x is mostly firstly emitted in the form of NO (Conrad, 1990). NO_x is a common anthropogenic pollutant (Duxbury and Mosier, 1993).

[43] Soil humidity favours denitrification up to N_2O reduction, while NO_3^- is preferred as an electron acceptor over N_2O (Granli and Bockman, 1994).

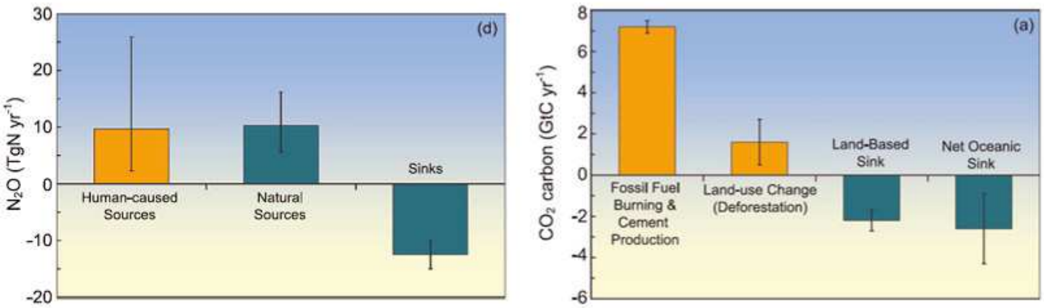

Figure 12. Global budgets of N_2O (14) and CO_2 (15) (Denman et al., 2007).

Considering that oceanic N_2O source may be underestimated[44] by at least two-fold (Bange, 2006), it is additionally likely that some N_2O source is overestimated or N_2O sink underestimated (Goldberg and Gebauer, 2008). Moreover, N_2O stratospheric lifetime seems to be shorter than previously thought, which also indicates that sinks may have been underestimated (Chapuis-Lardy et al., 2007). Recent bottom-up and top-down estimates of total N_2O sources in the 1990s agreed on averages of 17.7 (8.5–27.7) and 17.3 (15.8–18.4) MtN yr^{-1}, respectively (Denman et al., 2007). With a bottom-up approach, total direct and indirect annual N_2O emissions from agricultural soils in the 1990s, including synthetic fertiliser, manure and biological N fixation would average 3.5 MtN yr^{-1} (2% of N input) (Mosier et al., 1996), 4.2 MtN yr^{-1} (Mosier et al., 1998; i.e. IPCC, 1997) or 5.4 MtN yr^{-1} (Denman et al., 2007). A FAO statistical model, considering most of the factors[45] influencing median values of N_2O measurements, gives estimates for total direct N_2O emissions[46] of 3.5 MtN yr^{-1}, with a 34% share occurring in developed countries, respectively (i.e. 3.3–3.4% of N input) (FAO, 2001). Total top-down assessed agricultural N_2O emissions of 4.3 to 5.8 MtN yr^{-1} (3.8–5.1% of N input) (Crutzen et al., 2008) are to be compared with the bottom-up estimate 6.3 MtN yr^{-1} (Mosier et al., 1998) that also encompasses 2.1 MtN yr^{-1} from animal waste management. IPCC assessments of global N_2O budgets have been continuously evolving, notably due to improvement in considering the diverse direct and indirect sources and refining emission factors; still, the uncertainty on anthropogenic N_2O remains remarkably significant (Fig. 13).

[44] Bange emphasised that estimates used in global budgets are out of date. Moreover, due to increased release of anthropogenic Nr into the ocean, N_2O emissions by marine microorganisms could increase up to 1.6 MtN-N_2O per year (in Galloway et al., 2008).

[45] Climate, crop type, fertiliser type, application rate, mode and timing of application, soil organic C and N content, soil pH, soil texture and drainage, measurement technique, frequency of measurements, length of measurement period. This analysis does not include organic soils; neither did the one from Mosier et al. (1996). Organic soils are considered in the IPCC guidelines. They appear to be a great source of N_2O, because of high soil organic content and low drainage, which implies reducing conditions (IPCC, 2006). Total areas of organic soils (histosols) ~1.2% of ice-free land area (online 03.02.2009: http://soils.ag.uidaho.edu/soilorders/histosols.htm).

[46] Data for fertilisation IFA/IFDC/FAO (1999), land-use in FAO, 2001.

5.2.2.2 Origins of uncertainties

Uncertainties in estimating N_2O emissions from agricultural fields originate in the difficulties (1) of identifying all the primary sources; the contribution of biological N fixation[47] is especially hard to quantify (Mosier et al., 1998; IPCC, 2006), (2) of following the fate of nitrogen throughout the whole nitrogen cascade that implies several processes and "actors" (Duxbury and Mosier, 1993; Galloway et al., 2003), and (3) of capturing and characterising the spatial and temporal high variability in emissions (Parkin 1987; Mosier et al., 1996). This variability is due to multiple involved processes that each respond differently to various environmental and soil factors (Farquharson and Baldock, 2008). Moreover, these factors can interfere at three control levels: (1) in the rate of nitrification and denitrification, (2) in the proportions between the gaseous end products of these reactions, and (3) in the consumption of these gases in the soil before escaping to the atmosphere (Firestone and Davidson, 1989).

On a global basis, about 120 MtN from new Nr (fertilisers and cultivation-induced biological fixation) and 50 MtN from previously created Nr (crop residues, deposition, etc.) are added annually to agroecosystems (Galloway et al., 2003). Within the primary cycle of Nr (dashes lines in Fig. 14), only half of the N input is harvested in the crop (Duxbury and Mosier, 1993; Galloway et al., 2003), while the other half is lost by a combination of leaching (19–26% of input in Smil, 1999), run-off and gaseous losses through direct emissions (15–35% of input in Smil, 1999), primarily from denitrification (Duxbury and Mosier, 1993), volatilisation and nitrification. Secondary N flows, shown by the solid lines (Fig. 14), encompass N_2O emissions from nitrification and denitrification through two indirect pathways; (i) following volatilisation of

[47] Symbiotic rhizobia in root nodules are able to denitrify. This can lead to N_2O emissions, possibly 4 kg N.ha^{-1} for improved pastures; legumes could increase N_2O emission two- to three-fold compared with unfertilised fields in Mosier et al. (1996). This denitrification by rhizobia could also lead to net N_2O consumption depending on local factors.

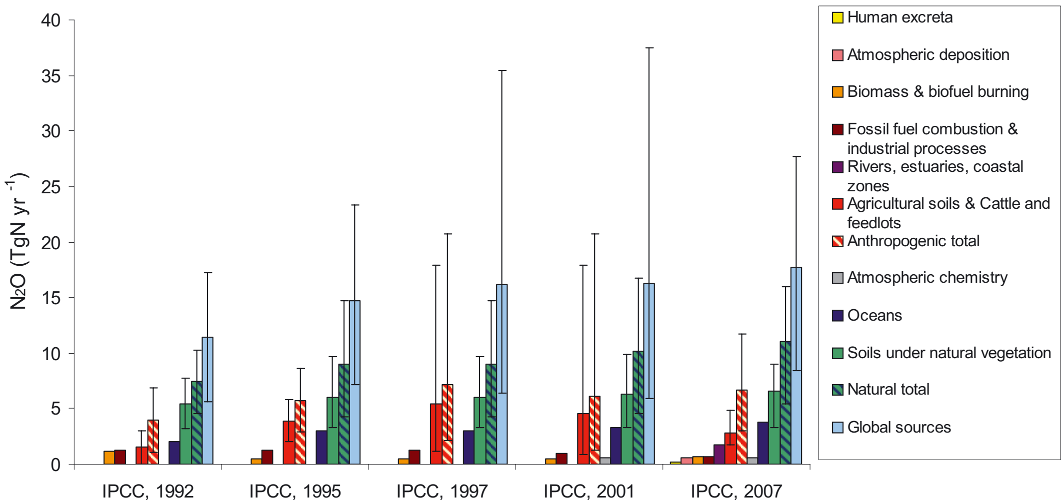

Figure 13. IPCC assessments of total annual N_2O emission during the 1990s. Drawn from IPCC data in Mosier et al. 1998 and Denman et al., 2007. Note: for the 1997 assessment 0.9 Tg yr^{-1}(Mt yr^{-1}) of emissions by agricultural soils are subtracted to the total 6.3 Tg yr^{-1}(Mt yr^{-1}) to prevent double counting with part of indirect agricultural emissions already accounted for within emissions by oceans. Emissions from human excreta, rivers and atmospheric deposition can also be partly considered as agricultural indirect emissions.

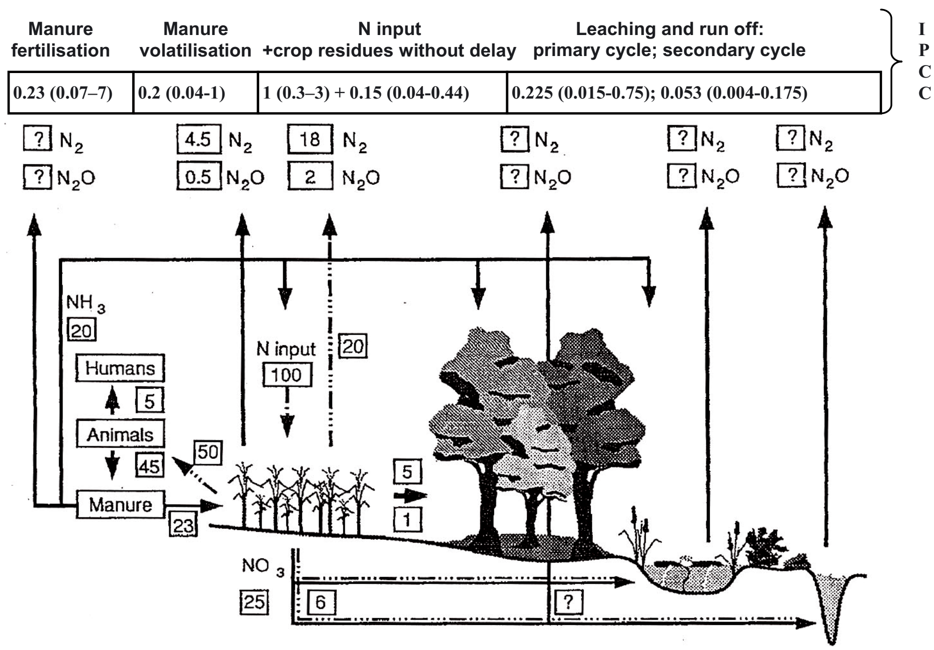

Figure 14. A simplified flow of fertilizer N through the environment (Duxbury and Mosier, 1993) and N_2O emissions with ranges into brackets (N_2O-Nkg) estimated with the IPCC Tier 1 method (IPCC, 2006) considering some US average data on the diverse fractions as given by Duxbury and Mosier (1993).

NH_3 from urea, ammonia or manure application, of NO_x, and the subsequent re-deposition of these gases and their products NH_4^+ and NO_3^- into soils and waters; and (ii) after leaching and run-off of Nr, mainly NO_3^-.

Throughout the whole Nr life cycle, only a small amount (about 4 MtN from the initial 170 Mt in Smil, 1999) will accumulate in the agroecosystems, while the rest will eventually transfer back into the atmosphere (Duxbury and Mosier, 1993; Galloway et al., 2003), including the 21 MtN temporarily stored through human consumption of grain (64%), and meat (CAFOs[48] 20%) (Galloway et al., 2003). Despite current knowledge, it is still not possible to reliably predict the fate of a unit of Nr that is applied or deposited in agroecosystems (Mosier et al., 1996), and the total amount of Nr lost

[48] Concentrated animal feeding operations.

through denitrification in agroecosystems is poorly known. Agroecosystems receive about 75% of the Nr created by human activity (Galloway et al., 2003). In the mid-1990s, the fate of only 35% of Nr inputs in the terrestrial biosphere was relatively well known: 18% was exported to and denitrified in coastal ecosystems, 13% deposited into the ocean, and 4% directly emitted as N_2O; the remaining 65% either accumulated in soils, vegetation and groundwater or was denitrified into N_2, but the uncertainty of those estimates remains large on every scale (Galloway et al., 2008), and further uncertainties appear when trying to assess all the direct and indirect N_2O emissions. In the field, direct N_2O emissions from N-fertilised agricultural fields have been found to vary between 0.001% and 6.8% of the N applied. Case studies combining diverse measurement techniques confirmed that uncertainty in N_2O fluxes found in the literature was indeed due to diverse combinations of controlling factors and not linked to the analytical methods (Mosier et al., 1996).

5.2.2.3 Emission factors

The 2006 IPCC guidelines to assess N_2O emissions from managed soils (IPCC, 2006) consider all identified direct N_2O sources, except direct emission from biological fixation due to the lack of experimental evidence, and the two above-mentioned indirect pathways. The Tier 1, 2 and 3 methods consist first of a comprehensive accounting of all N input into the fields, including inorganic and organic fertilisers, as well as mineralised N from soil organic carbon due to land-use change or crop residues that indicate background emission levels linked to recovery from past managements. These Nr amounts are then multiplied by default emission factors (Tier 1), emission factors related to country-specific data when available (Tier 2), or the emissions are estimated with process-based models (Tier 3). Following the same order, Tier 1, 2 and 3 methods guide the estimation of input fractions to be multiplied by indirect emission factors with less to more country specificity, respectively.

Following the Tier 1 method and the given default emission factors, the 100 kg of Nr input on a US field (assuming maize for crop residues) as characterised by Duxbury and Mosier (1993) (Fig. 14) would emit 1.86 kg N-N_2O with a wide uncertainty range between 0.47 kg and 12.4 kg. These estimates include emissions from crop residues, without time lag, and emissions due to a secondary cycle through manure recycling and further run-off and leaching due to its application. Using the 1.25% ± 1% of N input lost as N_2O (Bouwman, 1994) and further 0.75% of this input lost through indirect emissions, the same scenario would be expected to lead to 2 kg N-N_2O ± 1 kg emissions (Mosier et al., 1996). The rough 2.5 kg N-N_2O estimated by Duxbury and Mosier (1993) did not include all indirect contributions.

Statistical models aim at finding reproducible correlations, i.e. relationships representative of most data sets, between controlling factors and emissions, e.g. emission factors depending on the type and amount of applied fertiliser. In this sense, the more data collected on the different direct and indirect emissions, the more informative emission factors will be. The multiplication of data sets and the development of complementary techniques on various scales (aircraft measurements, micrometeorological techniques, chambers (see picture below), ^{15}N balance, C_2H_2 inhibition and lab work), have already made it possible to improve the understanding of controlling factors beyond a coarse linearity with N fertiliser. It has also been proven that measurements should be carried on throughout the year, because maximum fluxes were observed at different times for different treatments, and with a high frequency to capture temporal emission patterns (Mosier et al., 1996; Laville et al., 1997; Beheydt et al., 2007; Pattey et al., 2007). Frequent measurements appeared to yield lower total emissions (FAO, 2001; Pattey et al., 2007) due probably to less error with interpolation of punctual high emissions (Conen et al., 2000). Still, increased numbers of measurements that represent a wider range of agricultural systems are needed in order to improve statistical models (Mosier et al., 1996; Stehfest and Bouwman, 2006), as well as process-based ones.

Measurement of N_2O and CO_2 fluxes with automatic chambers in a sugar beet plot,
Estrees-Mons, June 2008 bessou©INRA

Given the complexity of N_2O emission control (Fig. 15), it is almost impossible to determine a quantitative relationship between the cause (a change in any ecological driver or environmental factors) and the consequence (N_2O fluxes) through simple correlation or regression analysis (Li, 2007). The extreme spatial and temporal heterogeneity of many primary drivers actually obscures the relationships between cause and effects for many of the biogeochemical processes, so that correlations between a change in primary drivers and linked changes in biogeochemical cycles are inherently non-linear (Li, 2007; Conrad, 1996). As regression models neglect several variables, because datasets used for developing the model did not distinguish these variables, for instance, emission factors cannot in essence lead to significant reduction of estimation uncertainty and cannot always be used to test different management or mitigation scenarios (Beheydt et al., 2007).

5.2.2.4 Process-based models

Process-based models make it possible to assess emissions with more accuracy, because of a better accounting for all involved processes and local conditions. Numerous models can nowadays simulate cropping systems and the associated fluxes between the soil-plant-atmosphere compartments. By simulating plant uptake, biomass growth and residues, nitrate leaching

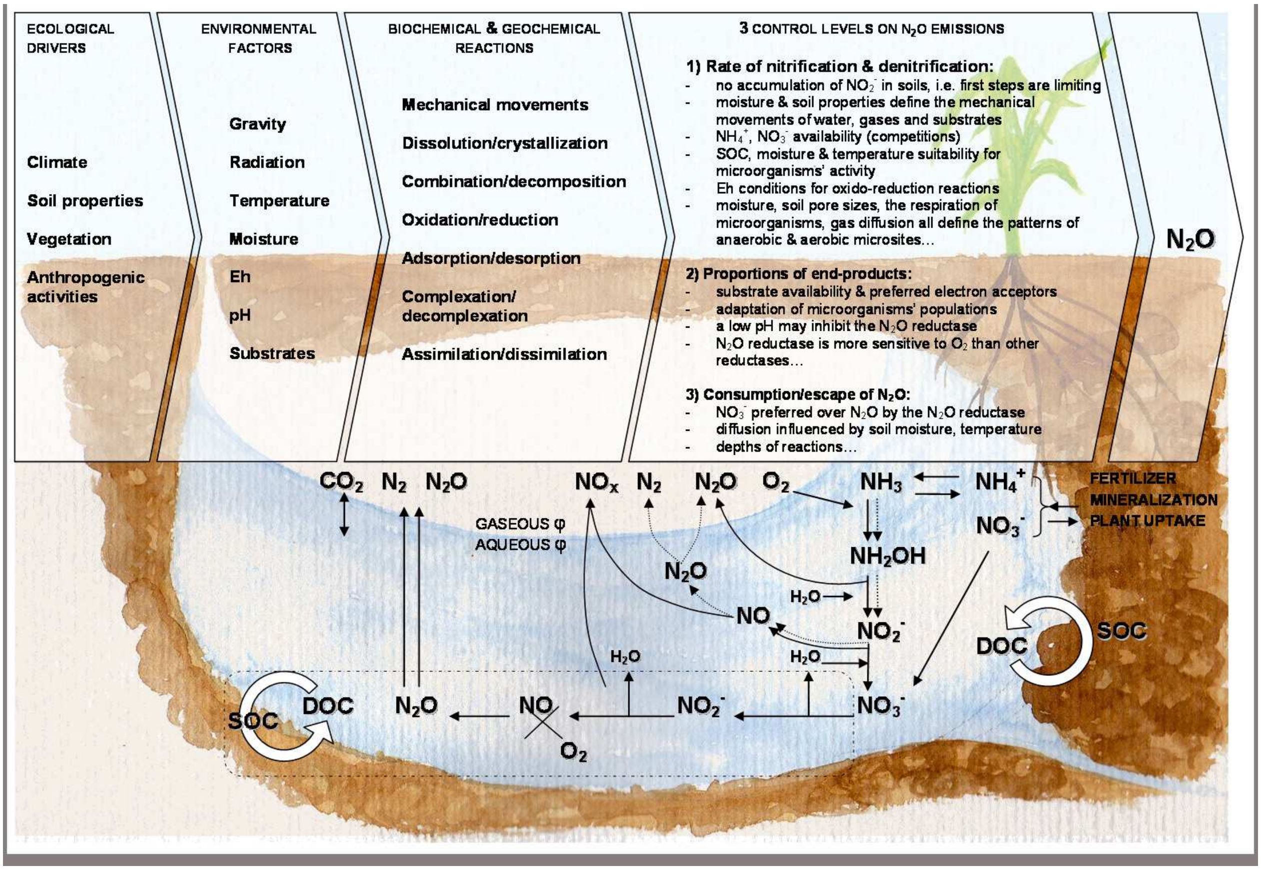

Figure 15. Scheme of the determinism of N_2O emissions by soils adapted from Li (2007); Wrage et al. (2005); Farquharson and Baldock (2008). Soil particles, gaseous and aqueous phases (ϕ) are artificially "well distinguished". The two extremities of the symbolised pore represent the pore continuum throughout the soil matrix. Reactions take place at the interface of soil particles and aqueous phase where microorganisms and substrates are. Denitrification takes place in the "anaerobic dashed-line rectangle". Dashed arrows lead to N_2O emissions by nitrifier denitrification occurring in low O_2 conditions, which implies that some NH_3 oxiders process the whole reaction chain (Wrage et al., 2001).

or volatilisation, for instance, models can provide insight into the amounts of Nr that might be directly or indirectly emitted as N_2O. Specific sub-models then simulate the part of these amounts that is expected to be emitted as N_2O. Dynamic models for N_2O emission in relation to soil processes have been available for a dozen years and in more recent years developed for different ecosystems and N species (Sutton et al., 2007).

N cycling models can be classified following three approaches: (1) simplified empirical process models in which N cycling processes are assumed to be determined by easily measurable parameters; (2) microbial growth models, where N dynamics is simulated by explicitly representing the dynamics of involved microorganisms; and (3) soil structural models simulating physical processes such as diffusion into and out of soil aggregates where occurring anoxia leads to denitrification (Parton et al., 1996). Heinen (2006a) compared some 50 process-based denitrification models. Most simplified models are comparably based on a potential denitrification (D_p) weighted by a product of reduction functions due to nitrate content, degree of saturation, soil temperature and soil pH. The potential denitrification represents the soil microorganisms' capacity to reduce nitrates under non-limiting conditions, i.e. depending on the soil organic carbon content and microorganism populations. It can be measured by reproducing these optimal conditions on intact soil cores, for instance (Hénault and Germon, 2000), or deduced from CO_2 measurements that show the microorganisms' activity.

There is no consensus on the diverse reduction functions that are empirical and were calibrated from site-specific studies. Hence a universal simplified denitrification model is unlikely to exist and a chosen simplified model can only be used provided that parameters are calibrated for each location, with particular attention paid to determining the parameters of the saturation function, to which the model is the most sensitive (Heinen, 2006a).

$$D_a = D_p\, f_N\, f_S\, f_T = D_p \underbrace{\frac{N}{K+N}}_{f_N} \underbrace{\left(\frac{S-S_t}{S_m-S_t}\right)^w}_{f_S} \underbrace{Q_{10}^{(T-T_r)/10}}_{f_T} \tag{2}$$

D_a is the actual denitrification rate (mg N kg^{-1} d^{-1} or kg N ha^{-1} d^{-1}),

D_p is the potential denitrification rate (mg N kg^{-1} d^{-1} or kg N ha^{-1} d^{-1}),

f_N is a dimensionless reduction function for N, N is the nitrate content (mg N kg^{-1} or L^{-1}), K is the nitrate content where $f_N = 0.5$, f_S is the dimensionless reduction function for the dimensionless degree of saturation S, S_m is S above which $f_S = 1$ (in the remainder of this paper $S_m = 1$), St is a threshold value for S below which no denitrification occurs ($f_S = D_a = 0$), f_T is a dimensionless reduction function for soil temperature T, T_r is a reference T where D_p is determined at (mostly) $T_r = 20$ °C, and Q_{10} is an increase factor for a 10 °C increase in T. (Heinen, 2006b).

Testing a common simplified model (Eq. (2)) with eight Dutch data sets, the latter author showed that parameters differed across location and that no aggregation could be done based on soil type. The optimisation could not result in perfect prediction on the point scale, and was only good for cumulative denitrification for sand and loam soils, as under- and overestimations seemed to counteract in the long term (Heinen, 2006b). The model is very sensitive to errors in the estimates of the parameters. These errors (computed for 250 soil conditions and 25 parameter conditions) propagate in the prediction of denitrification, so that defining parameters with 10% accuracy would lead to a coefficient of variation in the relative denitrification rate of about 10% (Heinen, 2006a). A test on error propagation on parameter estimation with an artificial data set showed that w and Q_{10} were overestimated, while S_t and K were underestimated, with large coefficients of variation; the greatest uncertainty was found for K (Heinen, 2006b).

The NOE model (Hénault et al., 2005) uses similar equations to equation (2) to simulate denitrification and nitrification with Michaelis-Menten functions of nitrate and ammonium, respectively, two different saturation functions, and a common temperature function. The nitrification potential is not introduced explicitly. The NGAS model (Parton et al., 1996, 2001) is also an empirical denitrification and nitrification model that was developed using laboratory- and field-observed gas fluxes from different soils. N_2O fluxes are simulated using simple relationships controlled by soil saturation, texture, temperature, pH, respiration, and NO_3^- and NH_4^+ contents. Comparing these two sub-models, both coupled within the CERES crop model, NGAS appeared to be easier to operate as no site-specific data are needed; however, it was therefore also less accurate than NOE (Gabrielle et al., 2006).

Another comparison of four models simulating N cycling, CENTURY-NGAS (Del Grosso et al., 2001), DNDC (Li et al., 1992; Li, 2000), Nexpert (Engel and Priesack, 1993) and NASA-CASA (Potter et al., 1997), also showed discrepancies amongst results. Although all four models generally agreed on global N cycling rates, they presented a wide range of results concerning the different gas fluxes for both cumulated totals and temporal patterns; even when models agreed on N_2O emissions, then N_2, NO_x or NH_3 fluxes diverged (Frolking et al., 1998). An accurate modelling of soil moisture dynamics and the response of modelled denitrification to soil moisture appeared to be a key for reliable N_2O emissions (Frolking et al., 1998). Site-specific NOE parameters thus explained the performances better than NGAS. Nevertheless, site-specific parameters are not easily available, and pedotransfer functions would be needed to infer these parameters from basic soil characteristics (Gabrielle et al., 2006).

Up-scaling empirical models may come at the expense of prediction accuracy, whereas mechanistic models rely more on deterministic relationships based on fundamental knowledge on the interaction of predictor variables and modelled processes, which does not depend on the site or the study scale. Although mechanistic model structure is defined from the process knowledge, numerical fitting is also often used to parameterise such models, adding elements of empiricism (Farquharson and Baldock, 2008). DNDC is a microbial growth model that has been widely used to simulate N_2O emissions from diverse ecosystems; it has even been upgraded to deal with peak N_2O production due to freezing and thawing events (Pattey et al., 2007). This model tracks microbial activities in soils by computing the Nerst and Michaelis-Menten equations that describe interactions between microbial activities and the driving factors: soil redox potential, soil organic carbon and electron acceptors. At each time step, the concentration of oxygen and other electron acceptors determines the soil Eh and the consequent anaerobic volumetric fraction into which substrates are proportionally allocated. This defines conditions for microbial activities and the following substrate consumption that ends up with a change in soil Eh looping to the next time step (Li, 2007). Extensive testing of DNDC versus field measurements has demonstrated its ability to simulate N_2O and NO emissions but still with some significant uncertainty, which is at least partly due to the still limited knowledge about the ecosystem processes involved in C and N cycling (Sutton et al., 2007). This is confirmed by a test of DNDC with 22 long-term N_2O measurements. When N_2O simulations were in agreement with field measurements, the patterns of NH_4^+ and/or NO_3^- were not captured by the model or vice versa. In general, DNDC gave higher and more frequent N_2O peaks, leading to an average overestimation taking all measurements into consideration, and both (large) under- and overestimations when looking at individual results. Although statistics indicated that simulations were not optimal, the general agreement between simulated and measured N_2O total losses was better than with the three tested regression models, including Bouwman's emission factor (1.25% ± 1) used in the IPCC Tier 1 (1996). Improvement of the model would be necessary to use easily available data such as NO_3^- as a response variable and test mitigation scenarios, without having to measure N_2O to validate the model (Beheydt et al., 2007) Moreover, there are still recognised factors for modelling N_2O emissions that are not adequately understood (Farquharson and Baldock, 2008).

More precision on N_2O emissions is essential for compliance with the precise mandatory reduction targets. How to consistently reach a 10 or 20% greenhouse gas reduction target when emission estimate uncertainty is larger than 10 or 20%? Within the framework of local LCAs, the simulation of N_2O emissions with a process-based model may drastically reduce the uncertainty compared with emission factors, provided that the model performs correctly on the local scale, i.e. that parameters are well calibrated for the cultivation sites. Local

LCAs might be the uniquely consistent scale to produce useful estimates of N_2O emissions, as long as the understanding of all controlling factors remains insufficient to improve process-based models, which would make it possible to produce robust estimates on various scales.

5.2.2.5 CO_2 *emissions and land-use change in local LCA*

Carbon may accumulate in soils, mainly in organic form. The removal of atmospheric C by plants and storage of fixed C in stable fractions of soil organic C (SOC) is termed "soil C sequestration" (Lal, 2004a). Hence, SOC comes from dead plant parts (leaves, roots, etc.), plant rhizodeposition and organic matter applications (animal waste, etc.). However, its storage is not definitive because dead organic matter undergoes a series of biogeochemical transformations, including decomposition, and is eventually mineralised by microorganisms and released as CO_2 or CH_4, through respiration or fermentation, respectively (Arrouays et al., 2002). Carbon stock in soil is the result of a dynamic balance between "inputs" of organic matter and "outputs" due to mineralisation, erosion, leaching or combustion. The soil may act either as a carbon source or as a carbon sink, according to the ratio between inputs and outputs. Carbon inputs depend on primary production (controlled by edaphic factors such as solar radiation, temperature, water and nutrient availability), and organic matter returned to the soils (e.g. crop residue management). Carbon outputs depend on biotransformation rates, controlled by the organic matter composition and local physico-chemical conditions (temperature, moisture, oxygen, etc.). Biotransformation is also slowed down when organic matter is associated with mineral particles (particularly clay) which provide "physical protection" against the activity of microorganisms (Balesdent et al., 2000).

5.2.2.6 Impact of land-use change on soil organic carbon

On the global scale, the SOC pool is about 1550 GtC (+950 GtC inorganic carbon), exceeding by far the atmospheric (760 GtC) and biotic pools (560 GtC) (Lal, 2004a, b). Comparatively, geologic and oceanic pools represent 5000 GtC and 38 000 GtC, respectively (Lal, 2004a). Over the last 200 years or so, soils may have lost between 55 and 78 GtC because of land-use conversion and soil cultivation. On a global scale, the SOC pool to 1-metre depth has a predominant range of 50 to 150 tC ha^{-1}, but can reach 800 tC ha^{-1} in organic soils (Lal, 2004b). The SOC pool varies widely among ecological regions, being higher in cool and moist than warm and dry regions, and among ecosystems, SOC pools to 1-metre depth average for croplands 80–103 tC ha^{-1}, temperate grasslands 141–236 tC ha^{-1}, tropical, temperate and boreal forests, 122, 96–147, and 247–344 tC ha^{-1}, respectively (Lal, 2004a). SOC stocks are higher in cool temperate forests and wetlands where plant productivity is relatively high but the activity of soil microorganisms is slowed down by the temperature, whereas SOC stocks are lower in the wet tropics where organic matter turnover is rapid, and lowest in dry regions where plant growth is limited (Cowie et al., 2006).

Arrouays et al. (2001) provided estimates of average SOC stocks in France (to a depth of 30 cm), according to land use. Arable lands are characterised by relatively low stocks: 43 tC ha^{-1} on average, whereas permanent grasslands and forests (excluding litter) exhibit average stocks of nearly 70 tC ha^{-1}. These differences can be explained partly by a greater supply of carbon to the soil under grassland and forest (mainly from the roots but also from shoot litter), and partly by a shorter residence time of carbon under arable land (Soussana et al., 2004). Increased biodegradation rates in arable land may be due to multiple factors, e.g. changes in soil climate, nutrient availability and pH. A major factor would be the double impact of soil tillage that directly enhances mineralisation through increased oxygenation and de-protection of the organic matter by soil tillage (Balesdent et al., 2000; Germon et al., 2007). Indeed, a fraction of the organic matter included in micro-aggregates is physically protected from biodegradation, and inversely contributes to the aggregate cohesion through the binding of mineral particles by organic polymers and to the water stability of aggregates due to increased hydrophobicity (Chenu et al., 2000). Disruption of soil structure by tools and subsequent disruption of micro-aggregates by the action of rain expose the hitherto encapsulated C to biodegradation (Balesdent et al., 2000; Lal, 2004a). As organic matter is removed and dissolved from top soils, aggregates also become less stable, which could lead to synergetic losses of organic matter (Germon et al., 2007). A great part of carbon supply under grasslands is also due to larger root turnover and rhizodeposition than under arable crops. This process favours carbon storage because direct incorporation into the soil matrix leads to a higher stabilisation by physical protection and root litter is also chemically more stable (Soussana et al., 2004). In particular, grasses whose roots reach the deep part of the soil profile well below the plough layer make it possible to sequester C that is less prone to oxidation and loss (Fisher et al., 1994).

Kinetics of SOC accumulation or release following land-use change is non-linear and asymmetrical (Arrouays et al., 2002; Seguin et al., 2007). Variations are more rapid during the first years after land-use or land-management change and reach a plateau after several decades. The time taken to reach this new equilibrium (sink saturation) is highly variable, around 100 years in a temperate location, up to several centuries in boreal regions. Moreover, if the land-use change is reversed, the accumulated SOC will be lost, usually more rapidly than it was accumulated (Smith, 2004). This could be partly explained by the synergetic effect mentioned above. Arrouays et al. (2002) provided an estimation of mean carbon changes due to land-use change in France (Fig. 16). This estimate is based on the use of an exponential function, fitted with data available in the literature and French average soil C stocks at equilibrium for the main types of land use. According to this study, the mean carbon change implied each year over a 20-year period by converting forest to annual crop is about −0.75 tC ha^{-1}, and -0.95 ± 0.3 tC ha^{-1} by converting permanent pasture to annual crop. On the contrary, conversion of arable land to forest or grassland leads to a mean annual

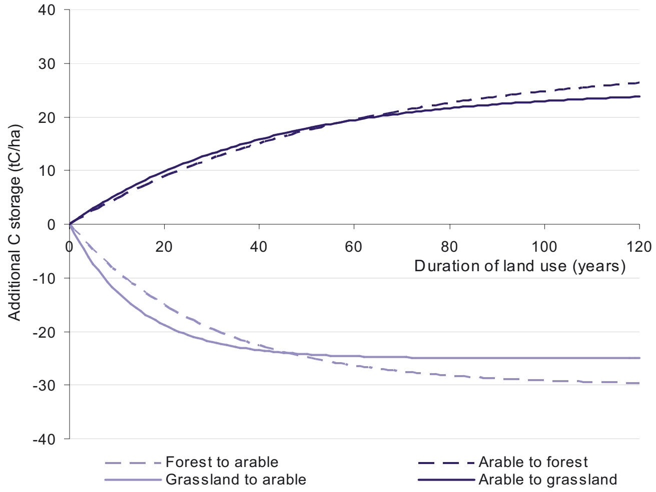

Figure 16. Change in soil carbon stocks associated with land use change. These are modal values for mainland France. The 95% confidence interval of these values is about ± 40% (Arrouays et al., 2002).

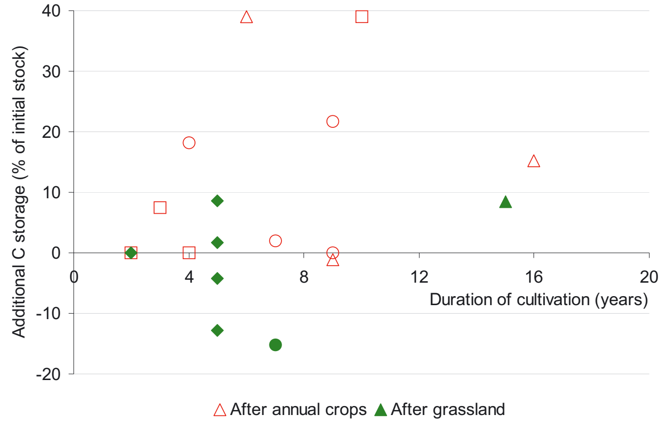

Figure 17. Observed changes in soil carbon stocks associated with the introduction of perennial energy crops (SRC poplar or willow: circles, miscanthus: triangles and switchgrass: squares/diamonds) after annual crops or grassland. Drawn from data in Garten and Wullschleger, 1999; Jug et al., 1999; Ma et al., 2000a, b; Kahle et al., 2001; Zan et al., 2001; Frank et al., 2004; Hansen et al., 2004; Clifton-Brown et al., 2007).

soil carbon storage of 0.45 ± 0.25 tC ha^{-1} for forest and 0.5 ± 0.25 tC ha^{-1} for grassland, over a 20-year period.

The variability of estimates is due mainly to the diversity in climatic conditions and soil characteristics (Seguin et al., 2007; Soussana et al., 2004). In a meta-analysis of 74 international publications, Guo and Gifford (2002) confirmed the high impact on SOC of grassland or forest conversion to cropland. According to this study, soil carbon stocks decrease on average by 42% after conversion of forest to crop and 59% after conversion of grassland to crop.

Introduction of perennial energy crops in current annual crop systems may increase carbon sequestration, due to the lack of soil tillage during the crops' growing cycle (typically 15–20 years), their high biomass production and pre-harvest losses, and their extensive root system (Lemus and Lal, 2005). In a field experiment in southern Quebec, Zan et al. (2001) measured a total root carbon content 4 to 5 times greater for 3-year-old SRC willow and switchgrass than for corn. High below-ground biomass (rhizomes and roots) was also measured for miscanthus. Below-ground biomass ranged, for example, from 15 to 25 tDM ha^{-1} for 4–9-year-old miscanthus in Germany, corresponding to 7.6–10.2 tC ha^{-1} (Kahle et al., 2001). It is thus expected that, as observed under grassland, root turnover and rhizodeposition should be a major carbon input under perennial energy crops. Perennial energy crops are usually harvested in late winter or early spring with high dry matter content. This practice causes pre-harvest losses, mainly by leaf senescence. Mean pre-harvest losses during 3 years of miscanthus cultivation were 4.5 tDM ha^{-1} yr^{-1}, corresponding to about 2 tC ha^{-1} yr^{-1}, in the same field experiment in Germany (Kahle et al., 2001). Several authors have evaluated impacts of perennial energy crops (short rotation coppice (SRC), miscanthus, switchgrass) on SOC, using field measurements in long-term experiments (Fig. 17).

The results of these experiments are highly heterogeneous, which is probably due partly to the diversity of climatic, pedological and agricultural conditions, and partly to differences in measurement methodology (e.g. soil sampling depth). However, in general terms, conversion of annual crops to perennial energy crops seems to increase carbon sequestration, which may not be the case when perennial energy crops are introduced after grassland. Also, there is no clear difference between perennial energy crops (short rotation coppice, miscanthus, switchgrass).

Land clearing (i.e. conversion of forest or grassland to arable crops) can lead to a large release of CO_2. To evaluate this effect, it is necessary to take into account the different carbon pools, i.e. not only the soil carbon pool but also the above-ground and below-ground biomass carbon pools. According to Fargione et al. (2008), the amount of CO_2 released during the 50 years following land conversion would be 737 tCO_2 ha^{-1} in the case of a tropical forest converted to soybean in Brazil, and 134 tCO_2 ha^{-1} in the case of natural grassland converted to corn in the US. It represents a "carbon debt" that should be included in LCA when biofuels are introduced after land clearing. The additional carbon sequestration by perennial energy crops compared with annual crops should also be taken into account in LCA. However, it is important to keep in mind that any carbon sequestration in soil is finite and reversible (Powlson et al., 2005).

5.2.2.7 Impact of agricultural practices on soil organic carbon

Agricultural practices can modify SOC levels in arable lands, by changing the amount of carbon supply to the soil or by changing the residence time of carbon in the soil. Effects of soil management practices on carbon sequestration have been widely studied. No-tillage generally implies an increase in soil carbon levels, compared with tillage (see review by Arrouays

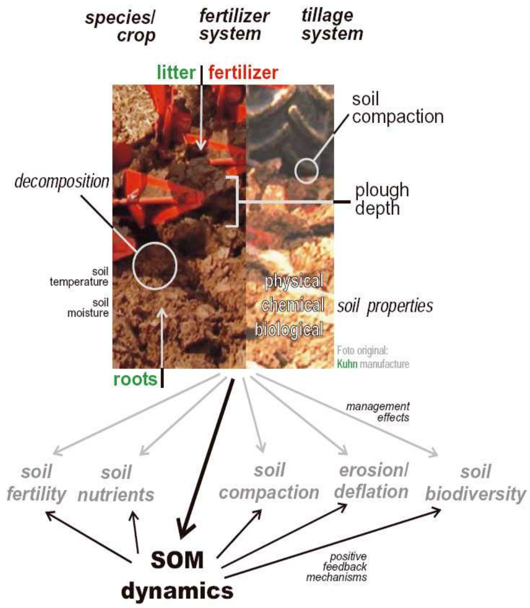

Figure 18. Soil organic matter is the key indicator in sustainable soil management (in Van-Camp et al., 2004).

et al., 2002; Germon et al., 2007). The mean increase in carbon sequestration due to no-tillage or reduced tillage was estimated at 0.2 ± 0.13 tC ha^{-1} yr^{-1}, over a 20-year period. However, there is no consensus about the magnitude of the differences between conventional tillage and no-tillage: some authors reported negligible differences; others found considerable differences (Arrouays et al., 2002). Origins of this variability are not well known.

According to Balesdent et al. (2000), comparing carbon stocks between tillage and no-tillage treatments causes some difficulties, due to changes in bulk densities and carbon repartition along the soil profile. Furthermore, no-tillage may refer to different degrees of "conservation tillage", such as direct sowing or non-inverting ploughing. Conservation tillage also encompasses intermediary techniques referred to as "reduced tillage", etc. Due to the complexity of soil organic matter (SOM) dynamics (Fig. 18), the varying depths and degrees of soil disturbance and the varying duration of the treatments across studies can also imply discrepancies amongst conclusions. The impact of conservation tillage on CO_2 emissions also varies amongst studies. Although CO_2 measurements directly after soil tillage mostly showed higher emissions than under direct sowing, impacts in the long term are less clear. CO_2 long-term measurements under conservation tillage are still lacking and little is known about CO_2 emissions when the new SOC equilibrium is reached under conservation tillage (Germon et al., 2007).

The use of catch crops over intercropping periods can represent an interesting option in terms of carbon sequestration. Arrouays et al. (2002) modelled a potential increase of 0.15 ± 0.08 tC ha^{-1} yr^{-1} over a 20-year period, for an annual incorporation of catch crops. In a field experiment with spring barley in Askov (Denmark), the mean difference in SOC between no catch crop and catch crop treatments was 1 tC ha^{-1} after 10 years[49], corresponding to an annual increase of 0.1 tC ha^{-1} yr^{-1} due to annual catch crop incorporation in soil (Thomsen and Christensen, 2004). Crop residue management can also impact soil organic carbon. Saffih-Hdadi and Mary (2008) compiled nine well-documented long-term field experiments, which compare effects of systematic removal or incorporation of cereal straws on SOC evolution. They differed in climate, soil type, carbon input and duration (from 12 to 35 years). The measured SOC increase due to straw return (as compared with straw removal) varied from 0.078 tC ha^{-1} yr^{-1} to 0.385 tC ha^{-1} yr^{-1}, corresponding to 4.2–19.1% of added straw carbon. Climate influenced the efficiency of straw incorporation in SOC. This incorporation is much more efficient under cold climates, where it can reach up to 0.90% of the initial SOC content compared with 0.53% under warm climates. Systematic removal of straw for bioenergy purposes will then lead to a decrease in SOC content. Using a simple carbon dynamics model called AMG, Saffih-Hdadi and Mary (2008) simulated the impact of straw removal one year out of two in nine experimental sites. After 50 years, it would reduce carbon stocks by 2.5–10.9% of the initial SOC, depending principally on the experiment (soil, climate, productivity).

As SOC sequestration is provisional, it can only play a minor role in climate change mitigation. The maximum global SOC sequestration potential of 0.9 ± 0.3 GtC yr^{-1} over 50 years (Lal, 2004a) could contribute to a maximum of 2–5% towards reducing the carbon emission gap under the highest emission scenarios (Smith, 2004). However, given the drastic CO_2 reduction needs to meet targets, it is already crucial that agricultural practices should aim to prevent carbon losses as much as possible, notably net CO_2 emissions due to land-use change, then to implement practices that enhance SOC sequestration. A better understanding of the SOC stabilisation in deeper soil layers could also open up new options in order to increase C sequestration.

5.2.2.8 Land-use impacts in LCA

"Land-use impacts are the 'amount' of land quality not present in a certain area due to the studied system, compared to a situation where the studied system had not been established" (Milà I Canals et al., 2007).

The major environmental importance of land-use impacts contrasts with the lack of consensus on this area within the field of LCA. As a result, the issue is seldom included in LCA and the credibility of LCA results is insufficient for many stakeholders. Lack of consensus comes at least partly from the failure to recognise the value judgements behind the assessment methodology. These value judgements include the following: what are the functions of land that need protection, which are the thresholds? What are the time perspective and reversible impacts? What are the future or alternative land uses? Which indicators represent the impact pathways? (Milà I

[49] Mean value of the 4 straw restitution treatments (Thomsen and Christensen, 2004).

Canals et al., 2007). Focusing on bioenergy chain LCAs, many studies have lately put emphasis on the necessity to develop a methodology within the LCA tool to take into account the impacts of land-use change on the ecological functions of land. Among others, the impact of deforestation on biodiversity and CO_2 emissions, the impact of straw removal on soil fertility, the impact of perennial crops for future land uses, etc., are examples that show the importance of such factors as part of sustainability criteria.

Nevertheless, potential impacts of land-use change are difficult to assess for mainly two reasons. First, impacts due to land-use change have to be characterised in comparison with unchanged land use. When considering agricultural land use, in principle only degradation caused by the management practice during the cultivation period should be allocated to the crop harvested (Mattsson et al., 2000). This implies defining a reference scenario and a time frame for the occupation of land or recovery period. But one reference scenario is sometimes not sufficient to cover the range of possibilities when it comes to dealing with the use of new land areas or longer time frames such as in the cases of crop rotations or the cultivation of perennial crops. Second, LCA methodology based on equivalency factors is hardly adaptable for land-use change. Indeed, aggregation of parameters such as soil organic matter and landscape values, for example, is difficult so that the land-use impact category should be less aggregated than other impact categories in LCA (Mattsson et al., 2000), leading to the complexity of dealing with two approaches and a resulting mix of quantitative and qualitative information. Other approaches and tools, such as Environmental Impact Assessment, may provide more detailed information than LCA on effects of different land managements. "However, LCA is the appropriate tool to bring a life cycle perspective to support complex decisions involving different land uses, and, consequently, it should incorporate a measure of the different impact pathways affected by land use" (Milà I Canals et al., 2007).

Many references focus on suggesting indicators to include the effects of land use on productivity and biodiversity, although the practical implementation of such sets of indicators is seldom checked with a consistent framework (Milà I Canals et al., 2007). Most proposed methodologies use a number of indicators that are largely submitted by the availability of data (Anton et al., 2007). Mattsson et al. proposed to divide "land-use change" into three sub-categories: (1) soil fertility, with a set of 7 indicators, (2) biodiversity, and (3) landscape values (Mattsson et al., 2000). Schenck and Vickerman (2001) also gave a list of indicators for the assessment of impacts on biodiversity. Impacts on biodiversity, for instance, are currently considered in LCA through damage to the biotic environment, or concurrence of species (Jolliet et al., 2004); the effects considered have been traditionally limited to those caused by changes in the chemical composition of the environment (toxicity, eutrophication, etc.) (Milà I Canals et al., 2007). The World Resource Institute[50] showed that the greatest biodiversity losses are derived from changes in land use, rather than to any chemical impacts (Schenck et al., 2001). Some of the latest methods for LCA thoroughly address land-use impacts, but fail to include effects of occupation or transformation on the resource aspect of land (Milà I Canals et al., 2007). Despite the availability of indicators, there is still a lack of consensus on which is the most ideal indicator for evaluation (Anton et al., 2007). Comparing indicators from Köllner (2001) and Weidema and Lindeijer (2001), it appears that further research is still needed to refine them so that they could deal with more specific ecosystems and geographical areas (in Anton et al., 2007).

CO_2 emissions due to biofuel combustion are commonly not included within the system boundary since the fuel is considered as carbon-neutral; indeed, the released carbon during combustion (CO_2) had been fixed from the atmosphere in the first place. This exclusion is hence justified when comparing a biofuel chain starting from the biomass production with a fossil fuel chain, because the actual carbon cycle is fully considered. However, when the delay between carbon capture and sequestration by the plants and re-emission is longer (land-use change, waste treatment, woody biomass, etc.) this assumption may lead to wrong conclusions. In this sense, Rabl et al. (2007) recommended that emission and removal of CO_2 ought to be counted explicitly at each stage of the life cycle. In this way, the LCA is furthermore consistent with the "polluter pays" principle, which implies that each greenhouse gas contribution should be allocated to the causing agent. For example, CO_2 from woodfuel for heating should be taxed as CO_2 from oil heating is, and a credit for CO_2 removal then only paid when and where the wood is replaced by new growth (Rabl et al., 2007). Furthermore, this stage-CO_2 accounting would be useful in the framework of an implementation of sustainability criteria for biofuel chains at an international level, including the accounting of a carbon debt or credit in the case of a change in SOC due to land-use or land-management change.

Beyond uncertainties linked to estimates of carbon stock and its changes due to land-use or management changes, the main difficulties arise from the definition of prospective land-use scenarios for comparative LCA. Areas impacted by land-use or management changes can be part of a crop rotation or more complex combined land-use patterns including indirect impacts through crop displacements. Soya in Brazil, for instance, is established on grasslands, pushing cattle pastures further into forests[51]. Therefore, dependence between impacted areas must be considered within a matrix of land-use changes across a sphere of influence that must be defined. This is particularly relevant in the case of SOC storage in order to take into account the fact that dynamics are long-lasting and often reflect transition states from past changes, and that kinetics are reversible and asymmetric. Methods to estimate the impacts must be adapted to these temporal scales, otherwise results could be biased by the approximation of impact differences between two instantaneous pictures of land-use patterns (Arrouays et al., 2002). A better accounting of the impact of land-use change on the soil quality and direct CO_2 emissions is

[50] WRI: Guide to World Resources 2000–2001: People and Ecosystems: The Fraying Web of Life, Elsevier, New York, 2002.

[51] Dr Emily Boyd, 25/11/2005, http://www.scidev.net/en/opinions/emissions-trading-cannot-solve-amazon-deforestatio.html.

necessary for biofuel LCA. Emphasis should be put on defining a harmonised methodology to include some indicators on soil quality. However, accurate estimates may be limited to local LCA given the complexity to encompass the temporal and spatial dimensions of the impact of land-use changes and the data needs. In order to simulate soil C change in bioenergy projects, it would be recommended to establish the baseline labile and recalcitrant C stocks through measurements and to model C dynamics over the land-use duration (Cowie et al., 2006). Furthermore, more research would be necessary to also introduce within LCA quantified impacts of land-use change on the albedo, surface energy balance and water cycle, and their consequences on climate change. Indeed, agriculture significantly affects climate through greenhouse gas emission and absorption, and modifications of surface properties, which act directly on different spatial scales. To date, however, the complete evaluation of the net impact of agriculture on climate through the modification of the natural environment is still not feasible (Seguin et al., 2007).

5.3. Biofuel greenhouse gas balances

Most studies have found that the use of 1st generation biofuels results in emission reductions of 20 to 60% of CO_{2eq} relative to fossil fuels. Expected reductions for future commercialised 2nd generation biofuels are in the range of 70 to 90% of CO_{2eq} relative to fossil fuels (FAO, 2008a). The large range of emission reductions for the 1st generation biofuels is due to various types of feedstock and conversion processes, and to the different sites of production and consumption. Varying LCA assumptions also explain that greenhouse gas balances of a given biofuel chain in one region may be variable (see part 5.1). Finally, field emissions in particular are complex to assess and imply further disparities amongst studies (see part 5.2). Greenhouse gas savings are therefore often presented as ranges; it does not make much sense to give a list of mean values for each biofuel chain. However, Table IV presents some of the main published studies to put in contrast varying results due to changes in co-product handling, within a study and amongst studies. This overview completes the data in Figure 11 and Table II.

A sensitive analysis of the N_2O emission factors showed that these assumptions critically impact the balance. Greenhouse gas emissions rise from 40 to 50% for methyl esters and pure vegetable oils by using Bouwman's IPCC (1995 Guidelines) emission factors instead of those from Skiba (1996). The resulting greenhouse gas savings fall, for instance, to −55.5% and −66% for the rapeseed pure oil and methyl ester, respectively. Although Bouwman's factors may be more accurate because the regression was based on more data sets on a wider range of soil diversity, whereas Skiba's factors were established for the UK's soils, extrapolation from any linear model implies a high uncertainty on the results due to the site- and time-dependence of field emissions. N_2O emissions in the JRC/EUCAR/CONCAWE study are likely to be more accurate as they were simulated with the DNDC model (version 82N) combined with the LUCAS land-cover survey model. The resulting emission factors, moreover, include N_2O indirect emissions from leached N. However, as the study assessed biofuel chains at the European level, the simulations were used to determine new emission factors through regression models. The averaged crop emission factors finally hardly give an approximate of total N_2O emissions at a national level, while emissions are too variable on such a scale to help distinguish between biofuel chains and co-product options at the local level. Nevertheless, biofuel chains with a valued co-product make it possible to save a lot of greenhouse gases, especially if biomass production is optimised to reduce field emissions as much as possible.

5.3.1. Prospects for reducing greenhouse gas emissions from biomass production

Agriculture greenhouse gas emissions increased by 10% between 1990 and 2000 (Stern, 2006); CH_4 and N_2O emissions alone increased by 17% between 1990 and 2005; 88% of these emissions are explained by three sources: biomass burning, enteric fermentation and soil N_2O emissions (Smith et al., 2007). Considering the increase in demand for agricultural feedstock, global agricultural greenhouse gas emissions are expected to rise by almost 30% in the period to 2020, with almost two-thirds of this increase coming from Africa, Latin America and China, half of it due to the use of fertiliser on agricultural soils (Stern, 2006). World nitrogen fertiliser demand is forecast to increase at an annual rate of about 2.6% until 2012, East Europe and Asia contributing to 81.9% of this increase (FAO, 2008b). N_2O emissions alone are projected to increase by 35–60% by 2030 due to increased use of fertilisers and animal manure production (FAO, 2002).

Drastic savings in agricultural greenhouse gas emissions are needed, and agricultural practices are the key to significantly reducing agricultural greenhouse gas emissions. Options for mitigating agricultural greenhouse gas emissions fall into three categories based on the underlying mechanism: (1) reducing emissions, (2) enhancing removals from the atmosphere, and (3) avoiding (or displacing) emissions (Smith et al., 2008). The global technical mitigation potential[52], including all gases for the two first categories, by 2030 is estimated to be some 5.5–6 $GtCO_{2eq}\ yr^{-1}$ ([53]), mainly through reduction of CO_2 emissions[54] (89%). The economic potential would vary between 1.5 and 4.3 $GtCO_{2eq}\ yr^{-1}$ at carbon prices from 20 up to 100 $US\$.tCO_{2eq}^{-1}$, respectively. At the same carbon price levels, some more 0.6–16 $GtCO_{2eq}\ yr^{-1}$ could be avoided, by substituting fossil fuels with bioenergy

[52] Mitigation potentials for CO_2 represent the net change in soil carbon pools which were derived from about 200 studies; the emission ranges for CH_4 and N_2O were derived using the DAYCENT and DNDC simulation models. All estimated potentials are followed by the mention *medium agreement, low evidence.*

[53] About 20% of 1990s global greenhouse gas emissions, or 5%, 9% and 14% for the three different economic potentials.

[54] Notably from SOC sequestration due to restoration of organic soils; 9% CH_4, 2% N_2O.

Table IV. Biofuel chains and greenhouse gas savings. Ethanol is compared with gasoline, and biodiesel with fossil diesel. The results are given as they were published and without any harmonisation in background assumptions, except for units, e.g. emissions for the fossil fuels per MJ vary across studies.

Biofuels	Regions	Feedstock	Co- product handling			GHG/MJ compared with fossil fuels	References
			Mass allocation (% applied to biofuels)	System expansion (substitutes)	Energy allocation (% applied to biofuels)		
Ethanol	Brazil + shipped to EU	Sugar cane	–			−66%	JRC/EUCAR/ CONCAWE, 2008*
				Excess bagasse for heat (diesel)		−81.6%	
	France	Wheat	DDGS (43) Straw 14% (96)	Straw 86% (IF)		−60%	ADEME/DIREM, 2002
		Sugar beet	Sugar, pulps (80)	Crop residues, molasses, slop (IF)		−52%	
	EU	Wheat		DDGS as animal feed (feed wheat and soya meal) + straw (IF)		−14.3%	JRC/EUCAR CONCAWE, 2008
				DDGS as fuel + straw (IF)		−28.6%	
				DDGS as feed (idem) + straw CHP		−62.7 %	
				DDGS as fuel + straw CHP		−77%	
				DDGS to biogas (fuel)		−60%	
		Sugar beet		Pulps to animal feed (soya meal)		−47%	
				Pulps to animal feed (idem) + slops to biogas (fuel)		−65%	
				Pulps + slops to biogas/heat (idem)		−80.5%	
	USA	Maize		DDGS (soybean meal, corn for cattle feed) Corn gluten, meal and feed, corn oil (whole corn, nitrogen-in-urea, soy oil)	15% of net energy allocated to fossil fuel co-products	−14%	Farrell et al., 2006
Biodiesel	France	Rapeseed RME	Meal (46), acid oils (97), glycerol (88)	Crop residues (IF)		−70%	ADEME/DIREM, 2002
		Sunflower SME	Meal (49), acid oils (97), glycerol (88)	Crop residues (IF)		−75%	

Biofuels	Regions	Feedstock	Co-product handling			GHG/MJ compared with fossil fuels	References
			Mass allocation (% applied to biofuels)	System expansion (substitutes)	Energy allocation (% applied to biofuels)		
Biodiesel	EU	Rapeseed RME		Glycerine to chemicals (propylene glycol) + meal to animal feed (soya meal from imported soybeans)		−45.5%	JRC/EUCAR/CONCAWE, 2008
				Glycerine and meal to animal feed (soya meal from imported soybeans)		−38.8%	
				Glycerine and cake to biogas (fuel)		−63%	
		Sunflower SME		Glycerine to chemicals (propylene glycol) + meal to animal feed (soya meal from imported soybeans)		−66%	
				Glycerine and meal to animal feed (soya meal from imported soybeans)		−59%	
				Glycerine and cake to biogas (fuel)		−80%	
Pure vegetable oil	France	Rapeseed	Meal (46), acid oils (97), glycerol (88)	Crop residues (IF)		−77.5%	ADEME/DIREM, 2002
		Sunflower	Meal (49), acid oils (97), glycerol (88)	Crop residues (IF)		−83%	
Cellulosic ethanol (pilots)	USA 2030	Maize stover			Electricity (79.6−91.2)	−86% −89%	Wu et al., 2006
		Forest residues			Electricity + chemicals (91.45)	−85%	
	USA	Switchgrass		Electricity (local grid electricity)		−88%	Farrell et al., 2006

Notes: IF = industrial fertiliser

* Horizon 2020−2030 + N_2O emission factors (kgN-N_2O ha^{-1}): sunflower: 1.11; wheat: 2.23; sugar beet: 2.79; rapeseed: 3.12.

** Horizon 2005 + Skiba's N_2O emission factors (%kg N fertiliser ha^{-1}) sunflower: 0.8; wheat: 0.5; sugar beet: 1.60; rapeseed: 0.5. Greenhouse balances consider complete combustion of the fuels and zero CO_2 emissions from biofuels linked to this combustion: so-called credit for renewable combustion CO_2.

Table V. A list of proposed measures (also referred to in literature as "Recommended Management Practices" RMPs) for mitigating GHG emissions from agricultural ecosystems, their apparent effects on reducing emissions of individual gases (mitigative effect) and an estimate of scientific confidence that the proposed practice can reduce overall net emissions (Smith et al. 2008).

		Mitigative effects[a]			Net mitigation[b] (confidence)	
Measure	Examples	CO_2	CH_4	N_2O	Agreement	Evidence
Cropland management	Agronomy	+		±	***	**
	Nutrient management	+		+	***	**
	Tillage/residue management	+		±	**	**
	Water management (irrigation, drainage)	±		+	*	*
	Rice management		+	±	**	**
	Agroforestry	+		±	***	*
	Set aside, land-use change (LUC)	+	+	+	***	***
Grazing land management/pasture improvement	Grazing intensity	±		±	*	*
	Increased productivity (e.g. fertilisation)	+		±	**	*
	Nutrient management	+		±	**	**
	Fire management	+		±	*	*
	Species introduction (including legumes)	+		±	*	**
Management of organic soils restoration of degraded lands	Avoid drainage of wetlands	+	–	±	**	**
	Erosion control, organic amendments, nutrient amendments	+		±	***	**
Livestock management	Improved feeding practices		+		***	***
	Specific agents and dietary additives		+		**	***
	Longer term structural and management changes and animal breeding		+		**	*
Manure/biosolid management	Improved storage and handling		+	±	***	**
	anaerobic digestion		+	±	***	*
	More efficient use as nutrient source	+		+	***	**
Bioenergy	Energy crops, solid, liquid, biogas, residues	+		±	***	**

[a] '+' denotes reduced emissions or enhanced removal (positive mitigative effect); '–' denotes increased emissions or suppressed removal (negative mitigative effect); '±' denotes uncertain or variable response.

[b] A qualitative estimate of the confidence in describing the proposed practice as a measure for reducing *net* emissions of GHGs, expressed as CO_2 equivalence. 'Agreement' refers to the relative degree of agreement or consensus in the literature (the more asterisks, the higher the agreement); 'Evidence' refers to the relative amount of data in support of the proposed effect (the more asterisks, the greater the amount of evidence).

generating electricity, i.e. the above-mentioned third category (Smith et al., 2008).

The uncertainty on the savings due to bioenergy is especially large because the net benefit in CO_2 reduction from fossil CO_2 displacement will depend on the greenhouse gas balance over the whole bioenergy life cycle, including direct emissions during the biomass production. Therefore, production systems of biomass for energy should necessarily contribute to the two first above-mentioned categories. Table V presents the main measures for mitigating greenhouse gas emissions from agroecosystems. It appears that the mitigative effects of these measures on N_2O emissions are mostly uncertain, due to the lack of knowledge related previously in this review. However, since N availability is the bottom line for N_2O emissions, management practices that will improve the fertilisation efficiency can help reduce N_2O emissions as detailed below. Other practices could further reduce the total agricultural greenhouse gases, but the trade-off between the different gases is still unclear.

5.3.2. *Improving fertilisation efficiency*

Direct field emissions can be reduced by improving fertilisation efficiency, i.e. combining reduced input and increased uptake and production. The nutrient balance expresses this difference between the total quantity of nutrient inputs entering an agricultural system, and the quantity of nutrient outputs leaving the system, in terms of kilograms of nutrient surplus (deficit) per hectare of agricultural land per year. Any surplus represents potential losses of nutrients into the environment and the correlated risk of polluting soil, water and air, whereas a deficit can reveal environmental pressures such as declining soil fertility (OECD, 2008).

Focusing on nitrogen balance, the situation is quite contrasted between OECD countries and developing countries where fertilisation inputs are much lower. Nitrogen balance is in surplus in all OECD countries, whereas in Sub-Saharan Africa notably it is in deficit, as in Kenya, Mali and Ghana, for instance (OECD, 2008; Roy et al., 2003). In two-thirds of OECD countries the nitrogen surpluses decreased between the 1990s and 2000s, whereas in a few countries such as Canada, New Zealand, Portugal and the USA it increased, notably due to the rise in fertiliser use and livestock numbers. The higher use of fertiliser is in part explained by the expansion in crop production together with a shift in cropping patterns to crops requiring higher inputs per kg of output, such as from wheat to maize in Australia and the United States. Some countries, whose surpluses diminished, still have amongst the highest surpluses. This is the case for Korea, Japan, Belgium, Denmark and the Netherlands, for instance, which have rather low nitrogen efficiency[55], between 30 and 50%, below the averages in the OECD and the EU-15, respectively, 55%–60% (OECD, 2008). Reduced nitrogen surpluses were notably correlated with the adoption of "nutrient management and environmental farm plans", and the improvement of N-use efficiency linked to reduced inorganic fertiliser input per unit of crop output, a closed storage system, and an optimisation of the timing and spreading of manure (OECD, 2008). Indeed, nitrogen efficiency can be increased by optimising the crop's natural ability to compete with processes whereby plant available N is lost, i.e. by matching the N supply with crop demand better [e.g. optimised split application schemes and doses, foliar application, application during stem elongation or later (Recous, 2001)] and adapt it to specific risks; e.g. avoiding nitrate application in case of leaching risk (Recous, 2001), applying fertiliser below the soil surface in case of possible volatilisation, choosing to apply nitrates when nitrification is more likely to happen, or ammonium-based fertilisers when it is denitrification; for instance, during seasonal precipitations (Mosier, 1996).

Integrated fertilisation management, including the introduction of catch crops or legumes in the crop rotation to uptake or fix nitrogen, respectively, and the incorporation of crop residues or manure spreading will also influence N_2O emissions. Indeed, fertilisation efficiency strategies must be developed considering both the cropping and intercropping cycles. Whereas high N inputs may be well correlated with high N_2O emissions during the vegetation period, over the year unfertilised plots can also emit high quantities of N_2O, notably depending on the amounts and the C/N ratio of crop residues in the soil (Kaiser et al., 1998). Organic amendments can influence N_2O emissions in three ways: (1) the amount and recalcitrance of the N supply, (2) those of the C supply, and (3) local increases in the oxygen consumption (Velthof et al., 2002). Through the crop residue C/N ratio, it is possible to influence the nitrogen mineralisation-immobilisation turnover[56] by microorganisms (Recous, 2001; Velthof et al., 2002), which determines the evolvement of the soil N pool, including the competition for the N substrates between plants and microorganisms, and potential N_2O emissions. A narrow C/N ratio and high contents of easily mineralisable N[57] in crop residues would favour N_2O emissions (Velthof et al., 2002). While organic amendments with high N content may accentuate N_2O, NH_3 and CH_4 emissions, they may also contribute to the rise in soil organic carbon, especially in the form of stabilised manure and recycled organic compost, that contain a greater fraction of recalcitrant carbon than fresh green manure, i.e. fresh crop residues (Larsson et al., 1998; Lal, 2004c; Cowie et al., 2006). Moreover, there could be possible reductions of N_2O and NH_3 emissions in the field, depending on the soil type, through digestion of the fresh green manure and slurries before application (Oenema et al., 2005). The initial N content of composts may be more determining for leaching risk and fertilising value than amendments' stability but this latter could play an interesting role in optimising fertiliser application timing and crop N recovery (Gabrielle et al., 2005).

Although nitrogen efficiency is not identically defined across the literature, authors agree that it could and should be widely improved in future (Crutzen et al., 2008; OECD, 2008; Galloway et al., 2008). In 2002–2004, nitrogen efficiency reached 70–78%, for instance, in Italy and Greece, respectively (OECD, 2008). This issue is especially important as N-intensive biofuels could cancel out any CO_2 savings due to N_2O and NO_x emissions. As critical examples, US corn and Brazilian sugar cane production have low N efficiency; only 30% of N input ends up in sugar cane tissues (Galloway et al., 2008). An increase in the nitrogen efficiency from 40% to 60%, resulting in the assumption that 3 instead of 5% of N input would be lost as N_2O over the whole nitrogen cascade, makes the maize ethanol and rapeseed biodiesel become carbon-neutral and beneficial, respectively (Crutzen et al., 2008).

A more efficient use of fertiliser would lead to direct reduction of field emissions, while at the same time it would also imply reduction of upstream industrial emissions during the fertiliser production and spreading in the fields. It could lead to a decrease in industrial Nr creation of about 15 MtN per year, i.e. 8% of total Nr created in 2005. The same amount could also be saved through improved animal management strategies (Galloway et al., 2008).

5.3.3. Other cultural practices

Basically, recommended management practices (RMPs) aim to improve the agroecosystem productivity while maintaining or reducing the input levels. In general, the choice of resistant varieties and an optimal adaptation of crop rotations to site-specific conditions will make it possible to reduce the greenhouse gases by combining high yields and low inputs. Farming operations should be limited as far as possible, since all inputs also imply an environmental cost starting with fossil fuel CO_2. In some cases though, the final greenhouse gas benefits will depend on the balanced gain in CO_2 sequestration due to enhanced biomass productivity over the cost in CO_2 emitted by the operations of irrigation, drainage or tillage, etc. and in other greenhouse gases. Reducing fallow will, for instance, imply higher energy, but this cost appears to be globally offset by greater benefits (Grant et al., 2004; Lal, 2004a). While some energy inputs are unavoidable, improved energy efficiency in agriculture could deliver an additional 0.77 $GtCO_{2eq}\ yr^{-1}$ mitigation potential by 2030 (Smith et al., 2008).

The opportunity to save energy input by reducing soil tillage is the major factor that has first fostered the change from conventional tillage to reduced/conservation tillage or

[55] Nitrogen efficiency measured as the percentage ratio of total nitrogen uptake by plants and forage (tonnes) over the total nitrogen available from fertiliser, livestock manure and other nitrogen inputs (tonnes).

[56] Through mineralisation N is made available for the plants, through immobilisation/organisation N is consumed for the development of the microorganisms.

[57] Easily mineralisable N is usually more abundant in fresh green material than in straw (Velthof et al., 2002).

no-tillage. In 1999, the worldwide area under no-tillage was approximately 50 Mha, representing 3.5% of total arable land (Smith et al., 2007). Conversion from conventional tillage to no-tillage can reduce emission of farming operations by 110 to 130 kg CO_2 ha^{-1} per season (Lal, 2004b). Since soil disturbance caused by soil tillage enhances SOC losses through decomposition and erosion (see part 5.2), reduced or no-tillage often also results in SOC gain besides fossil CO_2 savings (Robertson et al., 2000; Lal, 2004a; Seguin et al., 2007; Smith et al., 2008). Such practices are, however, frequently combined with periodical tillage, which reverses the SOC storage trend, thus making the assessment of the greenhouse gas uncertain (Smith et al., 2007). SOC sequestration through reduced soil tillage is an explicit illustration of greenhouse gas trade-offs, or hidden costs (Lal, 2004c) that are likely to obscure the real impact of a mitigation measure. Indeed, while reduced tillage may imply SOC sequestration and globally less CO_2 emissions, it adversely can lead to higher emissions of N_2O and CH_4, though not always (Robertson et al., 2000; Six et al., 2004; Lal, 2004b; Grant et al., 2004; Chatskikh and Olesen, 2007; Oorts et al., 2007). Enhanced CH_4 uptake in no-tillage systems has also been reported due to higher SOC stock and the presence of ecological niches for methanotrophic bacteria (Six et al., 2004; Lal, 2004b). No-tillage can increase N_2O and CH_4 fluxes because of higher bulk density and reduced porosity that diminish gas diffusion and increase water conservation at the surface, thereby increasing the likelihood of anaerobic conditions (Gregorich et al., 2006; Germon et al., 2007; Ball et al., 2008). Accumulation of organic matter and residue mulching at the surfaces of no-tilled fields can also favour N_2O emissions (Jørgensen et al., 1997; Ball et al., 2008). However, residue mulch can also limit N_2O emissions during freezing-thawing cycles by maintaining a warmer temperature that decreases the frequency or intensity of freezing events (Wagner-Riddle et al., 2007). Moreover, when N is a limiting factor, N_2O emissions may be enhanced by soil tillage due to an easier diffusion through the soil matrix without being further reduced (Gregorich et al., 2006; Chatskikh and Olesen, 2007).

The determinism of N_2O being especially complex, all changes related to the soil tillage system may influence the N_2O emissions. The tillage timing, as well as the cumulative effect of a tillage system in the long term, will also be determinant. Furthermore, dry-wet or freezing-thawing cycles can create cracks and enhance the sensitivity of compacted zones to fragmentation during tillage. Thus, seedbed preparation in spring will be more efficient in reducing the proportion of compacted zones, whereas seed bed preparation in autumn will only depend on the initial state (Boizard et al., 2002). Impacts of these weather cycles may also explain how the difference in N_2O fluxes between no-tillage and conventional tillage could change over time. In a review of 44 data sets, the higher N_2O fluxes trend in no-tillage systems compared with conventional tillage systems was reversed after 20 years in humid climates and fluxes became similar between tillage systems in dry climates (Six et al., 2004).

Finally, no-tillage could lead to other "hidden costs" due to possibly increasing use of herbicides and pesticides (Lal, 2004b) and decreasing yields (Chatskikh and Olesen, 2007). Indeed, tilling the soil just before sowing increases soil temperature and can favour germination (Richard and Cellier, 1998). Lower N uptake in no-tilled fields could result in higher gaseous losses, and differences between greenhouse gas balances of conventional tillage and no-tillage systems eventually further shrink (Chatskikh and Olesen, 2007).

Management practices that will reduce agricultural greenhouse gases can hardly satisfy all criteria, especially as determinisms for the diverse greenhouse gas emissions can be opposed, e.g. aerobic or anaerobic conditions that enhance CO_2 or N_2O production, respectively; draining rice paddy fields in order to reduce the emissions of CH_4 enhances N_2O emissions, etc. (Duxbury and Mosier, 1993). Moreover, trying to reduce N_2O emissions by preventing the optimum conditions from occurring could lead to compensating for N_2O emissions by decreasing the rate of reduction into N_2, while complete denitrification would be the less polluting pathway to close the N cycle (Galloway et al., 2003). As long as our understanding of all involved processes remains too incomplete, the best options to reduce the agricultural greenhouse gases are to improve its overall efficiency and to reduce all inputs, especially fertilisers. Perennials are therefore particularly interesting.

6. THE QUANTITATIVE POTENTIAL OF BIOFUELS

To avoid CO_2 emissions, substituting coal is at present a very effective way of using biomass. In the future, though, using biomass for transport fuels will gradually become more attractive from a CO_2 mitigation perspective because of the lower greenhouse gas emissions for producing second generation biofuels and because electricity production on average is expected to become less carbon-intensive due to increased use of wind energy, photovoltaic and other solar-based power sources, carbon capture and storage technology, nuclear energy, and fuel shift from coal to natural gas (IEA Bioenergy, 2007). In this context, land and biomass availability will become the primary limiting factors and the ceiling for contribution to global primary energy can already be foreseen.

6.1. Biomass availability: bottom-up approach

Biomass currently provides an annual amount of energy ranging from 40 EJ yr^{-1} (Parikka, 2004) to 45 ± 10 EJ yr^{-1} (IEA Bioenergy, 2007), of which roughly 7 EJ yr^{-1} are considered as modern biomass in opposition to traditional use of woodfuel (UNDP, 2000). In 2050, the total primary energy demand will vary between 800 EJ yr^{-1} and 1400 EJ yr^{-1} (IEA Bioenergy, 2007), and the share of biomass to meet that need is quite speculative. Due to the complexity of the numerous factors interacting to determine the potential and cost of bioenergy production, projecting future bioenergy consumption cannot be done by matching demand and supply. On the contrary, studies focus either on the supply-driven potential, i.e. resource assessment, or on demand-driven amounts required to meet exogenous targets without specifically defining the exploitable resources (Berndes et al., 2003).

Therefore, assumptions vary widely among studies and significantly impact resulting global potentials. Biomass supply could amount to 200–400 EJ yr^{-1} by 2050, i.e. 14% to 50% of total primary energy demand, without jeopardising the world's future food supply. Considering expected average conversion efficiencies, this primary bioenergy could correspond to 130–260 EJ yr^{-1} biofuels or 100–200 EJ yr^{-1} electricity (IEA Bioenergy, 2007). These future bioenergy potential estimates, which are rather larger than potentials for the current situation, are based on the assumptions of future higher yields, notably thanks to perennials and advanced conversion technologies, but also through an improvement in agricultural system efficiency and the use of marginal and degraded lands. They are average values of extreme supply scenarios in 2050, from a scenario with bioenergy exclusively from waste biomass (40 EJ yr^{-1}) up to a scenario with an intensive dedicated agriculture concentrated on the better quality soils (1100 EJ yr^{-1}). This wide range calls for scrutiny when examining potential assessments.

6.1.1. Bottom-up models

Supply-driven studies have proven, though, that technical biomass potential could meet the amount levels of bioenergy use reported in demand-driven studies. Most studies also agree on the fact that energy crops represent the main potential biomass source compared with forest products or residues; land availability for energy crops and biomass yields hence appears to be the main key assumption (Berndes et al., 2003; Smeets et al., 2007). Diverging assumptions on the energy crop yields alone lead to a 40% difference in the maximum bioenergy potential produced from woody energy crops on about the same amount of surplus land when comparing the studies Hoogwijk et al. (2003) with Smeets et al. (2007).

Most complex approaches use models, such as IIASA's BLS model or IMAGE 2.2 model, etc., to simulate land uses and biomass availability on a geographical grid, taking into account geo-climatic conditions, the types of soils and crops, and agricultural practices. Still, the definitions of land-use patterns, geographical aggregation, and further assumptions on the evolution of both the crop yields and the efficiency of agricultural systems differ, inducing a wider range of bioenergy potentials. For instance, the management factor defined as the gap between theoretically feasible and actual crop yields, i.e. introducing yield limitation by less than optimal management practices and technologies, varies between 0.7 and 1.5 when comparing the two above-mentioned studies. A management factor above 1 expresses an increase in the harvest index, the development of irrigation and biotechnologies (Hoogwijk et al., 2005), reflecting a growing confidence in future biomass production systems up to very optimistic scenarios above current theoretical optimal systems.

Land availability for energy crop depends on the competitive uses of land for food and feed, biomaterials, forest, conservation areas, and build-up. The background assumption in most studies is that land demand for food and feed production has to be fulfilled before land is allocated to bioenergy production. Future land demand for food and feed is then assessed, taking into account expected population growth and diet evolvement; diets having a dual effect on land use. Indeed, diet in MJ day^{-1} per capita increases as a function of the income in absolute quantity terms as well as in qualitative ones, tending to an increasing share of livestock and oilseed products in the global average diet. Growth in meat and dairy product production and consumption is expected to continue, especially in developing countries, where people eat only about 30 kg of meat per capita a year, whereas this rate is above 80 kg yr^{-1} in the industrial world. Experts predict that by 2050, nearly twice as much meat will be produced as today. The impact on land use will be severe as animal husbandry is very land-consuming. In 2002, more than 70% of the agricultural lands worldwide were dedicated to the production of animal products, while these only accounted for some one-fifth of the total calorie intake (FAO, 2003). Therefore, an analysis of the sensitivity of land availability to the evolvement of animal production systems is also necessary.

Nowadays, of the 13.4 Gha of land area in the terrestrial biosphere (Holmgren, 2006):

- 5 Gha are used for agriculture

(roughly 1.3 Gha food crops + 0.2 Gha fodder crops + 3.5 Gha pastures)

- 4 Gha are under forest cover (56% subtropical and tropical forests, overall 95% are natural forests while the remaining 5% are plantation forests)
- 4.4 Gha of the rest of the land encompass semi-natural vegetation types such as savannas, etc., barren land and about 0.26 Gha of build-up area (FAO, 2002, 2003; Hoogwijk et al., 2005; Smeets et al., 2007).

Table VI shows land-use change patterns in 2000 based on annual changes between 1990 and 2005.

Table VII gives an overview of past and future trends of population, average calorie intakes and agricultural production. World population growth and demand for agricultural products have been slowing down since the late 1970s. Indeed, although the world average calorie intake has been rising, especially in developing countries where incomes have increased, high levels of food consumption have now been reached in many countries. In particular, China has already passed its phase of rapid growth. In the past four decades, rising yields accounted for about 70% of the increase in crop production in the developing countries and yield growth, even if not as rapid as in the past, will continue to play the same role for the next 30 years. The contribution of irrigation to this yield growth and intensification of livestock production is also expected to increase. Hence, the expansion of agricultural land at the expense of forest is expected to be concentrated in the developing countries and limited to an overall 12.5% increase, i.e. half of the increase between the early 1960s and late 1990s. However, more than half the land that could be

Table VI. Global land uses (Gha) and annual conversions in 2000 (Gha yr^{-1}) (Holmgren, 2006).

Into: From:	Forest	Woodland/ Grassland	Cropland	Urban area*	Losses (% yr^{-1})	Gains (% yr^{-1})	Net change (% yr^{-1})
Forest	3.97	0.003	0.0098	0.0002	–0.33	0.14	–0.18
Woodland/ Grassland	0.0014	3.44	0.001	0.0002	–0.08	0.15	0.07
Cropland	0.0043	0.002	1.51	0.0016	–0.52	0.70	0.19
Urban area*	n.s.	n.s.	n.s.	0.038	–0.00	5	5

n.s. not significant; *urban area only encompasses here areas with more than 100 000 inhabitants, total area of 13.4 Gha including "other lands" for which no significant changes were identified.

opened up is in just seven countries of tropical Latin America and Sub-Saharan Africa, where 80% of land expansion is expected to take place, whereas other regions face a shortage of suitable land, e.g. in South Asia more than 94% of suitable land is already farmed.

Agricultural production can grow in line with demand, provided that regional shortages, notably in livestock products in developing countries, are reduced through international trade efforts. Net cereal imports by developing countries will almost triple over the next 30 years while their net meat imports might even increase by a factor of almost five. By 2030 about 440 million undernourished people may still remain. The FAO analysed that developing countries' farmers could gain a lot from lower trade barriers, provided domestic policies permit one to remove the domestic bias against agriculture and to improve productivity and product quality to the standards demanded abroad. Investments in transportation and communication facilities, upgraded production infrastructure, and improved marketing, storage and processing facilities could be particularly important. In resource-rich but otherwise poor countries, a more export-oriented agriculture could provide an effective means to fight rural poverty (FAO, 2002).

6.1.2. Availability of agricultural land

Estimation of land availability for energy crops finally essentially relies on the surplus land area that may be released in certain regions either because of higher yields and intensified production, or some agricultural lands being abandoned as they become no longer suitable for food and feed production.

First expectations are reinforced by the fact that in many countries average wheat yields, for instance, for the period 1996–2000 were below the agro-ecologically attainable levels; in India, Brazil, and even Australia and the US, they were calculated to be roughly half the maximum levels (FAO, 2002 in Smeets et al., 2007). Also, feed conversion efficiency[58] of bovine meat and dairy products was in 1998 three to four times higher in industrial countries than in developing countries, indicating that part of the land demand for livestock production could be outpaced by an increase in efficiency of livestock production systems. Moreover, the intensification of husbandry systems could permit one to spare grassland for other crops as some part of suitable croplands are currently used as pastures, especially in developing countries (Smeets et al., 2007).

Testing diverse scenarios of combined population growth, and change in diets and food and feed production systems (Tab. VIII), recent studies show that considerable parts of agricultural lands could be allocated to bioenergy production without jeopardising the food and feed supply: 0.15 to 2.4 Gha (Hoogwijk et al., 2003), 0.6 to 1.3 Gha (Hoogwijk et al., 2005) and 0.7 to 3.6 Gha (Smeets et al., 2007). The highest estimates, notably in the systems 3 and 4 of Smeets et al. (2007), are rather too optimistic though, combining very intensive agricultural production with high yield increases and landless animal production, together with a management factor of 1.5, that indicates even further possible improvements; more irrigation, more fertilisers, while system 4 already corresponds to a 25% addition to the yield levels at a very high rain-fed/irrigated level of agricultural technology. The landless animal production system especially leads to consequent surplus pasture areas, up to 613–820 Mha in some developing countries, which can partly serve for bioenergy production (Smeets et al., 2007). However, such a system could hardly be implemented in these regions by 2050. On the contrary, a low productive agricultural system (Y1 in Tab. VIII = "Low External Input" in the study Hoogwijk et al., 2003), which could be roughly compared with an organic production system, does not permit one to grow any energy crop on agricultural land. This scenario suggests yields in 2050 that would be around the same as today's, once again emphasising that agricultural intensification is a prerequisite.

Considering medium population scenarios with affluent diets and relatively high production systems across the three studies, surplus agricultural land areas allocated to energy crops would vary between 0.45 and 1.3 Gha. Maximum bioenergy potential on these surplus agricultural land areas would vary between 135 EJ yr^{-1} and 409 EJ yr^{-1} (LHV = 15 GJ tDM^{-1}), i.e. roughly 25% to 50% of total primary energy demand in 2050, considering a medium energy demand scenario of 837 EJ yr^{-1}. Nevertheless, this maximum potential is simulated considering a unique type of biomass source which is short rotation coppices (SRC). These crops provide high yields and are convenient for assessing maximum primary bioenergy potential based on gross energy contents, whereas potentials expressed based on first generation biofuel crops would not be as relevant without calculating conversion factors. However, SRC are not representative of the portfolio

[58] Feed conversion efficiency is defined as the amount of animal product produced per amount of animal feed input.

Table VII. World population and agricultural production projections at a glance (FAO, 2002).

	Time frame	1979–1981	1997–1999	2015	2030	2050
Population in billions	World	4.430	5.900	7.207	8.270	9.322
(annual growth rate %)		(1.6)	(1.5)	(1.2)	(0.9)	(0.6)
	Developing countries:	3.259	4.595	5.858	6.910	7.987
remaining countries		(1.9)	(1.7)	(1.4)	(1.1)	(0.7)
	Industrial countries:	0.789	0.892	0.951	0.979	0.986
26 countries: Western Europe, USA, Canada, Australia, New Zealand, Japan, South Africa, Israel		(0.7)	(0.7)	(0.4)	(0.2)	(0.0)
	Transition countries:	0.382	0.413	0.398	0.381	0.349
28 countries: Eastern Europe, former Yugoslavia SFR, CIS, Baltic States		(0.5)	(0.1)	(−0.2)	(−0.3)	(−0.4)
Calorie consumption in kcal capita^{-1} day^{-1}/in MJ.capita^{-1} day^{-1} (1 kcal = 4.186×10^{-3} MJ)						
(annual growth in demand for agricultural products in %) – [calories from animal products]						
	World average	2 552/10	2 803/11	2 940/12	3 050/12	3 302/13
		7	7	3	8	8
		(2.1)	(2.0)	(1.6)	(1.4)	[550]
	Developing countries	2 312/9.7	2 681/11	2 850/11	2 980/12	3 236/13
		(3.7)	2	9	5	5
			(4.0)	(2.2)	(1.7)	[549]
Calorie consumption in kcal capita^{-1} day^{-1}/in MJ.capita^{-1} day^{-1} (1 kcal = 4.186×10^{-3} MJ)						
	Industrial countries	3 135/13	3 380/14	3 440/14	3 500/14	3 448/14
		1	1	4	7	4
		(1.0)	(1.0)	(0.7)	(0.6)	[1 054]
	Transition countries	3 389/14	2 906/12	3 060/12	3 180/13	3 629/15
		2	2	8	3	2
		(−1.7)	(−4.4)	(0.5)	(0.4)	[941]
Total arable land in Gha (irrigated land in Gha)						
	World	1.608				
		(210)	(271)			
	Developing countries	–	0.956	1.017	1.076	–
			(202)	(221)	(242)	
	Industrial countries	–	0.387	–	–	–
			(42)			
	Transition countries	–	0.265	–	–	–
			(25)			

Time frame	1979–1981	1997–1999	2015	2030	2050
Cropland in Gha in developing countries (yields tons/ha)					
Wheat	0.096	0.111	0.113	0.118	–
	(1.6)	(2.5)	(3.1)	(3.5)	
Rice (paddy)	0.138	0.157	0.162	0.164	–
	(2.7)	(3.6)	(4.2)	(4.7)	
Maize	0.076	0.097	0.118	0.136	–
	(2.0)	(2.8)	(3.4)	(4.0)	
All cereals	0.408	0.465	0.497	0.528	–
	(1.9)	(2.6)	(3.2)	(3.6)	
% of total harvested land	60	55	53	51	–
Forests in billion ha (annual change in Gha)					
World	–	3.870 (−0.0094)	–	>2.940 (if < −0.0094)	–
Tropical and sub-tropical	–	2.168 (−0.0123)	–		–
Non-tropical	–	1.702 (+0.0029)	–		–
Build−up land worldwide in Gha	–	0.26	–	0.39–0.52*	–

Calculations from Smeets et al. (2007) with the data from FAO, 2002 are in italics.
* Data from the Global Environment Outlook, UNEP 2002 in Hoogwijk et al. (2005).

of energy crops and present higher yields than other energy crops. Considering the cereal yields of the High External Input system and assuming that the whole cereal crop is harvested for energy purposes, the 0.45–1.3 Gha would only provide 40–279 EJ yr^{-1} (LHV = 15 GJ tDM^{-1}) (Hoogwijk et al., 2003). Yields of energy crops assumed in other studies range from 7 to 49 tDM ha^{-1} yr^{-1} (Smeets et al., 2007). Moreover, SRC production systems require specific machines and overall high investments that are unlikely to be widely affordable in order to reach optimum yields all over the world by 2050.

Theses studies are often quoted throughout the literature, because of the scarcity of such global assessments. A deep insight into these studies' background assumptions would make it possible to apprehend the relevance of the results better. However, other assessments based on different models and hypotheses would be needed to test the robustness of the results.

6.1.3. Biomass from forest and residues

Studies also differ fundamentally in their conclusions about the availability of forest biomass for bioenergy purposes (Berndes et al., 2003). Assumptions diverge both concerning the projected forest plantations and forest growth rates, and the volume restrictions due to competitive wood industrial demand. According to Smeets et al., 2007, energy potential from surplus forest growth in 2050 ranges between 59 EJ yr^{-1} in the case of a low plantation scenario and a high forest product demand, and 103 EJ yr^{-1} in the case of a high plantation scenario and a low demand; meanwhile, it could reach 74 EJ yr^{-1} in a medium scenario. Woodfuel is one of the main forest products; about 60% of the world's total wood removals from forests and trees outside forests are used for energy purposes. Demand for woodfuel will remain strong for many years to come, although its share in total energy demand is expected to decrease, as most developing countries have adopted energy policies aimed to promote the use of other options by households, such as liquefied petroleum gas (LPG), bottled gas and kerosene. This decrease is being largely compensated for, though, by the increased woodfuel use for industrial energy in developed countries (Trossero, 2002). The overall demand for forest products will continue to grow as world population and income grow, but improvement in forest exploitation efficiency, increases in

Table VIII. Comparison of three selected assessments of surplus agricultural land areas available for energy crops.

Scenarios for 2050	Land available pool (Gha)	World population (billion inhabitants)			Diet per person in MJ day^{-1} (grain $_{eq}$ kg^{-1} day^{-1})			Production systems: yields in tDM gr_{eq} ha^{-1} yr^{-1} (irrigated)		Constraints	Available surplus agricultural area in Gha ($EJ\ yr^{-1}$ of primary energy from dedicated crops)
Hoogwijk et al., 2003	5 = 1.5 croplands (including 20% irrigated) + 3.5 grasslands	P1	P2	P3	D1	D2	D3	Y1	Y2	Total agricultural land is constant = 5 Gha No deforestation for bioenergy production Area needed for food and feed is doubled to take into account losses, risks, uneven accessibility to the resources, etc. Management factor (MF) for energy cr ops: 0.7	Yields weighted by MF = 20 tDM ha^{-1} yr^{-1} HHV* = 19 GJ t^{-1} D1 "not likely" Y1,D2–D3,P =>0 P1,D2,Y2 => 2.6 (988) P2,D2,Y2 => 2.4 (912) P3,D2,Y2 => 1.9 (722) P1,D3,Y2 => 0.8 (304) P2,D3,Y2 => 0.45 (171) P3,D3,Y2 => 0.15 (57)
		8.7	9.4	11.3	10.1 (1.3)	10.1 (2.4)	11.5 (4.2)	2.2 (4.0)	5.9 (14.3)		
Hoogwijk et al., 2005 interpreting the IPCC SRES: A1, A2, B1, B2	13 – excluded and allocated areas	A1 B1	B2	A2	A2	B2	A1 B1	A2 B2	A1 B1	Land-claim exclusion factors (% of global land area): agricultural land needed for food and feed production, forest areas, tundra, nature reserves: 5% (A1, A2), 15% (B1, B2), urbanisation (3–4%), extensive grassland areas, rest land areas sensitive to diverse ecological stresses such as scarce water resources: 50% (A1, A2) 90% (B1, B2) Impact of climate change is included Pressure on land quality is not included Management Factor (MF) for energy crops: A2, B2: 1.1; B1: 1.3; A1: 1.5	*Yields weighted by MF = 6–34 tDM ha^{-1} yr^{-1} [*mean yields tDM ha^{-1} yr^{-1}] LHV = 19 GJ t^{-1} A1 => 1.3 (409) [20] B1 => *1.3 (398) [20] A2 => 0.6 (129) [16] B2 => *1 (279) [18]
		8.7	9.4	11.3	12.5	12.8	13.2	Regional yields of 12 food crops weighted by MF			
								$MF^{\bullet}$ = 0.78	$MF^{\bullet}$ = 0.82		

Scenarios for 2050	dLan available pool (Gha)	World population (billion inhabitants)	Diet per person in MJ day^{-1} (grain $_{eq}$ kg^{-1} day^{-1})	Production systems: yields in tDM gr$_{eq}$ ha^{-1} yr^{-1}	Constraints				Available surplus agricultural area in Gha (EJ yr^{-1} from dedicated crops)
Smeets et al., 2007	13 −excluded and allocated areas	8.8	13 details by regions and animal products contribution given by the authors	Systems	S_1	S_2	S_3	S_4	Yields weighted by MF(=1.5) = 4–39 tDM ha^{-1} yr^{-1} [mean yields tDM ha^{-1} yr^{-1}] HHV = 19 GJ t^{-1}
				Animal production	mixed	mixed	landless	landless	
				Feed conversion efficiency	high	high	high	high	
				Technology for crop production	very high	very high	very high	super high	
				Irrigation	none	partly	partly	partly	S1 => 0.7 (215) [16]
				Average yield increases (19 crops) between 1998–2050	x2.9	x3.6	x3.6	x4.6	S2 => 1.15 (455) [21] S3 => 3.3 (1,101) [17] S4 => 3.6 (1,272) [20]
				The yields of the19 crops are calculated with IIASA model and weighted (0–100%) in function of the suitability (5 levels) of the land area allocated. Demand for feed crops, including increase in demand for feed from grasses and fodder compared with 1998, is added to total demand for food crops. Allocated land areas excluded from available pool for bioenergy crops: other land including uncultivated land, barren land, etc., build-up, plantations and natural forests, permanent crops, crops not in the model (13% of total harvested area), agricultural land needed for food and feed production including trade at regional levels to ensure food and feed security. No deforestation for bioenergy production Water is excluded as a limiting factor except in arid and semi-arid regions Irrigation is limited to areas in which climate, soil and terrain permit it Impact of climate change is excluded Management Factor (MF) for energy crops: 1.5					

Figures in italics are the closest to FAO simulations of business-as-usual scenario for 2050. "not likely" expresses the opinion of this article's authors and not any mentioned comment of the assessments' authors; *H/LHV: high/low heating value i.e. including/or not the energy recovered from vapour condensation; •Average management factors that affect the yields of the 12 food crops calculated with IMAGE 2.2 model at a geographical grid cell level of 0.5° × 0.5°; *read on the graphs.

plantation and an expansion of the role of trees outside forests should ensure sufficient wood supply.

The key questions now are where it should come from and how it should be produced (FAO, 2002). Ensuring a sustainable woodfuel supply for the poorest people of developing countries remains a serious problem. In places with intensive woodfuel use, for example around large urban centres and in zones with a high concentration of commercial activities such as brick-making, the pressure on woodfuel supply sources can be heavy, with consequent deforestation. Therefore, generalisation on sustainability of woodfuel use at the local level cannot be done without careful analyses in the field (Trossero, 2002). Because of the decentralised nature of wood energy systems, energy and forestry statistics seldom include the same level of detail about woodfuel consumption as for other conventional energy sources or forest products (Trossero, 2002). Some studies hence only mention forest biomass potential from residues.

Indeed, forest exploitation produces high amounts of residues, although the production of wood-based materials is continually increasing in efficiency, i.e. the ratio of residues over final products is decreasing (FAO, 2002). About 60% of the total harvested tree is left in the forest and the non-commercial species are often felled and left on site to ease the logging, providing a valuable source of bioenergy (Parikka, 2004), especially as industrial round wood production is expected to rise by 60% by 2030, from current levels to around 2400 million m^3, with one-third from plantations, about twice as much as today's plantation production of 400 million m^3 (FAO, 2002). After processing, about half of the log input becomes wastes that can have various alternative uses such as chips for pulp or chipboard, etc., and fuel for internal energy use or as compacted commercialised fuels, depending on the sale values on the diverse markets (Parikka, 2004).

Overall, biomass residues may be classified as follows: primary ones from agriculture (crop residues) or forest (logging residues), secondary ones from agriculture (from food processing, animal manure) and forest (mill and manufacture residues), and tertiary ones including all kind of final biomass waste. Although residues can provide a substantial source of biomass in a global energy context (Berndes et al., 2003), potential assessments are highly uncertain. First, residue generation is a multiplier factor of assumed total food and feed production. Fischer and Schrattenholzer (2001) thus stressed that this factor will furthermore decrease while the harvest index of crops is expected to increase thanks to progress in agricultural technology (e.g. bio-engineering, agricultural practices, etc.), which implies another degree of uncertainty. Second, only a fraction of total residues is recoverable in practice. Third, residue amounts available for energy production also depend on possible alternative uses of residues such as demand for feed, fertiliser (including the need to maintain soil quality), and for wood products, although these last may also partly become an eventual energy source. Most studies assume a recoverability fraction of 25% for primary residues, and higher fractions for secondary and tertiary residues between 75 and 100%. Estimates, from the literature, of potential contribution from biomass residues in 2050 vary between 38 and 245 EJ yr^{-1} (Hoogwijk et al., 2005). In Smeets et al. (2007), this range is reduced to 76 to 96 EJ yr^{-1} as residues needed for feed are excluded. These amounts should be further reduced though when compiling all residue demands at regional levels. Considering recoverability limits and some competing uses, the total maximum available residues for bioenergy accounts for roughly 6% of total bioenergy potential (Hoogwijk et al., 2003; Smeets et al., 2007).

6.1.4. *Geographical distribution*

Studies mostly agree on the fact that maximum bioenergy potentials, as a function of surplus non-food competitive land areas, are likely to be concentrated in a few regions. In Sub-Saharan Africa, Oceania, the Caribbean and Latin America, large areas suitable for crop production are currently used as pastures (Smeets et al., 2007). Intensification of animal production systems and overall yield increase in these regions would lead to the release of land areas for energy crops. Together with the CIS and Baltic States, these regions have the highest bioenergy potentials from energy crops on surplus agricultural lands (Hoogwijk et al., 2005; Smeets et al., 2007). The largest energy potential from surplus forest growth is found in the CIS and the Baltic States, the Caribbean and Latin America, and partially North America and Western Europe (Smeets et al., 2007). Figure 19 shows regional bioenergy future potentials of the four scenarios in Smeets et al. 2007. Figure 20 illustrates that considering current technologies and land and water availability as well as food insecurity, biofuel potential is nowadays still limited in Africa and Asia notably (Von Braun, 2007).

Oceania is the least land-stressed region, whereas Japan is the most land-stressed country. The Middle East and North Africa, and South and East Asia have relatively scarce agricultural land (Smeets et al., 2007). High land requirements for urbanisation were estimated in South and East Asia, mainly India and China, whose abandoned agricultural land areas should increase at the end of the century with the decrease in population growth (Hoogwijk et al., 2005). In industrial or transition countries, bioenergy production requires less drastic changes than in developing countries. Furthermore, in Eastern Europe, the CIS and Baltic States, food consumption and population are projected to decrease, which makes bioenergy potential in these regions more robust than in others. Regions with the highest potentials could turn into bioenergy exporters. In developing countries, bioenergy may provide new incentives for investments in agricultural research and, by providing new income, carry out a modernisation of the agricultural production systems, with a positive feedback on yields possible (Smeets et al., 2007). This is especially crucial for regions that are expected to have high bioenergy potential, such as Sub-Saharan Africa and Latin America, but which also suffer from soil erosion or nutrient depletion. In Sub-Saharan Africa, some 95 Mha of land is threatened with irreversible degradation if soil nutrient depletion continues (Henao and Baanante, 2006 in Agard, 2007). Furthermore, these regions are expected to be inversely impacted by climate change. According to Tubiello and Fisher (2007), cereal production would decrease

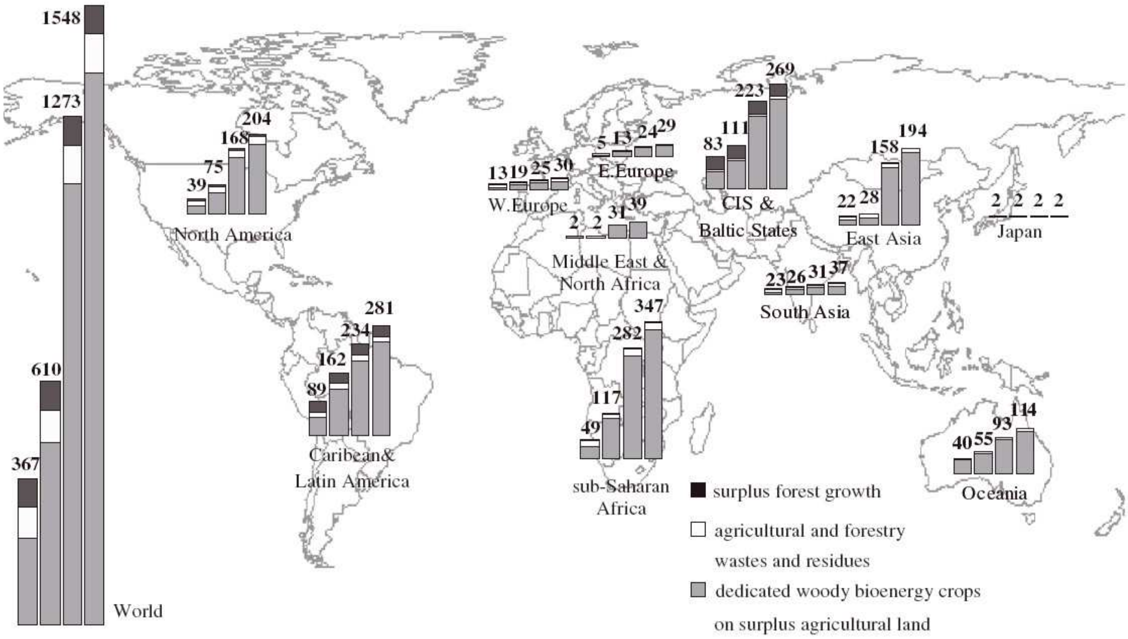

Figure 19. Total technical bioenergy production potential in 2050 based on systems 1−4 (EJ yr^{-1}; the left bar is system 1, the right bar is system 4) (Smeets et al., 2007).

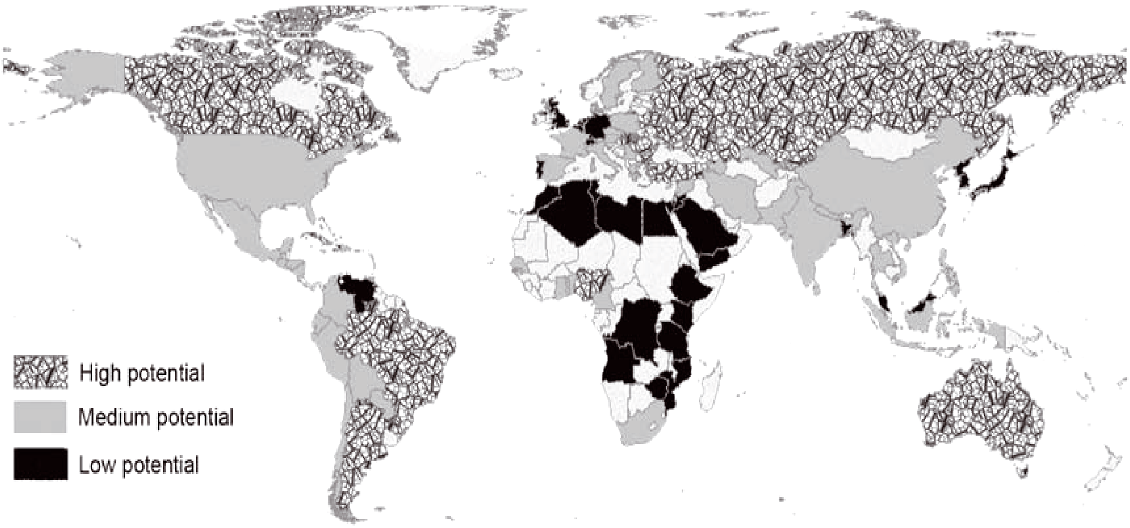

Figure 20. Biofuel production potential (Von Braun, 2007). Sources: data on food insecurity are from FAO (2006). The land availability index is derived by the author based on data from FAO (2007); IEA (2007a, b); and USDA (2006). Data on water availability are from WRI (2007).

by 3.9–7.5% between 1990 and 2080 due to climatic risks, whereas it would increase by 5.2–12.5% in Latin America (Tab. IX).

As shown in Figure 21, many areas are affected by degradation; Tropical Africa, Southeast Asia, North-Central Australia, Central America, the Caribbean, southeast Brazil, and boreal forests in Alaska, Canada and eastern Siberia are the most severely touched. Here, land degraded areas are defined as areas with a combined declining trend of Net Primary Production and declining Rain-Use Efficiency over the past 22 years, excluding the simple effects of drought. 23.5% of the world land area is in a state of more or less severe degradation (Bai et al., 2008). The most degraded areas are mainly associated with forest degradation, although the precise history of land degradation processes has to be investigated on a regional scale. The degraded areas represent a loss of NPP of about 800 million tons of carbon that were not fixed during this period, added to CO_2 emissions into the atmosphere of one or

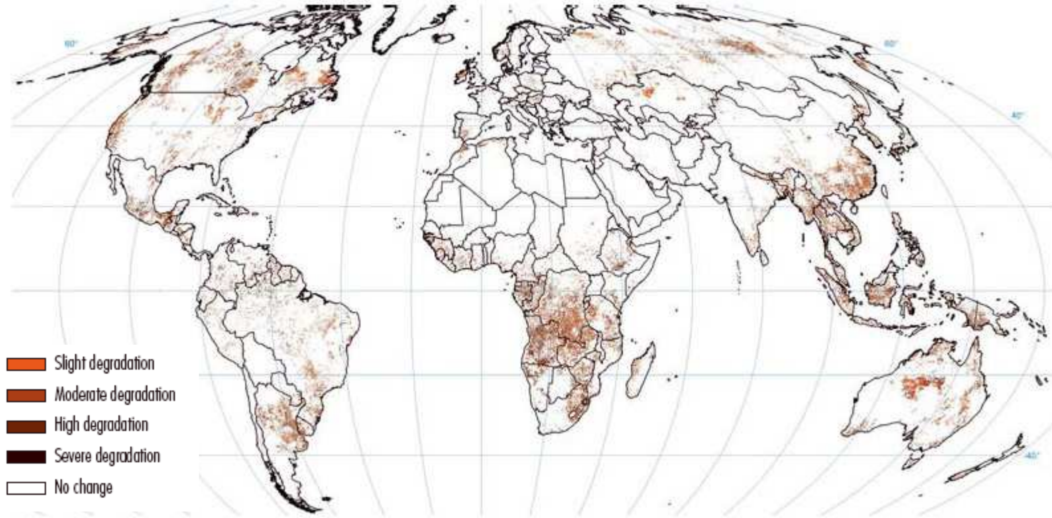

Figure 21. Global land degradation using biomass production and rain-use efficiency trends between 1981–2003 (Bai et al., 2007, online: http://www.fao.org/nr/lada/images/maps/ndvi_preliminar.jpg).

Table IX. Expected impacts of climate change on global cereal production. Adapted from Tubiello and Fischer (2007) in Von Braun (2007).

Localisation	1990–2080 (interval change %)
World	–0.6 to –0.9
Developed countries	2.7 to 9.0
Developing countries	–3.3 to –7.2
Southeast Asia	–2.5 to –7.82
South Asia	–18.2 to –22.1
Sub-Saharan Africa	–3.9 to –7.5
Latin America	5.2 to 12.5

Table X. Tropical degraded lands with potential for plantation establishment (Mha) in Grainger (1991). Total degraded land in the tropics could average 2 Gha (Grainger, 1988).

Degraded land areas	Forest fallows	Deforested watersheds	Degraded drylands	Total
Africa	59	3	110	172
Asia	59	57	110	226
Latin America	85	27	10	222
Total	203	87	330	620

two orders of magnitude more than this amount from the loss of soil organic carbon and standing biomass (Bai et al., 2007 in Agard, 2007).

From about 2 Gha of degraded areas in the tropics solely, some 420 to 620 million ha, could be suitable for afforestation or vegetation enhancement, respectively, subtracting or not the forest fallows that are part of shifting cultivation systems (Tab. X, Grainger, 1991). Houghton et al. (1991), also assessed that around 580 million ha of degraded land, formerly covered with forests or woodlands, may be available to be planted or managed as plantations. Parts of these areas could be used to produce energy crops and SRC could be a means of recovering parts of the lost former sequestered carbon. Although yields on these lands will not be optimum, some species with low needs, such as *Jatropha curcas*, could be used in agroforestry systems (see picture below) in order to limit further degradation of these areas and CO_2 emissions. On 420–580 million ha of degraded land, also referred to as "low-productive land", some 8–11, and 24–33 up to 80–110 EJ yr^{-1} could be produced considering yields of 1, 3 and 10 tDM ha^{-1} yr^{-1}, respectively (Hoogwijk et al., 2003; Smeets et al., 2007). Finally, Smeets et al. (2007) also mentioned that some further 247 EJ yr^{-1} could be produced on the 3.6 Gha classified as other land. This is an absolute finite potential, assuming that all other land could be allocated to energy crops. "Other land" or "rest land" areas include several kinds of natural vegetation and other sites remaining once agricultural, build-up and forest land areas are allocated that are not particularly suitable for production. They are therefore also classified as "low-productive lands". More specific studies would be necessary to evaluate on a regional scale the effective availability of rest land areas to be cultivated and the overlapping between these areas and degraded land areas.

Agro-forestry system: *Jatropha curcas*/chili
Belize, February 2003 bessou©INRA

To summarise, biomass availability for bioenergy can be assessed through five steps, each corresponding to a gradually decreasing potential (Hoogwijk et al., 2005). The first theoretical potential in 2050, some 3500 EJ yr^{-1} (Hoogwijk et al., 2005); 4435 EJ yr^{-1} (Smeets et al., 2007) takes into account the conversion of solar energy by vegetation (Net Primary Productivity), giving an upper limit of primary biomass energy potential on the total terrestrial surface. This indicative potential is severely reduced at a regional level by the multiple uses of land, which leads to a smaller geographical potential, also diminished due to losses through conversion processes of primary biomass to secondary energy carriers corresponding to a technical potential. Finally, economic and other socio-technical constraints may also drastically limit final economic and implementation bioenergy potentials.

Economic feasibility notably depends on raw material costs, conversion efficiency and incentives that will translate into political orientation choices. In the case study of woody biomass for energy production as a means of greenhouse gas reduction, Dornburg et al. (2007) showed that, in Poland, biomass potential is larger than the amount that could make possible cost-effective greenhouse gas savings at low costs. Biomass cost is notably strongly affected by land price elasticity; so that electricity and methanol from woody biomass remain interesting ways of cost-effective reductions in greenhouse gas emissions only as long as markets are large enough to absorb the supply without lowering market prices. In some cases, theoretical and technical potentials might be well above these thresholds. For biomaterials, market volumes are an even more critical issue. Hence economic potential highly varies depending on the supply curves of bioenergy and the internalised environmental costs, which also highly depend on market sizes.

Implementation potential, that actually defines the overall final bioenergy potential, is further limited by diverse constraints that can barely all be embraced in any assessment. For instance, the actual feasibility of the implementation of energy crops and the impacts on degraded lands are merely mentioned; studies refer to exogenous assessments of the actual extent of degraded land that could be suitable for plantation establishment (Berndes et al., 2003). Also, many environmental aspects are not taken into account, such as impacts on biodiversity, erosion, water and nutrient uses, etc. The issue of water and nutrient uses is especially crucial in SRF with species such as willow or eucalyptus that take up lots of water and nutrients from the soils. Large-scale energy crop implementation would in some countries, such as Poland or South Africa, lead to further exacerbation of an already stressed water situation (Hoogwijk et al., 2005).

Therefore, more research is needed in modelling interactions between the competitive land uses and ecological issues (Berndes et al., 2003). Notably, land-use change from forest area to bioenergy area is excluded in most studies, but land-use change from forest area to agricultural land for food and feed production is not, meaning that the actual forest areas decrease/shrink. In the scenario with high demand for food and low technology development (A2) a significant amount of forest is cut down: 45% of South American forest area could disappear within 100 years (Hoogwijk et al., 2005). Part of the abandoned agricultural land areas should therefore be allocated to reforestation and not to bioenergy production, in order to compensate for the loss in forest areas due to land clearing for agriculture.

6.2. Focus on Europe

A report of the European Environment Agency (EEA, 2006b) focused on how much bioenergy could be produced without harming the environment, leading to the following strict assumptions: the use of energy crops with low environmental pressure, the preservation of current protected forests and extensively cultivated agricultural areas (almost 6 million ha of grassland, olive groves and dehesas), a minimum 30% share of used agricultural area dedicated to *environmentally-oriented farming EOF*[59], 3% of intensively cultivated agricultural land set-aside by 2030, ambitious waste minimisation strategies, and the further liberalisation of agricultural markets with a reduction of 40% below the greenhouse gas emission 1990 level that would make the price for carbon emission permits increase.

The results show that primary biomass potential could rise to 7.9 EJ yr^{-1} (190 Mtoe yr^{-1}) in 2010 (target 6.3 EJ yr^{-1} ≈ 150 Mtoe yr^{-1} by 2010) up to around 12.3 EJ yr^{-1} (295 Mtoe yr^{-1}) in 2030, i.e. 17% of European current global energy consumption (EEA, 2006b). This *conservative estimate*[60] concludes that the largest potential comes in the short term from the waste sector, especially in Germany and France (all kind of waste included: around 4.2 EJ yr^{-1} ≈ 100 Mtoe yr^{-1}), whereas energy crops from agricultural land would overtake it in the long term (up to around 5.9 EJ yr^{-1} ≈ 140 Mtoe yr^{-1}). *Environmentally-compatible* bioenergy potential[61] from forestry is estimated to be almost constant throughout the period analysed (around 1.7 EJ yr^{-1} ≈ 40 Mtoe yr^{-1}) (EEA, 2006b).

The modelling of the released and set-aside land area was based on the CAPSIM model (EuroCare, 2004). The available arable land within the EU-22 that could be used for dedicated bioenergy production increases from 13 million ha in 2010 (8% of the total UAA) to 19.3 million ha in 2030 (12% of the UAA). Assumptions include that current areas used for energy crop production remain available for bioenergy production, but other drivers interact to influence this land availability. Most of the land is made available through release of land from food and fodder production. As maintaining the current European food self-sufficiency level was set

[59] EOF encompasses organic farming and high natural value farming (NHV).

[60] Quoted as written in the study: related to the overall value judgements in the study that limit the available potential including strict environmental assumptions.

[61] "*Environmentally-compatible*" bioenergy potential = the quantity of primary biomass that is technically available for energy generation based on the assumption that no additional pressures on biodiversity, soil and water resources are exerted compared with a development without increased bioenergy production (EEA, 2006b).

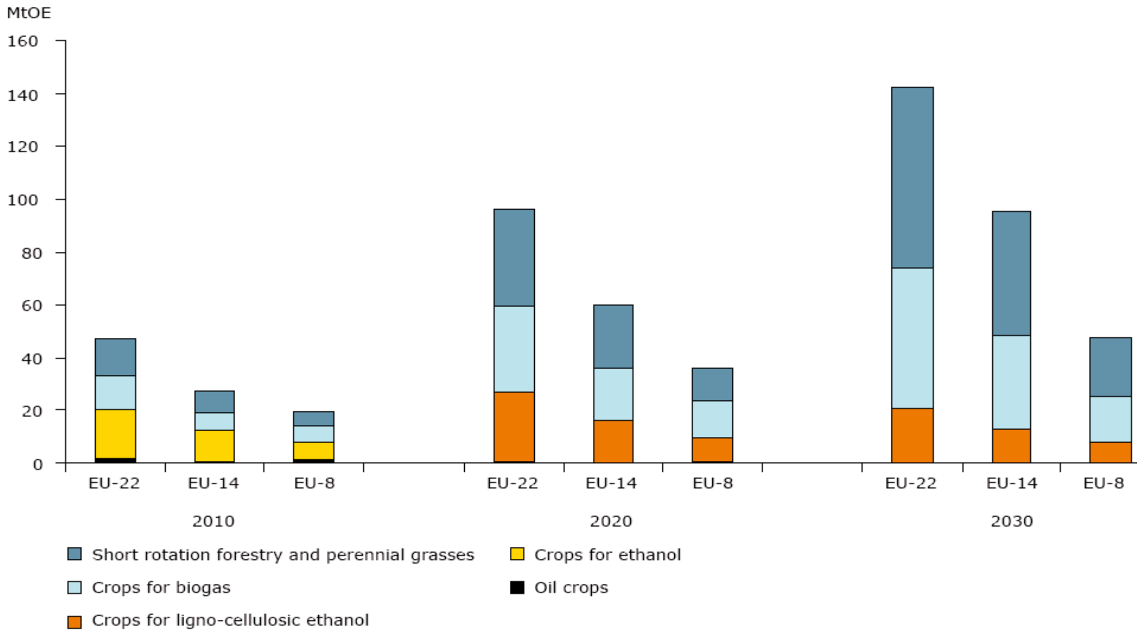

Figure 22. Environmentally-compatible agricultural bioenergy potential (EEA, 2006b).
Note: No data available for Cyprus, Luxembourg and Malta. 'Oil crops' comprise rapeseed and sunflower. 'Crops for ethanol' include the potential of grains from maize, wheat, barley/triticale. 'Crops for ligno-cellulosic ethanol' cover the energy value of the whole plant (corn and straw) for wheat and barley/triticale. 'Crops for biogas' are maize (whole plant), double cropping systems, switch grass and the grass cuttings from permanent grassland. 'Short rotation forest and perennial grasses' include poplar, willow, miscanthus, reed canary grass, giant reed and sweet sorghum, which may often be used in whole-plant conversion systems like gasification, or biomass-to-liquid processes.

as a framework condition, the competition between food and bioenergy production was assumed to be relevant only for the part of agricultural production that corresponds to projected food exports. Thus, consideration of the competition effect between bioenergy and food production was restricted to France and Germany, the only member states which are projected to combine a very high export surplus for cereals with a large agricultural land area (EEA, 2006b).

The main "suppliers" of available land for bioenergy production are Poland, Spain, Italy, the United Kingdom, Lithuania and Hungary. These countries, plus Germany and France, will produce more than 85% of the *environmentally-compatible* bioenergy potential in Europe. Population size and the economic competitiveness of the agricultural systems in each member state are the main factors determining land potential. On the basis of the available land and an environmental ranking of energy crops, sustainable crop mixes were determined for different environmental zones in Europe. Europe was divided into 13 zones with homogeneous pedo-geo-climatic characters. The *environmentally-compatible* agricultural bioenergy potential is shown in Figure 22, taking into account the LHV for the conversion of the harvested dry biomass into an energy potential (EEA, 2006b).

Figure 22 shows that high shares of bioenergy would be supported by a complete shift to second generation biofuels with notably a crop mix, not at all relying on oil crops any more. Following the scenario presented in Annex 3 of the Biomass Action Plan, the distribution of biomass used in 2010 would be: 50% for heating, 37% to produce electricity and 13% in the transport sector. The main drivers in the increase in bioenergy potential are productivity increases[62], notably due to the introduction of advanced technologies, and the assumed liberalisation of the agricultural sector, notably linked to the CAP reform that should result in additional land area available for dedicated energy crops. The EEA study shows that there is a significant development potential for bioenergy within Europe, even considering strict environmental assumptions (EEA, 2006b). It is, though, in agreement with the previous detailed studies, showing that this potential will be limited by land availability issues, especially in Western Europe.

6.3. Liquid biofuel potential: top-down approach

The top-down approach does not aim to assess maximum biomass potential but to study the feasibility of biofuel development plans, i.e. the land areas that would be necessary to produce certain amounts of biofuels (Fig. 23). Given the numerous types of feedstocks and processes that determine biofuel yields per unit of biomass input, there would be multiple scenarios for future global biofuel potentials and as many

[62] Assumed yield increases: 1% per year for conventional arable crops, 1–2.5% for dedicated energy crops. A lower yield increase of 1% for all crops would reduce the bioenergy potential by 2% in 2010, and by 13% in 2030.

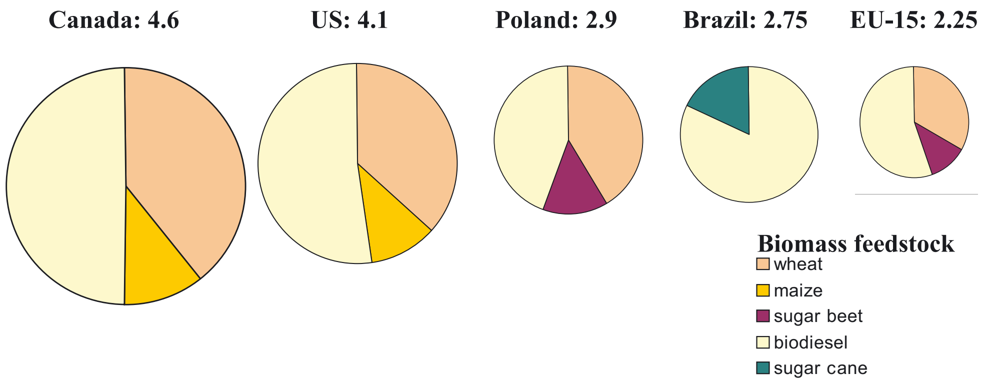

Figure 23. Land areas (ha) needed to produce 3000 L of gasoline equivalent of biofuel energy; 1000 L per type of biomass feedstock except for Brazil 2000 L of ethanol from sugar cane. Drawn from data in OECD (2006).

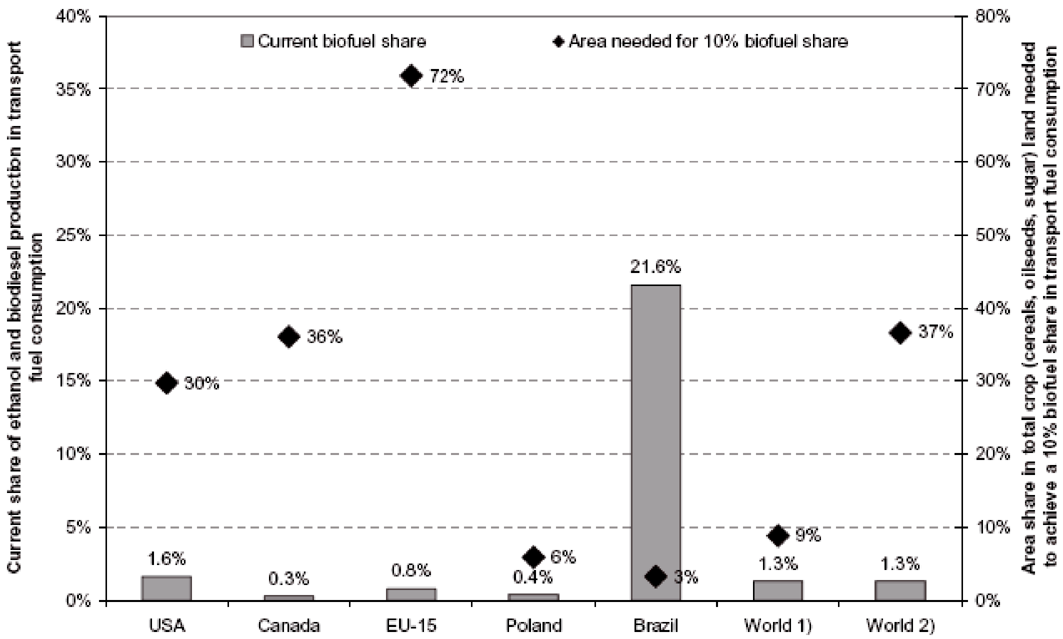

Figure 24. Biofuel shares in transport fuel consumption and land requirements for 10% biofuel shares in major biofuel producing regions (OECD, 2006).

Notes: Current biofuel shares include ethanol and biodiesel only – shares are on an energy basis. World area shares are calculated relative to land used for cereals, oilseeds and sugar globally (World 1) and within the five major biofuel producing regions only (World 2). All areas requirements are calculated on the basis of average crop area and yield data for 2000–2004 and transport fuel consumption in 2004. For these calculations, the 2004 shares in the feedstock mix are assumed to remain unchanged. Note that calculations for the EU exclude ethanol transformed from wine which represented about 18% of EU ethanol production in 2004.

possible studies. Global assessments therefore extrapolate current biofuel trends and implemented target policies.

In 2004, a study estimated that 37% of total harvested grains, oilseeds and sugar crops area would have been needed to reach a 10% biofuel share (on an energy basis) in world transport fuel consumption within the major biofuel producing regions (Brazil, US, EU[63], Canada) (OECD, 2006). Except in Brazil and Poland, such a share would require excessive amounts of cropland areas (Fig. 24). In Brazil, the combination of high sugar cane yields and low transport fuel consumption per capita make it possible to go beyond 10% shares up to even more than the current 40%. In the EU, on the contrary, the situation is especially critical as land endowment per capita is particularly low. Moreover, biodiesel represents the largest

[63] In both studies here (OECD, 2006 and Fulton et al., 2004) data concern the years 2000–2004, hence projections for the European Union encompass only the 15 Member States before the entrance of the other 12 members if not mentioned.

share of biofuels produced in Europe and also consumes much land area, about five times more than ethanol from sugar beet (OECD, 2006).

As an extrapolation of land areas allocated to biofuel production in 2004 and the correspondent transport fuel share of biofuels, 72% of the total cropland area of relevant crops harvested in 2004, would have been needed to reach the same target in the EU (OECD, 2006). Fulton et al. (2004) assessed the potential impacts on cropland areas if the US and the EU were to expand 1st generation biofuel production to reach targets of 5% in 2010 and 10% in 2020 displacement of both road transport gasoline and diesel future consumption (on an energy basis). Considering constant total cropland areas, including set-aside lands in the EU, and annual increases in crop yields (1% for all crops) and conversion efficiencies (1% for ethanol and 0.3% for biodiesel), by 2010 some 20 and 21% of total cropland area would be needed to produce enough biofuels in the EU and the US, respectively, rising to 38% and 43% by 2020. Especially for biodiesel to displace 10% of consumed transport diesel much higher land allocations would be necessary, even over 100% of projected EU oil-seed croplands as soon as by 2010 or by 2020 in the US (Fulton et al., 2004).

Despite fundamental differences between the two estimates, i.e. the time frame and calculation method, total land areas, assumptions on crop yields and fuel consumption increases, the share of sugar beet ethanol in the EU and the biodiesel production projection in the US, both estimates show very high requirement of land areas to reach the 10% biofuel target. They also emphasise that it may make sense to focus more on ethanol blending rather than on biodiesel, since in the medium term substantial amounts of biofuels can only be achieved if the feedstock mix is adjusted in favour of commodities with a higher biofuel output per hectare. This may also include imported feedstock and biofuels (OECD, 2006).

Within the EU (27 Member States) and Ukraine, a 10% biofuel share could be reached by 2020 relying on low-cost 1st generation biofuels without endangering food security or nature conservation, assuming that demand for other bioenergy sectors remains confined to forest feedstock, i.e. 100% of the agricultural feedstock for bioenergy is dedicated to biofuels, and provided that 30% of the target is met by imports from outside Europe, without imports a 9% share could be reached by 2030. Compared with previous estimates, higher biofuel potential within the EU is notably due to higher potentials in new member states. However, if 10% is not the end term for biofuels, 2nd generation biofuels will become more competitive (REFUEL, 2008).

The development of 2nd generation biofuels produced from residues and dedicated crops appears crucial to lower the pressure on cropland areas. Compared with 1st generation biofuels, 2nd generation biofuels can be produced from a wider range of feedstock, including agricultural and wood-related residues without direct use of land, or dedicated crops that can be grown on a wider spectrum of soils. Second generation biofuels also make two- to four-fold higher land-use efficiencies possible due to higher crop yields, less agricultural management inputs and better conversion efficiency.

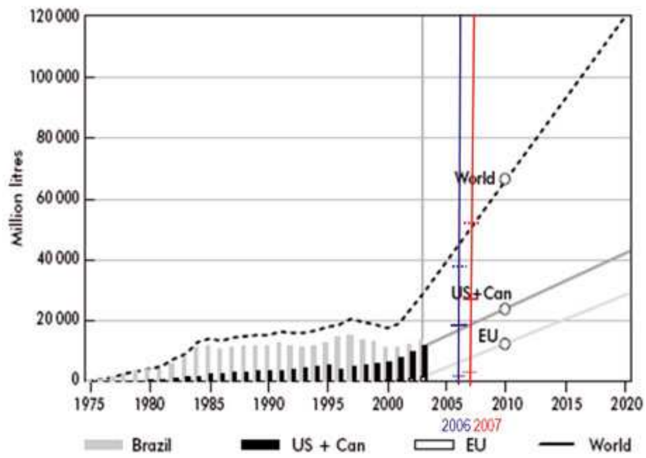

Figure 25. Fuel ethanol, production and projections to 2020 (Fulton et al., 2004) added production data for 2006 and 2007 from F.O. Licht.

By 2030, a 15 to 25% biofuel share could be met by a mix of 1st and 2nd generation biofuels produced in Europe only (REFUEL, 2008). Considering the estimated cellulosic feedstock from residues and bioenergy crops on marginal lands in the US (≈ 388.5 million dry tons per year at US\$50.ton^{-1}) and a conversion efficiency in a post-2010 scenario of 400 L ethanol per dry cellulosic feedstock ton, second generation ethanol could provide up to 26% of US annual motor gasoline consumption by 2020 without using dedicated cropland area (in Fulton et al., 2004).

Furthermore, yields of dedicated lignocellulosic energy crops are expected to grow much faster than those of conventional crops as research in breeding new varieties or adapting crop farming are still in the early stages. Finally, cooking oils and other municipal wastes could contribute to a lower extent to biofuel production, poor economics of scale being compensated for by low feedstock prices. In the EU and the US, 1 and 1.9 billion litres of biodiesel, respectively, could be produced annually, about one percent of diesel consumption in the US in 2010 (Fulton et al., 2004).

6.4. Projected worldwide biofuel production and consumption

In the reference scenario of the IEA with 2004 as the baseline and a global primary energy demand increase of 1.6% per year (IEA, 2006), total world production of 1st generation biofuels is projected to climb up to 3.85 EJ yr^{-1} (92 Mtoe) by 2030, expressing an average annual growth rate of 6.3%. Figure 25 gives an overview of past and projected ethanol production, showing that global trends follow the predicted scenario, the EU still produces less than expected, compensated for by the US where it is the contrary. In 2008, world biofuel production reached 46 Mtoe, 65 Gl of ethanol (beyond the prediction in Fig. 25) and 16 Gl of biodiesel (ENERS, 2009).

The IEA Alternative Policy Scenario incorporates 1400 different policies and measures that aim at enhancing energy

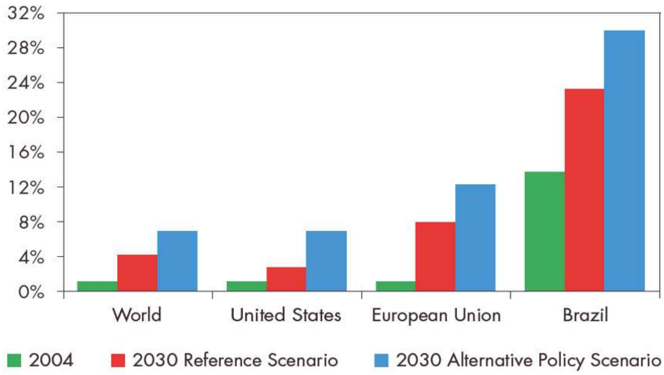

Figure 26. Share of biofuels in road-transport fuel consumption in energy terms (IEA, 2006).

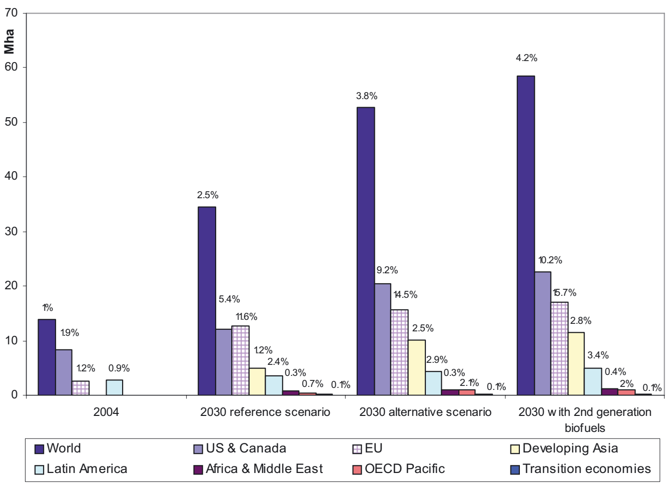

Figure 27. Land requirements in Mha and percent of respective agricultural land areas for biofuels production in the different scenarios from the IEA. Drawn from data in IEA (2006).

security and mitigating CO_2 emissions. Measures in the transport sector would produce close to 60% of all oil savings in this scenario, more than two-thirds linked to more efficient new vehicles, the rest being related to the increased use of biofuels. With a faster assumed growth of 8.3% per year, biofuel production could rise to 6.15 EJ yr^{-1} (147 Mtoe) by 2030. In the Reference or Alternative Scenarios, biofuels meet 4% or 7% of the world road-transport fuel in 2030, occupying 2.5% (34.5 Mha) or 3.8% (52.8 Mha) of world total agricultural land, respectively (Figs. 26, 27). If cellulosic ethanol were to be largely available by 2030, a larger share of biofuels (10%) could be possible with only a little extra land area (+ 5.7 Mha).

Ethanol is expected to account for most of the biofuel increase worldwide as production costs are expected to fall faster than those of biodiesel and as it is likely to become a more attractive option for fuel suppliers in Europe. The global share of biodiesel nonetheless will grow in both scenarios because of production increase in the US and Brazil; it could reach up to 15% of total biofuel use in both countries. In both scenarios, the biggest increase in biofuel production and consumption occurs in Europe and the US may become a sizeable net importer of biofuels. Brazil remains the biggest ethanol exporter but other countries such as Malaysia or Indonesia could also become biofuel exporters. However, the assumed development of today's still limited international trade in biofuels will depend on whether trade barriers are removed or not (IEA, 2006). Biofuel trade liberalisation would lower the prices of blended fuel and enhance total biofuel demand imports. The shift from domestic production to import from abroad would be rather significant in most European countries for ethanol, from around 95% of domestic production to an average 50% share in 2020 across member states. Trade amongst European member states would also decrease, from around 5% to 2%. For biodiesel, the impact should be much lower as import tariffs are much lower than for ethanol (Boeters et al., 2008).

If the IEA Alternative Policy Scenario was successfully implemented, energy-related CO_2 emissions could be cut by 1.7 Gt or 5% in 2015, and 6.3 Gt or 16% in 2030 relative to the Reference Scenario; 12% of these savings would come from renewables (including biofuels), 10% from nuclear power and the remaining from more efficient production and use of energy. Despite these savings, global CO_2 emissions would still be 8 Gt higher in 2030 than they are today. Going beyond the Alternative Policy Scenario to keep emissions at current levels, i.e. saving these 8 Gt, would mean increasing electricity efficiency by 50% over the alternative scenario and to implement new technologies such as CO_2 Capture and Storage. In this last scenario, 1Gt more would be saved in the transport sector thanks to more efficient and cleaner vehicles, notably using twice as much biofuel as in the Alternative Policy Scenario (IEA, 2006).

6.5. Impact of biofuels on agricultural commodity prices

As a background assumption in most potential assessments, energy crops should not compete with food and feed crops for land. In practice, land areas that are allocated in the supply-driven studies to food crops because of their high yield potential could also be allocated to energy crops by farmers depending on the markets. In a sensitivity analysis of the impact of land allocation on the bioenergy potential, 30% to 51% of the most productive land areas previously allocated to food and feed crops would then be allocated to energy crops, leading to an overall decrease in bioenergy potential in most cases due to the need for more agricultural land to produce enough food and feed on less productive areas. On the contrary, a global geographical optimisation of land-use patterns by allocating the most productive land areas to food crops would result in an increase in bioenergy potential in all scenarios (Smeets et al., 2007).

In 2006, global cereal stocks, especially wheat, were at their lowest levels since the early 1980s. As a result of reduced plantings and adverse weather in some major producing countries, world cereal production decreased by 2.4% between 2005 and 2006, coinciding with further expansion of the

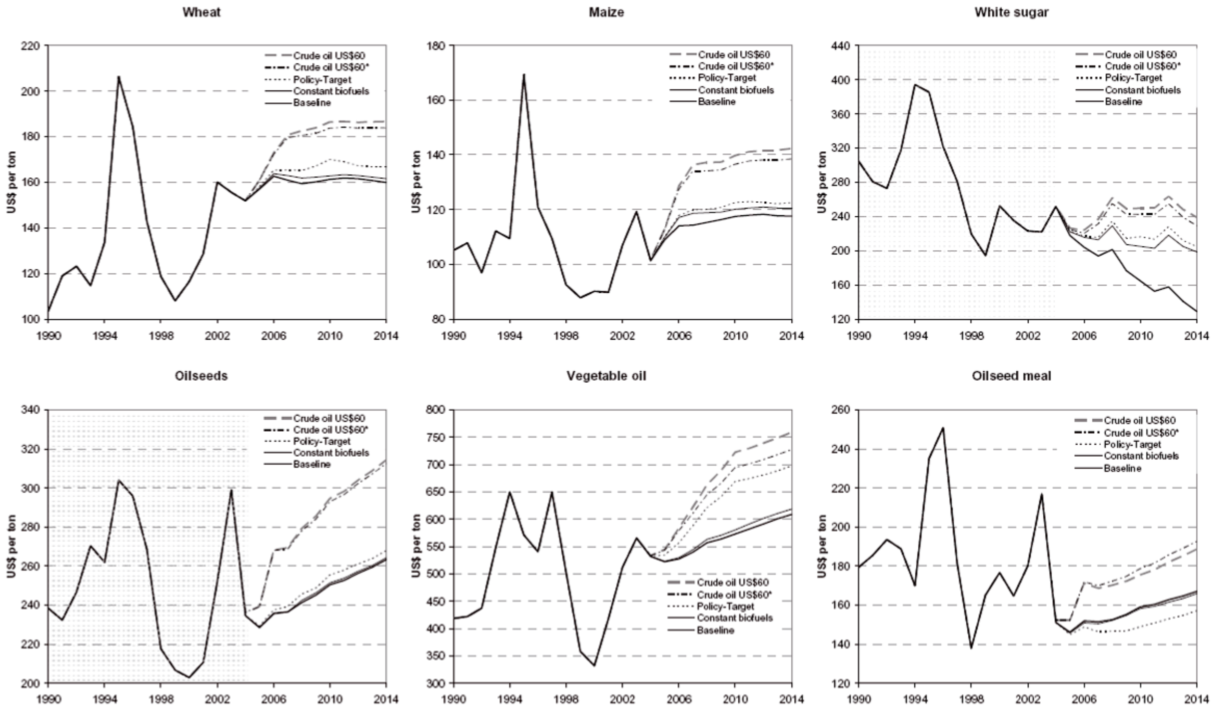

Figure 28. World crop market prices under alternative scenario assumptions (OECD, 2006).
Note: "Crude oil US$60*" denotes a scenario assuming higher crude oil prices, but unchanged petrol-based fuel prices (and hence unchanged biofuel prices) relative to the policy-target scenario.

demand. Commodity prices hence rose steeply, further pushed up by speculative transactions adding to increased commodity-price volatility (Von Braun, 2007). Despite a more favourable global production outlook for the coming years, prices are unlikely to return to the low levels of previous years due to a host of reasons, notably the escalation of cost of inputs and the need to replenish stocks (FAO, 2008a). Biofuel production has contributed to the changed world food equation. While cereal use for food and feed has increased by 4% and 7%, respectively, since 2000, the use of cereals for industrial purposes, such as biofuel production, has increased by more than 25%; in the United States alone, the use of maize for ethanol production increased 150% between 2000 and 2006 (Earth Policy Institute and FAO in Von Braun, 2007). As a consequence, the prices of commodities used in biofuel production are becoming increasingly linked to energy prices. The coefficient of variation of oilseed price in the past five years was 0.2[64], compared with typical coefficients in the range of 0.08–0.12 in the past two decades; that of maize has increased from 0.09 to 0.22 in the past decade (Von Braun, 2007).

Since feedstock represents the principal share of total costs, the biofuel sector will both contribute to feedstock price changes and be affected by them. Hence, the impact of biofuel expansion on projected food prices is not yet well established (Von Braun, 2007). Furthermore, due to competitive land uses, biofuels may also impact prices of other food commodities than their own feedstock. Higher prices for maize, for instance, caused food consumers to shift to rice and wheat, while it was more profitable for producers to preferentially grow maize over these same crops. Prices of rice and wheat therefore increased. Comparing the expected grain price increase between 2000 and 2007 with and without the biofuel growth that occurred at that time, IFPRI estimated that biofuels were responsible for 30% of the increase in weighted average grain prices (Rosegrant, 2008).

Using the OECD/FAO Aglink/Cosimo/Sugar model (OECD, 2006), impacts on commodity prices until 2014 were modelled with three biofuel development scenarios compared with a baseline scenario, i.e. status quo agricultural policies, normal weather patterns, and biofuel growth in the US and Brazil only: the constant biofuel scenario with no growth in biofuel production, the policy-target scenario with projected biofuel growth in line with officially stated goals (in 2004), and a third scenario considering the policy-target scenario and higher oil prices (Fig. 28).

The biofuel growth in the baseline scenario has a relatively small impact on coarse grain, wheat and vegetable oil prices, increases of 2.5%, 1% and 1.5%, respectively, compared with the constant biofuel scenario. Impact on livestock markets prices is also limited despite slight price decreases due to the production of co-products such as oilseed meal for feed. The impact on sugar price is much larger, a 37% price increase up to 60% in the policy-target scenario.

The policy-target scenario especially requires enhanced biofuel production in the EU and Canada added to increased production in the US and Brazil. As a consequence, substantial

[64] i.e. 20% around the annual mean price.

growth in feedstock needs implies significant trade pattern changes. EU imports of vegetable oils would increase threefold, while wheat exports would fall by 41%, and Canadian wheat and coarse grain exports would also decrease by 34% in 2014. Overall, world prices for most commodities increase substantially compared with the constant biofuel scenario, by 4% for wheat, and 15% for vegetable oils, also inducing an increase in butter price of 3% as a substitute for oil (OECD, 2006). Despite a more significant decrease in oilseed meal price (–6%), meat market prices increase with and without growth in biofuel production. With crude oil price at a sustained level of US$ 60 per barrel from 2005 to 2014 instead of US$ 45 to US$ 35 in the other scenarios, commodity prices first increase due to the high production costs. Biofuel costs also increase, but compared with the policy-target scenario, at the same time oil price provides a further incentive for biofuels, whose share would increase to 6% in 2014, compared with 5.5% in the policy-target scenario. The increase in biofuel production would lead to a further increase in commodity prices (OECD, 2006). In comparison, IFPRI modelled that in a 2007-baseline biofuel production, maize and oil prices would be higher by 6% and wheat and sugar by 4% in 2014 compared with a constant biofuel scenario (Rosegrant, 2008).

By 2020, taking current biofuel investment plans (2007) into account, international prices could increase by 26% for maize, by 18% for oilseeds, by 11% for sugar, and by 8% for wheat (Von Braun, 2007). While the large response of sugar price to biofuel production is suspected to be inaccurate (note 35 in OECD, 2006), the lower response in the IMPACT-WATER model (IFPRI) is also surprising since sugar price is highly related to ethanol and energy prices, unless sugar price increase is limited in IMPACT-WATER following the WTO sugar reform or other factors. In Boeters et al. (2008), the impacts on food prices of a 10%-biofuel target turn out to be negligible on the European Union scale compared with the policy baseline, i.e. the current economic welfare already impacted by distortionary taxes.

Market price projections and comparisons remain superficial, as many factors are still highly uncertain, such as food and feed demand, speculations on commodity prices, and biofuel international trade, including the future role of developing countries, and the development of 2nd generation biofuels. The latter are expected to be produced mainly from residues or dedicated energy crops that will not lead to an additional demand for food commodities as feedstock. The use of marginal lands could lower the pressure on agricultural lands and therefore the land prices. However, this potential remains limited due to low productivity, high costs and potential impacts on landscape.

Where dedicated crops would take over food crops for the use of land, the impact on food prices would also depend on local policies. For instance, within the framework of the US 1996 Farm Bill, a scenario of switchgrass production substituting 4–9.5 Mha allocated to food crops between 1996 and 2000 could have led to a price increase of from 4–14% of maize, sorghum, wheat, soybean, cotton and rice. However, through higher farm income and reduced loan deficiency payments, switchgrass promotion could have led to significant savings for the treasury (Ugarte and Walsh, 2002), the bottom-line cost being passed on to the consumer prices. As an example, according to OECD estimations, the CAP cost to ordinary citizens is around 100 billion € each year (US$ 125 billion), half from taxpayers and half from consumers owing to higher food prices. This is an average cost to an EU family (4 people) of around 950 € a year (US$ 1190), with only around 20 € (US$ 25) of this spent as EU money on targeted environmental programmes. The CAP has been estimated to be equivalent to a value added tax on food of around 15% and removing market price support would bring a one-off reduction in inflation of 0.9% (OECD, 2005).

7. CONCLUSION

Although bioenergy and biofuels in particular have recently been high on the policy agenda and subject to a lot of discussion, they still only contribute a marginal share in the global energy supply. Some key points may help to figure out how bioenergy can play a bigger part in the years to come.

- When reviewing biomass potential assessments, a rather modest assumption would be that the share of bioenergy in the total energy consumption can be multiplied by at least a factor of 2.5, but such deployment scenarios are extremely hard to predict since technology evolution is nonlinear[65]. This means breakthroughs may be expected along the way, notwithstanding the driving role of policies that are also constantly evolving. However, it remains certain that the contribution of biofuels will be limited because of the scarcity of available land, unless advances in technology make tremendously higher yields possible, in biomass production and conversion, with significantly lower costs. For second generation biofuels, a ten-fold increase in the plant productivity is still needed to reach commercial potential. A global 10% share of transportation fuels, excluding international aero-traffic, may be reached by 2020–2030, provided that biofuel chains are optimised in terms of both environmental and economic performance, and combined with changes in the automotive sector towards: lighter cars, hybrids, flex-fuel and city-vehicles. Taking the 13.5% contribution of transport to global CO_{2eq} emissions, total emissions in 2000 of 41.75 $GtCO_{2eq}$ (Baumert et al., 2005), and a range of greenhouse gas savings of 20–60% (FAO, 2008a), a 10% biofuel share would result in a reduction of 113–340 $MtCO_{2eq}$ per year. If second generation biofuels become available by 2030 and are combined with hybrid technology, biofuels could save an additional 1 $GtCO_{2eq}$ per year (IEA, 2006). This may represent a small contribution, but still a necessary one given the current trends and the scope of the GHG reduction targets (Fig. 29).
- The sooner second generation biofuels are commercially available, the more likely the 10% target will be met. Although they out-perform first generation biofuels, the latter

[65] The Schumpeterian vision of technology advances that evolve by plateaus punctuated by radical breakthroughs.

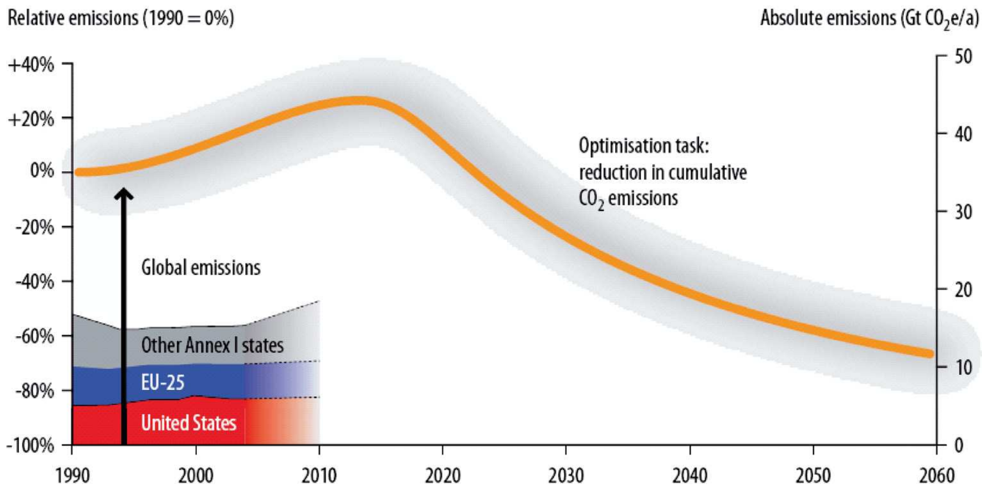

Figure 29. Necessary reduction in greenhouse gas emissions (including from deforestation and other land use) to stay within the 2 °C global warming target from today's perspective (Fischedick et al., 2007).

will develop further and serve as a springboard for the second generation biofuels to be quickly introduced within well-established biofuel chains. Therefore, attention must be paid to the diffusion of best available practices and to the attainment of sustainability standards of first generation biofuel chains.

- From a CO_2 perspective, the best mitigation pathway is to prevent emissions. A combusted litre of biofuel will never perform better for the environment that a non-combusted litre of fossil fuel. However, a significant part of global greenhouse gas emissions cannot be avoided and for those only CO_2 or C sequestration can have a mitigative effect. From a sequestration point of view, the interest of biofuels is to concentrate non-point sources from transport into facilities where they can be captured, as is the case with CO_2 from sugar fermentation, for instance. It can be expected that with the need for reducing industrial emissions, biofuel conversion facilities will keep improving in efficiency and gas savings or storage. Agricultural practices may also contribute to enhancing soil carbon sequestration; but while the residence time of stored soil carbon is not permanent and not easily controllable, the turnover of fixed atmospheric CO_2 by biomass and released through combustion can be quantified and in principle repeated indefinitely. Here, the intrinsic interest of biomass is that photosynthesis will be enhanced as atmospheric CO_2 concentration increases, provided that no other production factors are limiting, creating a negative feed-back loop. Finally, the overall interest of biofuels in terms of CO_{2eq} savings relies on the overall performances of agro-ecosystems, which in most cases and for all agricultural production can be largely improved.
- Soil organic carbon lost through deforestation negates the benefit of biofuel in terms of CO_2 savings. However, merely shifting the burden of deforestation and biodiversity losses onto biofuels will not stop land clearing for agricultural purposes. On the contrary, funds for biofuel development programmes could provide leverage to implement sustainability criteria for agricultural production worldwide, and to enhance the productivity of traditional slash and burn cultivation, thereby preserving the forest. Moreover, policies are also needed, notably in tropical regions, to empower local populations to prevent illegal logging and effectively urge forest preservation and investment in productive and environmental-friendly agroecosystems. In Brazil, for instance, it is cheaper to clear new land areas for the international beef and soya bean markets than to invest in already deforested regions[66].
- All bioenergy chains are not suitable for all locations. Bioenergy chains can bring benefits to the society in terms of fossil energy savings, as well as other positive environmental impacts, but only if the best appropriated bioenergy chain mix is chosen in accordance with local conditions, notably the biomass production systems and the types of primary energy inputs for conversion. Better knowledge of life-cycle greenhouse gas emissions from all energy uses of biomass, and strong sustainability criteria for biomass production, also addressing trade-off effects due to indirect land-use change, are still needed to fully assess the benefits and limitations of biomass use (EEA, 2008). Given the wide range of candidate biofuel chains, options will have to be identified and taken that minimise adverse environmental impacts while harnessing most of the advantages of biofuels. The resulting bioenergy mix will be systematically better for the environment than the business-as-usual scenario. In countries where land area is the main limiting factor, for instance, the priority should be given to biofuels from waste oils, animal grease, residues and municipal waste. The competition for the use of biomass for heat and power or biofuel should also be limited as far as possible by giving priority to the bioenergy chain that makes it possible to save the most greenhouse gases compared with the local substituted energy source.

[66] Dr Emily Boyd, 25/11/2005 http://www.scidev.net/en/opinions/emissions-trading-cannot-solve-amazon-deforestatio.html.

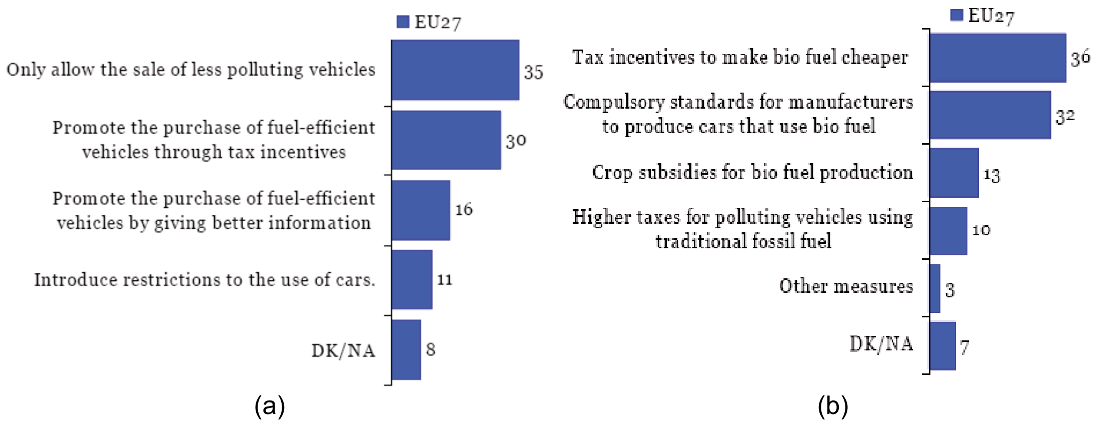

Figure 30. (a) The best way to reverse the rise of CO_2 emissions from road transport in response to question Q5: road transport generates about one fifth of the European Union's harmful emissions. Between 1990 and 2004, CO_2 emissions from road transport rose by 26%. Which is the best way to reverse this trend? (b) The best way to encourage the use of biofuels in response to question Q8: biofuels are renewable fuels that can reduce fossil oil dependence of vehicles. Which is in your opinion the best measure to encourage the use of bio fuels? %, Base: all respondents DK: Don't Know/NA: No Answer (Eurobarometer, 2007).

- In the case of sustainable bioenergy chains, positive externalities ought to be given economic values so that bioenergy could be more competitive. Overall benefits from bioenergy chains have to be considered, taking into account the value of all co-products. Considering the competition for natural resources, the principle of "zero waste" within integrated biorefinery appears to be the best economic and environmental choice. The use of contaminated or degraded lands for bioenergy purposes is also essential. Economic incentives should thus aim at fostering these priorities.
- The "success story" of ethanol in Brazil suggests that further growth in biofuel production can be expected through intensive breeding programmes to foster the development of second generation biofuel feedstock. Indeed, the high productivity of sugar cane has benefited from decades of research and commercial cultivation. Nowadays, cane growers in Brazil use more than 500 commercial cane varieties that are resistant to many of the crop diseases found in the country. Between 1975 and 2000, in the São Paulo state, the sugar cane yield per hectare increased by 33%, ethanol yield from sugar by 14%, and fermentation productivity by 130% (Kojima and Johnson, 2005). Another key element is the flexibility of the production unit, that has to be found as a balance between complete integration to reach maximum efficiency, minimum losses and economies of scale, while this optimum-oriented specialisation will not prevent the unit from being flexible enough to adapt the processes to various feedstocks and end-products. Most distilleries in Brazil are part of sugar mill/distillery complexes capable of switching between 60%–40% and 40%–60% sugar-ethanol, which makes it possible to take advantage of fluctuations in the relative prices of sugar and ethanol. In France for instance, mill/distillery complexes are optimised to produce 66% sugar and 33% ethanol on a year-run basis, with very little flexibility[67]. The success of the mill/distillery complexes also relies on good integration within both the nationwide ethanol supply system and the electricity grid to sell the co-produced electricity, once their auto-consumption is satisfied (Kojima and Johnson, 2005).
- Bioenergy can contribute to tackling part of the energy dependency and the depletion of non-renewable resources, but they alone will not suffice. Their development would also be vain if at the same time energy efficiency was not drastically improved and energy consumption behaviours did not change radically. As the transport sector is the main growing source of energy consumption and greenhouse gas emissions, biofuels play a critical role as long as the bulk of vehicles is not electrified. Despite improvements in the energy efficiency of various transport modes and the introduction of non-fossil fuels, increased transport demand, especially increased car usage and a reduced number of passengers per car, is outweighing these benefits. Present knowledge indicates that it will not be possible to achieve ambitious targets comparable with the Bali roadmap without limiting transport demand (EEA, 2008). In a recent survey, responses from 25 767 EU citizens indicated that 54% would be willing to pay more for using less-polluting transport. The best ways to reduce transport CO_2 emissions and to promote biofuels would be through an interdiction to sell "polluting vehicles" that do not achieve state-of-the-art emission standards, and tax incentives to foster both fuel-efficient vehicles and biofuels (Fig. 30, Eurobarometer, 2007). However, evidence also suggests that only a minority of individuals actually take action to reduce private transport energy consumption and fewer may intend to take action in the future. Analysing "Special Eurobarometer" surveys from 1984, 1993 and 2002,

[67] Personal communication by Dane Colbert, Director of Ethanol Union, 30/10/2008.

it appears that action concerning reducing car fuel use has not increased between 1993 and 2002, despite an increase in real fuel prices, and intentions to take action to reduce energy use were generally lower in 2002 than in 1993 or 1984. According to the 2002 survey, more than half of all respondents (55%) reported having taken no action on energy efficiency in any of the four broad transport energy efficiency measures examined (reducing travel, cutting fuel use, buying a more efficient vehicle and using public transport) while almost two-thirds of all respondents (64%) report that they do not intend to take further action in any of these areas of energy efficiency (Stead, 2007). Regarding the fact that awareness of transport pollution issues is, moreover, likely to be lower on a global scale than in Europe, radical measures may be necessary in order to radically change transport consumer behaviours. To address transport demand, measures and policy instruments must hence also go beyond the transport sector itself and be introduced into sectors of the economy such as households, industry and service, within which the demand for transport actually originates (EEA, 2008).

- Harmonisation at an international level is crucial in order to ensure the overall complementarity of bioenergy chains, to provide a coherent framework for the markets and to control the sustainability of the systems. In particular, the framework for international biofuel trade is complex: trade barriers should be removed so that no artificial competitiveness would hamper the development of biofuels, but on the other hand, control of sustainability is necessary on a global scale and instruments should be put in place to ensure it. Voluntary schemes for the certification of sustainable biomass already exist, e.g. the Forest Stewardship Council (FSC) for forest product or the Roundtable Sustainable Palmoil (RSPO), and are currently being discussed as implementation options for bioenergy sustainability standards. Hence most of the key elements for such standards are available as well as experiences from existing voluntary schemes. Although legally binding standards are superior, pragmatically, voluntary schemes might provide a well-needed start ("entry option") (Fritsche et al., 2006). Finally, cross-sector integration of agriculture, energy and transport policies is also mandatory to make biofuel incentives coherent and to send a clear message to the population.
- Concerns about GMOs and rising food prices are justified. However, these are not exclusive to biofuels, and should be addressed in a wider perspective. Not mentioning the debate on GMO, competition for land uses between food and non-food crops should be minimised as much as possible. Where competition for land uses is critical, market distortion should not, as in the case of the cotton market, spoil opportunities given to developing countries. It is important that governmental support for biofuels as an infant industry remains temporary, or else the policy will result in inefficient allocation of resources in the long run, once costs decline as output expands and production experience is acquired. The extent to which biofuel programmes can contribute to rural development is dependent on the industry characteristics and, ultimately, whether it is able to become financially viable without direct government support. However, if public funds are needed to support the industry, the question to be addressed in the first place is whether government resources will be diverted from other programmes and what would be the comparative impacts on rural development and the environment (Kojima and Johnson, 2005). From an agronomical point of view, promising options in developing countries are in particular those that introduce energy crops within agro-forestry systems. However, the need for an intensification of the production, and also the development of perennial plantations, may remain largely hampered as long as the lack of land property rights does not make it possible to empower the farmers. Biofuel programmes need to be integrated within a broader context of investment in rural infrastructure and human capital formation. Indeed, strengthening property rights, removing both international and domestic trade-barriers, access to education, water, electricity and networks, and developing transport infrastructure have proven better drivers of rural development than direct aid in providing farming equipment or inputs, for instance (Kojima and Johnson, 2005).

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