

# Greenhouse gas emissions of biofuels: improving Life Cycle Assessments by taking into account local production factors

Cécile Bessou

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# Greenhouse gas emissions of biofuels, Improving Life Cycle Assessments by taking into account local production factors

**Cécile Bessou**

**– Ph.D. Thesis –**

AgroParisTech
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that contributed to this Ph.D. work and whose names follow up.

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à ma famille et mes proches pour leur soutien.

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# General introduction

The need to quantify the environmental impacts of anthropogenic activities has never been more urging, as we started to observe their cumulative effects (e.g. the depletion of the ozone layer, global warming etc.). While actions should be taken immediately to try and mitigate these effects, our quantitative knowledge on the potential impacts of alternative production systems (in particular to substitute fossil resources) is still incomplete. Predicting the environmental impacts of such systems requires a capacity to model future scenarios by reproducing the underlying mechanisms leading to environmental pollution. It implies i) to understand and characterize these mechanisms; ii) to model them with an acceptable level of uncertainty; and iii) to use a standardized and holistic method to compare the potential environmental impacts of possible actions and avoid pollution trade-offs.

Biomass is the oldest and the most widely exploited renewable energy source. It can be grown in most areas and the diversity of plants makes it possible to provide biomass the whole year round. However, contrarily to flow resources such as solar or wind energy, biomass stock are constrained by feedstock productivity and land exploitation rates, and by a maximum environmental load. The latter corresponds to the maximum production that can be achieved within a given agroecosystem without breaking the balance between the resources that can be supplied and the amounts of pollutants that can be digested by the ecosystem. Agricultural biomass production systems are particularly constrained. On the one hand, agriculture must satisfy to an exponentially increasing demand through population growth (including changes toward more caloric diets), and the diversification of biomass uses. On the other hand, the intensification of production systems to match this demand leads to critically-high environmental loads.

Biofuels are fuels produced from biomass. Among renewables, they have been particularly fostered for their possible contribution in reducing greenhouse gas emissions by the transport sector, which is the major growing contributor to the greenhouse gas emissions. In this view, the interest of liquid biofuel relies on their ability to be blended and distributed with fossil fuels, and on the possibility to save diffuse greenhouse gas emissions from transport, whereas stationary emissions at site may be captured by other advanced technologies. However, biofuels are still source of pollutions that are linked to the production of the agricultural biomass and to its conversion and distribution. The overall interest of biofuels hence depends on the global savings they may enable compared to the fossil fuels they would substitute.

Many studies have been published that compare the environmental impacts of fossil fuel chains and biofuel chains. However, their results are highly variable, which has contributed to create some confusion on the overall interest of biofuels. Despite their common life-cycle based approach, not all these studies are complete Life Cycle Assessments (LCA), i.e. assessments that encompass all the potential environmental impacts throughout the product chain. LCA is a holistic and standardized tool that makes comparisons complete and less subjective [thus complying with the above-mentioned point iii)]. Nevertheless, biofuel LCAs are especially complex to establish and to interpret because of 1) the complexity of the environmental impact mechanisms of agricultural productions, and 2) the difficulty to deal with the diversity of co-products generated during the biomass conversion. Resulting from these constraints,

assessments are often lacking in transparency concerning the data quality and the underlying assumptions in co-product handling, which adds to the complexity to conclude on the impacts of a biofuel chain.

Good data quality standards are paramount to the reliability of LCAs, and are well defined in the LCA handbook (Guinée *et al.,* 2002). They include criteria such as transparency, completeness, and relevance. It is however problematic to gather the wealth of data required by the LCA that meet all these criteria. In particular, the data from the agricultural step are subject to high uncertainty, due to both 1) the variability related to local climatic, edaphic and management factors (often disregarded) and 2) limited understanding and modelling of the underlying mechanisms [*cf.* the above-mentioned points i, ii)]. Agricultural causes a diverse range of environmental impacts. They are linked to management practices and in particular to the use of inputs such as fertilizers or pesticides. The type and state of the ecosystem, i.e. the soil, climate, fauna and flora, and pollution background levels, also influence the way a given input will contribute to the diverse impact mechanisms. Among these impacts are the emissions of $N_2O$, which is the fourth contributor to global warming (Forster *et al.,* 2007). Arising mainly from agro-ecosystems through an enhancement of soil microbial activities by fertilizer N inputs, $N_2O$ emissions may seriously counterbalance the greenhouse gas savings of biofuels.

Although many studies have focused on measuring $N_2O$ emissions from agricultural soils and on analysing the underlying processes, quantitative models are still limited in their ability to simulate the dynamics of these emissions as affected by local production factors over the whole crop cycle. Given the importance of $N_2O$ emissions within the biofuel greenhouse gas balances, their uncertainty strongly impacts the results of biofuel LCAs and hampers their relevance to the selection of biofuel options as an alternative to fossil-based fuels, lest model performance is improved,.

## Research objectives

This Ph.D. thesis work is part of the research activities of INRA (French National Institute for Agricultural Research) within the department “Environment & Agronomy”. A main research field within this department focuses on the analysis of the impacts of agricultural activities on the environment and of abatement strategies. Two laboratories were involved: the Agro-Impacts (Laon/Mons), and the Environment and Arable crops research unit (EGC) jointly managed by INRA and AgroParisTech (the ParisTech institute of life and environmental sciences). The bio-atmosphere group of EGC focused on atmospheric pollutions and on how to characterize and model the underlying mechanisms. The underpinning scientific issues are to develop both appropriate techniques to measure the fluxes of polluting gases, and robust models to describe observed fluxes and to try predicting potential ones. In the field of bioenergy, these lines of research are expected to provide key results regarding agricultural biomass potential contribution to climate change mitigation.

The objectives of this Ph.D. thesis work were 1) to improve our knowledge on the determinism of $N_2O$ emission and to model the underlying processes and controls; 2) to produce biofuel LCAs at local scale with a higher accuracy on greenhouse gas emissions (based on part 1)), so that they can be used as decision-tool at this scale. We chose to focus on two types of agricultural feedstock that are interesting candidates for biofuel production in terms of yield per hectare, given that land availability is a crucial limiting factor for bioenergy development. These crops are sugar beet, *Beta vulgaris* (1$^{st}$ generation biofuel), and *Miscanthus x giganteus* (2$^{nd}$

generation biofuel). First-generation biofuels are produced oilseed, starch or sugar crops through common food industry processes, whereas second-generation ones are produced from the lignocellulosic of all plant material thanks to more complex and still under-development processes.

Sugar beet is a common head crop in the study area (Picardie region). Following the Sugar Reform (2005) and regulation on sugar free imports (Balkans Agreement, EBA Agreement) European sugar production has been decreasing. In France, the first European sugar producer, the sugar beet areas in France were reduced by 11% between 2008 and 2009 (SSP, 2009). Due to this reduction, sugar refineries have been closed, notably in the Picardie region, while the proportional number of combined refinery-distilleries has increased. Since it is not limited by quotas, sugar beet ethanol hence appears as an interesting opportunity for sugar beet producers to diversify their outlets.

Contrarily to sugar beet, *Miscanthus x giganteus*[1] has been only recently implemented in the study area, which is among the pioneer sites to introduce this crop in France within agricultural rotations. This perennial crop originates from East Asia; it was introduced to Europe as ornamental plant about 70 years ago. As a C4 plant, its high nutrient and water use efficiency makes it a good candidate to produce high biomass yield for bioenergy. It has been therefore under scientist scrutiny in the field of bioenergy for a dozen years. Nevertheless, references on both cultural and environmental aspects are still lacking, notably in the context of France. In line with the competitive objectives of the Picardie region in the field of "Industries and Agro-resources", field experiments have been implemented by the INRA AgroImpact research unit at Estrées-Mons to define agronomic and environmental references for some newly developed energy crops, upon which *Miscanthus*.

In the Ph.D. frame, we focused on two key questions concerning the determinism of $N_2O$ emissions for the two crops tested in the field. Previous results of the "soil tillage" experiments in Estrées-Mons had shown that anoxia created into some soil compartments influenced greenhouse gas emissions, but further analyses were needed to characterize the impact of soil structure on $N_2O$ emissions. We therefore chose to study first the impact of soil compaction on $N_2O$ emissions in the sugar beet plots. Soil compaction is notably due to mechanical operations in the field and compacted areas are partly fragmented during soil tillage. Modelling the impact of soil compaction on $N_2O$ emissions is hence part of the modelling of the multi-dimension impacts of varying soil tillage on $N_2O$ emissions. Concerning *Miscanthus*, for which references on most environmental impacts are scarce comparing to more established agricultural crops, we studied the effect of N-fertilization on $N_2O$ emissions. N-fertilizer input is a main factor influencing $N_2O$ emissions. Moreover, N-input has been found to have rather low impact on *Miscanthus* yields. It is therefore especially necessary to monitor the impact of N-fertilization on $N_2O$ emissions, since high amount of this N-input may be left available by the plants to microorganisms.

[1] NB: *Miscanthus x giganteus,* where not specified in this thesis, is further referred to as *Miscanthus*.

The thesis work consisted of three main parts:

- a monitoring of $N_2O$ and $CO_2$ fluxes and driving environmental factors during two crop cycles of sugar beet and *Miscanthus*;
- an analysing of how soil compaction influenced $N_2O$ emissions in sugar beet plots and an integration of the resulting relationships within a denitrification-nitrification (i.e. $N_2O$) sub-model;
- simulations with an agroecosystem model incorporating the modified $N_2O$ sub-model to produce the input data for the local biofuel LCAs to be compared: ethanol from sugar beet and from *Miscanthus*.

## Dissertation structure

This thesis manuscript is divided in three main chapters, following a general introduction and leading to overall conclusions and perspectives (Fig. 0):

Chapter 1. Literature review on biofuels and greenhouse gas emissions with a special focus on $N_2O$ emissions from agriculture. This review also explores some broader aspects of bioenergy, i.e. economic and political frameworks and biomass potential, which are presented in appendix 1. The complete review was accepted for publication in *Agronomy for Sustainable Development.*

Chapter 2. Results of the modelling work on the impacts of soil compaction on $N_2O$ emissions in the sugar beet plots. This chapter was accepted as a regular article in *European Journal of Soil Science.*

Chapter 3. Local LCAs of ethanol from sugar beet and lignocellulosic ethanol from *Miscanthus*. This chapter is a project of regular article.

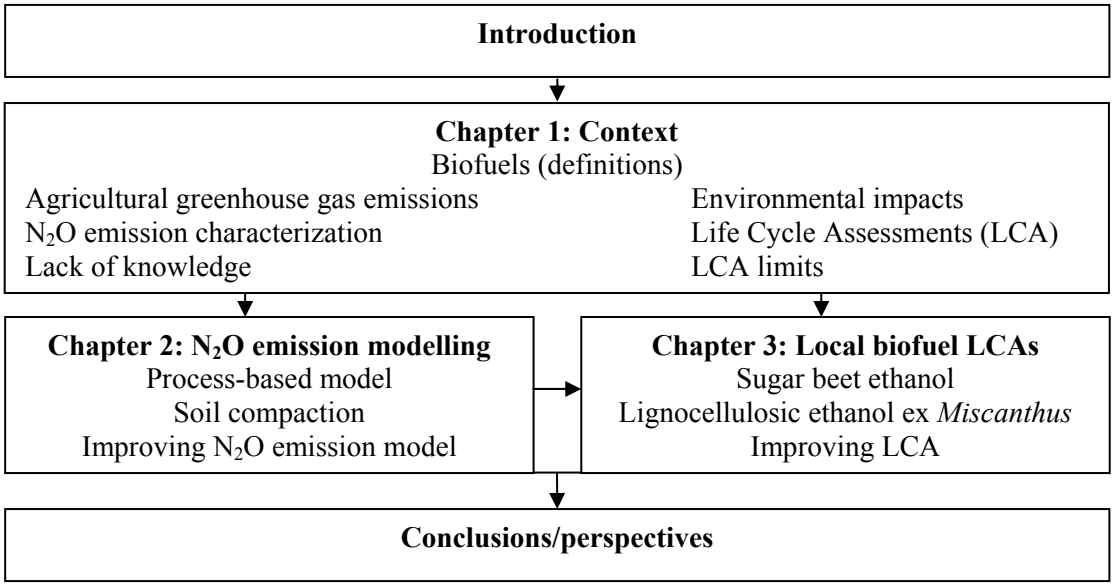



**Figure 0:** Dissertation structure and workflow

# Chapter 1

## Biofuels, greenhouse gases and climate change

**Review** (*Agronomy for Sustainable Development,* in press)

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**Key words**
Biofuels
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Land use change
Bioenergy potential
Perennials

## – Abstract –

Biofuels are fuels produced from biomass, mostly in liquid form, within a period 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 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 $2^{nd}$ generation biofuels from lignocellulose should be available on the market by 2020. Research is active on the improvement of their conversion efficiency. A 10-fold increase compared to 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 to fossil fuels, biofuels combustion can emit less greenhouse gases throughout their life cycle, considering that part of 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 standardized 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 many data, and whose methodology is still lacking harmonization. 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 neither at local nor 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 improve definitely 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 biofuels development will ultimately be land availability. Given available land areas, population growth rate and consumption behaviours, it would be possible to reach by 2030 a global 10% biofuel share in transport sector contributing to lower global greenhouse gas emissions up to 1 $GtCO_{2\ eq}.year^{-1}$ (IEA, 2006), provided that harmonized 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, 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.

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## 1. Introduction

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

In 2005, the world total primary energy supply approximated 11,430 Mtoe $yr^{-1}$ (479 EJ $yr^{-1}$ [2]), compared to 6,130 Mtoe $yr^{-1}$ (257 EJ $yr^{-1}$) in 1973 (IEA, 2007a). According to 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 barrel per day (Mb/d); the global worldwide current production averaging 86Mb/d (ASPO, 2008).

According to some experts, the peak-oil will occur in 20 years, whereas others argue that the world is already at peak production. If one might argue on the exact moment, it is though generally accepted now that it will happen soon and that an energy transition is unavoidable (Van der Drift and Boerrigter, 2006). ASPO, Association for the Study of Peak Oil, states that overall oil and gas productions will be at their peak by 2010, conventional oil peak would be already 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 of 1 to 2% each year since 1986 (EU DG-TREN, 2005). Cheap reserves will not be sufficient to fulfil the world growing energy demand and fossil fuels are pointed out to be the main anthropogenic cause of global warming. These increasing supply and environmental costs make petroleum no longer the only candidate as 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.

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

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 provide 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 by 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 population (IEA Bioenergy, 2007). The contribution of bioenergy to the global supply mix has almost not 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 the atmosphere 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 to 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 third of the projected increase in oil demand will come from transports 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 on the longer-term that fundamentally reduce $CO_2$ emissions, because energy will always be at a premium (Hansen, 2006). The automotive market logically evolves towards electric motors, whose energy efficiency is roughly 7.5 times higher than that of combustion ones. The compacity and lightness of liquid fuels still enable 50 fold higher energy storages than best current batteries (Roby, 2006). Fuel cells may in the future replace these limited electro-chemical accumulators. Development of these technologies and changes in the market and consumption behaviours are essential. Still electricity or $H_2$ are energy vectors that can be produced with different level of $CO_2$ emissions.

Biofuels can contribute to reduce the dependency on fossil fuels and lower greenhouse gas emissions from transports, provided that the savings of greenhouse gases through the use of bioenergy is not counteracted by an increase of 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 consumptions 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 biofuels sustainability. We therefore 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 woodfuel being produced and consumed in the developing countries (Parikka, 2004). The prefix "bio", from the Greek "βίος"(meaning "life")[3], 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[4]. The total sum of living organisms is called biosphere by 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 that can be used as fuel for transports, or energy source to produce industrial or domestic heat and electricity (Feedstock and conversion routes Fig. 1). Bioenergy come from biomass. In contrast, fossil energies are mineral resources, stocked in the lithosphere. The carbon fossil energy sources are the result of mineralization transforming organic matters into mineral matters. 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 in two main types of natural resources: flow resources and renewable stock resources. Flow resources, like 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.

Considering biomass, "renewable" indicates that it shall 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 resources management is technically appropriate, environmentally non-degrading, socially favourable and economically viable, then the renewable resource shall be exploited in a sustainable[5] 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

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

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

[5] Based on the sustainability's pillars concept in the Brundland report, 1987

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 relies on the fact that the emitted $CO_2$ from the plant originates from the atmosphere where it eventually goes back. 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 residues amount that would remain available for 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 percents 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 of this stock could critically affect 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, forest biodiversity are among precious resources, whose conservation for the future generations should not be jeopardized by land use changes. For the producers, "Renewable Biomass" shall not anymore just mean "which can be grown" but also implies conditions for a sustainable production.

**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 to release 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.

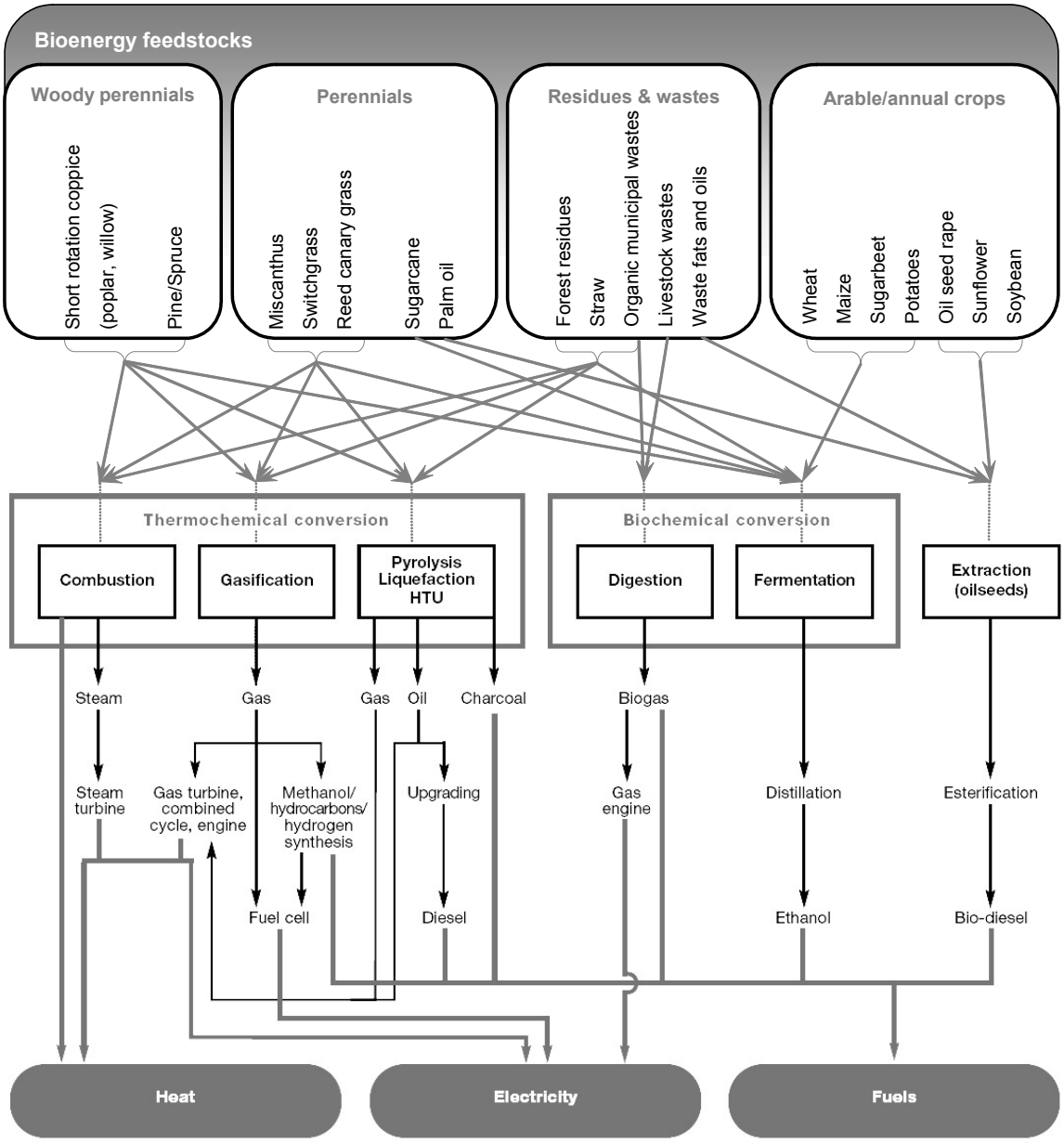



**Figure 1**: Bioenergy feedstock and conversion routes, adapted from Plassat, 2005 and UNDP, 2000
HTU: hydrothermal upgrading

Biomass gives way to a whole product chain in which residues or co-products can largely contribute to the environmental and economic optimization 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 summarized 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 to all the other renewables, bioenergy firstly present 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 shall aim at finding the best cost/benefit equations, 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 increase investment risks. Secondly, whereas bioenergy chains, like the other renewables, were intrinsically thought to be advantageous first of all at a local scale, biofuels especially give way to international trades that can raise the issue of externalities displacement and imply the need for specific political and economic frameworks.

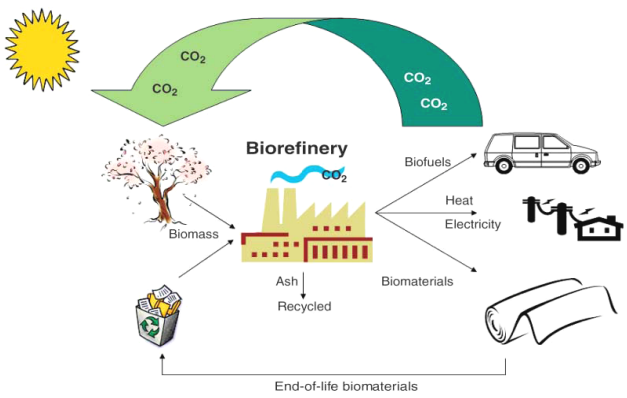



**Figure 2**: The fully integrated agro-biofuel-biomaterial-biopower cycle for sustainable technologies (Ragauskas *et al.*, 2006)

## 3. Transportation biofuels

Biofuels are nowadays commonly classed as 1st, 2nd or 3rd generation biofuels as shown in Table 1. First-generation biofuels refer to 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 cost account for the majority of a 1st generation biofuel's eventual price, 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 on the fundament that the last ones 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 proportional 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.

**Table 1**: 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)<br>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<br>Fischer-Tropsch (FT) diesel*<br>Dimethyl Ether (DME)*<br>Methanol*<br>Mixed Alcohols (MA)*<br>Substitute Natural Gas: SNG | $H_2$* |

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

### 3.1 First-generation biofuels

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

*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). Other potential plant oil feedstock includes mustard seed, linseed, castor oil, peanut, cottonseed, coconut, *Lesquerella* or micro-algae. There are as much 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* fruit, Belize, February 2003
bessou©INRA

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. However, if they contain many saturated fatty acids, oils and fats are solid at room temperature and cannot be directly used as fuel in a diesel engine except in warm climates. The disadvantages of vegetable oils compared to 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 methylester and 103 kg of glycerol (Graboski and McCormick, 1998). In this three-step reaction, triglycerides are converted to 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 are of the order of 10-20 times and twice that of diesel fuel respectively (Lang *et al.*, 2001). 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 in 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 to their high sulphur predecessors (Radich, 2004; Goodrum and Geller, 2005). Some fatty acids like ricinoleic (castor oil) and lesquerolic acids (*Lesquerella*) could be especially efficient to enhance 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 the 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).

*Ethanol*

Ethanol, on the contrary to biodiesel, is a single-compound biofuel whose final composition does not vary with the type of feedstock. Feedstock is sugar and starch crops, which are basically equally processed through pre-treatment, fermentation by yeasts and other microbes, and distillation. 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 stalk contains sugar. 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 to 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 the entry in the process chain. In the wet-milling, grains are soaked and chemically sub-divided into rich starchy parts of primer 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 in grinding the unprocessed heterogeneous seeds into granules. It is therefore less expensive but also leads to less diverse co-products productions. The main co-product is the dried distillers grain (DDG) fed to animals that can digest high proportions of fibres, and contributing 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 the storage and the 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 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 of 7% ethanol (ADEME/DIREM, 2002).

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[6] and particulates 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[7], 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.*, NOx emissions are reported to be in the range between ±10% as compared to diesel depending

[6] Sulphur oxides (SOx) contribute to acid rain and can be carcinogenic.
[7] NOx are precursor to the formation of tropospheric ozone.

on engines combustion characteristics (2008). Biofuels increase the octane level[8] (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.

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 fuels and vehicles 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 on 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 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 biofuels and blends standards were adopted worldwide by the various agencies, ABNT/ANP[9] in Brazil, ASTM International in the US or CEN[10] in Europe. The Tripartite Task Force has been working on implementing a road map to come up with international compatible biofuels-related standards to help increasing the use of biofuels and avoiding adverse trade implications in a global market.

The water content of ethanol shows how critical it may be to agree on standardized 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 up to 7% (Tripartite Task Force, 2007). Considering the additional environmental and economic costs related to biofuels upgrading, an international harmonization should urge a maximum level of blends flexibility correlated with a focus on a minimum processing and optimum engine modifications.

*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 the substrate contents, the greater the biogas yield is (Moras, 2007). Apart from about 55 to 70% of methane (52-65% in examples in table 2), the actual fuel, biogas also contains substantial amounts of $CO_2$, 30 to 45%, small quantities of hydrogen sulphide and other trace gases such as ammonia[11]. 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.

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[8] The high octane value of gasoline 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).

[9] Brazilian Petroleum, Natural Gas and Biofuels Agency

[10] Comité Européen de Normalization

[11] www.planete-energies.com (consulted on 10.03.2008)

Biogas is less considered as transportation biofuel, because its target vehicle fleet remains marginal due notably to on board gas storage constraints. Primarily interest of biogas remains its local development as fuel for heat and power plants in rural area. 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 lately 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 degrade finally just about 10 to 15% of the initial material, biogas permits to take advantage of cheap feedstock and diminish greenhouse gas emissions. Moreover, the solid residue of the process can be dried and used as fertilizer, which 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 the use in city fleets and trucks as it has been the case in Sweden and Switzerland for a long time. Table 2 gives the productivities of common feedstock and biofuels worldwide.

### 3.2 Current 1st generation biofuel supply worldwide

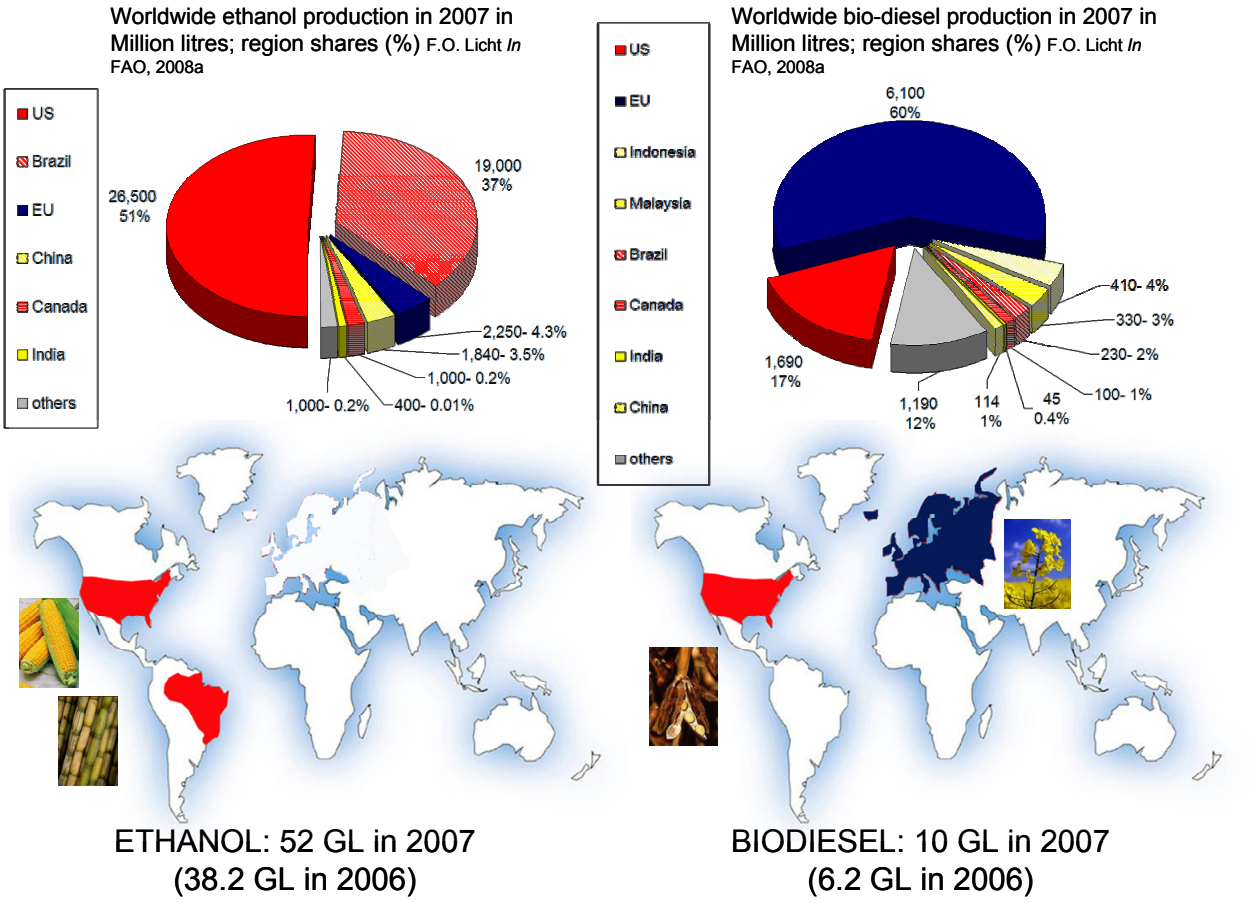
Combustible renewables and waste are mostly consumed directly; only about 1.8 % is consumed by the transport sector, about 17.6% by the industries and 80.6% by other sectors notably households (IEA, 2007a). Production of heat and electricity dominate current bioenergy use with two key industrial sectors 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 six fold. 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%[12] of its transport fuel consumption (OECD, 2006). The remainder 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[13] production worldwide, mainly from rapeseed and sunflower seeds, with Germany as leading producer (40%), followed by the US, France and Italy generating most of the rest (Worldwatch Institute, 2007). In 2007, the EU still accounts for 60% of global biodiesel production that amounts 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 producing countries in Asia or Oceania, and more than tripled in Brazil. Figures 3 & 4 show the production shares of ethanol and biodiesel worldwide in 2007.

[12] Various data: 13.2% in energy terms according to IEA 2006; 40% according to Xavier 2007.
[13] Biodiesel does not here take into consideration pure vegetable oils mostly directly consumed by farmers on the farm.

Despite the growth in biofuels consumption and a comparative 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 still minimal, only 10% of biofuels produced in the world are sold internationally; with Brazil accounting for roughly half of those sales (USDA, 2006). However, trades are 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).




**Figures 3 & 4**: Worldwide ethanol and biodiesel productions in 2007 in Million litres; region shares (%) Drawn from data of F.O. Licht[14] *In* FAO, 2008a

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

**Table 2:** Main feedstock, productivities per hectare, and co-products of 1st generation biofuels
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.

| **Biomass feedstock** | **Countries regions** | **Yields tFM 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=0.791kg) | 1l ethanol =**0.665l** 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<br>FNR, 2007 |
| SUGAR CANE | Brazil | 85 | 7,080 | 4,708 | ~8 [2.1-8.3] | Yeast as cattle feed supplement;<br>Bagasse as feedstock for feed, heat, electricity, and cellulosic ethanol;<br>Fructose as sweetener; invertase for food industry<br>23.8 t ha$^{-1}$ bagasse (50% $H_2O$) =7,854 kWh electricity | Brazilian Ministry *In* Xavier, 2007<br>Cardona and Sanchez, 2007 |
| | | | 6,500 | 4,323 | | | Fulton *et al.*, 2004 |
| | | | 6,800 | 4,522 | | | 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<br>Kadam, 2000 |
| | Thailand | 55.7 | - | - | | 104 kg sugar/t cane<br>+45 kg molasses/t cane (+5 kg rice husk) =10.2 l ethanol [national availability~1Mt yr$^{-1}$ molasses] | Nguyen and Gheewala, 2008<br>Prasertsri, 2006 |
| | Global | 65 | 4,550 | 3,026 | | | Rajagopal *et al.*, 2007 *In* FAO, 2008a |
| SUGAR BEET | France | 66.2 | 3,793<br>7,307 | 2,522<br>4,860 | ~2 [1.2-2.2] | 3.7 t ha$^{-1}$of sugar,<br>If only ethanol without sugar production<br>44t green manure (10% dry matter) | ADEME/DIREM 2002 |
| | | - | 7,500 | 4,988 | | (Stabilized) molasses as fertilizer<br>Sugar beet pulp and dried slop as animal feed<br>Slop to biogas<br>Heat | France Betteraves (FB)/Passion Céréales 2007 |
| | Germany | 55.5 | 5,439 | 3,617 | | | FNR, 2007 |
| | EU | -<br>68.9 | 5,500<br>- | 3,658<br>- | | | Fulton *et al.*, 2004<br>JRC/EUCAR/CONCAWE, 2008 |
| | Global | 46 | 5,060 | 3,365 | | | Rajagopal *et al.*, 2007 *In* FAO, 2008a |
| SWEET SORGHUM | China | 60<br>90 | 5,070<br>7,605 | 3,370<br>5,057 | ~1 [0.9-1.1] | 3-4t sorghum grains | USDA, 2007 |

| Biomass feedstock | Countries regions | Yields tFM $ha^{-1}$ | 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 1l = 0.96l diesel fuel sunflower oil 1l = 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 tFM 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 (1l=0.9kg) | 1l biodiesel= 0.88-0.95l diesel = 0.85l diesel =**0.891** diesel | | | Worldwatch Institute, 2007<br>ITDG, 2000<br>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<br>Glycerol sold or converted to biogas then steam | FNR, 2007 |
| | France | 3.34 | 2,184 | 1,944 | | | ADEME/DIREM 2002 |
| | EU | -<br>3.11 | 1,200<br>- | 1,068<br>- | | | Fulton *et al.*, 2004<br>JRC/EUCAR/CONCAWE, 2008 |
| SUNFLOWER | France | 2.44 | 1,690 | 1,504 | 3 | | ADEME/DIREM 2002 |
| | EU | -<br>2.44 | 1,000 | 890 | | | Fulton *et al.*, 2004<br>JRC/EUCAR/CONCAWE, 2008 |
| SOYBEAN | USA | -<br>2.7 | 500<br>552 | 445<br>492 | ~3 [1.5-3.5] | | Fulton *et al.*, 2004<br>Naylor *et al.*, 2007 *In* FAO, 2008a |
| | EU | - | 700 | 623 | | | Fulton *et al.*, 2004 |
| | Brazil | -<br>2.4<br>2.8 | 400<br>491<br>- | 356<br>437<br>- | | | Fulton *et al.*, 2004<br>Naylor *et al.*, 2007 *In* FAO, 2008a<br>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<br>Detoxified meal or cakes as animal feed,<br>Meal to produce biogas<br>Shell as fuel<br>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<br>FAO, 2008a |

| Biomass feedstock | | Countries regions | Yields $m^3$ $tFM^{-1}$ | 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 | - | $CO_2$ (25-45% of biogas) Digested biomass as fertilizer | 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 | | | | | | | | 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 | $402.10^3$ $l.t^{-1}$ | | - | | | |
| WASTED OILS | | Global | $800.10^3$ $l.t^{-1}$ | | - | - | | |
| *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.

### 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 in intricate assemblages of cellulose, hemicellulose and lignin, whose proportions and molecular organization 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 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, this last one also contributing to make cells walls hydrophobic and more resistant against enzymatic attack (Möller *et al.*, 2006). Other molecules such as for instance, waxes or inhibitors to fermentation that 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 primarily barrier to produce ethanol from lignocellulosic feedstock, commonly referred to as cellulosic ethanol. Indeed expensive pre-treatments are necessary to breakdown 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 solubilize 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 acids hydrolysis methods are more mature but very energy intensive and present the disadvantage to potentially also degrade 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 100-fold more enzymes than hydrolysis of starch (Tolan 2006 *In* Möller *et al.*, 2006).

Researchers have been therefore focusing, on the one hand on improving the yields of pre-treatment and lowering their costs, and on the other hand on developing integrated processes that make it possible to protect, separate and use the other compounds, like the C5-sugars and the lignin, or the co-products such as furfural and fermentation gases. Traditional fermentation

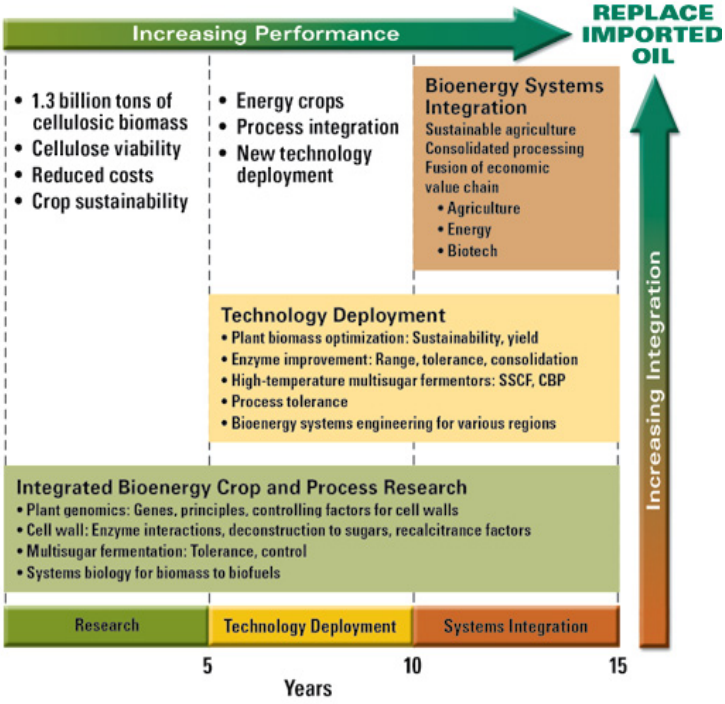
processes relied on yeasts and microbes that only convert C6-sugars (mainly glucose) to ethanol (Fulton *et al.*, 2004). Researchers have already succeeded in producing several new yeast strains and bacteria, such as engineered *E. coli*, *K. oxytoca* and *Z. mobilis* (Balat *et al.*, 2008) that exhibit varying degrees of ability to convert the full spectrum of available sugars to 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 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). The "consolidated biomass processing" (CBP), 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 yeast strain requires 1) screening thousands of combinations biomass-degrading enzymes to identify the appropriate set of enzymes, then 2) managing to ensure that this strain is capable of simultaneously express the genes for all necessary enzymes (Hector *et al.,* 2008).

Current biomass-conversion technologies were developed empirically based on a limited understanding of the biological and chemical properties of biomass (Himmel *et al.*, 2007). Therefore, all research efforts also rely in parallel on fundamental researches to better understand and characterize the cell walls of a very wide range of biomass feedstock. 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) 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. Researches worldwide include breeding programmes to develop new varieties with interesting phenotypes in terms of growth and resistances, or regarding specific biorefinery-related assets, i.e. regarding the cell wall composition. Researches also include 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 lead to reduce biomass by up to 40%, which emphasizes the need to determine wether cell wall manipulation

may compromise the plant's structure integrity or susceptibility to peasts and pathogens (Chapple *et al.,* 2007).

Figure 5 shows the expected timeframe 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 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 to sugars for fermentation, high yield, and robustness will be essential in contributing to achieve energy security.






*Miscanthus*: Estrées-Mons, November 2008 bessou©INRA

**Figure 5**: Phased Development of Bioenergy Systems over the next 5 to 15 years (US DOE, 2006). SSCF: Simultaneous Saccharification and Co- Fermentation; CBP: Consolidated Biomass Processing

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 of cellulosic ethanol commercial plants were being developed in 2007-2008 essentially in the US (Solomon *et al.*, 2007) or under discussion in Canada or China. Steam pre-treatment with the addition of a catalyst for hydrolysis and improved enzymatic digestibility is the closest technology to commercialization and has been widely tested in pilot-scale equipments (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 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's costs, respectively (Kaylen *et al.*, 2000). In recent simulations, production costs, mostly based on 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 governmental funds in a first development phase, appear to Murray Burke[15], 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, maximization of ethanol yields and optimization of the use or commercialization of co-products must be achieved. Actual 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 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). 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, 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 intrinsic 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 are 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 shall notably permit 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 worldwide produced syngas 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 vary depending on the process (Fulton *et al.*, 2004).

There are two major types of gasifier that were selected because of their high efficiency to produce $H_2$ and CO, although they still produce a different ratio of gases at different

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

temperatures and with a differing level of cleanliness (ITDG, 2000; Van der Drift and Boerrigter, 2006). Fluidised-bed gasifier, typically operated at 900°C, has been already 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, entrained flow gasifier, typically operated at 1300°C, makes it possible to produce syngas without catalytic reformer but needs energy intensive pre-treatment of the biomass, such as torrefaction or pyrolysis, in order to reach 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 to diverse hydrocarbon liquids and fuels (Fulton *et al.*, 2004).

The diverse options are compatible, 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, in particular of methane used as biogas to provide heat and power. Mature biorefinery (Fig. 6) is here the combination of biochemical and thermo-chemical treatments, which shall permit to produce as much biofuels (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 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 2$^{nd}$ and 3$^{rd}$ 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 complete integrated biorefinery complexes, as well as technological revolutions in the transportation sector to introduce hydrogen as competitive fuel for automotives. Hydrogen ($H_2$) is a fuel, whose combustion produces only water. Although water vapour is the most important 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 fertilizers, to upgrade lower quality fractions in the refining of fossil fuels, to produce also 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 must be produced from hydro-carbonates or water requiring considerable energy inputs.

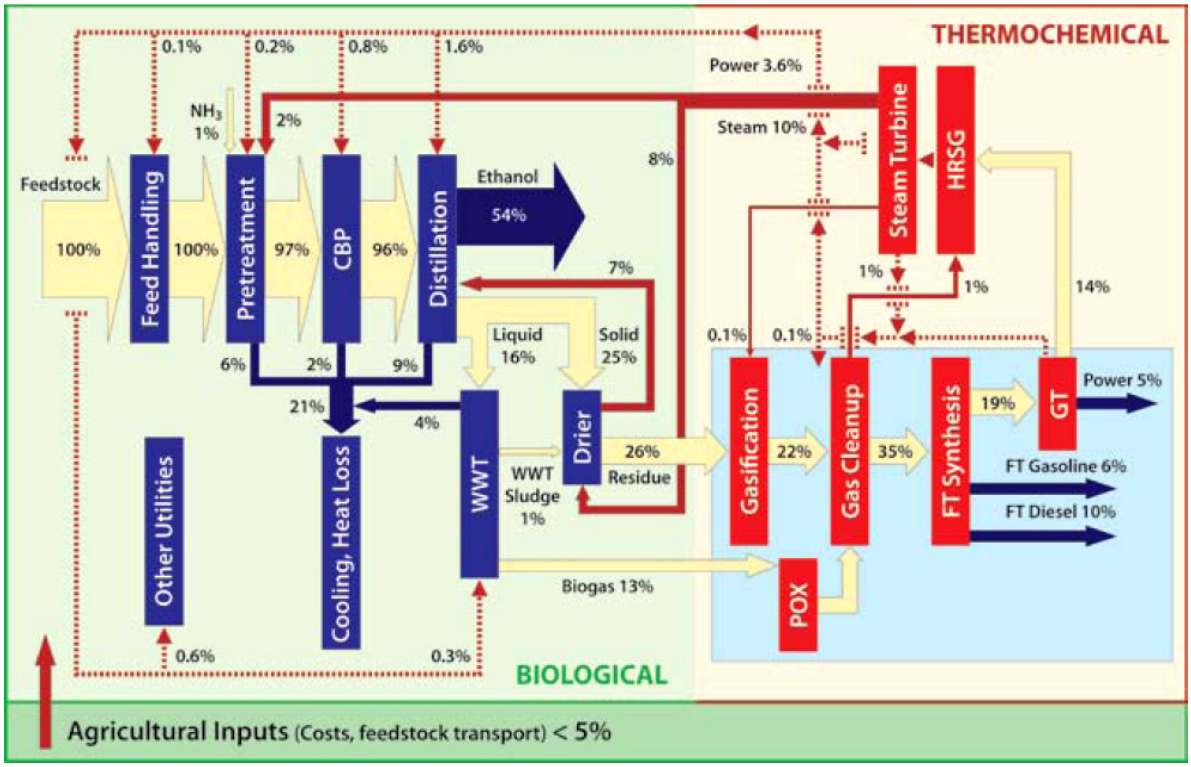



**Figure 6**: Mature Biomass Refining Energy Flows: Example Scenario *In* US DOE, 2006a "Envisioning Mature Biomass Refineries," presented at First International Biorefinery Symposium, Washington, D.C. (July 20, 2005)
Dotted arrows from above indicate energy inputs needed to run machinery.

Hydrogen is designated as 3rd generation biofuel, when it is produced from biomass via the thermo-chemical above-described processes. However, this term would not be appropriated when talking about hydrogen coming from the conversion of fossil fuels, even if the processes should ensure 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 that releases $CO_2$ in the atmosphere. Production via water electrolyse 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 rentable 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 energy as gasoline, due to its small 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 tree 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 trades, as economic incentives often appear as efficient levers to reach targets.

## 4. Political and economic frameworks

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## 5. Biofuels and greenhouse gases

### 5.1 Assessing the environmental impacts of biofuels

*Life Cycle Assessment of biofuel chains*

#### The Life Cycle Assessment tool

Life Cycle Assessment (LCA) is a holistic methodology developed in the 1980s[16], 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 optimized 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 environment problem merely by shifting it to another stage in the product's life cycle (Guinée *et al.*, 2002) and this is particularly crucial when it comes to decision on national and international levels on global issues like energy and climate change. The LCA methodology consists in 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 characterize correctly and completely 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, LCA tool has been standardized by the ISO norms 14 040 series (40/41, 43, 47-49, 1997/98, 2000/01[17]), there are several methods to apprehend the impacts and their characterization, 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 a lot performed during the last decade notably in order to compare the environmental performances of bioenergy chains between one another or to 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 10 shows a simplified scheme of a representative system boundary for 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

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

[17] 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.

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 infrastructures 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 to the chain, are not included in the system boundary, essentially solar energy for photosynthesis, but also the so-called indirect energies like 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 for energy chain. 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, what 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 optimizing the production yield of the one product of interest.

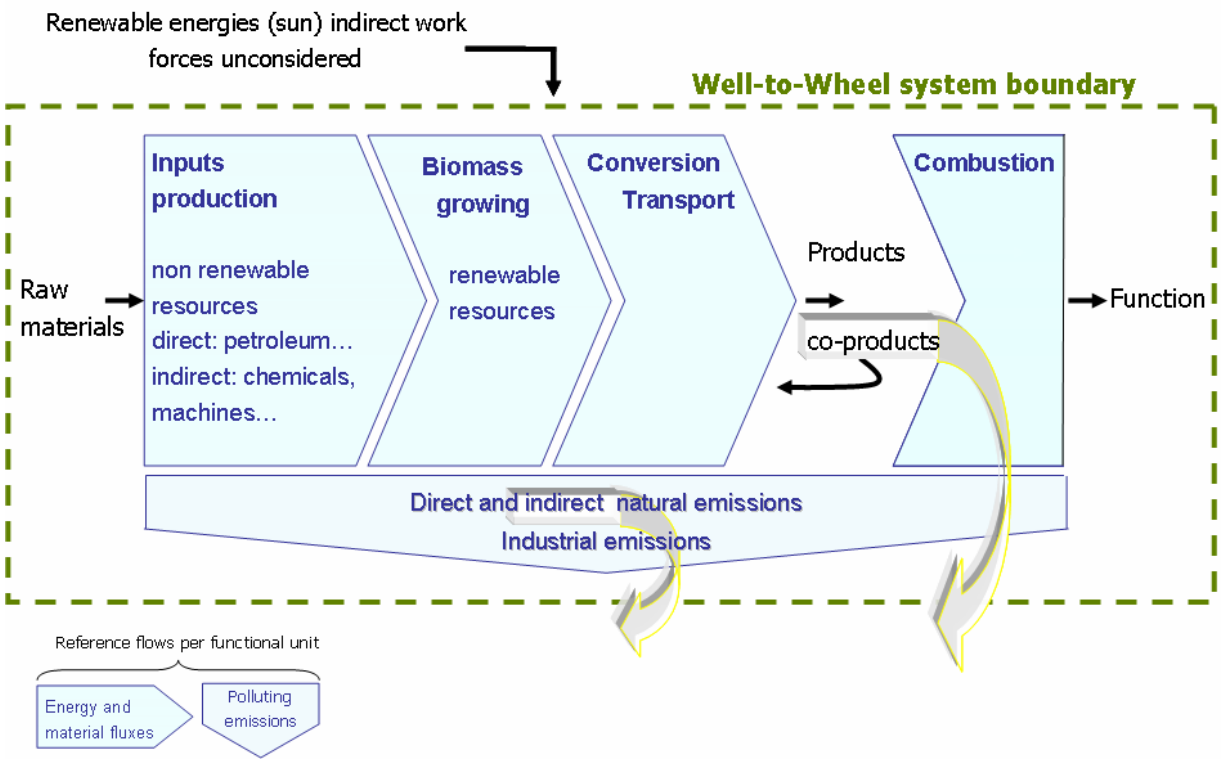



**Figure 10**: Biofuel LCA simplified Well-to-Wheel system

A distinction can be done 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 to 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 to these "natural second order" emissions, whereas emissions through cultural operations, transport, conversion processes, storage etc. will be referred to as industrial emissions (Fig. 11).

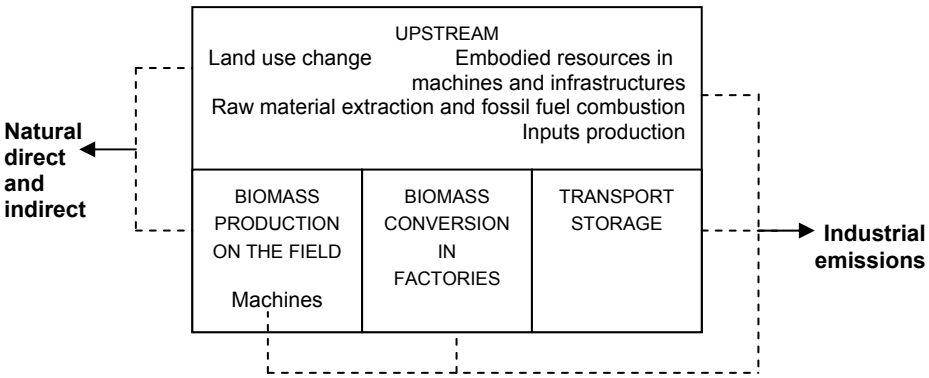



**Figure 11:** Natural and industrial emissions of a bioenergy chain (without combustion)

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 characterize renewable energy systems (Malça and Freire, 2006). The respective definition and use of among others "energy efficiency", "overall energy balance", "gross/net energy requirement", or "energy renewability efficiency" (Malça and Freire, 2006) shall also be clarified when comparing bioenergy chains assessments.

**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 to their fossil counterparts if competition for other uses of the resource is not considered (Quirin *et al.*, 2004), whereas taking into account the impacts like acidification, eutrophication or ozone depletion reverses this trend. Indeed, these drawbacks are linked to intensive agricultural productions that notably lead to nitrogen compounds emissions responsible for those bad impacts on the environment. Relative advantages of biofuel chains between one another depend on the feedstock. Figures 12 & 13 illustrate the results of a comparative study of 109 biofuel chains by the IFEU, i.e. the ranges of greenhouse gases and primary fossil energy savings by substituting fossil fuels by the biofuels assessed in some 64 studies (Quirin *et al.*, 2004).

The comparison based on a unit MJ energy content of biofuels (Fig. 12) makes it possible to compare all the chains, included biofuels from residues, whereas the comparison on a hectare basis (Fig. 13) introduces a land use perspective. Amongst 1$^{st}$ generation biofuels, bioethanol

from sugar cane, molasses and sugar beet, as well as biogas from wastes (here compared to gasoline, but also advantageous compared to natural gas) show the best combined performances in terms of both greenhouse gas and primary fossil energy savings (Fig. 12). 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 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 to a MJ of ethanol, whose distillation and dehydration are energy-intensive. In comparison, primary energy input in refining gasoline or diesel is not significantly different between the two fuels. Therefore, diesel substitutes can save the most of 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 perform better than the worst ethanol chains but are worst 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 then more on the feedstock types and the cropping systems. In the case of sugar crops, energy inputs are partly compensated 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 biodiesel compared to pure rapeseed oil. The latter is not refined and would alone 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 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 a 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 to use economic allocations for co-product handling.

From a hectare perspective (Fig. 13), 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 important savings possible than ethanol because of its high energy-related hectare yield that permits to displace higher amounts of fossil fuel; for each MJ of ethanol that replaces gasoline, 3 MJ of ETBE[18] replace MTBE. However, on an energy-unit basis ethanol is slightly more interesting than ETBE, as ETBE requires a further processing step compared to ethanol. ETBE is though here compared to MTBE, which is produced with more energy input than gasoline but emits a bit less greenhouse gases as bulk of the energy is provided by natural gas for 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 compensate for the ethanol higher density, ETBE permits more $CO_2$ equivalent reduction per MJ of ethanol

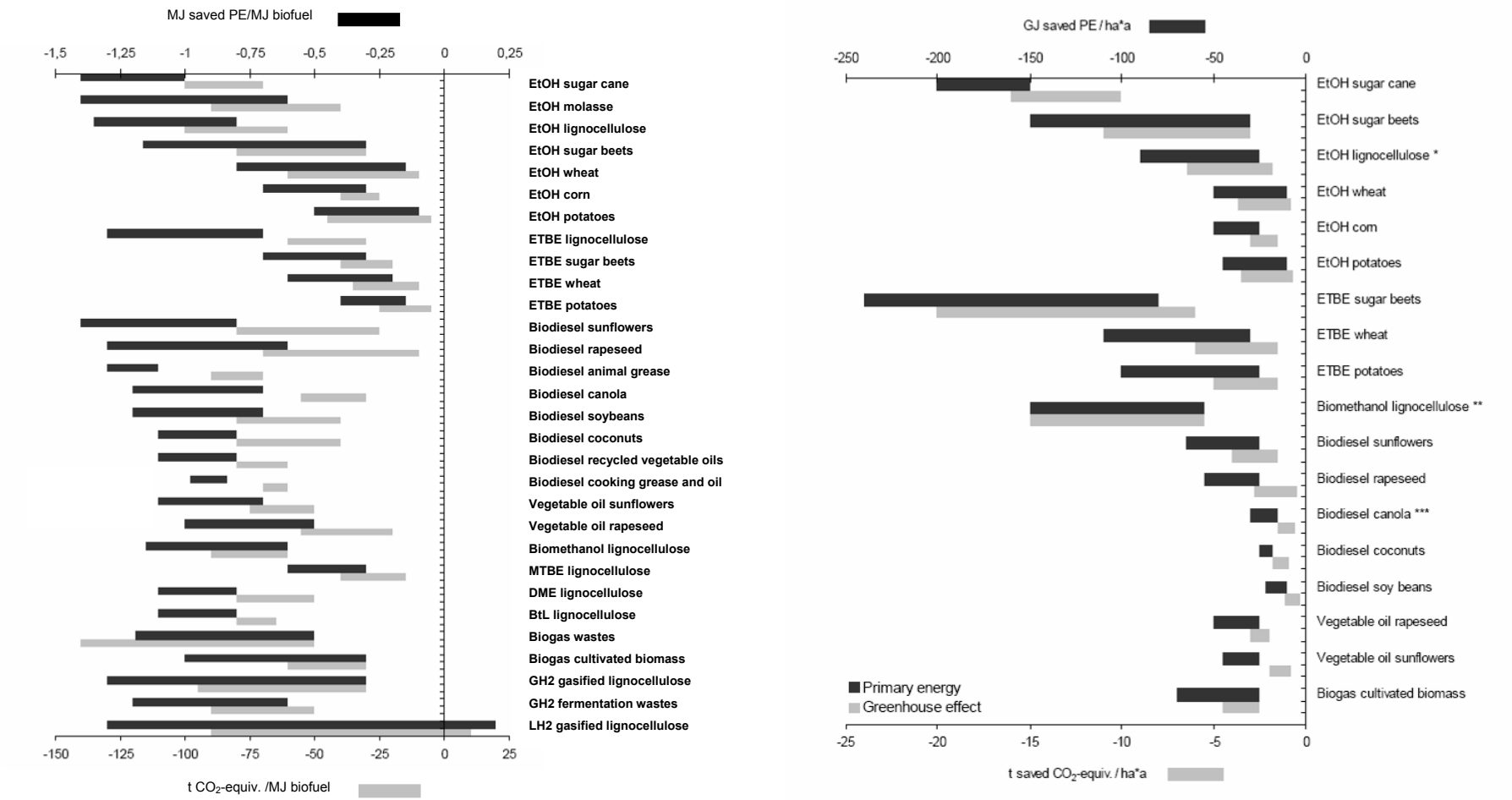
[18] 3 MJ of ETBE are produced from "1 MJ of ethanol and 2 MJ of isobutylene"

(Croezen *et al.*, 2007, Higgins, 2007). Blending in ethanol or ETBE reduces the petroleum base fuel requirement for butanes and reformate's octane numbers. Compared to an LCA in which ethanol and ETBE simply replace MTBE, these modifications in the refinery operations cause an additional 3% GHG savings in the ethanol case, and about 20% in the ETBE case. In the ethanol case, 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 ETBE case, less volume of petroleum base fuels is needed for the same volume of ethanol (5%) with an even lower octane numbers' requirement. Although this advantage 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 CO2-eq/GJ ethanol converted into ETBE against 37 kg CO2-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. 12). 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 lie on the fact that the whole plant is transformed, increasing the energy yield for a same mechanical and chemical treatment, which implies that the environmental load is lower for a 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 coppices (SRF/SRC) generally have less impact on soil erosion, nutrient inputs into 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 a more extensive root system present throughout the year, which provides 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 to 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 case add 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 fertilized.




**Figures 12 & 13**: Results of the energy and greenhouse gas balances of the biofuels as compared to their fossil counterparts (gasoline for ethanol, biomethanol, biogas and H2; 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 (Fig. 12) and per hectare a year (Fig. 13). Negative values imply advantages for the biofuels; the zero mark means that the $CO_2$ 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 restrictedly 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

*Limits of the LCA tool*

**Data quality**

When compiling results from diverse biofuels LCA, 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 firstly 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 from their local environment has lead to a site-independent methodology for LCA. However, the life cycles steps in close contact with the environment (such as agriculture or land filling) are site-dependent by nature (Kodera, 2007). Thus by essence, LCA implies a site-independent model of a biofuel chain at a given period, which is merely compatible with the dual agricultural-industrial nature of biofuel chains. First consequence is the difficulty to collect 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 amongst biofuel chains 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 on 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 of estimated emissions to the atmosphere, waterborne, and solid waste are approximately 50-150%, 1000%, and 30%, respectively (*In* Lo *et al.*, 2005). In addition, models based on international or national average values might not be useful not either adaptable to more specific local production conditions for decision-making on regional levels. Moreover, LCA steady-state model cannot take directly into account the variation of 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 neither encompassed in LCA.

The varying quality of input data makes the comparison of diverse scenarios outputs more difficult. The lack of transparency and homogeneity on background assumptions between different biofuel chain assessments may hide that data might not always be reliable. To deal with data quality and uncertainties, tools exist and could, if systematically associated to the results, enlighten comparisons between assessments. A “pedigree matrix” permits 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 shall 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 traduce 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 matrixes" for different specific types of data or domains could also serve as reference, 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).

**Co-products handling**

The complexity of bioenergy chains evidences limits of LCA. Indeed, LCA methodology leaves some liberty degrees, when it comes to deal 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 burdens allocation. This is, however, impossible in the case of bioenergy chains in which product and co-products come from the same feedstock, i.e. are chemically linked. In this case, methods of co-products 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 weights 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 in defining ratios in function of the market value of each product. This last option can be relevant in the sense that economics will still be the underlying driving factor of a biofuel chain development, while LCA does not take into account economic factors. Nevertheless, 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 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 to the fossil chain could vary between around 68% to 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 between 89.7-95.6% of total primary energy in the case of sugar beet, whereas it varies between 42.7-91.2% in the case of wheat (Malça and Freire, 2006). Kim and Dale (2002) also found that burdens to corn ethanol widely vary following the co-products handling, within 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 that 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 productions 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 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 would be 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 at the local scale.

**Impact characterization**

Many 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 to an exhaustive impact characterization also faces methodological as well as scientific constraints. Life Cycle Impact Assessment (LCIA) is the step that actually permits to quantify the potential environmental impacts. It consists in classifying and aggregating the results of the inventory into category indicator results (Eq. 1) that characterize how the environmental mechanisms for the chosen impact categories are modelled, and which are the contributions of the involved flows to these impacts.

**Equation 1**

$$\text{Impact Category Indicator}_i = \sum_j [\text{Characterization Factors } (_{i,j}) \text{ x Emission/Extraction Inventory } (_j)]$$

where “j” denotes the substances or resources (Brentrup *et al.*, 2004)

Traditionally in LCIA, characterization 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 logics and approaches underpinning the characterization (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 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). It exists different classifications of AoP that are actually connected; e.g. human health, natural environment, and manmade environment, or 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) or EDIP (Wenzel *et al.*, 1997), have been the main widespread historically, whereas endpoint ones, like EPS (Steen, 1999) or Eco-indicator 99 (Goedkoop and Spriensma, 2000) are more recently gaining interest, since they imply the analyse 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 designing 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 enough relevant 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[19] across

[19] Weighting and normalization are two non-mandatory steps in LCA methodology.

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 characterization, notably in order to better account for the location and time of the emissions, waste generated, and resources depleted, as well as the geographical zone and period over which the contributions to different impacts should be considered (Pennington *et al.*, 2004). Linear characterization (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 at a place where it will not cause any harm; as such its emission shall be considered as contributing to potential impacts independently from site and time (Heijungs *et al.*, 2003). It follows that impact categories are assumed independent from one another, and unless a precise scientific background permits to justify hypothesis to partition the contribution of a same substance to several impacts, the substance flow shall contribute in its entirety to each impact (Guinée *et al.*, 2002). Firstly, these assumptions only lead to calculate potential impact scores, not actual damages (Khalifa, 1999; UNEP, 2003). Potential impacts shall thus represent the worst-case scenario, where some redundancy of a substance contribution is preferred onto 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 influence 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 characterization 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 jeopardize 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 include other determinisms than chemical or physical ones. Khalifa (1999) emphasizes 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 by 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, then 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 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 characterization, 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 better simulate acidification discrepancy 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" consisting 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 as 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 of emission of substance (s) in 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 characterizations are integrated over an infinite period, since the variation of emissions at the regional scale during the period for integration is considered as marginal when compared to the total contributing emissions (Potting and Hauschild, 2005). Indeed, LCA steady-state assumption is funded on the "multiple sources-multiple receptors" character of present environmental problems, i.e. the temporal variation of 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 at 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 (like 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 emphasizes that, at the regional scale, these nitrate fate factors can imply large variations compared to 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 rate and locations define 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 could 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 emissions and meteorological data from 1985 through 1995. Regional $NH_3$ and $NO_x$ fate factors express the fractions of these airborne emissions that actually end up into the aquatic environment taking into account both direct deposition in freshwater and marine environment and run-off from terrestrial systems to the

aquatic environment. Now, for a short timeframe, these regional fate factors may vary depending on punctual variations in precipitation patterns compared to those used to model the fate factors. The characterization 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, 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 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 at 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 purpose, 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 on 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 on 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 on probability distributions etc.) and may, if completely represented, even entail 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 LCA of diverse commodity chains across different case studies is hampered by the lack of international harmonization 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 basis, and further extrapolation across regions shall be limited to LCAs that use harmonized methods.

## 5.2 Focus on greenhouse gas emissions from agriculture

"Climate change" is often presented as an example of well-established midpoint impact characterization, notably because its global dimension particularly fits with the "multiple sources-multiple receptors" background assumption. However, when assessing biofuel chains, it appears that greenhouse gas emissions on the field can show highly variable temporal and spatial patterns, and that these emissions contribute for an important share of the total greenhouse gas balance of the whole chain. Emission during agricultural productions contribute for instance to 34%-44% to the greenhouse gas balance of corn ethanol in the US (Farrell *et al.*, 2006) up to more than 80% in the case of pure vegetable oils (ADEME, 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 more detailed, i.e. the determinism of $N_2O$ emissions and $CO_2$ emissions due to land use change.

*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. fertilizers 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 fertilizers or pesticides. Briefly, the more intensive the use of machines and inputs are, the higher the overall emissions of pollutants. In Farrell *et al.* (2006), agricultural practices across six compared corn to 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, $CO_2$, $N_2O$ or $CH_4$ are by-products of the microbial activity, which is characterized 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 gases 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 amount includes approximately: a) 3.3 $GtCO_{2eq}\ yr^{-1}$ of $CH_4$ (50% of total $CH_4$ anthropogenic emissions[20]) 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 b) 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 to overall anthropogenic $CO_2$ emissions. The net flux between the atmosphere and the agricultural land,

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

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[21] 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 $GtCO_{2eq}$ $yr^{-1}$ in 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 accounted carbon sinks 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, thinning of forests for instance. Finally, the assessment does not consider the indirect effects of fertilization by N-deposition or increased atmospheric concentration of $CO_2$ that could partly counterbalance the raise 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 (Houghton, 2003; Baumert *et al.*, 2005). Sixty percent of the 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 to a net sink (Houghton, 2003). Nevertheless, anthropogenic $CO_2$ emissions from land use change and forestry sector are subject to *extraordinary uncertainties*[22] 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 managements (Houghton, 2003). Hence, estimates of the carbon fluxes are uncertain about ±150% for large fluxes and ±50MtC $yr^{-1}$ 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). In order to better assess the greenhouse gas emissions due to agricultural productions, 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 frame of biofuels from energy crops, on $N_2O$ and $CO_2$ emissions.

[21] The remaining 6% of agricultural greenhouse gases by subsector are undifferentiated sources of $CH_4$ and $N_2O$.
[22] As written by the authors Baumert *et al.*, 2005 p.91

*$N_2O$ emissions*

**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 fertilization. Second, all the Nr[23]-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 characterization 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 antagonist processes occurring between mid-point impact (e.g. acidification) and end-point damage (e.g. water toxicity), inducing further uncertainty on indirect emissions that are linked to primary direct emissions.

A wide range of experiments and studies has been focusing on how to improve N-fertilization efficiency in order to reduce firstly 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 or applications of fertilizers to the crop needs and pedo-climatic conditions. However, many questions remain especially concerning the determinisms of gaseous N-losses at 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 deficit in N fertilizer recovery that vary between 1% and 35% (Recous *et al.*, 1988).

A 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[24] 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[25]) (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). $N_2O$ also is the main source of stratospheric NO[26] 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 to nitrate, and denitrification is the anaerobic reduction of nitrate to nitrogen gas ($N_2$). $N_2O$ is an obligate

[23] Nr means reactive nitrogen compounds, i. e. all inorganic and organic N compounds except $N_2$ that is a nonreactive N compounds.

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

[25] Radiative forcing ($W.m^{-2}$), or global warming potential, traduces 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 increase, a negative radiative forcing (cooling) when precursors that lead to the destruction of greenhouse gases are released in the atmosphere. Halocarbons also are 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.

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

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 fertilizers 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 $N_2O$ releases by oceans. Compared to $CO_2$ (Fig. 14 & 15), 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[27] (Chapuis-Lardy *et al.*, 2007). $N_2O$ uptake is often masked by larger $N_2O$ production and may be indirectly accounted 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 better into consideration its contribution to global $N_2O$ budget; 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. Considering that oceanic $N_2O$ source may be underestimated[28] by at least two-folds (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 fertilizer, 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[29] influencing median values of $N_2O$ measurements, gives estimates for total direct $N_2O$ emissions[30] 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

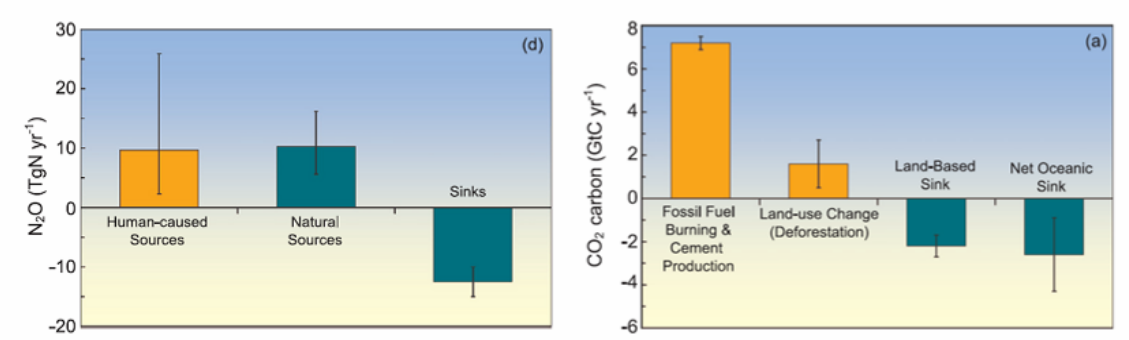
[27] Soil humidity favours denitrification up to $N_2O$ reduction, while $NO_3^-$ is preferred as electron acceptor over $N_2O$ (Granli and Bockman, 1994).

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

[29] Climate, crop type, fertilizer 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 that 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).

[30] Data for fertilization IFA/IFDC/FAO (1999), land-use *In* FAO, 2001.

$N_2O$ budgets have been continuously evolving notably due to improvement in considering the diverse direct and indirect sources and refining emission factors; the uncertainty on anthropogenic $N_2O$ remains remarkably important (Fig. 16).




**Figures 14 & 15:** Global budgets of $N_2O$ (14) and $CO_2$ (15) (Denman et al., 2007)

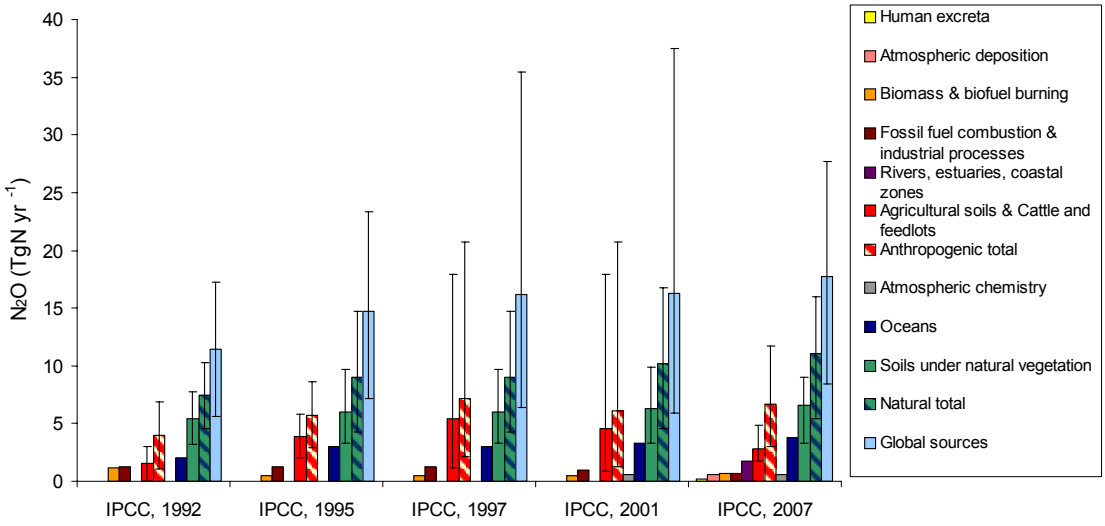



**Figure 16:** IPCC assessments of total annual $N_2O$ emission during the 1990s. Drawn from IPCC data *In* Mosier *et al*. 1998 and Denman *et al*. 2007. Note: for the 1997 assessment 0.9 Tg $yr^{-1}$(Mt $yr^{-1}$) of emissions by agricultural soils are subtracted to the total 6.3 Tg $yr^{-1}$(Mt $yr^{-1}$) 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.

**Origins of uncertainties**

Uncertainties in estimating $N_2O$ emissions from agricultural fields originate in the difficulties 1) to identify all the primary sources, the contribution of biological N-fixation[31] is especially hard to quantify (Mosier *et al.*, 1998; IPCC, 2006); 2) to follow 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) to capture and characterize the spatial and temporal high variability of 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) on the rate of nitrification and denitrification, 2) on the proportions between the gaseous end-products of these reactions, and (3) on 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 (fertilizers 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 on Fig. 17), 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), volatilization and nitrification. Secondary N-flows, shown by the solid lines (Fig. 17), encompass $N_2O$ emissions from nitrification and denitrification through two indirect pathways; (i) following volatilization of $NH_3$ from urea, ammonia or manure application, of $NO_x$, and the subsequent re-deposition of these gases and their products $NH_4^+$ and $NO_3^-$ to soils and waters; and (ii) after leaching and runoff of Nr, mainly $NO_3^-$.

Throughout the whole Nr lifecycle, 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%), or meat (CAFOs[32] 20%) (Galloway *et al.*, 2003). Despite current knowledge, it is still not possible to predict with reliability 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 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 to the terrestrial biosphere was relatively well known: 18% was exported to and denitrified in coastal ecosystems, 13% deposited to the ocean, and 4% directly emitted as $N_2O$; the remaining 65% either accumulated in soils, vegetation, and groundwater or was denitrified to $N_2$, but the uncertainty of those estimates remains large at 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-fertilized 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

[31] 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 Galbally *et al.* (1992), legumes could increase $N_2O$ emission 2 to 3 folds compared to unfertilized fields Duxbury *et al.* (1982) *In* Mosier *et al.* (1996). This denitrification by rhizobia could also lead to net $N_2O$ consumption depending on local factors.

[32] Concentrated Animal Feeding Operations

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).

**Emission factors**

The IPCC guidelines to assess $N_2O$ emissions from managed soils (IPCC, 2006) consider all identified direct $N_2O$ sources, except direct emission through biological fixation due to the lack of experimental evidence, and the two above-mentioned indirect pathways. The Tier -1, -2, -3 methods consist first in an comprehensive accounting of all N-input to the fields, including inorganic and organic fertilizers, as well as mineralized 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, -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 characterized by Duxbury and Mosier (1993) (Fig. 17) would emit 1.86 kg $N\text{-}N_2O$ 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 $N\text{-}N_2O$ ±1kg emissions (Mosier *et al.*, 1996). The rough 2.5 kg $N\text{-}N_2O$ estimated by Duxbury and Mosier (1993) did not include all indirect contributions.

Statistical models aim at finding reproducible correlations, i.e. relationships representative for most data sets, between controlling factors and emissions, e.g. emission factors depending on the type and amount of applied fertilizer. In this sense, the more data are collected on the different direct and indirect emissions, the more the emission factors will be informative. The multiplication of data sets and the development of complementary techniques at various scales (aircraft measurements, micrometeorological techniques, chambers (see picture), $^{15}N$ balance, $C_2H_2$ inhibition and lab works) have already made it possible to improve the understanding of controlling factors beyond a coarse linearity with N fertilizer. 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, Estrées-Mons, June 2008 bessou©INRA

Given the complexity of $N_2O$ emission control (Fig. 18), 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 for these variables for instance, by essence emission factors cannot lead to significant reduction of estimation uncertainty and cannot always be used to test different management or mitigation scenarios (Beheydt *et al.*, 2007).

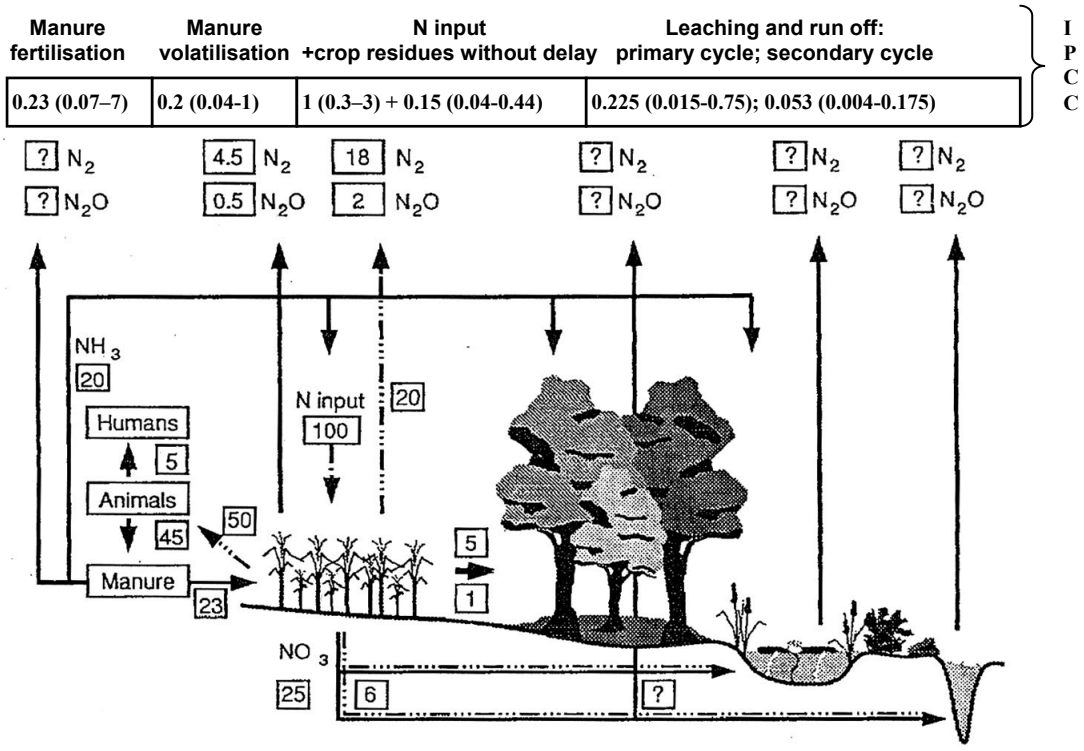
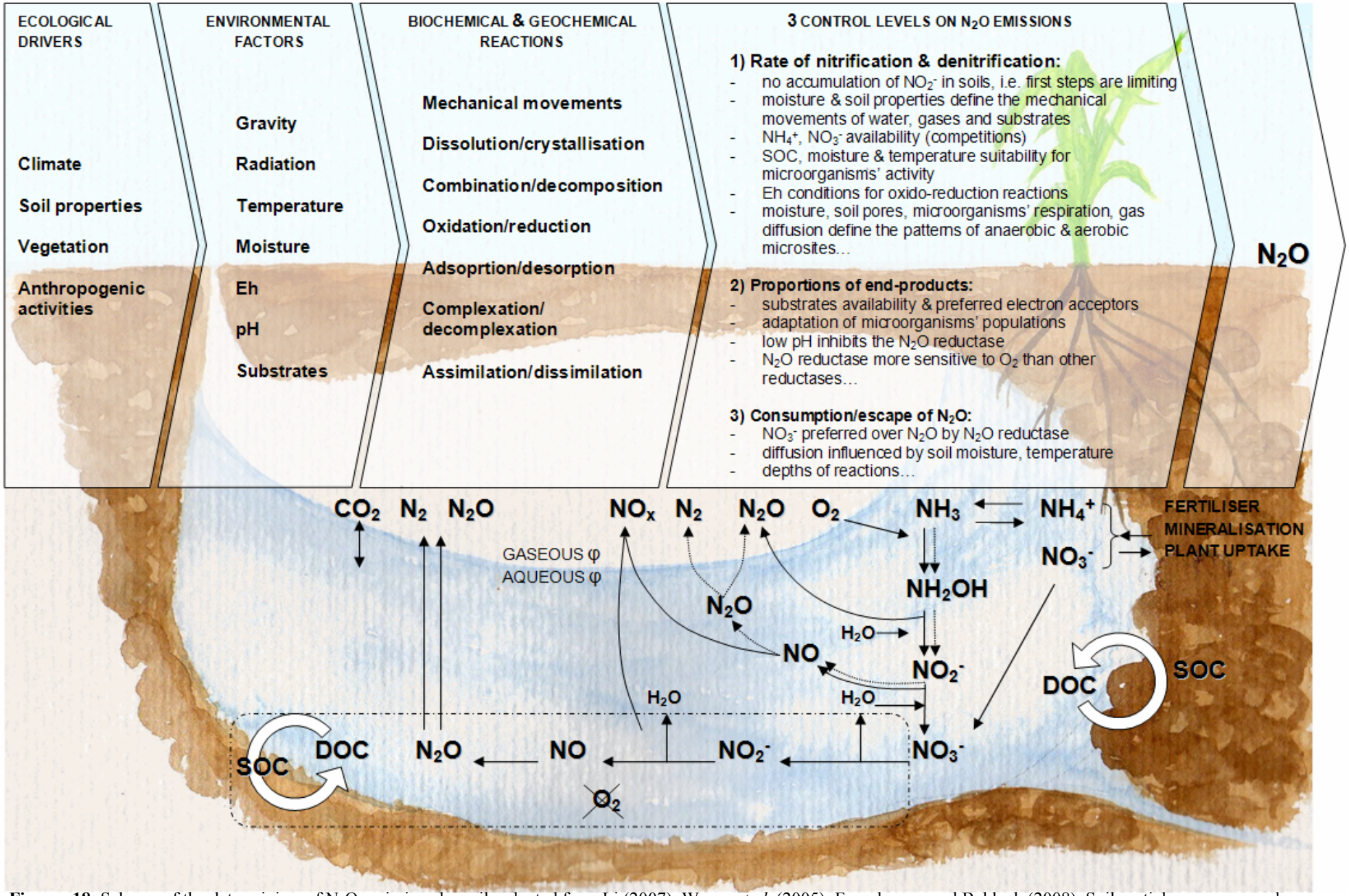



**Figure 17:** A simplified flow of fertilizer N through the environment (Duxbury and Mosier, 1993) and $N_2O$ emissions with ranges into brackets ($N_2O$-Nkg) estimated with the IPCC Tier 1 method (IPCC, 2006) considering some US average data on the diverse fractions as given by Duxbury and Mosier (1993).




**Figures 18**: 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).

**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 or volatilization for instance, models can provide insight into the amounts of Nr that might be emitted directly or indirectly 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 developed more recently 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. depends 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 *et al.*, 2001), or deduced from $CO_2$ measurements that traduce the microorganisms' activity.

There is no consensus on the diverse reduction functions that are empirical and 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 a particular attention in determining the parameters of the saturation function, for which the model is the most sensitive (Heinen, 2006a). 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 optimization could not result in perfect prediction at 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 parameters 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

**Equation 2**

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

$D_a$ 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 a10°C increase in T. (Heinen, 2006b)

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 coefficient of variation; the greatest uncertainty was found for $K$ (Heinen, 2006b).

The NOE model (Hénault *et al.*, 2005) uses similar equations to (Eq. 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) also is 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.*, 2006b).

Another comparison of four models simulating N cycling, CENTURY-NGAS (Del Grosso *et al.*, 2001), DNDC (Li *et al.*, 1992, 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 better performances 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.*, 2006b).

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 parameterize 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 been even 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 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 following substrates consumption that ends up with a change of 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 with still some significant uncertainty which is at least partly due to the still limited knowledge about ecosystem processes involved 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 the Bouwman's emission factor

(1.25%±1) used in the IPCC Tier 1 (1996). Improvement of the model would be necessary to use easy available data such as $NO_3^-$ as response variable and test mitigation scenarios, without having to measure $N_2O$ to validate the model (Beheydt *et al.*, 2007) Moreover, there are still recognized factors for modelling $N_2O$ emissions that are not adequately understood (Farquharson and Baldock, 2008).

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

*$CO_2$ 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 hence comes from dead plant parts (leaves, roots, etc.), plant rhizodeposition, and organic matter applications (animal waste, etc.). But, its storage is not definitive because dead organic matter undergoes a series of biogeochemical transformations, including decomposition, and is eventually mineralized 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 mineralization, 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 nutrients availability), and organic matter returns to the soils (e.g. crop residues 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).

**Impact of land use change on soil organic carbon**

At the global scale, the SOC pool is about 1,550 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 5,000 GtC and 38,000 GtC, respectively (Lal, 2004a). Over the last 200 years or so, soils may have lost between 55 to 78 GtC because of land use conversion and soil cultivation. On a global scale, the SOC pool to 1-meter 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-meter 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 characterized 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 mineralization 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 contribute to the aggregates 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 stabilization by physical protection and root litter is also chemically more stable (Soussana *et al.*, 2004). In particular, grasses whose roots reach deep part of 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 above-mentioned. Arrouays *et al.* (2002) provided an estimation of mean carbon changes due to land use change in France (Fig. 19). 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 $-0.95 \pm 0.3$ 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 soil carbon storage of $0.45 \pm 0.25$ tC $ha^{-1}$ for forest and $0.5 \pm 0.25$ 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 in average by 42% after conversion of forest to crop and 59% after conversion of grassland to crop.

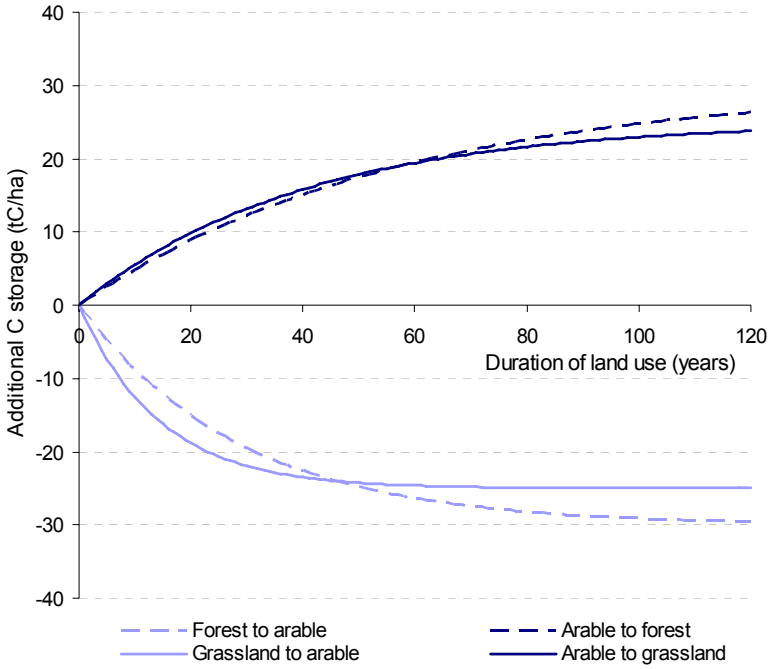



**Figure 19:** Change in soil carbon stocks associated with land use change. These are modal values for mainland France. The 95% confidence interval of these values is about ± 40%. (Arrouays *et al.*, 2002)

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 belowground biomass (rhizomes and roots) were also measured for *Miscanthus*. Belowground biomass ranged for example from 15 to 25 tDM $ha^{-1}$ with 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 at late winter or early spring with higher dry matter content. This practice causes pre-harvest losses, mainly by leaves senescence. Mean pre-harvest losses during 3 years of *Miscanthus* growing 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. 20).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 measurements 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. In addition, there is no clear difference between perennial energy crops (short rotation coppice, *Miscanthus*, switchgrass).

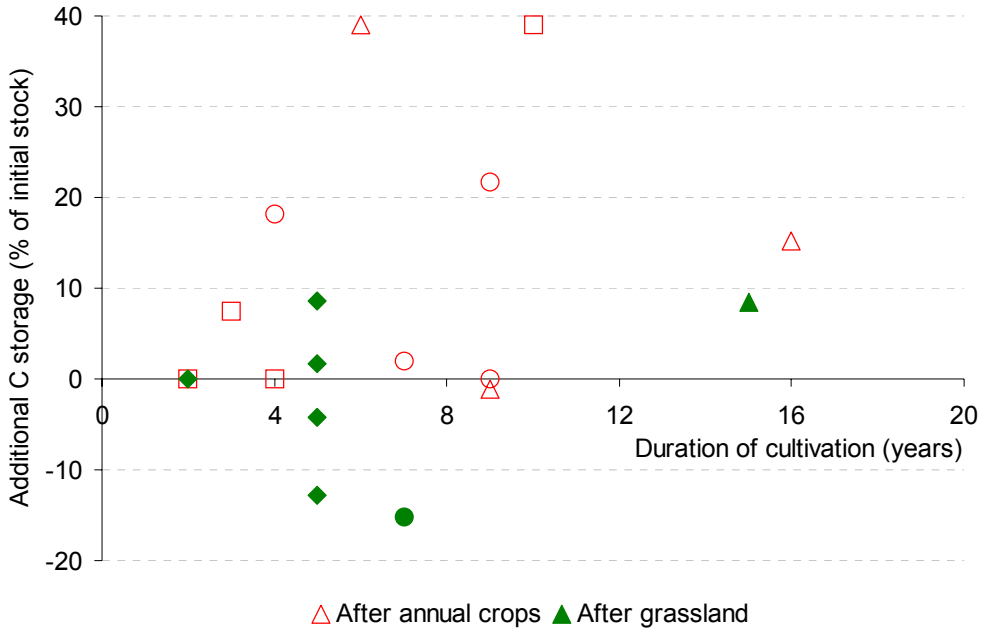



**Figure 20:** Observed changes in soil carbon stocks associated with the introduction of perennial energy crops (SRC poplar or willow: circles, *Miscanthus*: triangles and switchgrass: squares/diamonds) after annual crops or grassland. Drawn from data *In* Garten and Wullschleger, 1999; Jug *et al.*, 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).

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 aboveground and belowground 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 dept" that should be included in LCA when biofuels are introduced after land clearing. The additional carbon sequestration by perennial energy crops comparing to annual crops should be also 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).

**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 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, comparing to tillage (see review by Arrouays *et al.*, 2002; Germon *et al.*, 2007). The mean increase in carbon sequestration due to no tillage or reduced tillage was estimated at $0.2 \pm 0.13$ 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; other 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 degree of "conservation tillage", such

as direct sowing or non-inversing ploughing. Conservation tillage also encompasses intermediary techniques referred to as "reduced tillage" etc.

Due to the complexity of soil organic matter (SOM) dynamics (Fig. 21), 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).

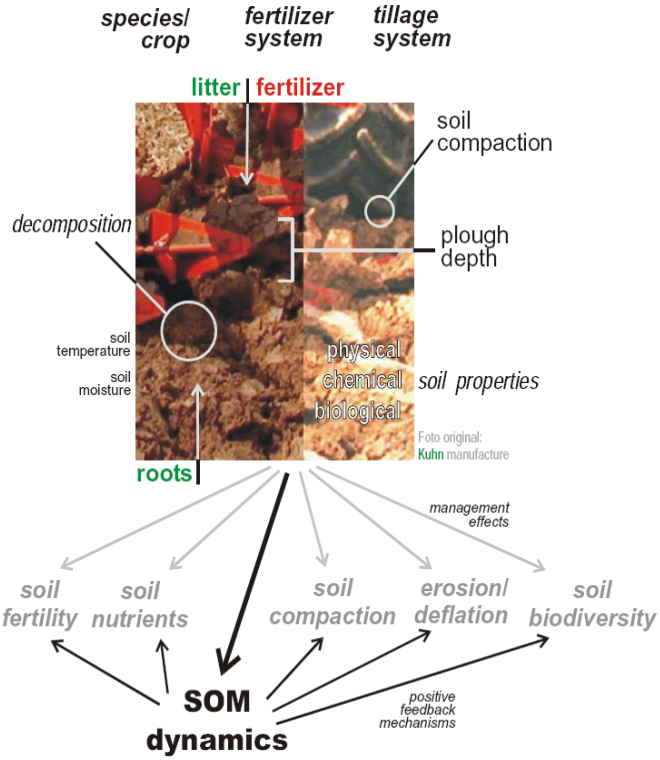



**Figure 21**: Soil organic matter is the key indicator in sustainable soil management.
(according to Baritz *In* Van-Camp *et al.*, 2004)

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 $0.15 \pm 0.08$ 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), mean difference in SOC between no catch crop and catch crop treatments was 1 tC $ha^{-1}$ after 10 years[33], 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 residues 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 year). The measured SOC increase due to straw return (as compared to 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 to 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 $0.9 \pm 0.3$ GtC $yr^{-1}$ over 50 years (Lal, 2004a)

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

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 than enhance SOC sequestration. A better understanding of the SOC stabilization in deeper soil layers could also open new options in order to increase C sequestration.

**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 to many stakeholders. Lack of consensus comes at least partly from the failure to recognise the value judgments 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 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 characterized 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 to define 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 deal 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 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 *et al.* (2001) also gave a list of indicators for the assessment of impacts on biodiversity. Impact on biodiversity for instance are currently considered in LCA through damages on biotic environment, or concurrence of species (Jolliet O. *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[34] 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 from 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 ideal indicator for evaluation (Anton *et al.*, 2007). Comparing indicators from Köllner (2001) and Weidema and Lindeijer (2001), it appeared that further research is still needed to refine them so that they could deal with ecosystems and geographical areas that are more specific (*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 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 conclusion. In this sense, Rabl *et al.* (2007) recommended that emission and removal of $CO_2$ to be counted explicitly at each stage of the life cycle. By 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 frame of an implementation of sustainability criteria for biofuel chains at an international level, including the accounting of a carbon-debt or credit in 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, 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[35]. 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 that dynamics last long 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 bias 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 necessary for biofuel LCA. Emphasis should be put on defining a harmonized 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

[34] WRI: Guide to the world resources 2000-2001, people and ecosystems: the fraying web of life, Elsevier, New York. 2002

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

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 introduce within LCA quantified impacts of land use change on albedo, surface energy balance, water cycle and their consequences on climate change. Indeed, agriculture significant affects climate through greenhouse gas emission and absorption, and modifications of surface properties, which act directly at 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 CO2eq relative to fossil fuels. Expected reduction for future commercialized 2nd generation biofuels are in the range of 70 to 90% of CO2eq 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 4 presents some of the main published studies to put in contrasts varying results due to changes in co-product handling, within a study and amongst studies. This overview completes data in Figures 12 & 13, and Table 2.

A sensitive analysis on 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, respectively, by using Bouwman's IPCC (1995 Guidelines) emission factors instead of those from Skiba (1996). The resulting greenhouse gas savings fall down for instance to -55.5% and -66% for the rapeseed pure oil and methyl ester, respectively. Although the 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 UK's soils; extrapolation from any linear model imply 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. 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 N2O emissions at a national level, while emissions are too variable at such a scale to help distinguishing between biofuel chains and co-product options at the local level. Nevertheless, biofuel chains with valuing co-product make it possible to save much greenhouse gases, especially if biomass production is optimized to reduce field emissions as much as possible.

**Table 4:** Biofuel chains and greenhouse gas savings. Ethanol is compared to gasoline, biodiesel to fossil diesel.
The results are given as they were published and without any harmonization in background assumptions, except for units, e.g. emissions for the fossil fuels per MJ vary across studies.

| **Biofuels** | **Regions** | **Feedstock** | **Co-products handling** | | | **GHG/MJ compared to 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 (fuel) | | -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.*, 2005 |
| **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-products handling | | | GHG/MJ compared to fossil fuels | References |
|---|---|---|---|---|---|---|---|
| | | | Mass allocation (% applied to biofuels) | System expansion (substitutes) | Energy allocation (% applied to biofuels) | | |
| **Biodiesel** | EU | Rapeseed RME | | Glycerin to chemicals (propylene glycol) + meal to animal feed (soya meal from imported soybeans) | | -45.5% | JRC/EUCAR/ CONCAWE, 2008 |
| | | | | Glycerin and meal to animal feed (soya meal from imported soybeans) | | -38.8% | |
| | | | | Glycerin and cake to biogas (fuel) | | -63% | |
| | | Sunflower SME | | Glycerin to chemicals (propylene glycol) + meal to animal feed (soya meal from imported soybeans) | | -66% | |
| | | | | Glycerin and meal to animal feed (soya meal from imported soybeans) | | -59% | |
| | | | | Glycerin 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 fertilizer

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

** Horizon 2005 + Skiba's $N_2O$ emission factors (%kg N-fertilizer $ha^{-1}$) 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 $CO_2$.

*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 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; almost two thirds of this increase coming from Africa, Latin America, and China, half of it due to the use of fertilizer on agricultural soils (Stern, 2006). World demand for nitrogen fertilizer 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 fertilizers and animal manure production (FAO, 2002).

Drastic savings in agricultural greenhouse gas emissions are needed, and agricultural practices are the key to reduce significantly 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[36] including all gases for the two first categories by 2030 is estimated to some 5.5-6 $GtCO_{2eq}$ $yr^{-1}$ [(37)] mainly through reduction of $CO_2$ emissions[38] (89%). The economic potential would vary between 1.5 and 4.3 GtCO2eq $yr^{-1}$ at carbon prices between 20 and 100 US$.$tCO_{2eq}^{-1}$; respectively. At the same carbon price levels, some more 0.6-16 GtCO2eq $yr^{-1}$ could be avoided, by substituting fossil fuels by bioenergy 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 $CO_2$ reduction benefit 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 moreover necessarily contribute to the two first above-mentioned categories. Table 5 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 fertilization efficiency can help reducing $N_2O$ emissions as detailed below. Others practices could further reduce the total agricultural greenhouse gases, but the trade-off between the different gases is still unclear.

[36] 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 potential are followed by the mention *medium agreement, low evidence.*

[37] About 20% of 1990s global greenhouse gas emissions, or 5%, 9%, 14% for the 3 different economic potentials.

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

**Table 5:** 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. fertilization) | + | | ± | ** | * |
| | nutrient management | + | | ± | ** | ** |
| | fire management | + | | ± | * | * |
| | species introduction (including legumes) | + | | ± | * | ** |
| management of organic soils | avoid drainage of wetlands | + | − | ± | ** | ** |
| restoration of degraded lands | 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).

**Improving fertilization efficiency**

Direct field emissions can be reduced by improving fertilization 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 correlated risk of polluting soil, water and air, whereas 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 fertilization inputs are much lower. Nitrogen balance is in surplus in all OECD countries, whereas in Sub-Saharan Africa notably it is in deficit, like in Kenya, Mali or Ghana for instance (OECD, 2008; Roy *et al.*, 2003). In two thirds of OECD countries the nitrogen surpluses decreased between 1990s and 2000s, whereas in a few countries like Canada, New Zealand, Portugal or the USA it increased notably due to the rise in fertilizer use and livestock numbers. The higher use of fertilizer 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 or the United States. Some countries, whose surpluses diminished, still have amongst the highest surpluses. This is the case for Korea, Japan, Belgium, Denmark or the Netherlands for instance, which have nitrogen efficiency[39] rather low between 30-50%, below the averages in the OECD or 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 fertilizer input per unit of crop output, closed storage system and an optimization of the timing and spreading of manure (OECD, 2008). Indeed, nitrogen efficiency can be increased by optimizing the crop's natural ability to compete with processes whereby plant available N is lost, i.e. by better matching the N supply with crop demand [e.g. optimized split application schemes and doses, foliar application, application during stem elongation or later (Recous, 2001)] and adapt it to specific risks; e.g. avoiding nitrates application in case of leaching risk (Recous, 2001), applying fertilizer below the soil surface in case of possible volatilization, choosing to apply nitrates when nitrification is more likely to happen, or ammonium based fertilizers when it is denitrification for instance during seasonal precipitations (Mosier, 1996).

Integrated fertilization 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, fertilization 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 unfertilized plot 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 of the oxygen consumption (Velthof *et al.*, 2002). Through the crop residues C/N ratio, it is possible to influence the nitrogen mineralization-immobilization turnover[40] by microorganisms (Recous, 2001; Velthof *et al.*,

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

[40] Through mineralization, N is made available for the plants, through immobilization/organization N is consumed for the development of the microorganisms.

2002), which determines the evolvement of soil N pool, including the competition for the N substrates between plants and microorganisms, and potential $N_2O$ emissions. Narrow ratio C/N and high contents of easily mineralizable N[41] 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 also may contribute to the raise of soil organic carbon especially in the form of stabilized 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 fertilizing value than amendments' stability but this latter could play an interesting role in optimizing fertilizer application timing and crop N recovery (Gabrielle *et al.*, 2005).

Although nitrogen efficiency is not identically defined across literature, authors agree that it could and should be widely improved in future (Crutzen, 2008; OECD, 2008; Galloway *et al.*, 2008). In 2002-4, nitrogen efficiency reached 70-78% for instance in Italy or 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 or Brazilian sugar cane productions have low N efficiency; only 30% of N input ends up in sugar cane tissues (Galloway *et al.*, 2008). An increase of 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 fertilizer would lead to direct reduction of field emissions, while at the same time it would also imply reduction of upstream industrial emissions during the fertilizer production and spreading in the fields. It could lead to a decrease of industrial Nr creation by 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).

**Other cultural practices**

Recommended management practices (RMPs) aim to improve the agroecosystems 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 no-tillage. In

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

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 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 frequence 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 cumulative effect of a tillage system on 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 seedbed 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 to conventional tillage systems was reversed after 20 years in humid climates and fluxes became similar between tillage systems in the dry climates (Six et al., 2004).

Finally, no tillage could lead to other "hidden costs" due to possible increasing use of herbicides and pesticides (Lal, 2004b) and decreasing yields (Chatskikh and Olesen, 2007). Indeed, soil tillage 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 oppose, e.g. aerobic or anaerobic conditions that enhance $CO_2$ or $N_2O$ productions, 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 to occur could lead to compensating $N_2O$ 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, best options to reduce the agricultural greenhouse gases are to improve its overall efficiency and to reduce all inputs, especially fertilizers. Perennials are therefore particularly interesting.

### 6. The quantitative potential of biofuels

Appendix 1 p. 157

### 7. Conclusion

Although, bioenergy and biofuels in particular have been recently 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 technologies evolution is non linear[42]. 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 technologies 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 optimized in terms of both environmental and economic performances, and combined with changes in the automotive sector towards: lighter cars, hybrids, flex-fuel, and city-vehicles. Taking the 13.5% contribution of transports 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. 32).

- The sooner second-generation biofuels will be commercially available, the more likely the 10% target will be met. Although they out-perform first-generation biofuels, the latter will develop further and serve as 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, an important 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

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

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 under the urge for reducing industrial emissions, biofuel conversion facilities will keep improving in efficiency and gas savings or storage. Agricultural practices may also contribute to enhance 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 feedback loop. Finally, the overall interest of biofuels in terms of $CO_{2eq}$ savings rely on the overall performances of agro-ecosystems, which in most cases and for all agricultural productions can be largely improved.

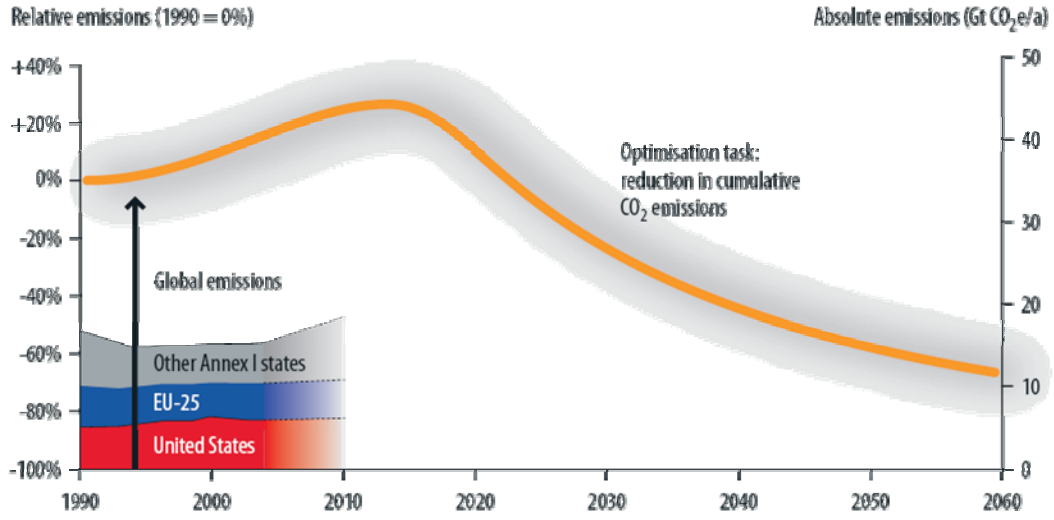



**Figure 32:** Necessary reduction in greenhouse gas emissions (including from deforestation and other land use) to stay within the 2°C global warming target from today's perspective. (Fischedick *et al.*, 2007)

- 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 on 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 population to prevent illegal logging and effectively urge forest preservation and investment in productive and environmental-friendly agro-ecosystems. 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[43].

- Not all bioenergy chains are 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 chains mix is chosen in accordance with local

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

conditions, notably the biomass production systems and the type 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, addressing also 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, or municipal waste. The competition for the use of biomass for heat and power or biofuel should be also limited as far as possible by giving the priority to the bioenergy chain that make it possible to save the most greenhouse gases compared to the local substituted energy source.

- 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 programs 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 sugarcane 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 a maximum efficiency, minimum losses and economies of scale, while this optimum-oriented specialisation shall not prevent the unit from being flexible enough to adapt the processes to various feedstock and end-products. Most distilleries in Brazil are part of sugar mill/distillery complexes capable of switching between 60%-40% to 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 are optimized to produce 66% sugar and 33% ethanol on a year-run basis, with very little flexibility[44]. The success of the mill/distillery complexes also relies on a well 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 tackle 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 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

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

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 to 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. 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. 33 & 34, Eurobarometer, 2007).

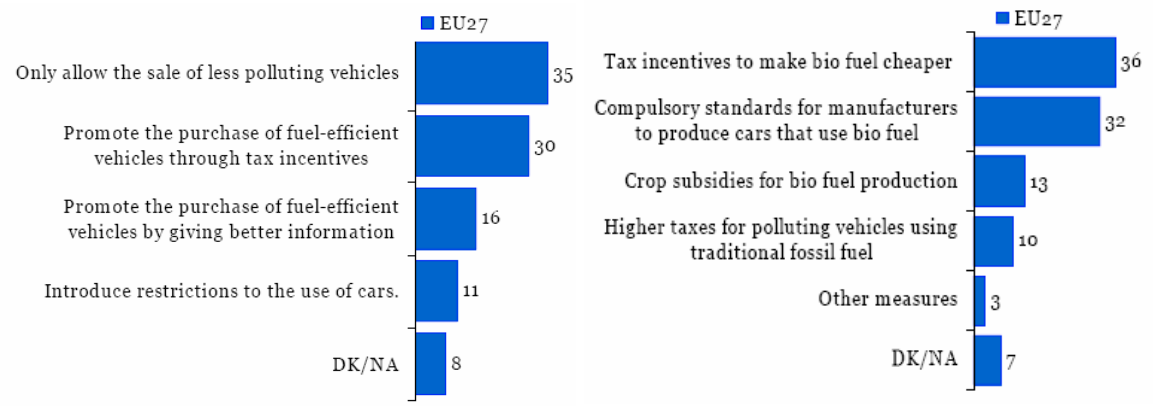



**Figure 33 & 34**: 33-left: The best way to reverse the rise of $CO_2$ emissions from road transport in response to question Q5: Road transport generates about one fifth of the European Union's harmful emissions. Between 1990 and 2004, CO2 emissions from road transport rose by 26%. Which is the best way to reverse this trend?
34-right: 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)

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, 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 that awareness on 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).

- Harmonization at an international level is crucial in order to ensure the overall complementarity of bioenergy chains, to provide a coherent frame for the markets and to control the sustainability of the systems. In particular, the framework for international biofuel

trades 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 the 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 to 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 programs 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 which would be the comparative impacts on the 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 productions, and also the development of perennials plantations, may remain largely hampered as long as the lack of land property rights does not make it possible to empower the farmers. Biofuel programs 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).

As discussed in Chapter 1, the great uncertainty in estimating agricultural $N_2O$ emissions lies on the complexity of the field emission determinism and the subsequent difficulties in quantifying and modelling the processes. Our objective, underlying the work presented in Chapter 2, was to contribute to better understand this determinism and to enhance model performances in predicting these emissions. A continuous and intensive monitoring of fluxes and main drivers was the adopted strategy in order to gather most of the information that could help explaining the observed emission patterns along two crop cycles.

Soil compaction is due to mechanical operations. Its intensity, notably assessed as the proportion of compacted zones and clods within the ploughed layer, depends on the type of cultural operations (frequency, total load and tyre inflation pressure), the soil structure and water content at the time of wheeling (Richard et al., 1999). No-tillage, as well as reduced tillage, notably implies to avoid ploughing, hence leaving compacted zones and residues near the surface. As described in Chapter 1, no-tillage can lead to increased $N_2O$ fluxes because of the reduced porosity of compacted zones that increase water conservation thereby increasing the likelihood of anaerobic conditions in soils (Gregorich et al., 2006; Germon et al, 2007; Ball et al., 2008). In a long-term experiment on cumulative effects of cropping systems on the soil structure, Boizard *et al.* (2002) have showed that there was no irreversible impact of the cropping system on the proportion of compacted zones in the ploughed layer. The proportion of compacted zones is a complex resultant of their creation, fragmentation and displacing by the diverse tillage operations, so that compacted zones could appear and disappear within a few years. Furthermore, dry-wet or freezing-thawing cycles can create cracks and enhance the sensitivity of compacted zones to fragmentation during tillage (Boizard *et al.*, 2002). Impacts of these weather cycles may also explain how a difference in $N_2O$ fluxes between no-tillage and conventional tillage could change over time. In the review by Six *et al.* (2004), the higher $N_2O$ fluxes trend in no-tillage systems compared to conventional tillage systems was reversed after 20 years in humid climates and fluxes became similar between tillage systems in the dry climates. The long-term reversibility of structure degradation depends on the texture and the swelling-shrinkage potential of the soil (Boizard *et al.*, 2002).

Modelling the impact of a cropping system on $N_2O$ emissions via changes in soil structure hence necessitates 1) to characterize and model the impact of the soil structure on the $N_2O$ emission drivers at the crop-cycle scale; 2) to use a mechanical model (Roger-Estrade *et al.,* 2004) to quantify the proportion of compacted zones at the cropping-system scale, considering the initial structural state (notably depending on the to the harvest of preceding crop), the type of operation and the weather conditions at the time of wheeling; and 3) to assess how the type of soil and climate may influence the long-term changes in soil structure and $N_2O$ emissions by the cropping-system. Within the frame of this thesis, we focused on this first point by studying the impact of soil compaction on $N_2O$ emission in sugar beet plots. As soil porosity and the water regime are primary control factors, we focused specifically on quantifying and modelling how they are affected by homogeneous soil compaction, whose intensity and remanence will in fact vary according to the different tillage modalities.

# Chapter 2

# Quantifying and modelling soil compaction impacts on $N_2O$ emissions in sugar beet fields

**Regular article** (submitted to *European Journal of Soil Science*)

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## – Abstract –

While $N_2O$ emissions contribute the lion's share of agricultural greenhouse gases, our understanding of what is actually happening in the field remains incomplete. This is particularly true for what concerns the multiple interactions between agricultural practices and $N_2O$ emissions through the alteration of the soil environment. Compaction induces major changes in the soil structure and key $N_2O$ control variables. Our objective was to characterize and better understand how it affects $N_2O$ emission kinetics by combining field experiments and modelling. We used automatic chambers to monitor $N_2O$ and $CO_2$ emissions on both an uncompacted and a compacted area in two sugar beet fields in northern France during two years. Soil compaction led to lower $CO_2$ emissions and higher $N_2O$ emissions by inducing favourable anoxic conditions for denitrification. Cumulative $N_2O$ emissions averaged 944-977 g N-$N_2O$ $ha^{-1}$ in uncompacted plots and 1 448-1 382 g N-$N_2O$ $ha^{-1}$ in compacted plots in 2007-2008, respectively. They did not differ significantly between years despite the different N fertilisations (100 and 150 kg N $ha^{-1}$). We ran the NOE model (Hénault et al., 2005) to simulate $N_2O$ emissions from nitrification and denitrification. It predicted 106-138 g N-$N_2O$ $ha^{-1}$ in uncompacted plots and 1 550-650 g N-$N_2O$ $ha^{-1}$ in compacted plots in 2007-2008 respectively, markedly underestimating the nitrification rates and associated $N_2O$ emissions. We modified the model based on published results to better simulate nitrification and take into account variations of $N_2O$ fractions of total end-products versus soil water-filled pore space and nitrate concentration. The modified model (NOE2) better predicted nitrification rates and $N_2O$ emission peaks following fertilization. Using a fine soil-layer discretization (1 cm) in soil water content and fertilizer depth also improved the simulations. NOE2 predicted 428-416 g N-$N_2O$ $ha^{-1}$ in uncompacted plots and 1 559-1 032 g N-$N_2O$ $ha^{-1}$ in compacted plots in 2007-2008, respectively. NOE2 underestimation by 56% in the uncompacted plots was notably related to the difficulty in simulating peak emissions during summer.

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## 1. Introduction

Arable soils are a major source of anthropogenic $N_2O$, via the microbial processes of denitrification and nitrification (Smith *et al.*, 2007). Nitrous oxide is an intermediate in the reaction sequence of denitrification, which is triggered by anaerobic conditions, and a by-product of nitrification, which takes place under aerobic conditions. These processes occur naturally in soils where the microorganisms involved are abundant. They may however be affected by agricultural practices that notably increase the concentrations of nitrogen substrates through fertilization and crop residue return, or induce changes in soil structure and water regime through which both the aerobic status of the soil and gas diffusion are affected. Both of these factors influence the production of $N_2O$ and the ratio of $N_2O$ emitted over the total nitrification and denitrification end-products, i.e. nitrate and $N_2$.

Characterizing and quantifying these influences is complex because of the high spatial and temporal variability of emissions at the plot scale (Farquharson and Baldock, 2008) due to diverse biogeochemical processes, numerous influencing factors and the heterogeneity of "primary ecological drivers" (Li *et al.*, 2000). Moreover, change in one factor affects different processes in interaction. Predicting the resulting global behaviour is all the more difficult since relationships between primary drivers and subsequent final impacts are not linear (Bouwman, 1996; Conrad, 1996; Li, 2007) and changes in these drivers due to agricultural practices are poorly characterized.

Process-based models make it possible to encompass most of the involved factors and can serve as tools for emission analysis or predictions. They have been available for a dozen years and more recently developed for different ecosystems and N species (Sutton *et al.*, 2007). Most simplified models of denitrification are commonly based on a denitrification potential weighted by a product of reduction functions linked to soil nitrate content, soil saturation, soil temperature, and soil pH. There is no consensus, however, on the diverse reduction functions, which are empirical and calibrated generally from site-specific studies (Heinen, 2006).

Models have been tested on their ability to reflect the order of magnitude of main $N_2O$ peaks rather than on their capacity to reproduce correct emission kinetics and the effect of changes in agricultural practices. Improved systematic measurement, better monitoring of drivers (Mosier *et al.*, 1996; Laville *et al.*, 1997; Conen *et al.*, 2000; FAO, 2001; Beheydt *et al.*, 2007; Pattey *et al.*, 2007) and enhanced models are all necessary to better characterize the effects of agricultural practices on $N_2O$ emissions.

As soil porosity and the water regime are primary control factors for $N_2O$ emission, we focused specifically on quantifying and modelling how they are affected by soil compaction resulting from different tillage modalities. During two sugar beet cycles, we continuously monitored $N_2O$ and $CO_2$ emissions as well as soil temperature and water content, and analysed soil mineral N content on several dates. This continuous monitoring is essential to characterize the temporal patterns of emissions, i.e. capture the background emission rate as well as emission peaks. We evaluated a simple model (NOE) and proposed improvements toward a more process-based approach, including a better representation of the vertical heterogeneity.

## 2. Field experiments

### 2.1 Experimental site and treatments

The study site was located at Estrées-Mons (Northern France, 49°80'N, 3°60'E) where mean annual precipitation and temperature are 667 mm and 9.6°C (Guérif *et al.*, 1996). The soil is an Orthic Luvisol (FAO *et al.*, 1998) containing 19% clay, 74% silt, 5% sand, 9.8 g $kg^{-1}$ organic carbon and with a pH of 7.8. Climatic data were collected from the Agroclim weather station[45]. Two sugar beet fields (F1 and F2) were studied in 2007 and 2008. The preceding crops were winter wheat (F1, 2006) and winter barley (F2, 2007). The straw was incorporated through mouldboard ploughing in mid winter. In both years, two treatments were considered: "uncompacted" (conventional technique) and "compacted". The compacted area (about 200 $m^2$) was obtained by running a loaded tractor in early March under wet conditions (soil potential ≈ -300 kPa). In April, the fields were fertilized with UAN (50% Urea, 50% Ammonium Nitrate) and immediately sown with sugar beet. The F1 field received 100 kg N $ha^{-1}$ on April 12, 2007 and F2 field 150 kg N $ha^{-1}$ on April 14, 2008. At harvest, plants and soils were sampled from 3 sub-plots for each treatment. Harvest took place on October 5, 2007 (F1) and November 4, 2008 (F2).

### 2.2. $CO_2$ and $N_2O$ flux monitoring

During the two crop cycles, $N_2O$ and $CO_2$ fluxes were continuously measured in automatic chambers (0.49 $m^2$, 0.25 m high). During the closing period (20 minutes), the air was circulated through the chambers and passed sequentially through a $CO_2$ analyser (LiCor 820) and a $N_2O$ analyser (Thermo 46C) with sensitivity levels of about 0.5 ppm and 2 ppb, respectively. This system made it possible to measure fluxes with a high frequency (4 times per day, i.e. every 6 hours) and throughout the growth cycle of the sugar beet (8-9 months per year). The $CO_2$ or $N_2O$ flux was calculated by fitting the kinetics of gas concentration to a linear or exponential model. Replicate chambers (two in 2007, three in 2008) were used in each treatment and placed transversally to the direction of field operations to gather most of the heterogeneity in the soil structure and crop residue distribution induced by tillage and harvest operations (Laville *et al.*, 1999; Oorts *et al.*, 2007). In 2007, soil profile observations were made just before harvesting to characterize the soil structure for each treatment.

Automatic chambers allowed us to measure fluxes with a high frequency and over long periods. The temporal integration of $N_2O$ fluxes is indeed a key point while most studies have (until now) considered punctual measurements. In fact, the integration between long time interval measurements may lead to important errors. For example, measuring once a week during one hour samples only 1/168 = 0.6% of the signal. Parkin (2008) indicates that "measurements at 3 weeks interval yielded $N_2O$ estimates between +60% and -40% of the actual cumulative $N_2O$ flux". Therefore concluding on statistical differences in cumulative $N_2O$ fluxes from punctual measurements only, even with an optimal spatial design, is not correct, whereas this may be a common practice. The originality of our method consisted in quasi-continuous measurements, allowing to integrate fluxes over time with little error and to test models for the ability to reproduce emission kinetics. The drawback of this method is the limited range of distance between chambers due to constraints in electricity supply, number of analysers, tubing lengths, etc. Consequently, even simply interspersed treatments are difficult to put in place with

[45] https://intranet.inra.fr/climatik

this technique of measurement. The consequence of a pseudo replicated statistical design (without treatment interspersion) depends on the spatial structure of the studied variable. It is of great importance if the variability at short distance is much lower than the variability at long distance. It is not so critical if the two variances are comparable; indeed this is the case for $N_2O$ emissions in arable soils. Jackson et al. (2007) found that "most of the variance in $N_2O$ fluxes was at the sub sample level". This observation was confirmed by Röver et al. (1999) and Van den Heuvel et al. (2009). The studies that established semi-variograms found a weak spatial dependency of N2O fluxes, i.e. a high ratio between nugget variance and sill variance (Clemens *et al.*, 1999; Yanai *et al.*, 2003; Yates *et al.*, 2006; Konda *et al.*, 2008; Nishina *et al.*, 2009). In our study, the distance between replicated chambers was about 3 m and the distance between treatments ranged from 10 to 20 m. According to the previous studies, the variances at these distances should have been rather close.

Starting from June, the upper leaves of sugar beet were regularly cut back in order to ensure that the chambers could still hermetically close. Despite some short interruptions, we managed to measure the gas fluxes throughout both crop cycles, giving 143 complete daily records out of 156 days in 2007, and 202 complete daily records out of 204 days in 2008. Flux measurement accuracy was less than 0.3 kg $CO_2$-C $ha^{-1}$ $d^{-1}$ and 1.0 g $N_2O$-N $ha^{-1}$ $d^{-1}$.

### 2.3. Soil measurements

Soil temperature and soil water content were recorded hourly in both treatments at two depths, -10 and -20 cm (2 replicates) using thermocouples and time-domain reflectrometry (TDR) probes, respectively. Bulk density and gravimetric water content were measured on undisturbed soil cores using steel cylinders (5 cm-diameter, 6 replicates) at sowing and harvest in 2007, and at sowing and during summer in 2008. The measured volumetric water content was used to calibrate TDR measurements.

Soil mineral N contents (SMN, i.e. $NH_4^+$ and $NO_3^-$) were regularly measured at different depths (6 replicates, each replicate being a mix of 3 soil cores). Initial SMN was measured in the 0-10, 10-20 and 20-30 cm soil layers. In 2007, measurements were carried out at days 5, 7, 14 and 21 after fertilization over only 0-10 cm because there was no rain. SMN was measured at day 33 (just after the first rain event) and day 62 over 0-10 and 10-20 cm. In 2008, due to wetter conditions, SMN was determined over 0-10 and 10-20 cm at days 3, 10, 16, 23 and 38 after fertilization, and once a month from June to September at 0-10, 10-20 and 20-30 cm. At sugar beet harvest, SMN content was measured every 30 cm down to a depth of 120 cm. Mineral N was extracted by shaking 100 g soil with 300 mL KCl M. Nitrate and ammonium in the soil extracts were analysed with a continuous flow colorimeter (Bran and Luebbe, Germany).

## 3. Modelling

### 3.1. Original NOE model

NOE is an empirical model developed by Hénault *et al.* (2005) that combines the NEMIS denitrification sub-model (Hénault and Germon, 2000) and a nitrification sub-model (Garrido *et al.*, 2002). Both processes are modelled as product functions and depend on the availability of mineral N, soil water content and temperature (Eq. 1). The denitrification potential ($D_p$) represents the site specific soil microorganisms' capacity to reduce nitrate under standard (20°C, water saturation) and non-limiting nitrate conditions (200 mg N $kg^{-1}$). $N_2O$ emissions are assumed to be proportional to the amounts of N denitrified and nitrified through the coefficients $r_{mx}$ and $z$ respectively.

$$N_2O = r_{mx}\left(D_P \cdot F_N \cdot F_W \cdot F_T\right) \quad \text{if } WFPS > 0.80 \quad (1a)$$

$$N_2O = z\left(f_A \cdot f_W \cdot f_T\right) \quad \text{if } WFPS < 0.62 \quad (1b)$$

$$N_2O = r_{mx}\left[\left(D_P \cdot F_N \cdot F_W \cdot F_T\right) + z \cdot \left(f_W \cdot f_A \cdot f_T\right)\right] \quad \text{if } 0.62 \leq WFPS \leq 0.80 \quad (1c)$$

Equations (1a), (1b) and (1c) yield the $N_2O$ emissions through denitrification alone, through nitrification alone and through both processes combined, respectively. The detailed function is given in Table 1. The symbols are similar to the original reference, except that the reduction functions for nitrification are designated by "f" instead of "N", in order to distinguish them from nitrate concentration.

We estimated $r_{mx}$ and $z$ using laboratory data from previous studies carried out by Khalil *et al.* (2004, 2005) on soil cores from the same site (Table 2). The $D_p$ value was set at the maximum denitrification rate previously observed on this site (unpublished results). Its value (0.3 mg $kg^{-1}$ $day^{-1}$) was in the lower range of published values (Myrold and Tiedje, 1985, Parkin 1987; Hénault *et al.*, 2005), but remained consistent with the low organic carbon and clay contents of the soil (Hénault, 1993). N losses from volatilization, which occur during a short period after fertilization, were estimated using the Volt'Air model (Génermont and Cellier, 1997; Le Cadre, 2004). NOE was run for each year and treatment using daily precipitations, mean daily soil temperature and water contents over 0-20 cm and substrate ($NH_4^+$ or $NO_3^-$) contents (interpolation in time was used when necessary for nitrate content).

### 3.2. NOE2 development and parameter estimation

The NOE model was modified in order to: i) achieve homogeneity of the equations; ii) describe some missing processes; iii) modify the reduction functions according to our knowledge of the underlying processes; and iv) introduce a vertical discretization of soil to better describe soil water and substrate distributions. These modifications led to a NOE2 model and are detailed below for the nitrification and denitrification sub-models (Table 1).

For nitrification, we introduced a nitrification potential *Np* and replaced the water content and temperature reduction functions by the dimensionless ones used in the soil-crop model STICS (Brisson *et al.*, 1998) (Eq. 15, 18). We removed the *WFPS* threshold above which nitrification was considered to cease (0.80), since this results is seldom observed in the field. Mineralization was absent from NOE whereas it is often the limiting factor for nitrification when the ammonium pool is small, i.e. far from fertilizer application periods. We thus introduced a mineralization rate function *Vm* based on a potential mineralization rate *Mp* and temperature and water content factors, which are similar to nitrification (Eq. 19). The nitrification rate (*Vn*, Eq. 21) is hence constrained by either soil environmental conditions when the substrate is available (*Vnp*, Eq. 11) or the mineralization rate when the ammonium substrate is limiting ($Vn_{max}$, Eq. 20). In the latter equation, we introduced a minimum value of exchangeable ammonium ($A_{min}$) that remains unavailable for microorganisms.

The most important modification to the nitrification model was to simulate the effect of oxygen availability on the nitrification rate ($f_{O2}$, Eq. 16) as well as on the reduction of $N_2O$ ($g_{O2}$, Eq. 23) using *WFPS* as an index of anoxia. Although the effect of oxygen availability on the

nitrification rate indirectly depends on soil water content, especially in its upper range, it is distinct and independent from the direct effect of the water content on microbial activity represented by $f_W$, which is active in the lower range of water content (Linn and Doran, 1984; Eq. 11). The functions $f_{O2}$ (Eq. 16) and $g_{O2}$ (Eq. 23) were established based on laboratory results published by Khalil *et al.* (2004). These authors showed that the nitrification rate decreases when the $O_2$ concentration decreases while the fraction of $N_2O$ emitted by nitrification increases as the probable result of nitrifier denitrification (Bollmann and Conrad, 1998; Wrage *et al.*, 2001; Mørkved *et al.*, 2006). Hence, the fraction of $N_2O$ emitted per unit of nitrified N ($z$) is no longer constant in the NOE2 model but depends on *WFPS* (Figure 1). Finally, $N_2O$ produced from nitrification was considered associated to aerobic zones in the soil connected to the atmosphere. We consequently hypothesized that $N_2O$ from nitrification escapes from the soil too rapidly to be reduced to $N_2$, a point in contrast to the NOE approach.

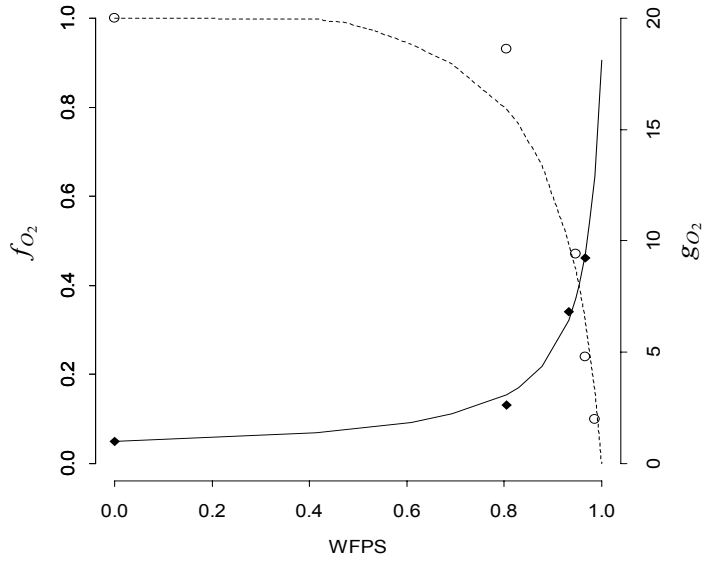



**Figure 1** Functions simulating the effect of anoxia (WFPS) on the nitrification rate ( $f_{O2}$ , dotted line) and the fraction of N2O emitted per unit of N nitrified ( $g_{O2}$ , continuous line) in model NOE2 (see Eq. 16 and 23 in Table 1). The curves were fitted to the observed values (open and filled-up points, respectively) reported by Khalil et al. (2004).

The denitrification sub-model was kept in its original form. However, many studies have shown that the fraction of $N_2O$ emitted from denitrification can vary over time since it may be influenced by: 1) factors that globally induce potential reduction of $N_2O$; 2) factors that more specifically inhibit the $N_2O$ reductase; or 3) factors that may induce transient shifts in microbial activity. Soil water and nitrate contents are important factors controlling this reduction rate (Focht, 1974; Firestone and Tiedje, 1979; Weier *et al.*, 1993; Maag and Vinther, 1996; Ruser *et al.*, 2006; Del Grosso *et al.*, 2000). We propose simple linear functions for the response of the ratio of denitrification end-products ($r$, Eq. 7) to *WFPS* ($G_{O2}$) and nitrate concentration ($G_N$) based on the results of Blackmer and Bremner (1978), Weier *et al.* (1993) and Khalil *et al.* (2005). Increasing *WFPS*, i.e. the anoxia, results in a linear decrease of $r$ (Eq. 8). Conversely, $r$ increases versus nitrate content (Eq. 9) according to two straight lines: a first steep increase up

to a value *Nc* followed by a much slower increase. The proposed model (Eq. 7-9) could explain 45% of the variance from the dataset of Weier *et al.* (1993). For comparison, we also implemented the corresponding NGAS function (DAYCENT model, Del Grosso *et al.*, 2000) which furthermore incorporates the $CO_2$ flux as an indicator of carbon availability.

The last major change to the original NOE model was to introduce a vertical discretization of soil layers of configurable but constant thickness. This modification was required in order to mimic the presence of ammonium derived from fertilizer near the soil surface and to limit volatilization at the very near surface of the soil. It also made it possible to test the impact of a finer description of temperature and water content, e.g. simulated by the HYDRUS 1D model (Šimůnek *et al.*, 2008), on nitrification, denitrification and $N_2O$ emissions.

Model parameterization was made mainly using published data. The whole set of parameter values is given in Table 2 with references.

### 3.3. Model evaluation

Evaluation of the models was conducted in three steps. Step 1: from the set of estimated parameters, NOE was run for each year and each treatment, using daily precipitations, mean daily 0-20 cm values of measured temperature and water content and daily-interpolated values of measured nitrate content. The evolution of the ammonium pool was simulated. This set of simulations was considered as our reference. Step 2: NOE2 was run using its own set of parameters, soil layers of 1 cm, and water contents and temperatures simulated from HYDRUS as inputs. HYDRUS parameters were first calibrated to ensure that measured values at 10 and 20 cm depths were correctly predicted. Step 3: a sensitivity analysis of NOE2 was carried out with respect to its main parameters, discretization in soil layers and details on water content and temperature evolution in the 20 cm depth soil layer.

As $N_2O$ emissions are subject to large spatial and temporal variability, we defined a limited set of homogeneous periods on the basis of main drivers of N emission: period I = progressive disappearance of ammonium; period II = progressive disappearance of nitrate; period III = full plant growth with almost no mineral N left; period IV = plant senescence (Fig 4). Model performance was then evaluated by comparing mean fluxes and their $10^{th}$, $50^{th}$ and $90^{th}$ percentiles for each time period rather than looking at highly variable (and uncertain) individual daily fluxes. The quality of prediction was assessed by calculating the mean deviation (MD) and root mean square error (RMSE), indicators of the bias and precision of the model, respectively.

**Table 1** Equations of NOE and NOE2 models. $N$ = nitrate concentration (mg N kg$^{-1}$); $A$ = ammonium concentration (mg N kg$^{-1}$); $W$ = gravimetric water content (g g$^{-1}$).

| Eq. | NOE | Eq. | NOE2 |
|---|---|---|---|
| Denitrification rate | | | |
| (2) | $Vd = D_P \cdot F_N \cdot F_W \cdot F_T$ | | |
| (3) | $F_N = \frac{N}{K_N + N}$ | | $F_N = \frac{N}{K_N \cdot W + N}$ |
| (4) | $F_W = 0,$ | | $WFPS < WFPS_C,$ |
| | $F_W = \left[\frac{WFPS - WFPS_C}{1 - WFPS_C}\right]^{1.74}, \quad WFPS \geq WFPS_C$ | | |
| (5) | $F_T = \exp\left[\frac{(T-11)\ln(Q_{10d1}) - 9\ln(Q_{10d2})}{10}\right], \quad T < 11\,°C$ | | |
| | $F_T = \exp\left[\frac{(T-20)\ln(Q_{10d2})}{10}\right], \quad T \geq 11\,°C$ | | |
| $N_2O$ from denitrification | | | |
| (1a) | $N_2Od = Vd \cdot r_{mx}$ | (6) | $N_2Od = Vd \cdot r$ |
| | | (7) | $r = r_0 \cdot (G_{O2} \cdot G_N)$ |
| | | (8) | $G_{O2} = 1 - c_W \cdot \max[0, (WFPS - WFPS_C)]$ |
| | | (9) | $G_N = \min[(d_{N0} \cdot N),\ (c_N + d_N \cdot N),\ 1]$ , $d_{N0} = \frac{c_N + d_N \cdot N_C}{N_C}$ |
| Nitrification rate | | | |
| (10) | $Vn = f_W \cdot f_A \cdot f_T$ | (11) | $Vnp = N_P \cdot f_T \cdot f_A \cdot \min(f_W, f_{O2})$ |
| (12) | $f_A = \frac{A}{K_A + A}$ | (13) | $f_A = \frac{\max[0,(A - A_{\min})]}{K_A \cdot W + \max[0,(A - A_{\min})]}$ |
| (14) | $f_W = a \cdot W + b$ | (15) | $f_W = \frac{W - a_W \cdot W_{FC}}{(1 - a_W) \cdot W_{FC}}; \quad 0 \leq f_W \leq 1$ |
| | | (16) | $f_{O2} = a_S\left(\frac{WFPS - 1}{WFPS - b_S}\right)$ |
| (17) | $f_T = F_T$ | (18) | $f_T = \exp\left[\frac{(T-20)\ln(Q_{10n})}{10}\right]$ |
| Mineralization rate | | | |
| | | (19) | $Vm = M_P \cdot f_T \cdot f_W$ |
| Maximum nitrification rate | | | |
| | | (20) | $Vn_{\max} = Vm + \frac{\max[0,(A - A_{\min})]}{\Delta t}$ |
| Effective nitrification rate | | | |
| | | (21) | $Vn = \min(Vnp, Vn_{\max})$ |
| $N_2O$ from nitrification | | | |
| (1b) | $N_2On = Vn \cdot z,$ $WFPS < 0.62$ | (22) | $N_2On = Vn \cdot z$, $z = z_0 \cdot g_{O2}$ |
| (1c) | $N_2On = Vn \cdot z \cdot r_{mx},$ $0.62 \leq WFPS < 0.80$ | (23) | $g_{O2} = \frac{c_S \cdot WFPS - d_S}{WFPS - d_S}$ |

**Table 2** Names, values and sources of NOE and NOE2 parameters. Reference conditions are:
* 20°C, *WFPS* = 100%; ** soil water = 0.27 g $g^{-1}$; *** 20°C, soil water = field capacity;
§ optimal aerobic conditions.

| Process | Eq. | Para-meters | NOE | Unit | NOE2 | Unit | Sources |
|---|---|---|---|---|---|---|---|
| Denitrification | | | | | | | |
| | (2) | $D_P$ | 0.327* | mg $kg^{-1}$ $d^{-1}$ | 0.327* | mg $kg^{-1}$ $d^{-1}$ | Optimized for NOE2 |
| | | | | | 0.158* | mg $kg^{-1}$ $d^{-1}$ | Optimized for NOE2+NGAS-*r* |
| | (3) | $K_N$ | 22** | mg $kg^{-1}$ | | | Hénault *et al.* (2000) |
| | | | | | 215 | mg $l^{-1}$ | Optimized for NOE2 |
| | (4) | $WFPS_C$ | 0.62 | - | 0.62 | - | Grundmann and Rolston (1987) |
| | (5) | $Q_{10d1}$ | 89 | - | 89 | - | Stanford et al. (1975) |
| | (5) | $Q_{10d2}$ | 2.1 | - | 2.1 | | Stanford et al. (1975) |
| $N_2O$ from denitrification | | | | | | | |
| | (1) | $r_{mx}$ | 0.11 | - | | | Khalil *et al.* (2005) |
| | (7) | $r_0$ | | | 0.63 | - | |
| | (8) | $c_W$ | | | 2.05 | - | Blackmer and Bremner (1978) |
| | (9) | $c_N$ | | | 0.44 | - | Weier *et al.* (1993) |
| | (9) | $d_N$ | | | 0.0015 | - | Khalil *et al.* (2005) |
| | (9) | $N_c$ | | | 3 | mg $l^{-1}$ | |
| Nitrification | | | | | | | |
| | (11) | $N_P$ | | | 27.3*** | mg $kg^{-1}$ $d^{-1}$ | Khalil *et al.* (2004) |
| | (12) | $K_A$ | 10 | mg $l^{-1}$ | | | Hénault *et al.* (2005) |
| | (13) | | | | 24 | mg $l^{-1}$ | Khalil *et al.* (2004) |
| | (13) | $A_{min}$ | | | 5 | mg $l^{-1}$ | |
| | (14) | $a$ | 0.0571 | kg N $ha^{-1}$ $d^{-1}$ | | | Garrido *et al.* (2002) Châlons |
| | (14) | $b$ | -0.37 | kg N $ha^{-1}$ $d^{-1}$ | | | Garrido *et al.* (2002) Châlons |
| | (15) | $a_W$ | | | 0.30 | g $g^{-1}$ | Brisson *et al.* (1998) |
| | (15) | $W_{FC}$ | | | 0.24 | g $g^{-1}$ | Boizard *et al.* (2002) |
| | (16) | $a_S$ | | | 1.16 | - | Khalil *et al.* (2004) |
| | (16) | $b_S$ | | | 1.09 | - | Khalil *et al.* (2004) |
| | (18) | $Q_{10n}$ | | - | 3.17 | - | Brisson *et al.* (1998) |
| Mineralization | | | | | | | |
| | (19) | $M_p$ | | | 0.30* | mg $kg^{-1}$ $d^{-1}$ | Brisson *et al.* (1998) |
| $N_2O$ from nitrification | | | | | | | |
| | (1) | $z$ | 0.0016§ | - | | | Khalil *et al.* (2004) |
| | (22) | $z_0$ | | - | 0.0016 | | Khalil *et al.* (2004) |
| | (23) | $c_S$ | | - | 0.40 | - | Khalil *et al.* (2004) |
| | (23) | $d_S$ | | - | 1.04 | - | Khalil *et al.* (2004) |

## 4. Results

### 4.1. Compaction effects on soil and crop

Soil compaction had a significant effect on bulk densities ($p<0.001$) since it varied from 1.35-1.44 g $cm^{-3}$ on uncompacted plots to 1.57-1.66 g $cm^{-3}$ on compacted plots (Table 3). Bulk density was rather homogeneous throughout the profile (0-30 cm).

Figure 2 shows the temperatures and water contents at two depths in both treatments and the rainfalls in 2007 and 2008. Soil temperatures were practically unaffected by depth or compaction. Undifferentiated temperatures below 5 cm under different kinds of soil tillage had already been observed (Richard and Cellier, 1998). Volumetric soil water content on the contrary was significantly greater in the compacted plots ($p<0.001$), and usually higher in the deeper soil layer for both treatments and both cycles. In 2007, water content varied less in the compacted plot than in the uncompacted one, especially during dry periods. During the crop cycle, the moisture content over 0-20 cm varied between 0.28 and 0.38 $m^3 m^{-3}$ in the compacted plot and from 0.15 to 0.36 $m^3 m^{-3}$ in the uncompacted plot. In 2008, it varied within the same range in both treatments, between 0.12 and 0.39 $m^3 m^{-3}$.

Sugar beet growth was affected by compaction both years ($p<0.01$), particularly in 2007 ($p<0.001$) (Table 3). Indeed, in 2007 we observed a germination delay and a lower seedling emergence in the compacted plot, probably due to the combined effect of compaction and dry weather that may have caused a coarse seedbed (Dürr *et al.,* 2001). The high proportion of forked taproots in the 2007-compacted plot indicated that the high bulk density had increased resistance to penetration and hampered root growth (Guérif *et al.,* 1996). Reduced water uptake by plants was probably a contributory cause of higher soil water content in compacted plots. The compaction effect was perhaps accentuated in 2007, since the 20-30 cm soil layer was more severely compacted and drainage was thus limited. Indeed, we observed gleyic zones in the layer, indicating anoxic conditions. Yield difference between treatments in 2007 was due to both a higher number of taproots per $m^2$ and a greater mean weight per taproot in the uncompacted treatment. In 2008, yields did not vary with treatment since taproot numbers and mean taproot weight variations between treatments compensated one another.

Compaction did not visibly affect the kinetics of fertilizer nitrification. Ammonium contents went back to the background concentration (≈1 mg N $kg^{-1}$) within 50 and 30 days after fertilization in 2007 and 2008 respectively, independently of treatment. Nitrate contents reached their maximum within the same time frame. They were lower in the compacted plots, suggesting that N losses had been greater in the compacted soil. Nitrate concentration reached its background level (<1 mg N $kg^{-1}$) towards the end of June (DOY 160) in both treatments.

### 4.2. Compaction effect on $N_2O$ and $CO_2$ fluxes

$CO_2$ fluxes measured cumulatively were higher in the uncompacted plots, 5 419-6 061 kg C $ha^{-1}$ in 2007-2008 respectively, versus 2 701-5 226 kg C $ha^{-1}$ in compacted plots (Table 4). The difference between treatments was much larger in 2007. These emissions, including both plant and soil respiration, followed a seasonal kinetics increasing with the temperature and crop growth. However, in the uncompacted plots, these fluxes reached a first local maximum around the end of April (DOY 120), indicating a strong heterotrophic microbial activity in soil.

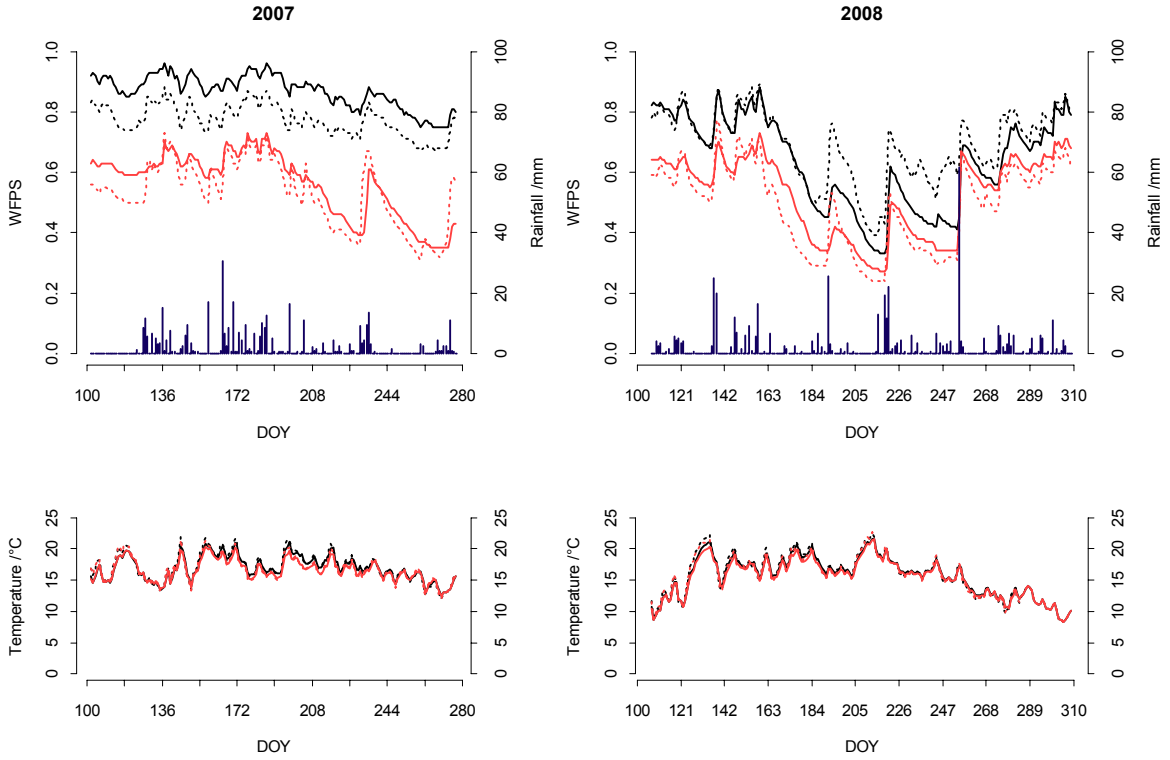



**Figure 2** Evolution of rainfall, water-filled pore space (*WFPS)* and temperature in the soil at -10 cm (full lines) and -20 cm depth (dotted lines) in the compacted soil (black line) and uncompacted soil (red line) cropped with sugar beet during 2007 and 2008.

**Table 3** Overview of field results.

| | 2007 | | 2008 | |
|---|---|---|---|---|
| | Uncompacted | Compacted | Uncompacted | Compacted |
| Crop cycle durations /days | 156 | | 204 | |
| Cumulative rainfall over the cycles /mm | 385.2 | | 466.3 | |
| Mean temperatures over the cycles /°C | 15.5 | | 14.5 | |
| Bulk densities /g $cm^{-3}$ | | | | |
| 0-10 cm | 1.37 | 1.57 | 1.35 | 1.58 |
| 10-20 cm | 1.37 | 1.61 | 1.39 | 1.59 |
| 20-30 cm | 1.44 | 1.66 | 1.39 | 1.56 |
| Yields of the experimental plots /tDM $ha^{-1}$ | 19.4 | 12.8 | 22.7 | 21.7 |

**Table 4** Summary of $N_2O$ and $CO_2$ flux measurement data.

* mean coefficient of variation (spatial variability)

** mean coefficient of variation (temporal variability)

| | 2007 | | | | 2008 | | | |
|---|---|---|---|---|---|---|---|---|
| | Uncompacted | | Compacted | | Uncompacted | | Compacted | |
| | Mean | CV/% | Mean | CV/% | Mean | CV/% | Mean | CV/% |
| Cumulative $N_2O$ flux /g N $ha^{-1}$ | 944 | 7* | 1 448 | 4* | 977 | 25* | 1 382 | 16* |
| Cumulative $CO_2$ flux /kg C $ha^{-1}$ | 5 419 | 23* | 2 701 | 14* | 6 061 | 4* | 5 226 | 12* |
| Mean daily $N_2O$ flux /g N $ha^{-1}$ $d^{-1}$ | 5.8 | 117** | 8.9 | 74** | 4.8 | 99** | 6.8 | 159** |
| Mean daily $CO_2$ flux /kg C $ha^{-1}$ $d^{-1}$ | 33.2 | 44** | 16.6 | 46** | 29.7 | 41** | 25.6 | 47** |
| $N_2O$ diurnal amplitude /g N $ha^{-1}$ $d^{-1}$ | 4.5 | 168** | 5.6 | 157** | 3.9 | 126** | 4.7 | 177** |
| $CO_2$ diurnal amplitude /kg C $ha^{-1}$ $d^{-1}$ | 17.5 | 71** | 7.9 | 75** | 23.7 | 68** | 19.1 | 71** |

Conversely, $N_2O$ emissions measured cumulatively were roughly 50% greater in compacted plots, reaching 1 448-1 382 g N $ha^{-1}$ versus 944-977 g N $ha^{-1}$ in uncompacted plots in 2007-2008, respectively. Including interpolations for the few missing measurement dates did not change much these total emissions (Table 5). However, these cumulative emissions were slightly underestimated, since the fluxes could not be measured during the first 4 and 2 days after fertilization in 2007 and 2008, respectively. The range of these emissions is consistent with other published values of cumulative $N_2O$ emissions in sugar beet fields: 1 240 g $N_2O$-N $ha^{-1}$ between April and October according to Kaiser *et al.* (1998) and in the lower range of annual emissions in fertilized soils that can emit between 0 and 30 kg $N_2O$-N $ha^{-1}$ $y^{-1}$ (Bouwman, 1996). $N_2O$ emissions followed a seasonal pattern with high temporal variation, as commonly observed (Bouwman, 1996; Hénault *et al,* 1998; Kaiser *et al,* 1998). They were particularly high after rainfall events, soon after fertilization but also during summer and autumn. In compacted plots, high emissions were more frequent and continuous during longer periods, whereas they were more punctual and isolated in uncompacted plots. The highest peaks reached 96 g N $ha^{-1}d^{-1}$ in compacted plots and 45 g N $ha^{-1}d^{-1}$ in uncompacted plots. Along the crop cycles, we observed daily baseline $N_2O$ emissions of 2-3 g $ha^{-1}$, indicating that $N_2O$ producing processes generally did not stop. Net $N_2O$ consumption also occasionally occurred as some negative fluxes were recorded on the hourly basis, but only one averaged daily flux was negative (in the uncompacted plot in September 2007, DOY 249). There was large diurnal amplitude in fluxes (Table 4), with higher fluxes occurring between 10 a.m. and 4 p.m. (GMT). The range of temporal flux variations was substantial at both the daily and the crop cycle scales, emphasizing the necessity to measure $N_2O$ fluxes several times each day and over long periods of time (Laville *et al,* 1997; Wagner-Riddle *et al.,* 2007). The spatial variability was much lower than the temporal, as the mean coefficient of variation between chambers varied from 4 to 25% (Table 4). The area covered by the chambers was sufficient to gather at least the spatial variability due to emission hot-spots in microsites, confirming results of Kaiser *et al.* (1998) and Laville *et al.* (1999).

Despite the different N fertilizer rates applied in 2007 and 2008 (100 and 150 kg N $ha^{-1}$, respectively), $N_2O$ emissions were better correlated to compaction treatment than to fertilizer rates. In 2007, 1.45% and 0.94% of the fertilizer dose was emitted as $N_2O$ in the compacted and uncompacted plots, respectively, versus 0.94% and 0.65% in 2008. This result also emphasizes that fertilization alone cannot be sufficient to estimate $N_2O$ emissions.

### 4.3. Simulated $N_2O$ fluxes with NOE2

Cumulative simulated fluxes were within the order of magnitude of observed fluxes and correctly reproduced the treatment effect (Figure 3). Simulated fluxes reached 428-416 g N-$N_2O$ $ha^{-1}$ in uncompacted plots and 1 559-1 032 g N-$N_2O$ $ha^{-1}$ in compacted plots in 2007-2008, respectively (Table 5). They were on average lower than observed values, i.e. NOE2 underestimated the actual $N_2O$ emissions. In fact, the simulated fluxes in compacted plots were close to the levels of those observed, whereas they were largely underestimated in uncompacted plots. Peak emissions were better captured in compacted plots, but high emissions were globally underestimated. Whatever parameter is considered, the mean difference on the 90% percentiles of daily fluxes was always greater than for the other percentiles or mean fluxes.

Cumulative fluxes as well as their temporal dynamics within both treatments were correctly simulated as long as substrates (ammonium for nitrification and nitrate for denitrification) remained abundant. However, these fluxes became systematically lower compared to measurements starting around the beginning of August (DOY 215-220) with the first heavy summer rainfalls (Period IV).

### 4.4. Nitrification modelling

The kinetics of ammonium disappearance in the 0-10 cm soil layer was well simulated by NOE2 both years, although with too low a rate in 2008 (Figure 4). This good agreement suggests that the nitrification sub-model of NOE2 was satisfactory and could be trusted for predicting $N_2O$ emissions. The model indicated that nitrification contributed to a major part of $N_2O$ emissions, between 47 and 92% of the total across the four simulations (Table 5). During period I, i.e. when ammonium derived from fertilizer nitrification was nitrified, the $N_2O$ emissions due to nitrification corresponded to a great part of total simulated emissions in compacted plots and to the totality of those simulated in the uncompacted plots (Table 5). During period IV, background $N_2O$ emissions from nitrification were simulated, since the introduction of mineralization made it possible to take into account new ammonium inputs from mineralization throughout the cycle. These background emissions constituted the quasi-total simulated fluxes in both compacted and uncompacted fluxes during this last period, when measured nitrate concentrations were very low.

Compared to the original model, the modifications (parameters and formalisms) introduced in NOE2 had a marked effect on output with the nitrification rate simulated with NOE2 2 to 7 times greater than with NOE (Figure 4) and $N_2O$ emissions 4 to 150 times greater across years and treatments. This difference is due to changes implemented in the functions of both the nitrification rate and the ratio of $N_2O$ from nitrification. Results of the sensitivity analysis of NOE2 when the values of the nitrification parameters were changed are presented in Figure 5. They show first that any change in parameters leading to higher precision (lower RMSE) would be accompanied by an accentuated bias (higher MD), indicating that the parameters (estimated independently of the experiment) were correctly estimated. The only exception is the nitrification potential $N_p$ that could be reduced by about 30%. Such a reduction would improve the RMSE without degrading the MD.

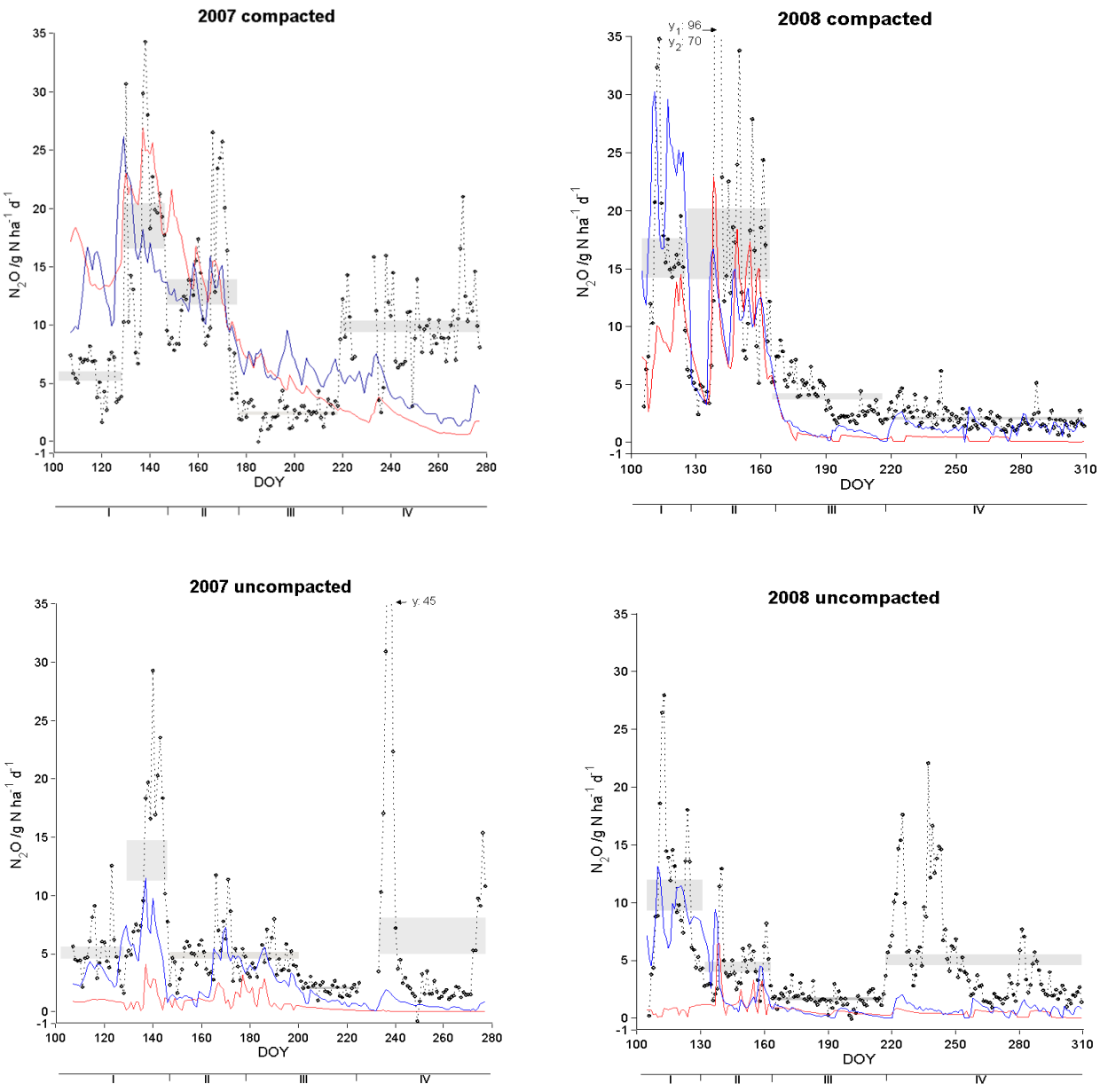



**Figure 3** Observed (dotted lines) and simulated (full lines) daily $N_2O$ fluxes in the two treatments during the two years. Simulations were run with NOE (red lines) and NOE2 (blue lines). Shaded areas correspond to the four periods (I to IV indicated below x axis) and sub-periods (see text). Their thicknesses represent the mean flux ± 1 standard deviation.

**Table 5** Process rates and contributions to $N_2O$ emissions simulated with NOE2 during each period compared to measured fluxes. Values in brackets are the fraction of nitrified and denitrified nitrogen emitted as $N_2O$.

| | Period I | Period II | Period III | Period IV | Total |
|---|---|---|---|---|---|
| **2007 uncompacted** | | | | | |
| Duration /days | 45 | 30 | 49 | 52 | 176 |
| Simulated nitrification /kg N $ha^{-1}$ | 70 | 13 | 23 | 12 | 118 |
| Simulated denitrification /kg N $ha^{-1}$ | 0.05 | 0.2 | 0.2 | 0.003 | 0.4 |
| $N_2O$ from nitrification /g N $ha^{-1}$ | 192 (0.28%) | 40 (0.31%) | 71 (0.30%) | 29 (0.25%) | 332 |
| $N_2O$ from denitrification /g N $ha^{-1}$ | 15 (29%) | 42 (25%) | 37 (23%) | 1 (19%) | 95 |
| Total simulated $N_2O$ /g N $ha^{-1}$ | 207 | 82 | 108 | 30 | 427 |
| $N_2O$ observed /g N $ha^{-1}$ | 338 | 155 | 145 | 323 | 961 |
| **2007 compacted** | | | | | |
| Duration /days | 45 | 30 | 42 | 59 | 176 |
| Simulated nitrification /kg N $ha^{-1}$ | 78 | 17 | 27 | 30 | 152 |
| Simulated denitrification /kg N $ha^{-1}$ | 2.1 | 1.7 | 0.9 | 0.6 | 5.3 |
| $N_2O$ from nitrification /g N $ha^{-1}$ | 324 (0.42%) | 105 (0.60%) | 151 (0.55%) | 148 (0.50%) | 728 |
| $N_2O$ from denitrification /g N $ha^{-1}$ | 374 (18%) | 266 (15%) | 120 (13%) | 72 (12%) | 832 |
| Total simulated $N_2O$ /g N $ha^{-1}$ | 698 | 371 | 271 | 220 | 1 560 |
| $N_2O$ observed /g N $ha^{-1}$ | 443 | 390 | 101 | 589 | 1 523 |
| **2008 uncompacted** | | | | | |
| Duration /days | 27 | 32 | 53 | 93 | 205 |
| Simulated nitrification /kg N $ha^{-1}$ | 93 | 25 | 12 | 25 | 155 |
| Simulated denitrification /kg N $ha^{-1}$ | 0 | 0.1 | 0 | 0 | 0.1 |
| $N_2O$ from nitrification /g N $ha^{-1}$ | 227 (0.24%) | 61 (0.25%) | 26 (0.22%) | 70 (0.28%) | 384 |
| $N_2O$ from denitrification /g N $ha^{-1}$ | 0 | 32 (24%) | 0 | 0 | 32 |
| Total simulated $N_2O$ /g N $ha^{-1}$ | 227 | 93 | 26 | 70 | 416 |
| $N_2O$ observed /g N $ha^{-1}$ | 275 | 142 | 88 | 472 | 977 |
| **2008 compacted** | | | | | |
| Duration /days | 21 | 39 | 52 | 93 | 205 |
| Simulated nitrification /kg N $ha^{-1}$ | 96 | 29 | 19 | 32 | 176 |
| Simulated denitrification /kg N $ha^{-1}$ | 0.3 | 2 | 0.08 | 0.003 | 2 |
| $N_2O$ from nitrification /g N $ha^{-1}$ | 361 (0.38%) | 114 (0.40%) | 52 (0.28%) | 124 (0.39%) | 651 |
| $N_2O$ from denitrification /g N $ha^{-1}$ $d^{-1}$ | 73 (23%) | 291 (19%) | 18 (22%) | 0 | 382 |
| Total simulated $N_2O$ /g N $ha^{-1}$ | 434 | 405 | 70 | 124 | 1 032 |
| $N_2O$ observed /g N $ha^{-1}$ | 312 | 667 | 210 | 193 | 1 382 |

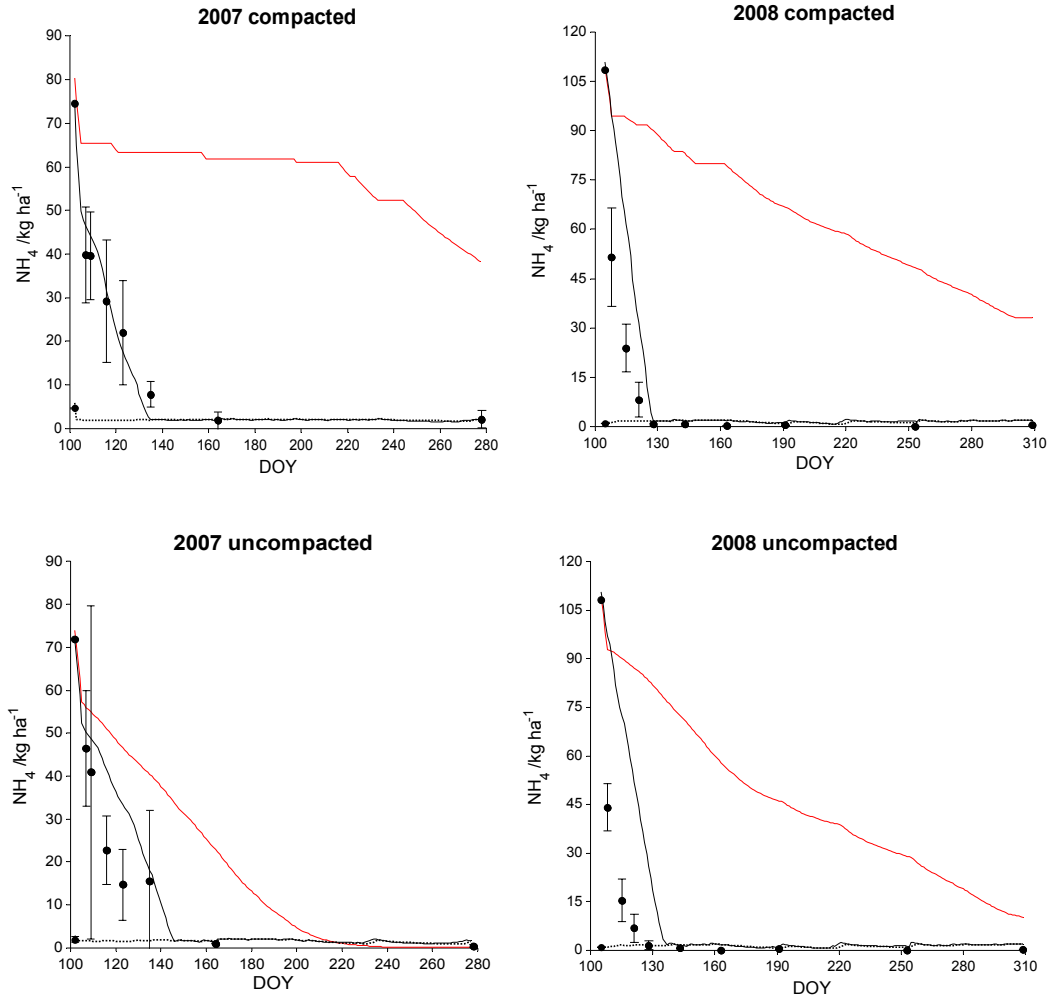



**Figure 4** Observed (symbols) and simulated ammonium contents in soil over 0-10 cm (full lines) and 10-20 cm (dotted lines) in the two treatments during the two years. Simulations were run with NOE (red lines) and NOE2 (black lines). The error bars indicate the confidence intervals on measurements ($p<0.05$).

The sensitivity analysis also showed that the simulated $N_2O$ fluxes with NOE2 varied little when the parameters defining the rate of nitrification ($K_A$, $Q_{10n}$ and $A_{min}$) were changed (Figure 5). The enhanced nitrification resulted first from the large increase in the potential nitrification rate $N_p$ (which was implicitly set at 1 kg N $ha^{-1}$ $d^{-1}$ in NOE). The change in the water function and the suppression of the 80% *WFPS* threshold led to a more continuous and quicker nitrification, especially in compacted plots where soil water content was higher. The introduced function, $f_{O2}$, taking over the water function when oxygen was more limiting than water, had a small counteracting impact on the increase of the nitrification rate, since its influence was limited to a small range of high water content values. Hence, NOE2 was not very sensitive to the parameters of this function, i.e. $a_S$ and $b_S$. The introduction of mineralization potential (*Mp*), to which NOE2 was very sensitive, greatly influenced the overall $N_2O$ emissions along the crop cycle as it enhanced the nitrification rate once the fertilizer-N had disappeared from the mineral N pool. We also introduced changes in the $N_2O$/N nitrified ratio (*z)* which increased the simulated $N_2O$ emissions coming from nitrification. First, this ratio increased versus *WFPS* according to the $g_{O2}$ function with NOE2 outputs quite sensitive to the three parameters of this

function ($z_0$, $c_S$, and $d_S$). Second, we did not consider any reduction by denitrification of $N_2O$ produced by nitrification.

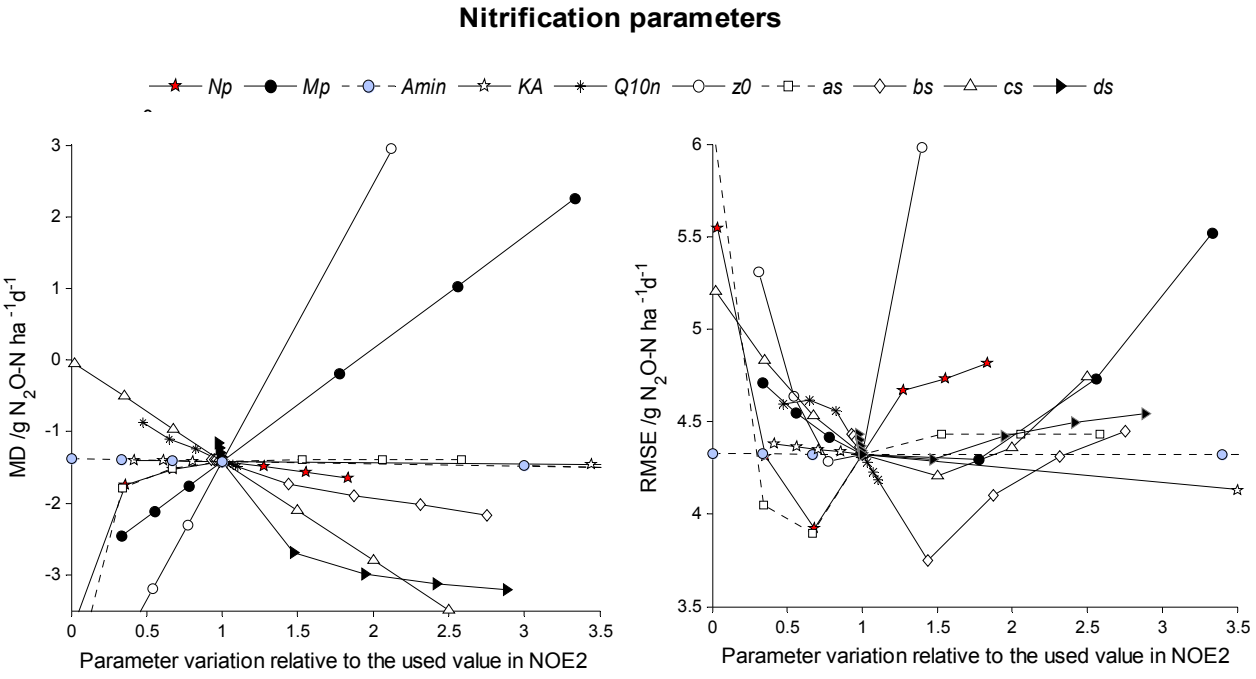



**Figure 5** Sensitivity analysis of NOE2 when the values of nitrification parameters of vary: mean difference (MD, left graph) and root mean square error (RMSE, right graph). MD and RMSE were calculated for the 16 studied sequences (= 4 periods * 2 treatments * 2 years). The x axis represents the ratio of the parameter value to its nominal value.

The last change in NOE2 was the discretization in soil layers. Unexpectedly, the refined description of soil water content and temperature across the profile did not change the simulation much. The main effect of the change was relative to the nitrification rate soon after the application of fertilizer. Figure 6 shows the effect of varying the depth of placement of the ammonium derived from the fertilizer on $NH_4^+$ disappearance and $N_2O$ fluxes, for the 2007-compacted treatment. Indeed this depth may vary with rainfall since UAN fertilizer contains both $NH_4^+$ ions and urea. $NH_4^+$ ions are strongly adsorbed onto the clay-humus complex and therefore rather immobile, while urea is highly soluble and can move downwards before being hydrolyzed into $NH_4^+$. In 2007, the better simulation of ammonium kinetics was obtained with a 1 cm only fertilizer depth (Figure 6), whereas it was 4-6 cm in 2008 (results not shown). This is consistent with the different rainfall patterns between the two years. Nevertheless, $N_2O$ fluxes were only better simulated in the uncompacted treatments. In compacted treatments, improving the ammonium kinetics led to slightly overestimate the $N_2O$ fluxes.

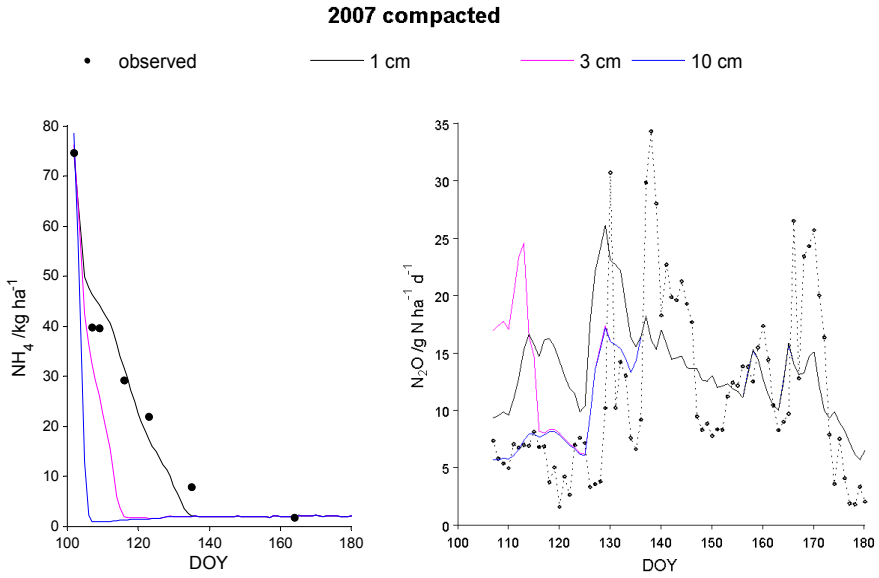



**Figure 6** Sensitivity analysis of NOE2 when the depth of location of the ammonium derived from the fertilizer vary: ammonium content (left graph) and daily $N_2O$ emissions (right graph). Full lines correspond to NOE2 simulations in the 2007 compacted treatment during period I.

### 4.5. Denitrification modelling

Across treatments and years, high $N_2O$ fluxes from denitrification were predicted by NOE2 only when nitrate content was high, in May (DOY 135-140), as a combination of residual nitrate from fertilizer plus the nitrate produced by fertilizer nitrification. During this period (Period II) the simulations were correct even though peak emissions and more generally emission amplitudes were underestimated. Globally, denitrification simulated with NOE2 contributed to 37-53% of total $N_2O$ emissions in the compacted plots, versus 8-22% in the uncompacted plots (Table 5).

In the 2007-compacted plot, total $N_2O$ emissions were overestimated during periods I and III. We believe that simulated $N_2O$ fluxes from denitrification were exaggerated for two reasons: i) the dry conditions in period I (between DOY 102 and DOY 124) and the low nitrate contents during period III were not favourable for denitrification; and ii) simulated $N_2O$ from nitrification closely followed observed fluxes. Part of these simulated emissions was produced in deeper soil layers and might actually have been reduced in the upper layer before escaping from soil surface.

During period IV, emissions from denitrification were close to zero in all simulations since nitrate was no longer available. Given their amplitude, it is not likely that the fluxes observed during that period were attributable solely to nitrification, particularly in the uncompacted plots. Since the fraction of $N_2O$ emitted per unit of nitrified N ($z$) was low, such peaks would require high amounts of ammonium that were neither measured nor simulated. Denitrification, resulting in higher proportions of emitted $N_2O$ per unit of denitrified N ($r$), might thus also have occurred although it was not simulated.

The denitrification rate simulated with NOE2 was in average 1.2 to 3.6 times lower, but $N_2O$ emissions were roughly equal, twice as much or twice as low due to the varying ratio. We did not change the denitrification rate of the original NOE model fundamentally. Differences

between denitrification rates simulated by NOE and NOE2 are therefore related only to discretization of the soil water content. In contrast to nitrification, this discretization had a strong impact on $N_2O$ emissions by denitrification that were more reduced during dry conditions than increased following rainfall events.

The sensitivity analysis of denitrification parameters showed that the model was sensitive to almost all parameters (Figure 7). In view of the lack of reliable information on some key parameters ($D_p$, $K_N$ and $r_0$) we proceeded by simultaneously optimizing them to check the validity of our previously estimated values. Our estimated value for $D_p$ (Germon *et al*, 2007) was among the optimal ones. However, the strong correlation (hyperbolic relationship) between $D_p$ and $r_0$ made it difficult to fix a precise value for each parameter individually. $K_N$ optimized value was much higher than that calculated using the available data on laboratory measurements (Khalil *et al.,* 2005). All the parameters were finally close to their optimal values, at least for the RMSE criterion (Figure 7).

The main modification brought to the denitrification module was the introduction of the *r* function. This function modulated the maximum ratio $r_0$ according to water and nitrate contents and included four parameters ($c_W$, $c_N$, $d_N$, $N_c$), NOE2 being very sensitive to two of them, i.e. $c_W$ and $c_N$. (Figure 7). We compared $N_2O$ emission simulations by NOE2 with either a constant ratio or the implemented function. This function enhanced $N_2O$ emissions when environmental conditions were propitious, i.e. *WFPS* greater but close to 62% and high nitrate concentrations. It thus improved NOE2 capability to recover part of the peak amplitude. Mean *r* value across the simulations was 21%, whereas the fixed NOE $r_{mx}$ value was 11%. It was, in average, greater in the uncompacted plots than in the compacted ones (Table 5). However, the improvement in capturing the peak emissions remained marginal within our explored ranges of water and nitrate contents.

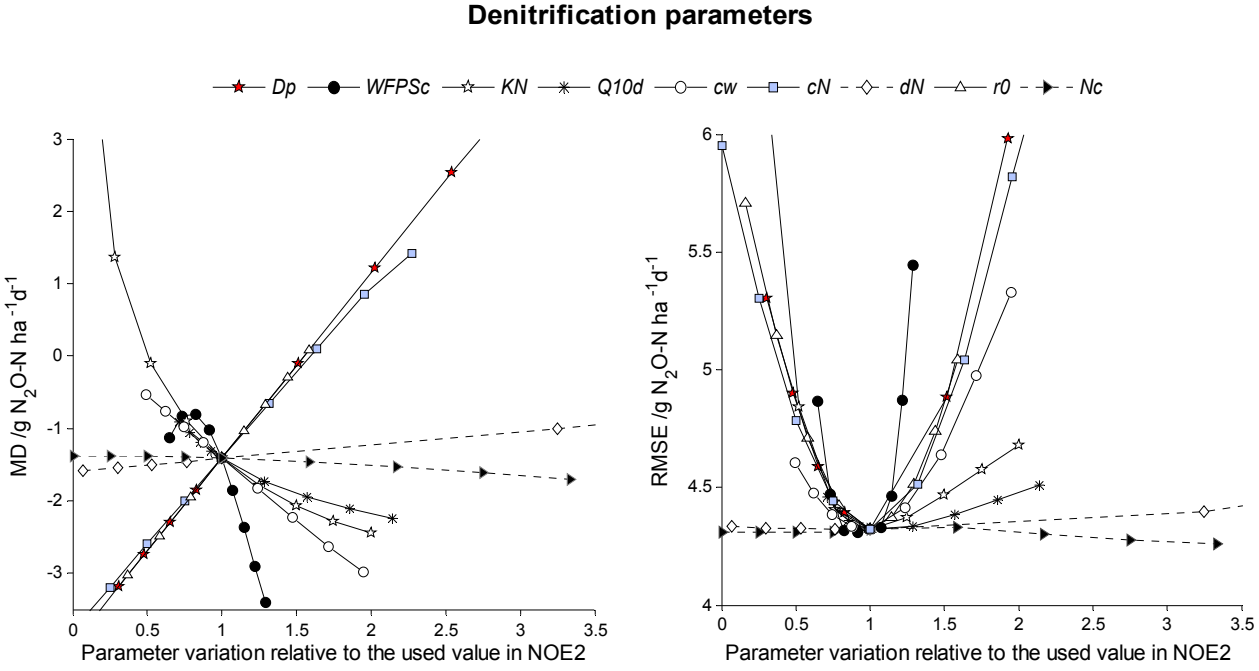



**Figure 7:** Sensitivity analysis of NOE2 when the values of denitrification parameters vary: mean difference (MD, left graph) and root mean square error (RMSE, right graph). MD and RMSE were calculated for the 16 sequences (= 4 periods * 2 treatments * 2 years). The x axis represents the ratio of the parameter value to its nominal value.

When comparing the NGAS-*r* and NOE2-*r* functions, simulated $N_2O$ emissions were close. NOE2-*r* performed slightly better when nitrate was more abundant, and $N_2O$ emissions by denitrification were higher with NOE2-*r* compared to NGAS-*r*. Finally, we tested the widely used hypothesis (Delgrosso *et al.,* 2000) that soil respiration could be used as an indicator of the dynamic denitrification potential. Indeed, the heterotrophic respiration rate from organic matter decomposition components is used as a proxy for labile C availability (Heinen, 2006; Chen *et al.*, 2008) and anaerobic $CO_2$ production is closely related to total denitrification (Swerts *et al.*, 1996). We tested this hypothesis with our measured $CO_2$ fluxes during the first two periods, i.e. when $CO_2$ derived from plant respiration was much lower than that derived from soil. We replaced the constant site-specific $D_p$ value by a daily denitrification potential ($D_{pi}$) proportional to the daily $CO_2$ flux of each treatment. The proportionality coefficient was calculated so that the mean $D_{pi}$ was equal to $D_p$. This method made it possible to slightly better mimic the kinetics of high emissions, when emissions by denitrification prevailed, i.e. during period II in the compacted plots. However, it did not improve significantly the overall simulation of $N_2O$ emissions, particularly concerning the under-estimation in uncompacted plots.

## 5. Discussion

The risk of soil compaction in sugar beet cropping systems is particularly important during harvest (Guérif *et al.*, 1996) and even more severe when harvest occurs in wet conditions (Boizard *et al.*, 2002). Soil compaction is an important environmental issue because it leads to greater water-filled pore space, greater denitrification rates and subsequently higher $N_2O$ emissions. Reduced tillage notably implies that be ploughing avoided, thus leaving compacted zones and residues near the surface. This practice, with a view to reducing energy inputs and erosion risks, has remained marginal in sugar beet cropping systems up to now in France (ITB, 2003), but it is widely fostered in diverse cropping systems worldwide. No-tillage can increase $N_2O$ fluxes because the reduced porosity and macro-pore connectivity of compacted zones increase water conservation and limit the oxygen diffusion rate resulting in an increased volume of soil in an anaerobic state (Dexter, 1997; Gregorich *et al.*, 2006; Germon *et al*, 2007; Bhandral *et al.*, 2007; Ball *et al.*, 2008). Pore bottlenecks control water retention properties and soil hydraulic conductivity (Dexter, 1997). Czyż (2004) found that a 5% increase in bulk density (from 1.47 to 1.54 g $cm^{-3}$) led to a 63% decrease of the oxygen diffusion rate in a silt loam soil. In our experiments, as expected, compacted plots with higher bulk densities and greater *WFPS* emitted more $N_2O$ than uncompacted ones. The observed increase in $N_2O$ emissions due to compaction was in the lower range of published *in* and *ex situ* studies. Such studies describe cumulative emissions increasing by from 1.3 to 8 times over periods of one to three months (Bhandral *et al.,* 2007; Uchida *et al.,* 2008).

NOE2 simulated $N_2O$ fluxes within the range of observed fluxes and made it possible to predict the differences between compacted and uncompacted treatments. Enhanced denitrification due to higher *WFPS* in compacted treatments was simulated and showed that differences in $N_2O$ emissions between treatments were due mainly to a higher denitrification rates in compacted plots. While simulated nitrification rates remained within the same order of magnitude across treatments and years, simulated denitrification rates rose by a ten-fold in compacted plots. Simulated $N_2O$ emissions from denitrification in uncompacted plots contributed to only 8-22% of total emissions compared to 53-37% in compacted plots, although the ratios of $N_2O$ emitted over total denitrified N were in average higher in uncompacted plots. To a lower extent and despite similar nitrification rates between treatments, the contribution to total simulated emissions of nitrification $N_2O$ emissions was roughly twice as high in compacted plots as in uncompacted plots due to higher *z* ratios. Simulated nitrification rates and

nitrification $N_2O$ emissions were much higher with NOE2 than with NOE. Despite the improvement of the nitrification simulation and higher $N_2O$ ratios compared to NOE, NOE2 globally underestimated $N_2O$ emissions, especially during period IV.

During the first three periods (I-III), NOE2 could well reproduce average fluxes, but was less able to recover the amplitude of flux variations due to denitrification. Our hypothesis is that the main driver of this variation was water content, since nitrate content and temperature did not vary much at the scale of a few days. In this context, the vertical discretization of water content within finer soil layers should have been decisive. However, NOE2 still did not simulate the amplitude of $N_2O$ emissions with high enough accuracy. Either the model did not react sufficiently to variations in soil water content, or the simulated water contents were not close enough to the field conditions, notably within the first 5 cm. In fact NOE2 reacted well to the simulated drier soil water content near the surface (e.g. in April 2007), but the discretization did not change enough the soil water content profile in moist conditions to induce any substantial change in NOE2 outputs. Our results suggest that the relationships between denitrification rate and *WFPS* and nitrate content could be improved, as also suggested by the remaining large unexplained variance in laboratory studies (e.g. Weier *et al.,* 1993).

During period IV, the underestimation of $N_2O$ fluxes by NOE2 was probably related to several combined factors, since any single factor could not explain the underestimation. These factors are: i) unconsidered substrate inputs, ii) possible increased ratio $N_2O$/N-denitrified, and iii) possible decrease in apparent $K_N$. First, measured substrate contents ($NH_4^+$ or $NO_3^-$) were low and strongly limiting. In sugar beet fields, senescence starts early and fallen leaves may constitute a significant nitrogen input since their C/N ratio is low (Kaiser *et al.,* 1998). Furthermore, wetting of dry soil in summer can induce a so-called "Birch effect" that releases both carbon and nitrogen substrates (Borken and Matzner, 2009). The ammonium formed by these two processes would be rapidly nitrified and subsequent nitrate also rapidly denitrified following rainfall events. Regarding the fast turnover of these pools (e.g. due to active plant uptake), intensive soil samplings would be required to observe any accumulation of mineral N. The second hypothesis is a possible increase of the $N_2O$/N-denitrified following dry-wet cycles, due to the delay in recovering the capacity to reduce $N_2O$, especially for taxa "adapted" to cropped soils (Bergsma *et al.*, 2002, Khalil *et al.*, 2005). The intensity and repetitions of drying-out periods may lead to the highest $N_2O$ emissions (Borken and Matzner, 2009), as observed in the uncompacted plots that were more severely impacted by drought. The third hypothesis is a change in the apparent nitrate affinity constant of soil denitrifiers ($K_N$), which could be due to the wide variation in nitrate diffusion rates in soil during dry-wet cycles. Indeed, we found a very wide range of values for this nitrate affinity constant ($K_N$) across the literature, depending on the tested initial nitrate concentrations and the experimental protocol. Low $K_N$ values within a spectrum of low nitrate concentrations in laboratory studies may hide analytical bias due to a much reduced diffusion *in situ* compared to laboratory studies (Conrad, 1996). Reducing $K_N$ value would increase denitrification rate particularly when nitrate concentration is low. One model improvement could consist in introducing a variable $K_N$ value accounting for variation of the soil diffusion rate versus water content.

Further improvement in modelling the proportion of $N_2O$ reduced to $N_2$ should take into account both, microbial adaptation to dry-wet cycles and diffusion constraints across the vertical soil profiles. Better simulation of the probability of $N_2O$ reduction depending on the locus of its production, in particular on its depth within the soil profile (Clough *et al.,* 2004) and on the migration of the aerobic-anaerobic interface (Tiedje *et al.*, 1982), should also enhance the model. It is worth noticing that diverging results on higher $N_2O$ emissions in non-tilled may be related partly to this probability of $N_2O$ reduction to $N_2$. Indeed, when nitrogen availability is

limiting, $N_2O$ emissions may be favoured by an easier diffusion through the soil matrix in uncompacted plots, whereas in compacted plots it is more likely to be further reduced to $N_2$ (Gregorich *et al.*, 2006; Chatskikh and Olesen, 2007).

The objective of our study was to obtain two contrasted treatments with regard to compaction, and evaluate our ability to explain both the kinetics of emission and the differences between the two treatments, using the measurements of the main variables driving $N_2O$ emissions. Our experimental design made it possible to test for a significant difference between the compacted and uncompacted sites, which is adequate for model evaluation. We have done the same experiment during two years, on two different fields, which gave more confidence in our interpretation of the observed effects in terms of compaction effect. However, we could not intend to derive general conclusions, from statistical inference tests, about the effect of compaction on $N_2O$ emissions in our experiments, since the treatment itself (compaction) should have been fully replicated (at least with interspersed treatments) in order to conclude from statistical inference on the local effect of soil compaction on $N_2O$ emissions. This approach, used in various climatic and soil conditions, would be particularly useful to derive general conclusions. Optimization of sampling technique by considering both spatial and temporal variability, while dealing with the severe constraints to measure fluxes with automatic chambers, should be considered for future research.

## 6. Conclusion

Greater soil compaction led to higher $N_2O$ emissions, and the impact of soil compaction and climate conditions interacting prevailed over the nitrogen fertilizer doses. While contradictory results on the effects of reduced tillage on $N_2O$ emissions may be due to the numerous factors associated with changes in tillage (including soil structure and organic matter), the main drivers that enhance $N_2O$ emissions in compacted zones have been identified and can be modelled. The accuracy in predictive modelling will depend on the accuracy in modelling these drivers.

NOE2 made it possible to simulate the impact of soil compaction on $N_2O$ emissions. The discretization into fine soil layers notably improved nitrification modelling and would be relevant for testing diverse fertilization scenarios, i.e. the impact of fertilizer type or fertilization depth. It also brought to light that a precise modelling of water content across the vertical soil profile is a prerequisite for any increase in model accuracy. Despite the introduction of dynamic functions for the ratios of end-products ($z$, $r$), further work is needed to model the impact of diffusion constraints and dry-wet cycles on these ratios.

NOE is already coupled with CERES and STICS agroecosystem models. Coupling NOE2 with such models will make it possible to better account for nitrate dynamics and other impacts of plant growth on $N_2O$ production, such as the return of organic material to the soil due to root or leaf senescence. Moreover, the simulated $CO_2$ emissions due to soil respiration will give a dynamic indicator for potential denitrification throughout the crop cycle. Finally, modelling the impact of the cropping system on $N_2O$ emissions via changes in soil structure could be done by introducing a mechanical model (e.g. Roger-Estrade *et al.,* 2004) to quantify the proportion of compacted zones considering the type of operation and weather conditions at the time of wheeling.

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The final thesis expected result was to produce Life Cycle Assessments (LCAs) of biofuels that would be sufficiently precise to be used as a decision-tool. Regarding the synthesis proposed in Chapter 1 on the LCA of biofuels and their current limits, we worked on building local LCAs that aim at overcoming these limits and reduce the overall uncertainty in biofuel LCAs. The focus was notably to reduce the uncertainty in the global biofuel greenhouse gases by reducing uncertainty on $N_2O$ emission estimates. Agricultural greenhouse gas emissions can contribute to almost half of the biofuel greenhouse gas balance (Farrell *et al.,* 2006), this share being even greater in the case of pure vegetable oils. $N_2O$ then contribute to the great majority of these agricultural greenhouse gases.

Within LCA, $N_2O$ emissions are commonly characterized with the help of emission factors such as those recommended by the IPCC' Tier 1 method. These emission factors were produced from statistical regressions compiling monitored emissions from all over the world. Some studies on biofuel greenhouse gas balance assessments have also used models to predict these emissions, but still in the view to define emission factors at a country level. This characterization based on emission factors introduces a high uncertainty on results, since it does not take into account the variability in $N_2O$ emissions due to the local production factors, i.e. local soil properties, climatic conditions, and management practices. On the contrary, agroecosystem models, such as CERES for instance, make it possible to predict $N_2O$ emissions, while considering climate, soil properties, and management practices as input variables. We therefore ran CERES to model the flows of biomass and polluting compounds to be used as entry data for the local biofuel LCAs.

Within CERES, NOE is the implemented sub-model that simulates $N_2O$ emissions by nitrification and denitrification. We therefore implemented the changes in the NOE functions that resulted from the work on NOE2 (Chapter 2) to enhance the model performances notably concerning the ratio of $N_2O$ over final products (see Appendix 9 for comparison of NOE/NOE2 within CERES). However, we could not implement the finer vertical soil discretization that had shown to improve also the simulations. Indeed, this discretization requires further modifications in other routines of CERES, especially in the modelling of water balance and the distribution of fertilizers. These modifications were outside the reach of this Ph.D. frame.

Since our local biofuel LCAs are to be useful for decision-making purposes, we paid attention to establish complete LCA rather than focusing only on the greenhouse gas assessments. We therefore used the Simapro software and Ecoinvent database to characterize all potential impacts. We decided to use local scenarios for the co-product substitutes. These LCAs are described in Chapter 3. Our final recommendation would be to try as much as possible to produce local LCAs in order to choose the right option for local development of bioenergy and biofuels. The impact of biofuel production on land use change is hardly assessable, especially trade-offs due to land competition for food and feed. This issue requires considering social and economic qualitative criteria. Since LCA does not deal with such criteria, it should be completed by another assessment, such as EIA (Environmental Impact Assessment) that would allow for the selection of a local priority order in criteria.

# Chapter 3

## Local Life Cycle Assessments of biofuels: comparison of ethanol from sugar beet and *Miscanthus.*

**Manuscript in preparation to submit a regular article to *Environmental Science and Technology***

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## – Abstract –

Renewable energy sources have been fostered to contribute to mitigate climate change and fossil resource depletion. Among these renewables, biofuels have been largely promoted worldwide. In Europe, the proposal directive (CEC, 2008) aims at substituting 10% (vol.) of consumed fossil fuel with biofuels by 2020. These biofuels should be produced according to criteria of sustainability, which notably include a minimum of 35% greenhouse gas savings compared to substituted fossil fuels. Published Life Cycle Assessments (LCAs) of biofuels have shown a great variability of potential impacts of biofuels, both due to the diversity in biofuel chains and to the varying quality and accuracy in the assessments themselves. We suggested that the best option to maximize the accuracy of biofuel LCAs is to produce local LCAs, i.e. encompassing the impacts of local edaphic, climatic and agricultural management factors. We hence produced local LCAs for two biofuel chains, 1st generation ethanol from sugar beet and 2nd generation ethanol from *Miscanthus,* and tested the impact of changes in local factors on the LCA results. We used the CERES-EGC model to produce inventory data for the agricultural production and the Simapro/Ecoinvent program/database to compile our LCAs. *Miscanthus* ethanol had overall much lower potential environmental impacts than gasoline or sugar beet ethanol, notably allowing for some 82-85% greenhouse gas savings and around 90% of fossil resources compared to gasoline. Sugar beet ethanol would lead to smaller savings 28-42% greenhouse gases and 20-28% of fossil resources per MJ compared to gasoline. The wide range of greenhouse gas savings of sugar beet ethanol illustrated that LCA results were sensitive to changes in local conditions during biomass growth and to management factors. Impacts of these factors were particularly noteworthy for acidification and eutrophication to which field emissions contribute to a great part. Trade-offs between low impacts and low ethanol yields per hectare emphasized the difficulty to define an optimum reference scenario. An assessment of the impacts on land as a resource should complete local LCAs to be used as keys for decision-making purpose. Finally, further modelling work is needed to better simulate *Miscanthus* production and environmental impacts, and to enhance the model ability to mimic management practices.

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## 1. Introduction

Facing the urging challenge of climate change mitigation, with no obvious alternative for fossil fuels while the world energy demand continuously increases, requires rapid decisions and actions. Renewable energy sources, so called "renewables", have been quickly fostered. While none of them alone may constitute an optimal solution, their diversity can help diversifying the energy mix and adapting it to local resources. Among these renewables, biofuels have been largely promoted worldwide. In Europe, the proposal directive (CEC, 2008) aims at substituting 10% of its consumed fossil fuel volume with biofuels by 2020. In order to account for this share and benefit from fiscal incentives, biofuels should be produced according to the following criteria of sustainability: they should not be produced in protected areas or in soils rich in organic carbon such as continuous forest areas or peat soils, and should allow a minimum of 35% greenhouse gas savings compared to equivalent fossil fuels. European biofuels should furthermore be produced with respect to European agreements on environmental-friendly production systems, although such criteria may be hardly applicable within the frame of international sustainability obligation (Pons, 2008). These criteria shall address the issue that not all biofuel chains may have a net benefit in terms of environmental impacts when compared to fossil fuels, notably because of polluting emissions during the agricultural production of the biomass or its conversion to liquid fuels (FAO, 2008).

Life Cycle Assessment (LCA) is an unavoidable tool to assess and compare potential environmental impacts throughout the life cycle of product chains while avoiding pollution trade-offs. However, the complexity of biofuel chain has led to some limits of this tool. Many studies on biofuel chains are therefore more based on a life-cycle approach focusing on specific impacts (e.g. global warming) than being actual LCA considering the all potential impacts (Quirin *et al.*, 2004; Blottnitz and Curran, 2007). The variability in data quality and the lack of harmonization on system boundaries across published studies also have led to very diverse results and conclusions. Confusion and controversy therefore arose on the environmental interest of biofuels. Moreover, LCA is a site and time independent methodology, so that the characterization of an environmental impact is linearly correlated to the sum of the contributory pollutants. This linear model is hardly relevant when assessing the environmental impacts of an agricultural production that result from complex interacting biogeochemical processes, themselves depending on the local edaphic, climatic and management factors.

Finally, the LCA methodology does not encompass a quantitative analysis of potential impact on land use as a resource, i.e. on the soil quality (organic matter, erosion…), on habitats or landscape etc. Given the potential expansive demand for biofuels and the limited available land area, worries have risen that biofuels may lead to land use changes, which should be considered when assessing their potential environmental impacts. Second-generation biofuels are more interesting in term of land use, because conversion processes deal with the bulk of the biomass raw material instead of using only part of the plants (i.e. seed oil, starch or sugar reserves). Being less selective, these processes make it possible to use more diverse biomass material and give higher yields per used land area unit. They are still not commercially widespread though, and data on pilot plants are scarce. Hence, LCAs of 2$^{nd}$ generation biofuels are still lacking.

Assessing the potential environmental impacts of a biofuel chain with enough accuracy so that a decision can be made/taken with respect to the sustainability criteria requires: 1) to internationally agree on common system boundaries; 2) to reduce model uncertainty; and 3) to complete the LCA with an assessment of the impacts on land as a resource. Many studies have

recently focused on harmonizing the LCA system boundaries for biofuel chains and especially concerning the handling of co-products (Quirin *et al.,* 2004; Farrell *et al.*, 2006). The Well-to-Wheel system including the energy embedded in farm machinery up to the fuel combustion and considering the substitution for co-products is among the most complete and recognized system boundaries. Some studies have also used agroecosystem models to generate input data for the LCA and diminish the overall uncertainty. Nevertheless, these model outputs were to the best extent related to country average data (JRC/EUCAR/CONCAWE, 2008; Smeets and Stehfest, 2009). We suggest that the best option to maximize the accuracy of biofuel LCAs is to produce local LCAs, i.e. LCAs encompassing the impacts of local edaphic, climatic and agricultural management factors, as well as local scenarios for co-product substitutions.

Our objective was hence to produce local LCAs for two biofuel chains and to test the impact of changes in local factors on the LCA results. We wanted to compare first and second-generation biofuels, so that we chose to establish local LCAs of the ethanol from sugar beet and from *Miscanthus,* both produced in the same agricultural region and promising high yields per hectare (Worldwatch Institute, 2007; Möller *et al.,* 2007).

## 2. Goal and scope definitions

### 2.1 Functional units and system boundaries

The aim of this study was to perform a full comparative Life Cycle Assessment (LCA) of the production and combustion of two biofuels: ethanol from sugar beet and lignocellulosic-ethanol from *Miscanthus x Giganteus* (later on referred to as *Miscanthus*). We compared both biofuels with the corresponding fossil fuel, gasoline, eliciting if the production of these biofuels offers environmental benefits. The chosen functional unit was 1 MJ (low heating value) of potential energy that would be delivered by the fuel at the refinery door if its combustion was complete (calculated on the basis of its C%, LHV). We did not account for biogenic $CO_2$ fixation and emissions. This system boundary is called Well-to-Tank system, i.e. including all flows of resources, energies and pollutants from the extraction of raw materials up to ethanol dehydration. Indeed, since the focus was the impacts of the fuel production and more specifically of the agricultural production, we did not include the steps of fuel blending and distribution to the service station.

Energy embedded in farm machinery and capital equipment was included. On the contrary, the caloric intake and transportation of farm worker were not included. Agricultural production is modern in the study area and does not require specific diets or extreme farm labour. Hence, worker diets should not induce any noticeable differences among the studied biofuel chains (e.g. Farrell *et al.*, 2006). Missing reliable inventory data prevented a complete inventory of some packaging, enzyme and yeast inputs for the fermentation processes. Figure 1 shows the limit of the system boundaries for the three fuel chains.

### 2.2 Co-product handling

The system was extended to take into account savings due to the substitution by co-products as recommended by LCA-ISO standards. Considered co-products of the sugar beet ethanol were pulps, lime-carbonation residue, vinasses and sugar. For each of these co-products an alternative product chain was designed according to local practices and based on the expertise of professionals from the region. Corresponding saved production of mineral fertilizers (Figure 1), were calculated based on the product contents and mineralization rates depending on the site

and time of spreading (Decloux *et al*. 2002; ITB, 2006). Pulps replaced maize silage, given close dry matter contents (around 25%DM) and low protein values that lead farmers to combine both compounds in feeding rations according to supply and needs (Comité National des Co-Produits/ADEME, 2001). For sugar, we used mass allocations, since a local substitute product would lead to consider the same sugar beet chain. The case study considered as proxy for the *Miscanthus* conversion, i.e. grass conversion to ethanol (Fromentin *et al.*, 2000), used both substitution and economic allocation for the diverse co-products. Substitution concerned the energy costs avoided thanks to biogas production from vinasses. Economic allocation factors were then used for fibres and proteins from the biomass waste. No sensitivity analysis to co-product handling was performed, since this study focused on the impact of local factors, which implied to consider the most probable scenarios for local co-product handling.

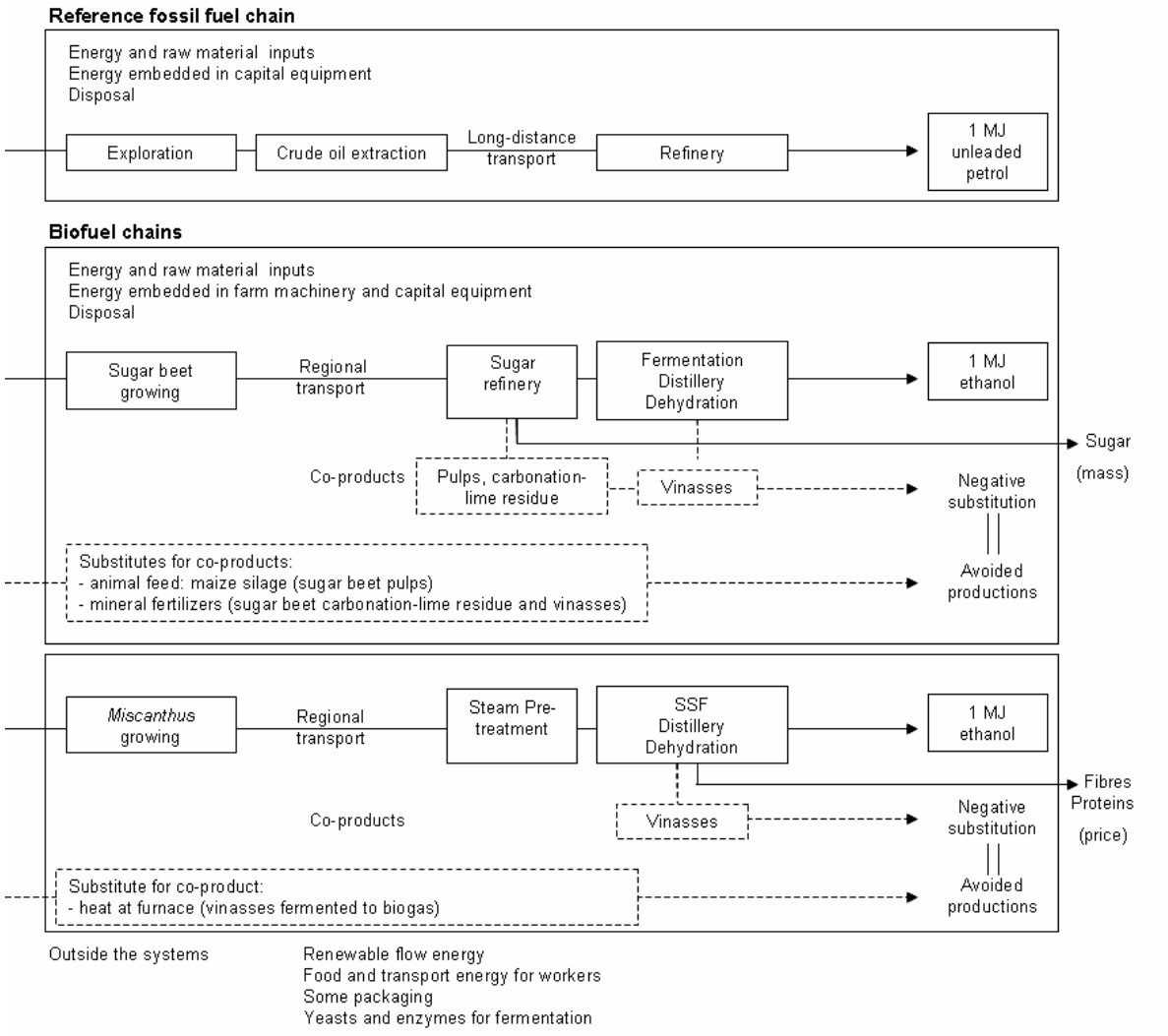



**Figure 1**: Well-to-Tank system boundaries of studied fuel chains.

### 2.3 Temporal and geographical scopes

LCA typically provides a "single snapshot in time" of studied systems (Sheehan et al, 2004). The aim of the study was to look at the possible impacts of state of the art technology concerning ethanol from sugar beet and near-future technology for the ethanol from *Miscanthus,* since commercial production of lignocellulosic ethanol should become available within a few years (Sheehan et al, 2004; USDOE, 2006a). Collected data are relevant of current production systems. Reference period corresponds to the 2000s. No prospective scenarios were studied about possible improvements of established technology in terms of energy efficiency or avoided pollution. In particular, we did not consider any possibility of fermentation $CO_2$ capture and we only assessed one lignocellulosic pathway, which is "often considered as the most rational technique for the production of ethanol from grass" (Jungbluth *et al.*, 2007) regarding the technological state of the art.

The agricultural production site was the INRA Agro-Impact experimental station at Estrées-Mons in the Picardie region (Northern France, 49°80'N, 3°60'E) where mean annual precipitation and temperature are 667 mm and 9.6°C respectively (Guérif *et al.*, 1996). Data on local production factors were relative to this specific site. The soil is an Orthic Luvisol (FAO, 1998). Climatic and agricultural management data were collected from this station during the last 22 years. None of the involved crops, sugar beet or *Miscanthus*, were irrigated at this site. In this area, sugar beet is a common head crop in the sugar beet-wheat-corn-wheat rotation, whereas *Miscanthus* has been first implemented in 2006, within the REGIX field trial. Data on *Miscanthus* production were hence both collected from this field trial and from literature to get some long-term perspectives.

### 2.4 Problem-oriented approach

We used the CML 2002 Baseline v2.04 method (Guinée *et al.,* 2002) that encompasses the impact categories listed in table 1. An overview of impact mechanisms is presented in Appendix 8. This method elaborates the problem-oriented (midpoint) approach without aggregating these impacts into damages (end-point impacts) to human health, to ecosystem quality or to resources (Goedkoop and Spriensma, 2000). This aggregation notably requires setting normalization and weighting factors, which adds to the subjectivity and uncertainty in interpreting the LCA results. Given some remaining shortcomings in the modelling of environmental relevance linking mid-point and end-point impacts, the problem-oriented approach is currently considered the "best available practice" for impact assessment (Guinée *et al.,* 2002). Weighting is furthermore not allowed by LCA-ISO standards when producing LCA for public product comparison. The scope of our study focusing on product comparison in terms of potential mid-points impacts, the problem-oriented approach was more relevant. We also had a look on the sensitivity of Climate Change indicators according to the used radiative forcing equivalents, i.e. IPCC equivalent factors from the third (2001) or the fourth (2007) Assessment Reports.

**Table 1**: Baseline impact categories in CML2 2000 v2.0 (Guinée *et al.*, 2002)

| Impact categories | Reference substances | Main contributing substances |
|---|---|---|
| Global warming (GWP20, GWP100) | $kg_{eq}$ $CO_2$ | $CO_2$, $N_2O$, HCFC22* |
| Stratospheric ozone depletion | $kg_{eq}$ CFC-11 | HCFC22* |
| Human toxicity | $kg_{eq}$ 1,4-DB | $N_2O$, $NO_x$, $NH_3$, Particules, $SO_2$, Acrylonitrile, Cadmium |
| Ecotoxicity: freshwater aquatic, marine aquatic, terrestrial | $kg_{eq}$ 1,4-DB | Acrylonitrile, Cadmium, Particles, $N_2O$, $NH_3$, $SO_2$ |
| Acidification | $kg_{eq}$ $SO_2$ | $SO_2$, $NO_x$ |
| Photo-oxidant formation | $kg_{eq}$ $C_2H_4$ | CO, Toluene, $NO_x$ |
| Eutrophication | $kg_{eq}$ $PO_4^{3-}$ | $PO_4^{3}$, $NH_3$ |
| Depletion of abiotic resources | $kg_{eq}$ Sb | Iron ore, Crude oil |

* HCFC 22 will be prohibited in January 2010 (A. Zoughaib pers.com. 11/2009)

## 3. Methodology and life cycle inventory

### 3.1 Industrial productions

*Fossil fuels and transports*

The Life Cycle Inventory of all processes involved in the production of the comparative fossil fuel, i.e. gasoline, was established using the SimaPro 7.1 software (PRé Consultants bv, NL) and Ecoinvent v2.0 databases (Swiss Centre for CLI). All data on primary energy sources, infrastructures, machines, industrial production and disposal were taken also from this database. Data on French energy mix were available, as well as European average data for crude oil supply, oil refinery and energy embedded in infrastructures and machines. For the second comparison, data inventory for the production of blended E10 fuels and the life cycle of cars was also based on Ecoinvent v2.0 database. Ecoinvent inventory for the production of ETBE was based on French data from IFP that were used in the ADEME/DIREM study (2002).

Transportation means and distances were also relevant for the average European case. Only distances directly related to input transport to farm and output transport from farm to refinery were specified for our case study. Emissions from transport included 2/3 of those due to empty way backs (ETH-Ecoinvent methodology). Transports of inputs to the agricultural field encompassed that of co-products from the refinery to the field, as well as saved transports for the products they substitute. Only substituted maize silage was assumed as produced on farm. Distances were recorded in accordance with current practices at the study site, i.e. 65 km to provide agricultural inputs to the farm, 22 km between the farm and the sugar refinery, and 200 km to provide raw material to conversion plants. In the case of *Miscanthus* further hypothesis were needed, since no lignocellulosic-based ethanol plant is operating in France. However, a project is ongoing (FUTUROL) that should give the way to such a construction in the north of France within the next ten years or so. As a credible expectation, we assumed that this virtual plant would be located within a radius of some 100 km to the experimental site[46]. We considered that rhizomes would be provided by ADAS in the United Kingdom (Bullard and Metcalfe, 2001; DEFRA, 2002), i.e. 300 km per freight and 160 km per lorry. No storage of rhizome was inventoried assuming that rhizomes would be planted within a few days after delivery.

[46] Pers. Comm. S. Cadoux, manager of REGIX trial and member of the FUTUROL project, 10/2009

*Biomass conversion*

Data on sugar refinery-distillery were taken from the ADEME/DIREM study (2002). These data were given for the Arcis-sur-Aube sugar refinery-distillery, which is located in the Champagne-Ardennes region nearby the Picardie region. The considered distance between the field and the refinery-distillery was in fact from the study site to another refinery-distillery at Eppeville (22 km away), since the producer is used to supplying this unit. We still considered the data from Arcis-sur-Aube, since this sugar-ethanol production unit was considered as representative for the state of the art of this kind of production units in France (ADEME/DIREM, 2002). Moreover, the usual distance to deliver sugar beet to conversion units remains within a radius of 30 km in the northern part of France[47].

Data for lignocellulosic conversion to ethanol are those of a pilot plant in Switzerland (Fromentin *et al.,* 2000), which is part of the Ecoinvent database v2.0 (Jungbluth *et al.*, 2007). This plant produces lignocellulosic ethanol from grass with steam pre-treatment and simultaneous saccharification and fermentation (SSF). Although this process has been a springboard to more efficient and cheaper technologies that are still under development, it provides a complete insight in environmental impacts of such conversion. Moreover, the accounting for biogas production from vinasses makes the process energy efficient. We considered that grass could be replaced by *Miscanthus* biomass due to relative close compositions. In the Ecoinvent inventory, the grass of meadow (45% cellulose, 29% hemicellulose, 26% others) was used, with 44%-C content. Recorded compounds of *Miscanthus* (43% cellulose, 27% hemicellulose, 24% lignin) are quite similar, with a C content of 48% (Lewandowski *et al.,* 2000). Inputs to the fermentation plant were given for a dry matter unit of biomass, so that we could take into account the difference in water content of both feedstock, i.e. 80% for grass and 20-25% for *Miscanthus* harvested in spring as done at the study site. Details of referenced characteristics on *Miscanthus* are given in Appendix 3.

### 3.2 Agricultural productions

*Modelling of agricultural emissions*

In order to take into account the local production factors in the assessment of biofuel production impacts, we modelled the agricultural production using the CERES agroecosystem model (Jones and Kiniry, 1986). This mechanistic model simulates the crop growth by considering the type of soil, climate conditions, plant variety, and management practices at the field scale. It also predicts substance losses in the ecosystem such as nitrate leaching, ammonium volatilization, or nitrous oxide emissions for instance.

CERES-EGC model (Gabrielle *et al.,* 2006a) simulates $N_2O$ emissions by nitrification and denitrification via the algorithms of NOE2 sub-model (Hénault *et al.,* 2005; Bessou *et al.,* submitted in 2009). In order to reduce the uncertainty on $N_2O$ emissions, we proceeded to a Bayesian calibration of the 22 parameters of NOE2. This site-specific calibration makes it possible to reduce significantly the spread in parameter distribution and the subsequent uncertainty on $N_2O$ emissions by updating these distributions against *a priori* probability distributions of parameter values gathered from different sites (Lehuger *et al.,* 2009). Based on data sets of field measurements on sugar beet plots at the study site, we hence defined the best set of parameter values for the nitrification and denitrification functions (see Appendix 9). This

[47] Pers. comm. D. Colbert, Director at Cristal Union, 10/2008

calibration reduced the RMSE on dynamic and cumulated fluxes by 15% and 63% respectively. Calibrated parameters were then used for the simulations of both cropping systems with CERES.

For sugar beet, we ran CERES-EGC with algorithms of CERES-beet (Leviel *et al.,* 2000; 2003). We used daily climate data from the Agroclim weather station at the site, and entered the data collected on local soil characteristics and cropping system management for the last 23 years (1986-2009). According to climate records over this period and with regard to cumulated rainfalls, we identified 1) median years (R50), 2) inferior quartile years (R25), and 3) superior quartile years (R75) (see Appendix 10). Mean values for yields and polluting flows were calculated per class of climatic year according to these rainfall patterns: R25, R50, and R75.

For *Miscanthus,* we ran an adapted version of the CERES-Sorghum model (Petter Medeiros, 1997) on an annual basis. The sprouting of shoots (ratoons) from the rhizomes was simulated with an annual sowing and tillering from the main stems at the same period. Yields and environmental emissions were averaged on an annual basis accounting for all inputs and outputs throughout the whole cropping system production, i.e. 15 years. The CERES-simulated outputs were fed into the life cycle inventory.

*Sugar beet*

In order to account for background emissions, we considered sugar beet as part of the crop rotation sugar beet-winter wheat-maize-winter wheat. This crop rotation was in place at the study site for more than 20 years. We hence ran CERES-EGC for the whole rotation (5 repetitions) during 1988-2009. For all crops, residues were incorporated by ploughing. Background emissions due to residue mineralization and possible leaching should be allocated to each crop so that the whole simulation period is covered. Considering that in between two crop cycles, mineralization is the main source of N-substrate, we first defined the boundary for this allocation from the sowing up to the fertilization of the following crop. Ploughing allocated to the crop is the one incorporating its own residue. Since the studied rotation induced some 8 months of bare soil between the wheat harvest until the sowing of sugar beet or maize, we then added part of this bare soil period to these two spring crops. This resulted in a final subdivision of the whole rotation period of one year for each crop splitting on February 15. N-inputs were calculated considering the balanced needs and returns of each crop within the rotation. Hence, sugar beet green residues left over in the field after harvest were not accounted for co-product within the ethanol LCA.

Beside our baseline scenario, we tested an "optimized rotation". This "optimized rotation" consisted in two aspects: 1) the incorporation of the first fertilizer application rate together with the seeds below the soil surface, which should make it possible to save inputs of 15 kg N $ha^{-1}$ $yr^{-1}$ (in both sugar beet (ITB, 2002) and maize fields; and 2) the introduction of a cover crop within the rotation. Cover crop between winter wheat and sugar beet reduced the bare soil period by some 85 days (ITB, 2002). In the study, we extend the cover crop, i.e. mustard, from August 15 until January 5, i.e. 144 days, so that it would be killed by the frost without needing to use herbicides. This cover crop shall reduce N-losses during the periods were bare soil occurred in the previous rotation, and return caught nitrogen when incorporating its residues. The effect of reduced leachable nitrate is remarkable for all crops within the rotation (ITB, 2002). Incidences on sugar beet were notably accounted within the LCA regarding reduced N-losses between February 15 and the sowing in April. Costs of cover crop establishment, i.e. sowing and mechanical destruction were divided between sugar beet, winter wheat and maize. Annual P and K fertilizations in sugar beet plots were 70 kg $ha^{-1}$ $P_2O_5$, and 110 kg $ha^{-1}$ $K_2O$. These inputs, as

well as pesticides and cultural operations (except for the introduction of the cover crop) did not vary between the two scenarios. Details of the sugar beet managements and key CERES outputs are given in table 2.

**Table 2:** Key data on the sugar beet production modelled with CERES-EGC (standard deviation); yearly basis splitting on February 15; four simulations over 21 years to grow sugar beet under each climatic year. R25: mean of 6 years; R50: mean of 9 years; R75: mean of 6 years

| | Baseline rotation | | | Optimized rotation | | |
|---|---|---|---|---|---|---|
| | R25 | R50 | R75 | R25 | R50 | R75 |
| N-fertilization /kg N $ha^{-1}$ | 140 | 140 | 140 | 125 | 125 | 125 |
| Mean yields /tFM (17% sugar) | 71 (41%) | 89 (18%) | 91 (16%) | 60 (44%) | 74 (15%) | 75 (17%) |
| Mean $N_2O$ /kg N-$N_2O$ $ha^{-1}$ $yr^{-1}$ | 1.08 (43%) | 1.34 (20%) | 1.23 (13%) | 0.97 (42%) | 1.12 (11%) | 1.16 (12%) |
| Mean $NO_3^-$ /kg N-$NO_3^-$ $ha^{-1}$ $yr^{-1}$ | 5.3 (149%) | 43.7 (81%) | 41.5 (65%) | 8.2 (66%) | 15.4 (43%) | 21.3 (57%) |
| Mean $NO_x$ /g N-$NO_x$ $ha^{-1}$ $yr^{-1}$ | 0.85 (42%) | 0.80 (14%) | 0.81 (18%) | 0.71 (42%) | 0.65 (11%) | 0.66 (14%) |
| Mean $NH_3$ /kg N-$NH_3$ $ha^{-1}$ $yr^{-1}$ | 5.4 (63%) | 4.9 (84%) | 7.3 (38%) | 3.5 (59%) | 3.3 (93%) | 4.9 (33%) |

*Miscanthus*

*Miscanthus x giganteus* is probably a natural hybrid involving *Miscanthus sacchariflorus* (diploid) and *Miscanthus sinensis* (tetraploid) (Lewandowski *et al.,* 2003). Because of its triploidy, *Miscanthus* is sterile, which is an interesting character in order to prevent unexpected spreading. On the counterpart, its implementation has to be done starting from rhizomes or micro-propagated plantlets, which is more expensive than sowing (Venendaal *et al.,* 1997; Lewandowski *et al.,* 2003). Moreover, risk of reduced over-winter survival and weed competition may further increase the cost of planting (Venendaal *et al.,* 1997; Atkinson, 2009; Smeets *et al.,* 2009). The quality of planting material and the establishment phase are therefore critical factors conditioning the future benefit from the cropping system over two decades. We hence paid much attention to detail this phase in the life cycle inventory and tried to optimize management factors taking into account diverse research results on this issue (Lewandowski, 1998; Christian *et al.,* 2009; Atkinson, 2009; DEFRA, 2002). We preferred rhizomes over micro-propagated plantlets since rhizomes are less expensive (DEFRA, 2007; Atkinson, 2009), have a better over-winter survival (Jørgensen, 1995; Lewandowski *et al.,* 2000) especially if irrigated the first year (Lewandowski, 1998), and were found to be more resistant on a longer term (Clifton-Brown *et al.,* 2007).

Weed control is also essential during crop establishment and can be done via chemical or mechanical treatments (Venendaal *et al.,* 1997), whereas it may not be necessary when the stand is three-year old and onwards (Christian and Haase, 2001; Lewandowski and Schmidt, 2006). Neither fungicides nor pesticides are otherwise recommended for *Miscanthus* production, since there are to date no reports of plant diseases significantly limiting the productivity of *Miscanthus*, and it has so far not suffered from any severe pests (Venendaal *et al.,* 1997; Lewandowski and Schmidt, 2006).

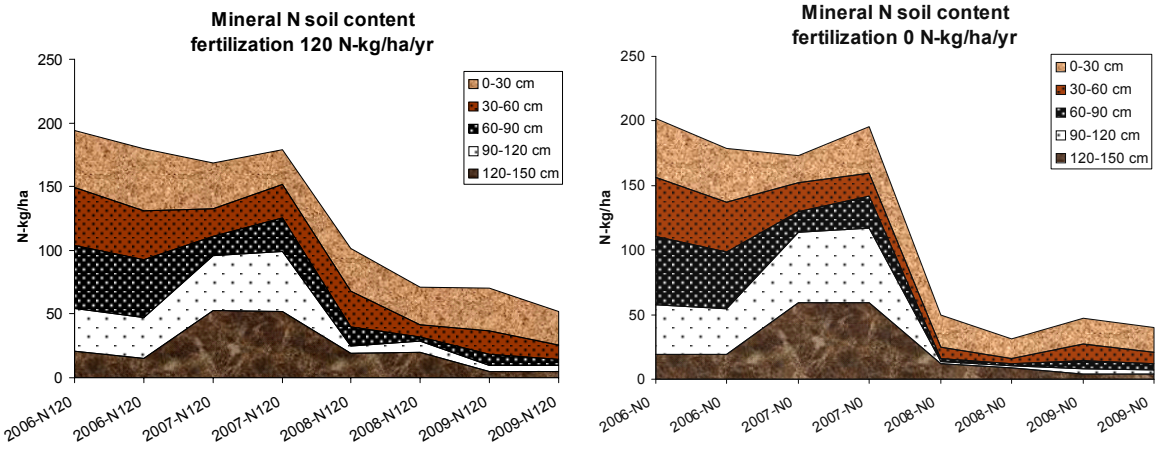
Reported potential life span of a *Miscanthus* productive field varies between 15 and 20 years (Lewandowski *et al.,* 2000; DEFRA, 2007; Miguez *et al.,* 2008; Atkinson, 2009). Yields increase during the two first years or may keep increasing up to the fifth year (Clifton-Brown *et*

*al.,* 2007; Himken *et al.,* 1997; ref). Ceiling yield can then last up to the 10th year before decreasing (Clifton-Brown *et al.,* 2007; Miguez *et al.,* 2008; Angelini *et al.,* 2009). Since *Miscanthus* did not respond to N-fertilization on several sites in Europe (Himken *et al.,* 1997; Christian *et al.,* 2008), it was concluded that N-fertilization is necessary mainly on soils with low N contents (<50 kg N $ha^{-1}$ Lewandowski and Schmidt, 2006) and can be avoided or limited to 50-70 kg N $ha^{-1}$ $yr^{-1}$ on other soils (Lewandowski *et al.,* 2003). This low effect of N-fertilization on yields may be due to combined factors. First, roots can uptake nutrients found in soil layers up to 2-2.5 m depth (Himken *et al.,* 1997; Christian *et al.,* 2006). The total N availability in agricultural soils is hence likely to be large and fertilization would affect N uptake marginally during crop establishment. Then, developed rhizomes act as storage organs accumulating nutrient from autumn until the end of the growing season and providing these nutrients back to shoots at sprouting time in April-May (Beale and Long, 1997; Christian *et al.,* 2006). Between September and March, remobilization of 47% N, 50% P, 30% K and 27% Mg of shoot nutrients toward rhizomes were measured. These amounts notably represented some 100 kg N $ha^{-1}$ (Himken *et al.,* 1997). As long as this internal N cycling remains efficient, N-fertilization might not be profitable. Long-term experiments have shown though that N-fertilization might become more efficient at the end of the *Miscanthus* life span, notably when N soil N stocks have diminished (Clifton-Brown *et al.,* 2007; Christian *et al.,* 2008; Miguez *et al.,* 2008). Bullard and Metcalfe (2001) recommended providing 80-40-60 kg $ha^{-1}$ of N-P-K only the first year. Christian and Haase (2001) recommended, starting from the third year and onwards, fertilizer applications of 40-100 kg N $ha^{-1}$, 10-20 kg P $ha^{-1}$ and 40-100 kg K $ha^{-1}$ depending on the nutrient off-take at harvest. Fertilization should be applied when root development begins, i.e. one month after stem growth has started (Christian *et al.,* 2008).

The date of cut is a crucial factor for bioenergy production from perennial grasses. Indeed, this may affect the biomass yield as well as biomass characteristics. Spring harvest implies lower moisture content and yield decrease due to leaf senescence by up to 30% relative t peak biomass production in fall (Miguez *et al.,* 2008; Richter *et al.,* 2008). Nevertheless, the translocation of nutrients to rhizomes is likely to be essential for maintaining high yields over the production period and to prevent excessive exportation of soil nutrients. During the first two years of establishment, the biomass is usually not exported and residue mulching from aboveground biomass contributes to the recovery of soil nutrients. For the complete production cycle (table 3), we hence only considered spring harvest starting from the third year and onwards. We also encompassed the costs of recovering the previous state of the arable land before implementing the perennial.

As in our field experiment, we considered for the LCA that *Miscanthus* was planted in an agricultural land. Preceding crops in *Miscanthus* plots had been sugar beet-wheat-peas-wheat. In April 2006, just before planting, mineral N soil content over 150 cm averaged roughly 200 kg N $ha^{-1}$. For the purpose of our study, we compared two treatments: with N-fertilization of 120 kg N $ha^{-1}$ (scenario N120), and N-fertilization of 50 kg N $ha^{-1}$ (scenario N50). We used our experimental data to compare with the model outputs. At the study site, fertilization treatments were with 120 kg N $ha^{-1}$ (scenario N120), or unfertilized (scenario N0). First fertilizer application occurred in April 2007 after the residual mineral N stocks had been measured in February (Figures 2 & 3). After the first year, total mineral N had not changed much, but mineral N stocks had slightly increased in the deepest measured layers. It suggests that mineral N migrated to deeper layers while mineralization maintained N levels within the first 30 cm. After the second year, stocks then decreased in both treatments especially in the deeper layers. They diminished more severely in the unfertilized plots. If we consider that, with the initial N-stocks, rhizome growth was not limited in neither of both treatments and that the difference between treatments was provided by the fertilizer (i.e. roughly 60 kg N $ha^{-1}$), it would

mean that the equivalent to half of the fertilizer got lost. These losses may have been even greater if accounting for mineralization inputs.




**Figures 2 & 3**: Evolution of mineral N-content in two fertilized (Fig 2 left) and two unfertilized plots (Fig. 3 right) between 2006 and 2009. On each date (*x* coordinates), points represent the measured stocks in one plot over 5 layers of 30 cm each. (data of the REGIX trial)

The results of our two-year long monitoring of $N_2O$ emissions in automatic chambers in these plots indicate that cumulated emissions between April 2008 and February 2009 amounted to almost 2.5 kg N-$N_2O$ $ha^{-1}$ and 0.7 kg N-$N_2O$ $ha^{-1}$ in the fertilized and unfertilized plots respectively (see Appendix 6). These important fluxes may have occurred because 1) the uptake of N-fertilizer by the plants was reduced, 2) high SOC due to important underground biomass enhanced microbial activity, and 3) according to bulk densities (1.55-1.58 g $cm^{-3}$ over the 0-150 cm soil profile), the inter-rows, where chambers were located, were compacted. Concomitantly, dry matter yields did not vary much between both treatments. In February 2008 ($2^{nd}$ year) yields were roughly 16 tDM $ha^{-1}$ and 12 tDM $ha^{-1}$ in the fertilized and unfertilized plots respectively. In February 2009, yields reached 18 tDM $ha^{-1}$ in both treatments.

With CERES-Sorghum, we simulated annual mean yields over 15 years of 11.18 tDM $ha^{-1}$ and 13:85 tDM $ha^{-1}$in the N50 and N120 treatments, respectively. These yields are in the lower range of published values on late harvest (see Appendix 3). Observed yields of the field trial are moreover expected to increase further. However, modelled yields appeared to be consistent with mean annual yields over a whole 15-year cycle, when accounting for low yields during the years and possible decrease at the end of the cycle (Christian *et al.,* 2008, Miguez *et al.,* 2008). CERES-Sorghum predicted mean annual $N_2O$ emissions of 1.75 kg N-$N_2O$ $ha^{-1}$ and 1.13 kg N-$N_2O$ $ha^{-1}$ in the N120 and N50 *Miscanthus* treatments, respectively. When comparing measured and modelled flux in our field trial, it appears that we probably under-estimated the seasonal emissions of $N_2O$ in the LCAs for the N120 treatment. In the long term, a lowering in $N_2O$ emissions in the N120 field may though be expected if fertilizer is better up-taken at the end of the cycle (Christian *et al.,* 2008).

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**Table 3**: Management of *Miscanthus* over a production cycle of 15 years (number of passes per ha)

| | Nursery 3 years | Establishment 2 years | Ceiling yields 8 years | Lower yields 5 years |
|---|---|---|---|---|
| **Soil preparation** | | | | |
| ploughing | 1 | 1 | | |
| tillage, rotary cultivator | 1 | 1 | - | - |
| tillage, harrowing | 1 | 1 | - | - |
| **Planting** (in spring after frost) | | | | |
| input for the LCA /rhizome production costs | | 1/10 | | |
| density /rhizomes $ha^{-1}$ | 20,000 | 20,000 | - | - |
| potato planter | 1 | 1 | | |
| irrigation | 1 | 1 | - | - |
| **Weed control** | | | | |
| hoeing | 1 | 1 | - | - |
| herbicides | 1 | 2 | 2 | 2 after last harvest |
| **Fertilization** | | | | |
| lime /3t $ha^{-1}$ | 1/5 | 1 | - | - |
| P /40 kg $ha^{-1}$ $yr^{-1}$ | 1 | 1 | - | - |
| K /60 kg $ha^{-1}$ $yr^{-1}$ | 1 | 1 | - | - |
| N50 /kg $ha^{-1}$ $yr^{-1}$ | - | 1 | 8 | 5 |
| N120 /kg $ha^{-1}$ $yr^{-1}$ | - | 1 | 8 | 5 |
| **Harvest** (in spring after nutrient translocation) | | | | |
| hoeing and earthing-up | 1 | - | - | 1 |
| rotary cultivator | 2 | - | - | 2 |
| potato complete harvester | 1 | - | - | 1 |
| potato grading | 1 | - | - | - |
| mulching | 2 | 2 | - | - |
| mowing | 1 | | | |
| baling | | | | |
| loading bales | - | - | 8 | 5 |
| **Yields** | | | | |
| t rhizomes $ha^{-1}$ (≥50 g weight) | 10 | - | | |
| N50 t DM $ha^{-1}$ $yr^{-1}$ | - | - | 11.18 | 11.18 |
| N120 t DM $ha^{-1}$ $yr^{-1}$ | | | 13.85 | 13.85 |

sources: Bullard and Metcalfe, 2001; DEFRA, 2002, 2007; Lewandowski and Heinz, 2003
These studies have notably proven that potato machines were operational for the rhizome harvest, as well as the use of silage maize ones for the harvest of *Miscanthus* biomass. One hectare of rhizome nursery provides 10 t of rhizomes averaging 50 g (out of 40 t $ha^{-1}$ material including soil and mid-sizes rhizomes)

### 3.3 Mechanisms of environmental impact

*Environmental processes*

The term *environmental process* refers to the chain of physical, chemical and biological events in the natural environment that link a particular environmental intervention to a particular impact (Guinée *et al.,* 2002). This first characterization modelling aggregates assigned emissions into mid-point category indicators. These impact categories should be selected as to limit over-lapping our double-counting (Guinée *et al.,* 2002). However, several substances may contribute to diverse impacts (e.g. $NO_x$, $NH_3$ etc.). The chain of cause and effect underlying environmental impacts is complex, so that assessment parameters in LCA are merely indicators providing an indication of potential impacts. Many of them are so-called *stressors,* i.e. indicators referring to the start of the causal chain (Guinée *et al.,* 2002). By modelling

agricultural emissions, we aimed at lowering the uncertainty in the life cycle inventory of these stressors for biofuel LCAs. Indeed, CERES made it possible to predict most of the emissions of these stressors in a non-linear way by accounting for the local climatic condition, soil type and agricultural management. Nonetheless, the characterization from stressors up to mid-point indicators follows the linear models provided by the CML 2001 method, and ones must recall that LCA results can therefore only be considered as potential ones. In the view of double counting especially, they represent the worst scenario from the inventory data set, not taking into account the initial sensitivity of environmental receptors or possible thresholds in many aspects whose stressors are especially not modelled (phosphate leaching, pesticides etc.).

*Direct and indirect emissions*

Natural field emissions, i.e. linked to physico-chemical or microbial reactions occurring in the soil, can be classified as direct and indirect emissions according to the role they may play in the causal chains. Primary emissions of leached nitrate for instance are characterized as direct field emissions, whereas $N_2O$ emissions that may derivate from the same nitrate within an extended temporal and spatial scale are considered as indirect field emissions. This scale changes make it particularly difficult to assess indirect emissions. Mechanisms are poorly characterized and indirect emissions are therefore not modelled by CERES-EGC. We still considered main indirect emissions to account for a more complete part of the causal chains, using the IPCC-factors. We did not consider $CH_4$ emissions from soil, since our soil would not be a net emitter or consumer of $CH_4$ (BIOIS, 2008). Table 4 finally summarized the modelling of direct and indirect emissions linked to the agricultural production in this study compared to their modelling in Ecoinvent database. We also accounted for heavy metals brought to the soil by fertilizers, following the corresponding inventory method of Ecoinvent. Due to the lack of reliable data on the heavy metals brought to the soil in pesticides or exported with the biomass, we decided not to account for these last terms.

**Table 4:** Modelling of field natural emissions in our case study [%range modelled by CERES-EGC in the case study], i.e. with the following inputs: N input=ammonium nitrate, P, K, lime from IPCC guidelines (2006), heavy metals emissions were adjusted from the LCA ode sugar beet production in Ecoinvent database.

| **Emissions /kg ha$^{-1}$ yr$^{-1}$** | **Modelled in this study with CERES-EGC** | **Modelled in Ecoinvent (*In* Nemecek and Käki, 2007)** |
|---|---|---|
| **Direct** | | |
| $N\text{-}N_2O$ | produced via denitrification and nitrification as influenced by soil temperature, soil water content, soil compaction, substrate availability, and microbial potentials. (Hénault *et al.,* 2005; NOE2) [0.8 – 1% of N-input] | 1.25% of N-input (less volatilized N) in kg ha$^{-1}$ (Schmid *et al.,* 2000) |
| $N\text{-}NO_x$ | 0.5% of nitrified N (Rolland *et al., In press)* [0.5-0.6% of N-input] | $NO_x$ = 21% of $N_2O$ |
| $N\text{-}NH_3$ | produced via volatilization (Volt'air sub-model: Génermont and Cellier, 1997; Le cadre, 2004) as influenced by soil energy balance and pH [2.6-5.2% of N-input] | 2% of N-input in kg ha$^{-1}$ (Asman, 1992) |
| $N\text{-}NO_3^-$ | through leaching as influenced by soil type and depth, soil water content, plant uptake [3.7-31.2% of N-input] | through leaching as influenced by soil type and depth, soil water content, plant uptake (Richner *et al.,* 2006) 24% of N-input in kg ha$^{-1}$ for sugar beet |

| Emissions /kg ha$^{-1}$ yr$^{-1}$ | Not-modelled with CERES-EGC | Modelled in Ecoinvent (*In* Nemecek and Käki, 2007) |
|---|---|---|
| **Direct** | | |
| $C\text{-}CO_2$ | from lime application 12% of $C\text{-}CaCO_3$ yr$^{-1}$ with a large uncertainty (50%), 12% being the maximum possible fraction | not available |
| $C\text{-}CH_4$ | not considered | not available |
| $P\text{-}PO_4^{3-}$ | In kg P ha$^{-1}$yr$^{-1}$<br>-through leaching in arable land (to groundwater): 0.07; in pasture or meadows 0.06<br>-through run-off in open arable land (to surface water): 0.175*(1+0.2/80)*kg $P_2O_5$<br>-through erosion to rivers: function of eroded soil amount, P soil content and eroded particle enrichment, fraction of eroded soil that reaches the river (Prasuhn, 2006 *In* Nemecek and Kägi, 2007) Emissions due to erosion in the *Miscanthus* | |
| **Indirect** | | |
| $N\text{-}N_2O$ | IPCC factor: 0.75% of leached $NO_3^-$N<br>IPCC factor: 1% of volatilized $NO_x$-N and NH3-N (IPCC, 2006) | 2.5% of leached $NO_3^-$-N+ 1% of volatilized $NH_3$-N (Schmid *et al.,* 2000) |

## 4. Results of the impact assessment

### 4.1 Biofuels versus fossil fuels

Figure 4 shows the comparative potential environmental impacts of the production and the virtual complete combustion of a MJ ethanol from sugar beet and *Miscanthus* with the fossil counterpart as reference, petrol low sulphur (i.e. gasoline). First, *Miscanthus* performed much better than sugar beet ethanol in all categories, and better than gasoline in almost all categories except eutrophication and terrestrial ecotoxicity. One MJ of combusted *Miscanthus* ethanol notably would save 82-85% greenhouse gases and around 90% of fossil resources compared to gasoline. Important savings were enabled by the biogas-cogeneration at the conversion plant (e.g. -60% of total greenhouse gases per MJ). Sugar beet ethanol would lead to smaller savings 28-42% greenhouse gases and 20-28% of fossil resources compared to gasoline.

The low impacts of *Miscanthus* were mostly due to high ethanol yields per hectare. The conversion of one hectare gave between 6,900 and 8,000 kg ethanol (5,450-6,320 l at 20°C). In comparison, one-hectare sugar beet provided between 2,600 and 3,970 kg ethanol (2,050-3,140l at 20°C). Better ethanol yields with *Miscanthus* were both because lignocellulosic conversion used a greater part of the total feedstock, and because both ethanol and sugar were produced from a sugar beet hectare. Ethanol represented only some 35% of total sugar and ethanol outputs from sugar beet input. Differences between biofuels hence were reduced when comparing on a hectare basis (Fig. 5). Still *Miscanthus* impacts remained by roughly two-third lower across most categories compared to sugar beet. Since it is a perennial, *Miscanthus* made it possible to save much energy from annual cultural operations. Remaining recurrent inputs were N-fertilizers and herbicides that contributed to a great part to environmental impacts directly linked to $N_2O$ and $NO_3^-$ emissions, such as global warming and eutrophication. Moreover, due to the long distance to reach the future lignocellulosic ethanol plant, transports influenced more the overall balance compared to sugar beet (Fig. 6 & 7). *Miscanthus* from the lower fertilization treatment (N50) had lower impacts that the N120 treatment on both MJ and ha bases, despite its lower yields, indicating that environmental impact savings largely compensated for yield losses. Lower yields also led to lower environmental costs from transports (15% less in each impact category for N50 compared to N120), since part of fuel consumption (1/3) depends on the load. As annual transports accounted for an important part of the impacts (Fig. 6), this reduction accentuated differences between both *Miscanthus* treatments.

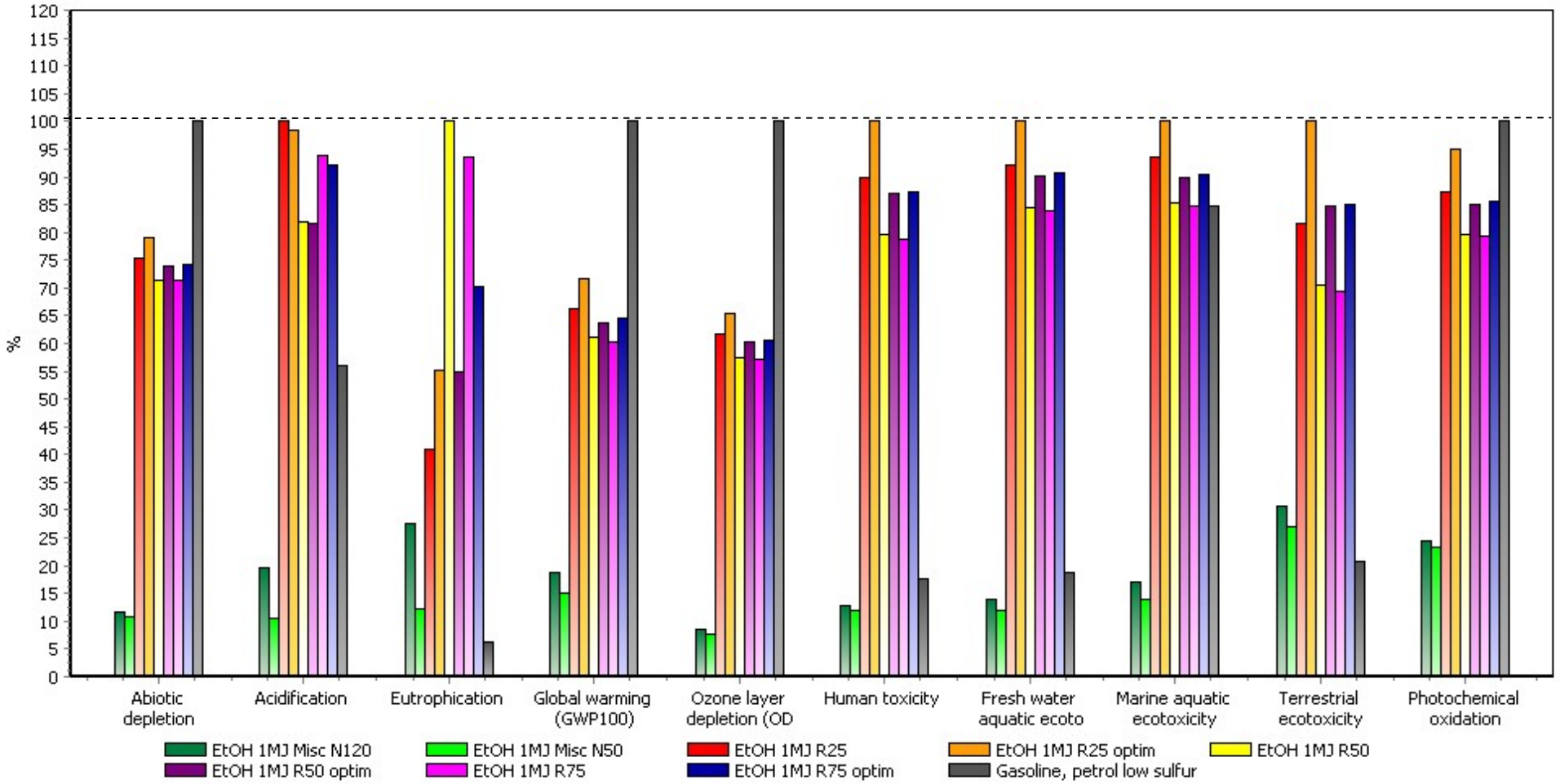



**Figure 4:** Comparison of the impact assessment with CML2 Baseline method on a MJ basis. Ethanol from sugar beet are in red, orange, yellow and pink (EtOH 1MJ R25-50-75, with optimized scenarios), *Miscanthus* two fertilizer levels are in green, and gasoline at the refinery doors in grey.

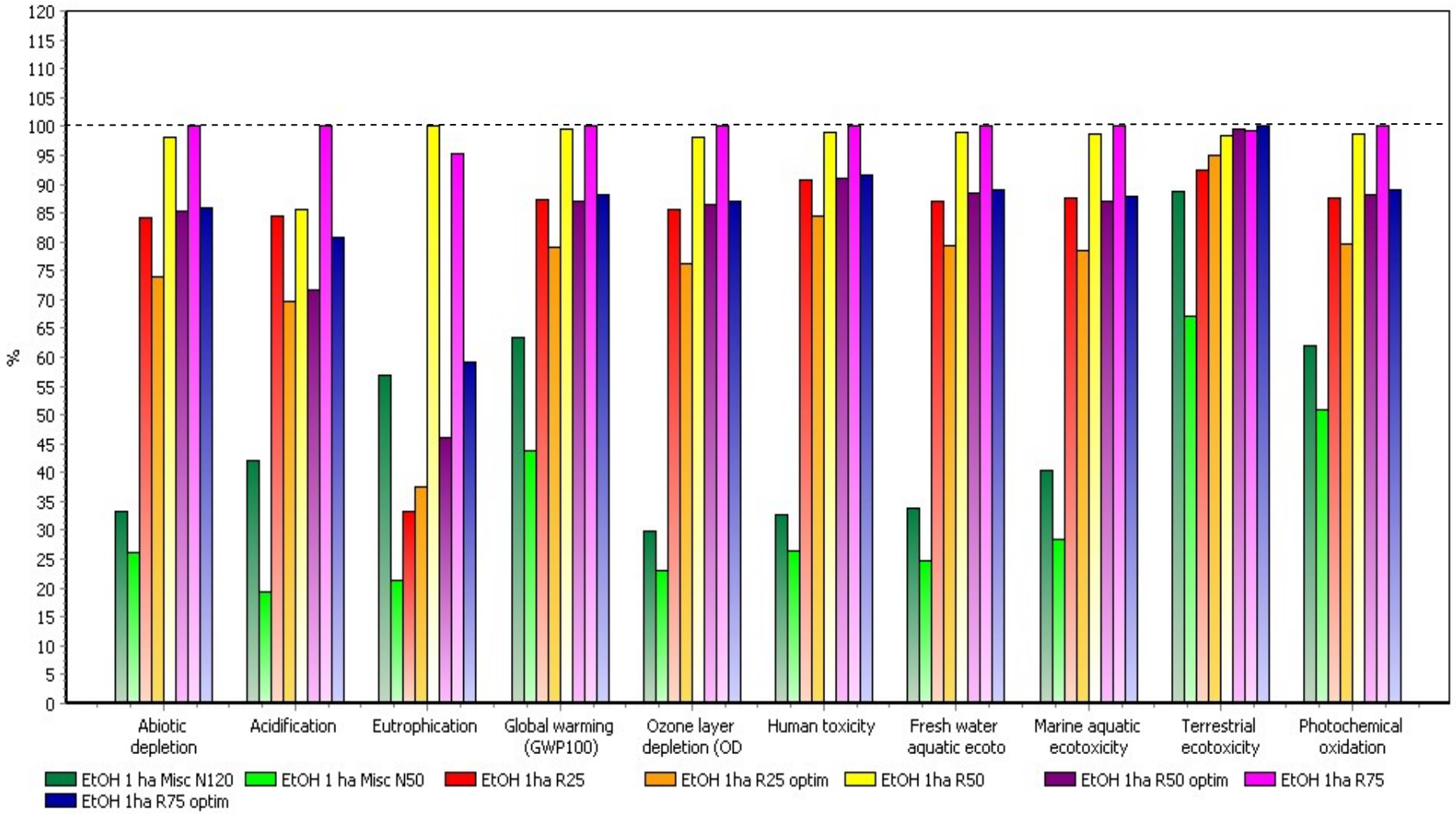



**Figure 5:** Comparison of the impact assessment with CML2 Baseline method on one-ha basis. Ethanol from sugar beet are in red, orange, yellow and pink (EtOH 1MJ R25-50-75, with optimized scenarios), *Miscanthus* two fertilizer levels are in green.

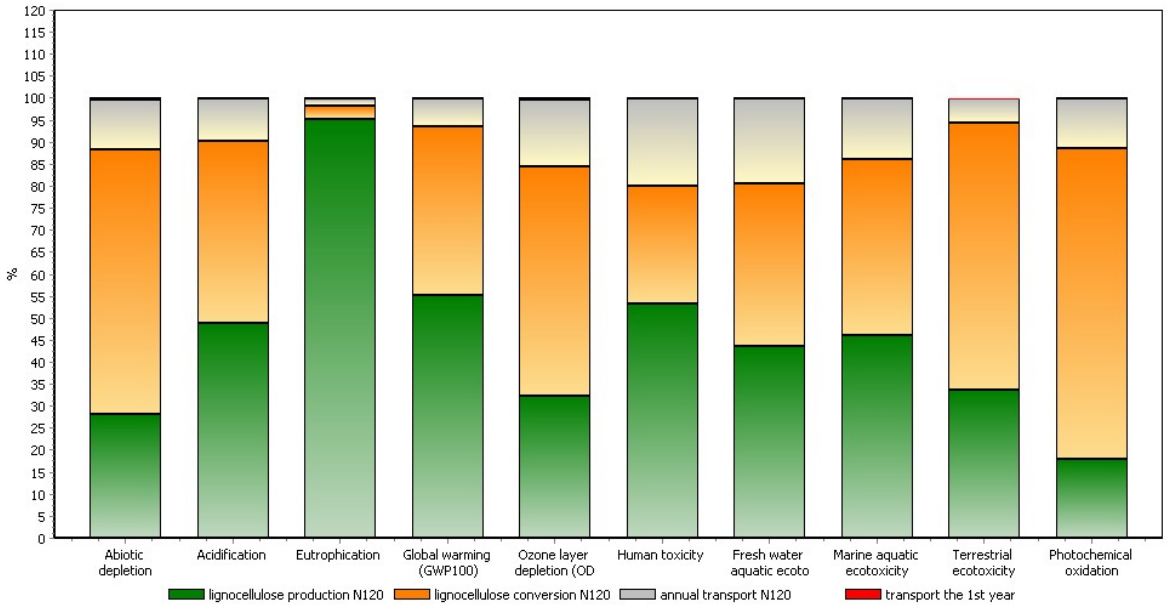



**Figure 6:** Process contributions to environmental impacts of ethanol from 1 ha of *Miscanthus* .

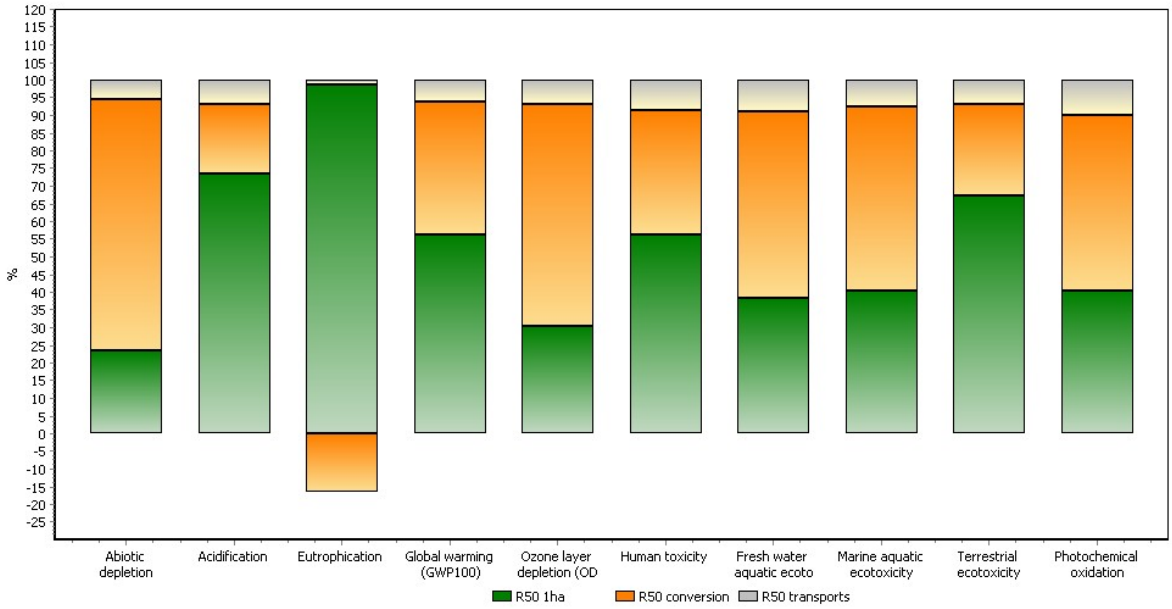



**Figure 7:** Process contributions to environmental impacts of ethanol from 1 ha of sugar beet.

Sugar beet ethanol had lower impacts than gasoline for abiotic depletion, eutrophication, photochemical oxidation and ozone layer depletion. Fig. 7 illustrates the contribution of the main phases to total impact indicators per category. The case of sugar beet production during a R50 year (median rainfall pattern) emphasized, like for the other scenarios (figures not shown), that high environmental impacts on acidification, eutrophication or global warming were mostly due to N-losses in the field. Despite consequent nitrate savings due to the substitution of maize silage by co-produced over-pressed pulps, the impact on eutrophication during this mean climatic year was still very high.

### 4.2 Impact of local production factors

Across impact categories, smallest to largest variations on mid-point indicators comparing the three scenarios on rainfall patterns (R25, R50, R75) ranged from 1% to 65% of savings, with an average difference approximating 5-15% of savings. LCA results were sensitive to changes in climatic conditions during biomass growth. As detailed in tables 2 & 4, climatic conditions influenced fresh matter yields, but also $NO_3^-$, $N_2O$, and $NH_3$ emissions that contribute to several impact indicators.

Drier years (R25) were more radically distinct from R50 and R75 scenarios. On a hectare basis, potential impacts of sugar beet ethanol under R25 climatic pattern were lower than for other both climatic patterns, whatever category indicator was observed. Regarding modelled data, it coincided with low nitrate leaching and following indirect emissions that led to reduced notably eutrophication, acidification, and global warming impacts. However, drier years (R25) were also characterized by lower yields, hence allowing for less ethanol production per hectare. Ethanol yields (kg) were 20% lower in average for the R25 scenario compared to R50 and R75 scenarios, these last ones showing similar ethanol yields per hectare (±2%). On a MJ basis, R25 ethanol had therefore more severe environmental impacts, except for eutrophication, which was the most sensitive to the change in rainfall. Between R50 and R75 scenarios, eutrophication indicators greatly varied, which emphasized the sensitivity of this impact category to climatic conditions. On both hectare and MJ bases, impact indicators of R50-R75 and R50 optim-R75 optim followed the same trends, except for acidification and eutrophication. For these two last impact categories, mid-point indicators for R50 scenario indicated lower impacts.

When comparing baseline and optimized cropping systems, it was not possible to draw consistent conclusions based on both hectare and MJ units because of the trade-off between yield levels and environmental impacts. On average, yields were 16% lower with optimized scenarios (R25 optim, R50 optim, R75 optim) because of the reduced fertilizer inputs. Indeed, effective benefit from fertilizer incorporation at sowing could not be modelled, since CERES did not make it possible to distribute fertilizer inputs into sub-divided soil layers where they would be provisionally stored because of less exposed to volatilization notably. This incorporation should make it possible to save some 15 N-kg $ha^{-1}yr^{-1}$ given our type of soil and the preceding crop residues (ITB, 2002). Resulting modelled effect was finally related to the savings in producing the fertilizer, to reduced sugar beet yields and emissions due to less N-input. On the contrary, the impact of the introduction of the cover crop was better modelled, since nitrate leaching was reduced by more than 50% in R50 and R75 compared to initial crop rotation (table 2). Differences in LCA results between baseline and optimized rotations were hence the largest for the eutrophication indicator (Fig. 3 & 4). This category was also the only one in which optimized rotations performed better than baseline ones on both hectare and MJ bases, i.e. saved impacts compensated for reduced yields. Comparing performances across impacts of the three optimized scenarios, R50 optim led to less global impacts when considering both hectare and MJ bases.

### 4.3 Sensitivity to global warming equivalent factors

Since global warming impact of biofuel chains is essential for decision-making purpose, we looked closer at the sensitivity of our assessments to the chosen equivalent factors. Used factors in LCA methodology are those from the IPCC that have been re-evaluated in each of the four Assessment Reports. In the third one (TAR in 2001) equivalent factors on a 100-year perspective were 23 kg $CO_{2\,eq}$ per kg $CH_4$ and 296 kg $CO_{2\,eq}$ per kg $N_2O$. In the fourth one (2007) equivalent factors on a 100-year perspective were 25 kg $CO_{2\,eq}$ per kg $CH_4$ and 298 kg $CO_{2\,eq}$ per kg $N_2O$. Despite the common use of the 100-year perspective, some argue that cost assessment should be done on a shorter time perspective since people are more concerned by immediate consequences. We therefore also looked at the 20-year equivalent factors (Fig. 8). We completed this sensitivity analysis by half-reducing the $CO_2$ emission factor of applied lime, as suggested in the IPCC Guidelines (2006) in comment to the maximum 12%-factor.

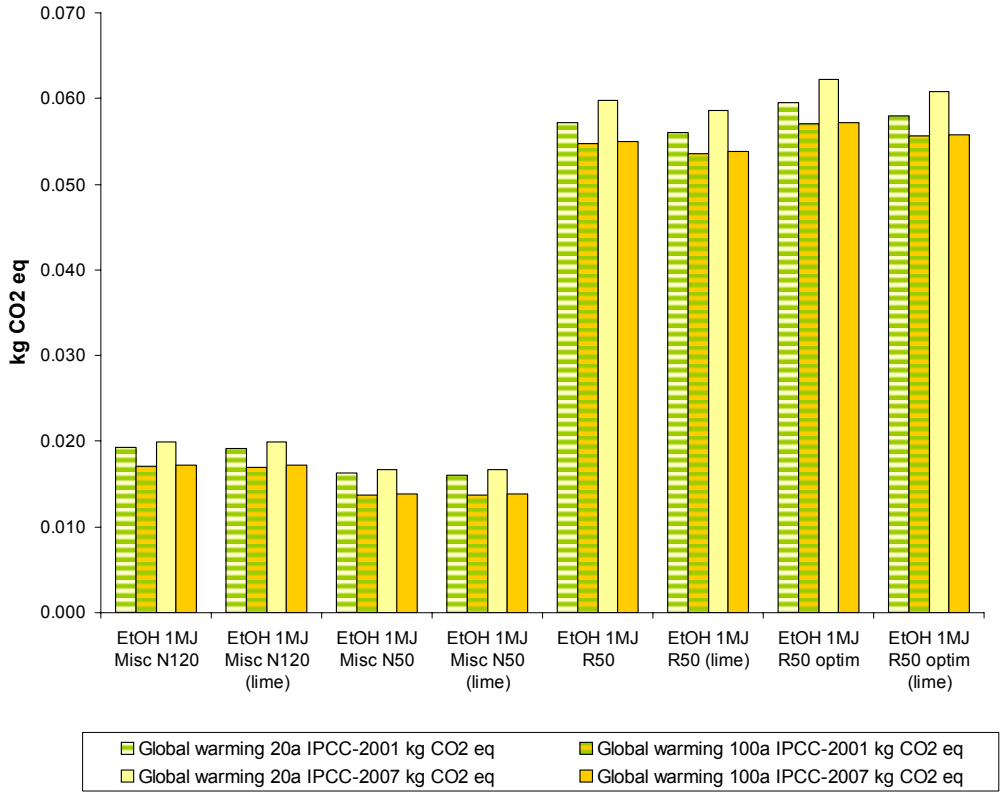



**Figure 8:** Sensitivity of global warming impacts to changes in the chosen equivalent factor.

There were no discernable differences between the 2001 and 2007 equivalent factors in our LCAs, on a 100-year perspective (other figures not shown). However, the effect was more visible on the 20-year perspective (Figure 8). With this example (R50 scenarios), the largest difference in resulting indicator was between the 20-year (2007) and 100-year (2001) perspectives, 0.062 and 0.057 kg $CO_{2\,eq}$ per MJ, respectively. This difference would correspond to either 32.6% or 36.8% greenhouse gas savings compared to gasoline, respectively. The assessment results were little sensitive to changes in the emission factor from lime. It is particularly true in the case of *Miscanthus* because the quantity of applied lime was three times lower than on sugar beet. Sugar beet development is sensitive to pH, and to risk of surface crust in the soil of Estrées-Mons, so that care must be taken to adapt well the lime dose. In our study,

we considered minimum doses for prevention, because lime has been regularly applied and hence risks are minimal. However, in cases with higher risks or insufficient historical record, liming rates could easily double and the consequences on global warming impact become more important.

## 5. Discussion

Our local LCA results showed that *Miscanthus* ethanol had overall much lower potential environmental impacts than gasoline or sugar beet ethanol. The low impacts of the *Miscanthus* cropping system are notably due its being a perennial, thereby saving annual cultural operations such as tillage and sowing. Moreover, pollutions linked to conversion processes were limited thanks to the biogas co-produced, as modelled for the lignocellulosic plant design used in the Simapro-Ecoinvent database. The local LCAs we obtained give a complete picture of total potential impact of *Miscanthus* ethanol with two fertilizer treatments. However, these results might be hardly comparable to other *Miscanthus* LCAs that would not assess the same type of conversion plant. In our study case, the type of conversion plant, the distance to this plant, or the production site of rhizomes were highly uncertain hypotheses. Moreover, attention must also be paid since the CERES-sorghum model could not account for remobilization of nutrients from the rhizome at rooting (BG?). Improvement in modelling *Miscanthus* growth and environmental impacts would be needed to reduce the LCA uncertainty and test other scenario such as zero N-fertilization treatments, which are common practice in the UK. Finally, the REGIX field trial should also provide more references on long-term perspectives to better account for production factors and environmental impacts at the end of the crop cycle and beyond.

Greenhouse gas savings of sugar beet ethanol compared to gasoline ranged between 28% and 42% with a mean global warming indicator across scenarios of 0.0578 kg $CO_{2\,eq}$ per MJ, i.e. 35.7% of greenhouse gas savings compared to 1 MJ. This indicator is greater than that of the ADEME/DIREM study (2002), i.e. 0.0336 kg $CO_{2\,eq}$ but is very close to 0.0575 kg $CO_{2\,eq}$ per MJ value published by JRC/EUCAR/CONCAWE (2008), where sugar beet pulps also were used for animal feed. In the latter study, it was also showed that these greenhouse gases could be halved if the pulps were used to produce heat. However, this use of pulps is still rather marginal at our study site. These 35.7% savings are critical because they are closed to the limit of 35%, prerequisite of the European Directive (CEC, 2008), and lower than the 50% limit to be enforced in 2010. In the EU Directive, biofuel chain performances were calculated with energy allocation for co-product handling, which might allow for more savings in the sugar beet case. Moreover, this indicator could sensibly vary according to which time-scale is chosen for the equivalent factor.

Local production conditions influenced all categories of the final potential impacts of the biofuels we tested. These results emphasized that an average LCA for a given biofuel chain can only be indicative. Under diverse conditions for feedstock production, the biofuel may affect the environment more or less severely, within a span that could overlap thresholds such as around the 35% greenhouse gas savings in the case of sugar beet for instance. The magnitude of these effects was particularly important for eutrophication and acidification that are strongly dependent on field emissions. Analysis on both hectare and MJ bases showed important trade-offs between low impact scenarios and low ethanol yields, which made it more complex to define optimal choices. Climatic years with extreme rainfall patterns (the R25 and R50 scenarios) presented the worst potential impacts. R25 led to more severe impacts on a MJ basis due to poor yields per hectare. R75 tended to increase several impacts such as eutrophication due to more nitrate leaching, or acidification and global warming. This effect may be related to

N cycling processes and microbial activities that are generally enhanced in wet conditions. Overall, the optimized scenarios had lower environmental impacts than the other scenarios, although lower yields counteracted this positive effect. In the case of eutrophication in particular, optimized rotations would be very beneficial. If the risk of drought or heavy rainfalls were higher, optimized rotations would decrease the risks of high impacts, the indicators of these scenarios varying within a narrower range than those of baseline scenarios. The range of impact assessment results on local conditions allow for a better precision on the probability of the impacts around site-specific baseline results. It should help as decision-making tool for a given scenario and prospective ones, such as optimized management or changes in climatic conditions for instance. Improvement of these local LCAs would encompass a more detailed characterization of these local conditions, such as rainfall patterns that take into account the time of rainfall compared to the crop cycles and fertilization dates.

The results on 'ozone depletion' impact category leaves little scope for discussion here, since the method we used to characterize this impact (see Table 1) does not account for any contribution of $N_2O$ to stratospheric ozone depletion, while it is recognized that $N_2O$ plays an important part in ozone chemistry (IPCC, 2007). This should be corrected to reflect better the potential impacts of the biofuel chains we tested here. This also applies to the ecotoxicity impacts, since we could not account for the complete balance of heavy metals in our agroecosystem. It stresses that despite the holistic nature of LCA, many environmental mechanisms are still not fully understood, nor clearly assessed with the current methodology.

Ethanol yields per hectare of sugar beet are much lower than for *Miscanthus*, whereas dry matter yields per hectare are quite similar. Indeed, sugar represents more than half of the refinery outputs. If only ethanol was produced from the total fresh matter harvested from the sugar beet fields, ethanol yields would be nearly twice higher (some 5.78 t $ha^{-1}$ ADEME/DIREM, 2002). Impact of sugar beet ethanol would be then less severe per MJ of ethanol produced. The biofuel chain assessment assumed that the biomass production aimed at producing the ethanol. In the LCA methodology, the whole life cycle inventory is built towards a functional unit fulfilled by a product of interest that justifies the costs of production and pollution. In the sugar beet case, given current operating refinery, sugar also is a product of interest. Up to now, sugar has been notably sold by producers at higher prices than ethanol. In current LCAs of sugar beet ethanol, as in our study, the costs of the conversion of sugar beet fresh matter into sugar and ethanol were allocated to each product, but not those of the upstream agricultural costs. In the case of long-date sugar beet production, where the biofuel has not led to land use change, we could assess the opportunity costs of both chains by considering shared responsibilities of both products for the field emissions, as it is already done for the conversion costs. A complete accounting for the carbon balance from the field to the refinery doors would be necessary to assess respective impacts of sugar and ethanol and to further help orientating development choices regarding the local needs and constraints. This carbon balance would also indicate the carbon that may be stored in the soil as a result of crop photosynthesis. This storage could be particularly important in the case of perennials such as *Miscanthus*, and contribute to lower significantly the impact of *Miscanthus*-based ethanol on climate change. Assessing this effect would also require an estimate of the turnover time of this carbon pool in the soil.

## 6. Conclusion

Local LCAs provided precise assessments of potential environmental impacts of ethanol from sugar beet and *Miscanthus* produced at Estrées-Mons in the northern French Picardie region. LCA results for sugar beet were comparable to those found in literature, while the results on the lignocellulosic ethanol from *Miscanthus* chain were novel, barring such comparison. Model development would be furthermore needed to improve the reliability of these *Miscanthus* LCAs. Effects of local soil type, climatic conditions and management practices were visible for all impact categories and on both hectare and MJ bases. They emphasized the need to take into consideration these local factors when making a decision at the local level to implement a particular biofuel chain. These local LCAs should be completed with a local assessment of environmental impacts, encompassing a qualitative assessment of production impacts on land as resource, as well as social and economic criteria.

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# Conclusions/perspectives

The hypothesis (thèse) underlying this work was that local characteristics have a major influence in the environmental impacts and performance of biofuel chains, via the agricultural feedstock production step. Thus, reducing the uncertainty on the greenhouse gas balance of biofuels implied a better estimation of the effect of these characteristics on the field emissions of GHG. The methodology we set up to demonstrate and tackle these effects also contributed to a broader goal of improving our knowledge and modelling of ecosystem processes and controls driving the relationships between crop management and environmental impacts.

We first set up field experiments to collect complete sets of data with $N_2O$ and $CO_2$ fluxes and the main environmental drivers in sugar beet and *Miscanthus* plots during two years. Automatic chambers allowed us to measure fluxes with a high frequency and over long periods, which was mandatory to fully take into account the temporal variability of fluxes and test the model's ability to reproduce their kinetics. In sugar plots, we observed that greater soil compaction led to higher $N_2O$ emissions, and the impact of soil compaction and climate conditions interacting prevailed over the nitrogen fertilizer doses. On the contrary, $CO_2$ fluxes were higher in the uncompacted plots. This is in accordance with the occurrence of anaerobic sites that is favoured by soil compaction through a reduction of total porosity. Anaerobic conditions enhance both $N_2O$ emissions by denitrification and the proportion of $N_2O$ emitted by nitrification over total nitrification end-products, whereas the nitrification rate itself decreases with decreasing oxygen partial pressure. In the *Miscanthus* plots, we observed high fluxes in the fertilized plots. These important fluxes may have occurred because 1) the uptake of N-fertilizer by the plants was reduced, 2) high SOC due to underground biomass and high N cycling enhanced microbial activity, and 3) according to bulk densities, the inter-rows, where chambers were located, were compacted.

To limit $N_2O$ emissions, N-fertilizer inputs should be reduced as much as possible. It is particularly true for *Miscanthus*, but lowering of application rates would also be feasible for sugar beet by incorporating fertilizer at sowing for instance. Cover crops would also reduce nitrate leaching and subsequent indirect $N_2O$ emissions. When assessing the trade-off between reduced $CO_2$ emissions through reduced tillage and enhanced $N_2O$ emissions, sugar beet does not appear as a good candidate for reduced tillage. Indeed, sugar beet harvest leads to high soil compaction, while a high quantity of residue with a lot of nitrogen in leaves and sugar in left over beet chips constitute a pool for intense mineralization. This combination would be very likely to produce high $N_2O$ emissions. Moreover, part of this $N_2O$ could be produced at deeper layers and bubble out to the soil surface during freezing-thawing events. Emissions would be especially important if no winter crop is sown right after sugar beet harvest. Hence, no-tilled cropping systems including sugar beet should be rather limited, either to cold areas where mineralization rate might be very low and if residue mulching reduces the risk of freezing events, or to optimized crop rotations, within which cover crops reduce the periods of bare soils notably between winter wheat and sugar beet. In terms of soil carbon storage, *Miscanthus* should be a more promising option.

NOE2 made it possible to reproduce the higher $N_2O$ emission fluxes in the sugar beet compacted plots and to capture part of their dynamics over the cropping cycle. The model,

however, still underestimated total emissions. The main drivers that enhance $N_2O$ emissions in compacted zones are known and can be modelled. The accuracy in predictive modelling hence depends on the accuracy in modelling these drivers. In particular, the discretization in fine soil layers in NOE2 improved nitrification modelling by mimicking a finer vertical distribution of N-fertilizer. Modelling results brought to light that a precise modelling of water and nitrate contents across the vertical soil profile is a prerequisite for any increase in model accuracy. Despite the introduction of dynamic functions for the ratios of end-products (*z, r*), further research work is needed to model the impact of diffusion constraints and dry-wet cycles on these ratios. The range of highest emission peaks during summer could not be rendered nor even approached. Tests revealed that several factors had to concur in order to produce such fluxes, including possible microbial adaptation mechanisms that are not modelled yet. Further research work is needed to elicit these mechanisms at both field and laboratory scales.

Among the factors that might explain high emissions during dry-wet cycles, Michaelis-Menten constants appeared to be determining since N-substrates at that time were low. These constants' values vary widely across literature (by a 100-fold factor), probably because diffusion constraints across the soil matrix largely differ between field and laboratory experiments. Even if intact soil cores are used, homogeneous substrate concentrations as control variable in laboratory experiments hardly reveal the effect of same concentrations measured in the field. Moreover, Michaelis-Menten constants might in fact vary along the crop cycles due to varying diffusion constraints during dry-wet cycles notably. This example emphasizes the difficulty to define robust parameters, while it is a prerequisite to upscale models.

Modelling outputs were used to produce local Life Cycle Assessments (LCAs) of two biofuel chains: 1$^{st}$ generation ethanol from sugar beet and 2$^{nd}$ generation lignocellulosic ethanol from *Miscanthus*. These LCAs made it possible to test the sensitivity of LCA results to change in local conditions of the biomass growing. Effects of soil type and rainfalls, as well as those of tested management practices could be observed for each impact category with varying amplitude. Combining agroecosystem model and LCA made it possible to provide an assessment of potential environmental impacts considering site-specific factors, allowing for more precise results than based on national or international average on production performances, or emissions factors. A further step will be to complete these local LCAs with an assessment of the impacts on soil as a resource, i.e. the change in soil organic matter content, soil erosion, and land use competition for instance. This complementary indicator would be needed to address the issue of land availability, which is critically rising with increasing needs for biomass resources. Since indirect impact on land use through trade-offs between forest and arable lands or within arable land uses for instance is hardly assessable within the LCA frame, economic and social factors should also be considered and priorities defined within a broader Environmental Impact Assessment for decision-making purposes at a local (bioenergy project) scale.

Biofuels, more than cars themselves, are nowadays the target of debates on anthropogenic greenhouse gas emissions. If the picture is not true in absolute terms since most of biofuels have lower greenhouse gas balances than their fossil fuel counterparts, emphasize has been put on the need to identify the best biofuel candidates in terms of environmental balance. The price of biofuels in terms of land use should also raise awareness on the real value of fossil resources that have been depleted because they could be cheaply exploited. Evaluating the land areas needed to drive our cars on biofuels stresses that we must change our consumption behaviours because fossil resources cannot be re-grown.

Beyond biofuels, it also reminds us that intensive agriculture has led to huge spreading of chemicals in the environment for a few decades now. As $N_2O$ emissions embody the long-term effects of such spreading, the drawback of agricultural inputs cannot be more easily avoided than $N_2O$ emissions themselves. Researchers in agricultural sciences are hence facing a triple problem. First, the mechanisms of agricultural pollutions are not all identified and characterized (nitrogen cascade, pesticides, GMO etc.), so that models cannot predict the global environmental impact of agriculture. Second, demand for agriculture products is fast growing, so that alternative production systems with low input should still produce at least as much as intensive ones. Third, technology breakthroughs in the field of living organisms raise severe ethical questions, so that a radical bio-technological change is not much likely to reverse the trend. In any case, new cropping systems that aim to simplify and generalize the use of noxious chemicals cannot be part of the solution. This context urges for researches and answers. Environmental assessment is no longer an option. It is a need, and is becoming a prerequisite for any development decision.

## Research perspectives

The determinism of field emissions is subject to many interrogations. In particular, research is needed to better apprehend and simulate the functions that define the ratios of emitted $N_2O$ over total end-products. These researches shall imply both field experiments and laboratory work. Indeed, the ratios may depend on specific changes in microbial activities combined with changes in diffusion constraint in soil. These results should be critical, since wide variation in these ratios could lead to very high emissions, although the factors determining the initial emission rates are not optimum. This issue emphasizes that reducing $N_2O$ emissions cannot only be achieved by limiting the factors that enhance the reaction rates. It is for instance the case with soil compaction. Soil compaction may lead, within some strictly anaerobic microsites to denitrification up to $N_2O$ reduction, while in some other microsites nitrification could be very restricted but $N_2O$ emissions by nitrification increased.

For life cycle assessments, more research is needed on long-term emissions; concerning indirect $N_2O$ emissions, and residence time of soil carbon. Local life cycle assessments, coupled with agroecosystem models, should allow us to encompass the impacts of biomass production on land as a resource. The assessment would account for $CO_2$ fixation through photosynthesis and return to soil of organic carbon, while considering the potential quantity and stability of stored carbon. To conclude with global warming impact assessment, more understanding and references would be needed on the emissions of $CO_2$ following lime application in the field.

In terms of model development, we pointed out that consequent improvement in predicting the impact of agricultural practices can be achieved by modelling the soil physical behaviour, biological processes and environmental drivers within finer soil layers across the vertical profile. This finer modelling would probably require running the agroecosystem model on a finer time scale (hour time step) in order to avoid that fine soil layer saturate too quickly. On this finer scale, it should be possible to mimic better the distribution of substrates available for microorganisms, as well as the water content and aerobic status of micro pores. A better simulation of ecological drivers would be particularly beneficial within the first centimetres of the soil surface, where soil temperature and water content may more quickly. Such model development would also make it possible to better account for the effect of specific management practices, like fertilizer incorporation or mulch effect due to organic fertilizer. Finally, model development for *Miscanthus* would consist in implementing a “rhizome-routine” to better simulate *Miscanthus* N cycling, biomass yields, and environmental impacts.

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# Appendix 1

# Biofuels, greenhouse gases and climate change

## 4 Political and economic frameworks

### 4.1 Climate change and greenhouse gas[48] 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 reference Brazilian Pró-álcool Programme launched in 1975, it was not until the awareness on 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 were expressed within the frame 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 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 gases concentrations. Today, anthropogenic greenhouse gas emissions are overwhelming and the order was reversed so that greenhouse gases are driving temperature increases. 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[49] 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 per 1.5 to 2% per year, doubled-$CO_2$ will be reached in approximately the year 2050.

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[48] By default examining greenhouse gas emissions include "six" gases: $CO_2$, $CH_4$, $N_2O$, $SF_6$, PFCs and FCs.
[49] Pre-industrial concentrations/in 2005: $CO_2$ (280/379 ppm); $N_2O$ (270/319 ppb); $CH_4$ (715/1,774 ppb), IPCC, 2007.

Encompassing the whole range of the six IPCC emissions scenarios[50] 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 sea level to change as feedbacks can lead to non-linear responses. Nevertheless, "it is almost inconceivable that under 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 of 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[51]. 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 $CO_2$ peak at 475 ppm in 2100 before it should slowly decline thereafter and 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 rise in 1990 emissions alone produce 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 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. Emissions 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%), or 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

[50] SRES: Special Report on Emissions Scenarios IPCC

[51] Sea ice melting does not directly cause sea rising 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.

2003–2004 period, the $CO_2$ growth of 50% in China accounts for more than half of the worldwide $CO_2$ increase.

The Kaya Idendity[52] model (Fig. 7) gives some clues to understand the energy-related $CO_2$ emissions by using four factors. Emissions intensity or "carbon intensity" is 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 coal industry, was more than offset by substantial GDP growth. Likewise, the U.S. decline in carbon intensity (17%) was offset by increases in population and GDP, giving a significant growth of greenhouse gas emission in the U.S of 13% over the 1990 to 2002 period.

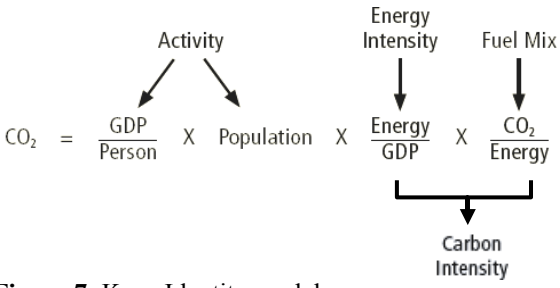



**Figure 7**: Kaya Identity model

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 emissions levels. This is the case in countries as diverse as the US, India, Australia or Iran (Baumert *et al.*, 2005). Given the diversity of large emitting countries, it is simply not possible to address adequately 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 emissions "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). As well, 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[53] 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 managed to stabilize transport emissions in recent years. Nevertheless, EU-15 greenhouse gas emissions from transport are still expected to increase

[52] 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.

[53] 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.

further 35% above 1990 levels by 2010 if only existing policies and measures are used (EEA, 2006a). Transport sector represents the most significant climate policy challenge at two levels. First, transports contribute the lion's share of 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 a 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 of the average greenhouse gas emissions per capita in $CO_{2e}$ 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 to 2003 and were 5% below 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 1990 level by 2010, meaning that EU-15 Kyoto commitment of 8% emissions 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 these 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 to 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).

### 4.2 Biofuel-related policies

*European policies*

In Europe, several directives have been released notably in 1997, the Energy White Paper and Action Plan, 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), or 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 was binding and 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 to 1997, including a triple of biomass use. In 2003, the total amount of used renewables averaged only 6% of the EU gross inland energy consumption (EU DG-TREN, 2006a), about two-third of this contribution coning 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 globally achieved (Fagernäs *et al.*, 2006). In 2005, the indicative target[54] 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 sharing within the EU-25 (EU DG-TREN, 2006b). This share was better than in 2003 (0.6%) but if this trend goes on, the 2010 target of 5.75% share will not be achieved, prevision indicates a 4.2% share by 2010 (CEC, 2008).

Considering the urge for a drastic reduction of greenhouse gases from the transport and the still very low incorporation of biofuels, European Union has decided to put into force a new directive that fixes mandatory targets. During the European Union Summit in March 2007, it was agreed that 20% of the global energy consumption of the European Union must 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 shall replace the former 2001/77/CE and 2003/30/EC directives after December 31, 2011. As a further incentive for investors, the Directive indicates that the share contribution by 2$^{nd}$ 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 productions, so that their consumptions can be taken into account to fulfil the binding target and give way to financial supports (Article 15), although no sanction has been planned for the non-compliance to these criteria or the non-fulfilment of both targets:

- a minimum saving of 35% of greenhouse gas emissions compared to the substitute fossil fuels,
- biomass feedstock must not be produced on soils within ecosystems considering 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 set out environmental sustainability criteria have been fulfilled. In particular, the Directive stipulates the method to calculate the greenhouse gas emissions throughout the production chain and fixes that emission reduction due to the co-products shall be handled either by system extension in the case of co-generated electricity from agricultural, or by energy allocation[55] in all other cases. It also gives minima of emission reductions for each biofuel chain (Annex VII) that shall serve as reference.

[54] 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 based on the energy content. (about 3% and 8.6% for ethanol; 2.2% and 6.4% for biodiesel when calculated on a volume basis).

[55] Based on the Low Calorific Values

However, the method does not explicit 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 criteria respect, and biofuels are not included in the guarantee of origin system that shall 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 contrasted as 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, greenhouse gas emission calculation method and certification control.

*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 programs 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[56], which has been banned[57] in a growing numbers of US States due to its toxicity in high blends (formerly 15% in California for instance), has also contributed to underpin ethanol blends. However, biomass currently accounts for merely 4% of total energy consumption (BRDI, 2006), 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,

[56] Methyl Tertiary Butyl Ether: fossil oxigenate additive to gasoline.
[57] Contamination of groundwater by MTBE due to leaking tanks is especially severe in the US. Despite the decision to phase it out, quantities of used MTBE in the US have not decreased due to its technical advantages that actually help to produce a cleaner burning gasoline (www.acfa.org).

the American Competitiveness Initiative and the Advanced Energy Initiative (2006), on providing an aggressive strategy for tackling long-term energy challenges (US DOE, 2006b). It has shown ambitious goals in terms of energy efficiency and diversification, i.e. the increase of domestic production of conventional fuels as well as the development of a 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 bioproducts 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 emphasize the role of technology development and innovations. The President’s Advanced Energy Initiative provided for 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 commercialization.

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 former 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 blends 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 biofuels supply and were massively followed by the member states as illustrated by the Governors’ Ethanol Coalition that include 32 member states upon 50, as well as international representatives from Brazil, Canada, Mexico, Sweden, and Thailand (US DOE, 2006a).

Although biofuels were initially thought to contribute to lower US energy dependence on imports, the US still imported about 1.7 billion litres of ethanol in 2007[58]. Even if all maize

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

grain grown in the US were converted to 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 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 mean for the US to lower drastically its greenhouse gas emissions, while it is besides not particularly willing to commit itself within international emission reduction targets plan.

*Chinese policies*

As Chinese economy and consumption levels boom, Chinese energy policy is likely to affect significantly 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's, 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 to 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 further 15% of total energy consumption by 2010 and 2020 (SCIO, 2007). The Renewable Energy Law of the People's Republic of China[59] (09/11/2005, Article 16) as well as the 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 the China's National Climate Change Programme, indicating that current measures are expected to lower by 2010 greenhouse gas emissions 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 totalised 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 consumptions during the period from 1990 to 2004 reached 6.8% and 10.1% respectively. Thus, China views biofuels as a necessary strategic component to reach independency on imported oil (Latner *et al.*, 2006). The development of biofuels, started in the late 80s, lead to the first ethanol production in 2002. In 2004, the first recorded ethanol production was 300 000 tons and it more than increased four

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

fold within two years to reach 1.3 million tons in 2006, estimated to 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 to a net importer (Latner *et al.*, 2006). Therefore, the government more focuses 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 shall 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).

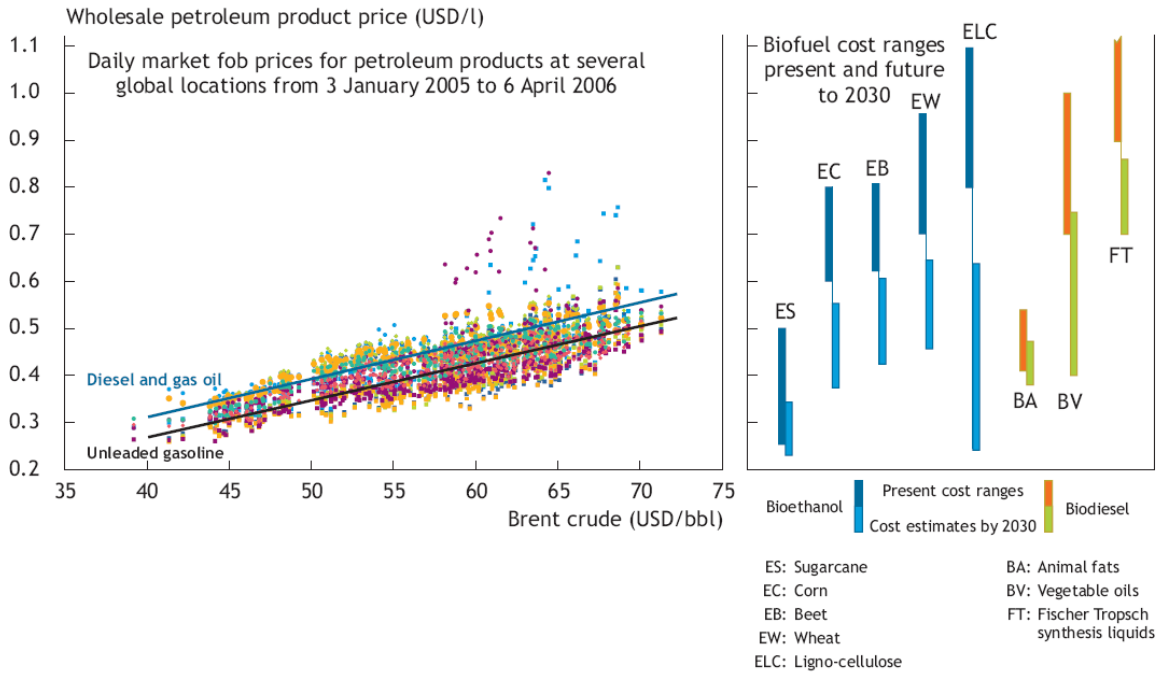
The European Union, the United States of America and China are major emitters of greenhouse gases, and therefore show growing political wills to reduce their emissions notably by developing cleaner energy sources. Nevertheless, many other countries, even some minor polluters, are implementing national 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 or Peru, but also in Asian countries such as India, Japan, and Thailand. Industrialised countries may more emphasize their role in greenhouse gas reduction and energy diversification, while developing countries promote biofuels especially as an opportunity to foster rural development and save foreign exchange. In all cases, biofuels are perceived as a mean 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 commercialization is their high cost of production compared to conventional fuels, two to four times higher (VIEWLS *In* Pelkmans *et al.*, 2006). Thereby, biofuel policies notably consist in 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 between two thirds up to 90% of the total costs of 1st generation biofuels, Wiesenthal *et al.*, 2007) and transformed, while petroleum has just to be looked for and exploited, which until resources have started to become scarce was largely cost effective. Moreover, prices of agricultural commodities, especially those of crude materials, are highly volatile due to fluctuations of 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, 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 the 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's, Brazil continues to maintain a significant tax differential between gasoline and ethanol (Kojima and Johnson, 2005).

Figure 8 gives an overview of the current and foreseen production costs of biofuels compared to 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.




*Daily wholesale gasoline and diesel prices (USD/l free on board) in 12 locations versus the crude oil price compared with the production cost ranges for biofuels without any government support schemes, both now and as anticipated by 2030.*

**Figure 8**: Biofuels current and prospected production costs without support schemes and petroleum products price evolution (IEA, 2007b)

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 to compensate for the higher biofuel production costs (Table 3) and create or enlarge a favourable price for biofuels relative the fossil ones, providing a strong incentive for the consumer to prefer biofuels to 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 showed 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 detaxed partly or completely biofuels: Austria, Czech Republic, France, Germany, Italy, Lithuania, Spain, Sweden and 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 1,140 million Euros in 2005 ($US 1,419 million[60]) 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 28 € ($US 35.2 [25]) 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 sells, 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 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, authorised quota system for biofuel producers or filling station obligation (e.g. 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 though relative 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-blends biofuels. Fuel standards and low blends do not make any biofuel visibility possible, and obligations do 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

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

second-generation biofuels shall count for double compared to 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 focussing 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 shares policies (European Commission, 2007). As an example, the E5 mandate in India has been suspended in 2004, two years after its implementation, due to the lack of ethanol supply.

**Table 3**: 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)<br>0.45 (maize) | 0.54 (with tax)<br>0.38 (without tax) | 0.62 (vegetable oils) | 0.57 (with tax)<br>0.37 (without tax) |
| **Canada** | 0.85<br>0.50 | 0.68<br>0.40 | 0.50 | 0.68<br>0.39 |
| **EU-15** | 0.88<br>0.68 | 1.32<br>0.40 | 0.68 | 1.29<br>0.40 |
| **Poland** | 0.80<br>0.51 | 1.2<br>0.39 | 0.82 | 1.1<br>0.38 |
| **Brazil** | 0.35 (sugar cane) | 0.84<br>0.39 | 0.65 | 0.49<br>0.38 |

Notes: * Biofuel prices are accommodated for 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 than for the US, i.e. 1st: wheat, 2nd: maize; and 1st: with tax, 2nd: without tax.

Supply-side instruments, mainly capital grants and feedstock support, have had though limited success in pushing biofuels so far (Wiesenthal *et al.*, 2007). Indeed, support to production facilities for 1st generation biofuels do not drastically impact the cost of biofuel due to the fact that capital only represents a marginal part of the total cost in comparison with feedstock (0.01€ per litre with a 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 to 1st generation biofuels, some 60% and above (JRC/EUCAR/CONCAWE, 2006). The same pattern is observed with feedstock supports. Although feedstock represents a large part of the total production costs; energy crop premium indeed only lowers 1st generation biofuel costs by 0.01 to 0.04€ per litre. However, as capital grants or loan programmes, feedstock subsidies appear to be an efficient instrument in order to support special type of biofuel (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 overproductions 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 in 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 to 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 included 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[61] 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[62].

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[63]'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 shall also emphasize 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[64] 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. In addition, biofuels producers need a self-running market with long-term visibility and a 2-3 year perspective based on yearly-tailored fiscal budget does not offer that. Internalising the external benefits of fuel ethanol is the greatest help of all, which could solve the problem (*In*

[61] http://ec.europa.eu/agriculture/capreform/infosheets/energy_en.pdf
[62] European Commission Press releases, IP/06/1243, Brussels, 22 September 2006
[63] URAA, Uruguay Round Agreement on Agriculture in Marrakech 1994
[64] Communication of Charles E. Hanrahan, Senior Specialist at the Congress Bibliotheca in Washington, on 01/2008 at the AgroParisTech.

EurObserv'ER, 2006). However, these externalities are not easy to quantify and to be given any value and the cost/benefit analysis also includes many 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, still lacks in 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. 9: 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. 9: 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 to green heat and power if permits were the only incentive (Fulton *et al*., 2004). This system makes emissions 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 to 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. As well reductions of tariff barriers within international trades 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 potentials 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, included 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 harmonization within ministries and governments at all levels. All over the world projected or implemented mandatory blends are on the agenda: E10 mandate in Thailand and China, planed B2 and E10 mandates in Latin America, possible ethanol blending mandate in Japan, 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 harmonization in terms of bioenergy strategy on a global level.

Furthermore, difference in blends standards can affect international trades as biofuels producers from different countries might not be supported by subsidies or taxes the same way when considering various blends characteristics needed to get these helps, as claimed by the European Biodiesel Board (EBB) In Bioenergy Business, 2007. This also stresses the need for harmonization 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 biofuels producers and consumers guarantee (Wiesenthal *et al.*, 2007). The introduction of policies on vehicle technology standard, so that all new vehicles would be compatible with specific mixture of biodiesels, would permit to lower the production costs of such vehicles and further incite to biofuel consumption (Fulton *et al.*, 2004). It would also pave a way to more flexibility and more coherence with other policies, which is lacking as the example of Spanish biofuels domestic surplus can show (Bioenergy Business, 2007).

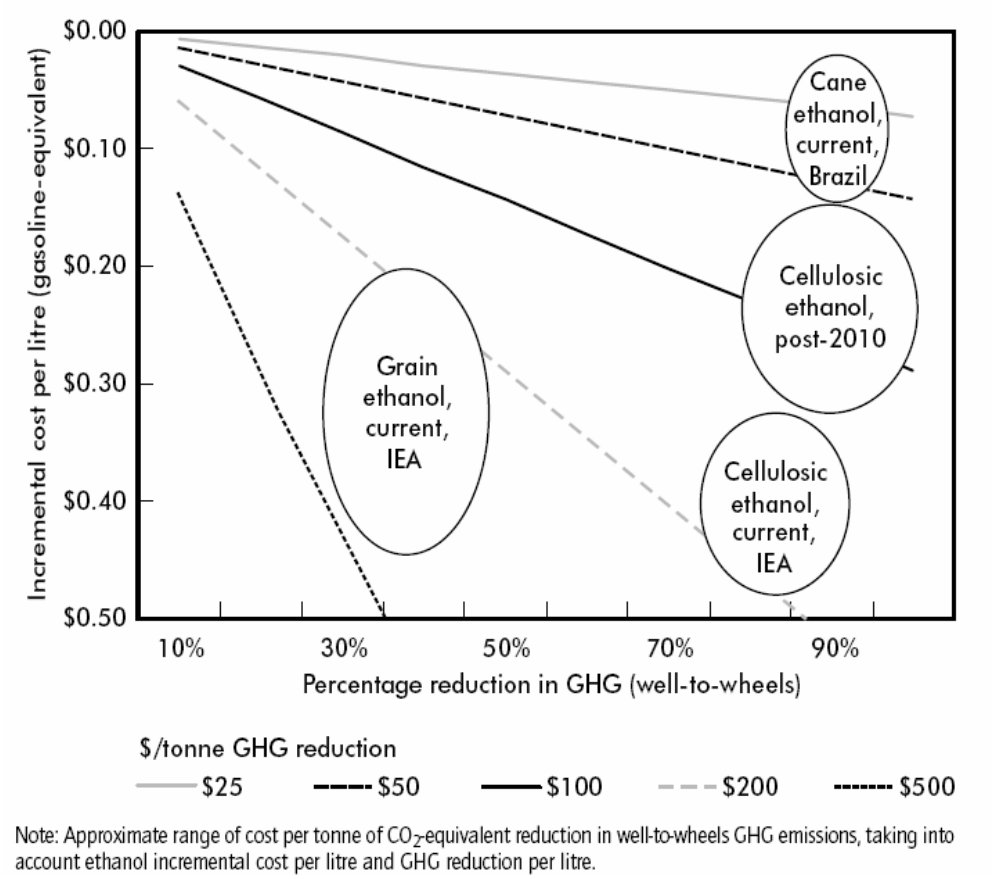



**Figure 9**: Biofuels incremental cost per ton of greenhouse gas reduction (Fulton et al, 2004)

Despite existing policies and measures, the major uncertainty factor, to decide which biofuel path should be fostered or not and to better define 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 hypothesis 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, addressing also knock-on effects due to indirect land-use change, are needed to fully judge the benefits and limitations of biomass use (EEA, 2008).

### 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 ceiling 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 by opposition to traditional use of woodfuel (UNDP, 2000). In 2050, the total primary energy demand will vary between 800 EJ $yr^{-1}$ and 1,400 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 made by matching demand and supply. On the contrary, studies either focus 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 jeopardizing 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 of agricultural systems 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.

*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 to forest products or residues; land availability for energy crops and biomass yields hence appear to be the main key-assumptions (Berndes *et al.*, 2003, Smeets *et al.*, 2007). Diverging assumptions on the energy crop yields alone lead to a 40%-difference on 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 yields 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. 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 evolvements; 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 products production and consumption is expected to go on especially in developing countries, where people still 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 from 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 rest land encompass semi-natural vegetation types such as savannas etc., barren land and about 0.26 Gha of build-up area (FAO, 2002 and 2003; Hoogwijk *et al.*, 2005*;* Smeets *et al.*, 2007).

Table 6 shows land use change patterns in 2000 based on annual changes between 1990 and 2005.

**Table 6**: Global land uses (Gha) and annual conversions in year 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 where identified.

Table 7 gives an overview of past and future trends of population, average calorie intakes and agricultural productions. 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 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 to 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 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 shortage of suitable land, e.g. in South Asia more than 94% of suitable land are already farmed.

*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 are 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, or 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[65] 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 to spare grassland for other crops as some part of suitable crop lands are currently used as pastures especially in developing countries (Smeets *et al.*, 2007).

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

**Table 7**: World population and agricultural production projections at a glance (FAO, 2002)

| **Timeframe** | **1979-81** | **1997-99** | **2015** | **2030** | **2050** |
|---|---|---|---|---|---|
| **Population in billions** | 4.430 | 5.900 | 7.207 | 8.270 | 9.322 |
| World | (1.6) | (1.5) | (1.2) | (0.9) | (0.6) |
| **(annual growth rate %)** | | | | | |
| 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$^{-1}$ (1kcal=4.186 10$^{-3}$ MJ)** | | | | | |
| **(annual growth in demand for agricultural products in %) - [calorie from animal products]** | | | | | |
| World average | 2 552/10.7 (2.1) | 2 803/11.7 (2.0) | 2 940/12.3 (1.6) | 3 050/12.8 (1.4) | *3 302/13.8 [550]* |
| Developing countries | 2 312/9.7 (3.7) | 2 681/11.2 (4.0) | 2 850/11.9 (2.2) | 2 980/12.5 (1.7) | *3 236/13.5 [549]* |
| **Timeframe** | **1979-81** | **1997-99** | **2015** | **2030** | **2050** |
| **Calorie consumption in kcal capita$^{-1}$ day$^{-1}$/in MJ capita$^{-1}$ day$^{-1}$ (1kcal=4.186 10$^{-3}$ MJ)** | | | | | |
| Industrial countries | 3 135/13.1 (1.0) | 3 380/14.1 (1.0) | 3 440/14.4 (0.7) | 3 500/14.7 (0.6) | *3 448/14.4 [1 054]* |
| Transition countries | 3 389/14.2 (-1.7) | 2 906/12.2 (-4.4) | 3 060/12.8 (0.5) | 3 180/13.3 (0.4) | *3 629/15.2 [941]* |
| **Total arable land in Gha (irrigated land in Gha)** | | | | | |
| World | 1.608 (210) | (271) | | | |
| Developing countries | - | 0.956 (202) | 1.017 (221) | 1.076 (242) | - |
| Industrial countries | - | 0.387 (42) | - | - | - |
| Transition countries | - | 0.265 (25) | - | - | - |

| Timeframe | 1979-81 | 1997-99 | 2015 | 2030 | 2050 |
|---|---|---|---|---|---|
| **Crop land 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.

Testing diverse scenarios of combined population growths, change in diets and food and feed production systems (Table 8), recent studies show that considerable parts of agricultural lands could be allocated to bioenergy production without jeopardizing 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), 0.7 to 3.6 Gha (Smeets *et al.*, 2007). Highest estimates, notably in the systems 3 and 4 of Smeets *et al.* (2007) are rather too optimistic though, combining very intensive agricultural productions with high yields increases and a landless animal production, together with a management factor of 1.5 that indicates even further possible improvements, more irrigation, more fertilizers, while system 4 already corresponds to a 25%-add to the yield levels in 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 to bioenergy production (Smeets *et al.*, 2007). However, such a system could hardly be implemented in these regions by 2050. On the contrary, low productive agricultural system (Y1 in Table 8 = "Low External Input" in the study Hoogwijk *et al.*, 2003), which could be roughly compared to an organic production system, does not permit 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 emphasizing that agricultural intensification is a prerequisite.

Considering medium population scenarios with affluent diets and relative 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 are short rotation coppices (SRC). These crops provide high yields and are convenient to assess maximum primary bioenergy potential, whereas potentials expressed based on first-generation biofuels' crops would not be as relevant without calculating conversion factors. However, SRC are not representative of the portfolio 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 purpose, 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 in these studies' background assumptions made it possible to better apprehend the relevance of the results. However, other assessments based on different models and hypotheses would be needed to test the robustness of the results.

*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 concerning both 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 case of a low plantation scenario and a high forest-products demand, and 103 EJ $yr^{-1}$ in 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 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 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 like 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.

**Table 8:** Comparison of three selected assessments of surplus agricultural land areas available for energy crops

| **Scenarios for 2050** | **Land available pool (Gha)** | **World population (billions 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 = | P1 | P2 | P3 | D1 | D2 | D3 | Y1 | Y2 | Total agricultural land is constant = 5 Gha | **Yields weighted by MF = 20tDM ha$^{-1}$ yr$^{-1}$** |
| | | 8.7 | *9.4* | 11.3 | 10.1 | 10.1 | *11.5* | 2.2 | 5.9 | No deforestation for bioenergy production | **HHV*=19GJ t$^{-1}$** |
| | 1.5 croplands (including 20% irrigated) | | | | (1.3) | (2.4) | (4.2) | (4.0) | (14.3) | Area needed for food and feed is doubled to take into account losses, risks, uneven accessibility to the resources etc. | D1 "not likely"<br>Y1,D2-D3,P => **0**<br>P1,D2,Y2 => **2.6 (988)** |
| | + 3.5 grasslands | | | | | | | | | Management Factor (MF) for energy crops: 0.7 | P2,D2,Y2 => **2.4 (912)**<br>P3,D2,Y2 => **1.9 (722)**<br>P1,D3,Y2 => **0.8 (304)**<br>P2,D3,Y2 => **0.45 (171)**<br>P3,D3,Y2 => **0.15 (57)** |
| Hoogwijk *et al.*, 2005 interpreting the IPCC SRES: A1, A2, B1, B2 | 13 | A1<br>B1 | B2 | A2 | A2 | B2 | A1<br>B1 | A2<br>B2 | A1<br>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).<br>Impact of climate change is included.<br>Pressure on land quality is not included.<br>Management Factor (MF) for energy crops: A2, B2: 1.1; B1: 1.3; A1: 1.5 | ***Yields weighted by MF = 6-34tDM ha$^{-1}$ yr$^{-1}$ [*mean yields tDM ha$^{-1}$ yr$^{-1}$] LHV=19GJ t$^{-1}$**<br>A1 => **1.3 (409) [20]**<br>B1 => ***1.3 (398) [20]**<br>A2 => **0.6 (129) [16]**<br>B2=> ***1 (279) [18]** |
| | - excluded and allocated areas | 8.7 | *9.4* | 11.3 | 12.5 | 12.8 | 13.2 | Regional yields of 12 food crops weighted by MF<br>MF• = 0.78 | MF• = 0.82 | | |

| **Scenarios for 2050** | **Land available pool (Gha)** | **World population (billions 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<br><br>- excluded and allocated areas | 8.8 | 13<br><br>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=19GJ $t^{-1}$**<br><br>S1 => **0.7 (215) [16]**<br>S2 => **1.15 (455) [21]**<br>S3 => **3.3 (1,101) [17]**<br>S4 => **3.6 (1,272) [20]** |
| | | | | 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 | |
| | | | | Average yield increases (19 crops) between 1998-2050 | x 2.9 | x 3.6 | x 3.6 | x 4.6 | |
| | | | | 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.<br>Demand for feed crops, included increase in demand for feed from grasses and fodder compared to 1998, is added to total demand for food crops.<br>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.<br>No deforestation for bioenergy production.<br>Water is excluded as limiting factor except in arid and semi-arid regions.<br>Irrigation is limited to areas in which climate, soil, and terrain permit it.<br>Impact of climate change is excluded.<br>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°x 0.5°; *read on the graphs

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 2,400 million m3, with one third from plantations, about twice as much as today's plantation production of 400 million m3 (FAO, 2002). After processing, about half of log input becomes wastes that can have various alternative uses such as chips for pulp or particleboard etc., and fuel for internal energy use or as densified commercialized fuels, depending on the sale values on the diverse markets (Parikka, 2004).

Overall, biomass residues may be classified as following: primary ones from agriculture (crop residues) or forest (logging residues), secondary ones from agriculture (from food processing, animal manure) and forest (mill and manufactures residues), 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 productions. Fischer and Schrattenholzer (2001) thus stressed that this factor shall 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, fertilizer (including the need to maintain soil quality), or 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 residues 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).

*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 yields increase in these regions would lead to the release of land areas for energy crops. Together with 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 and Faaij, 2006 *In* Smeets *et al.*, 2007). Figure 22 shows regional bioenergy future potentials of the four scenarios in Smeets *et al.* 2007. Figure 23 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).

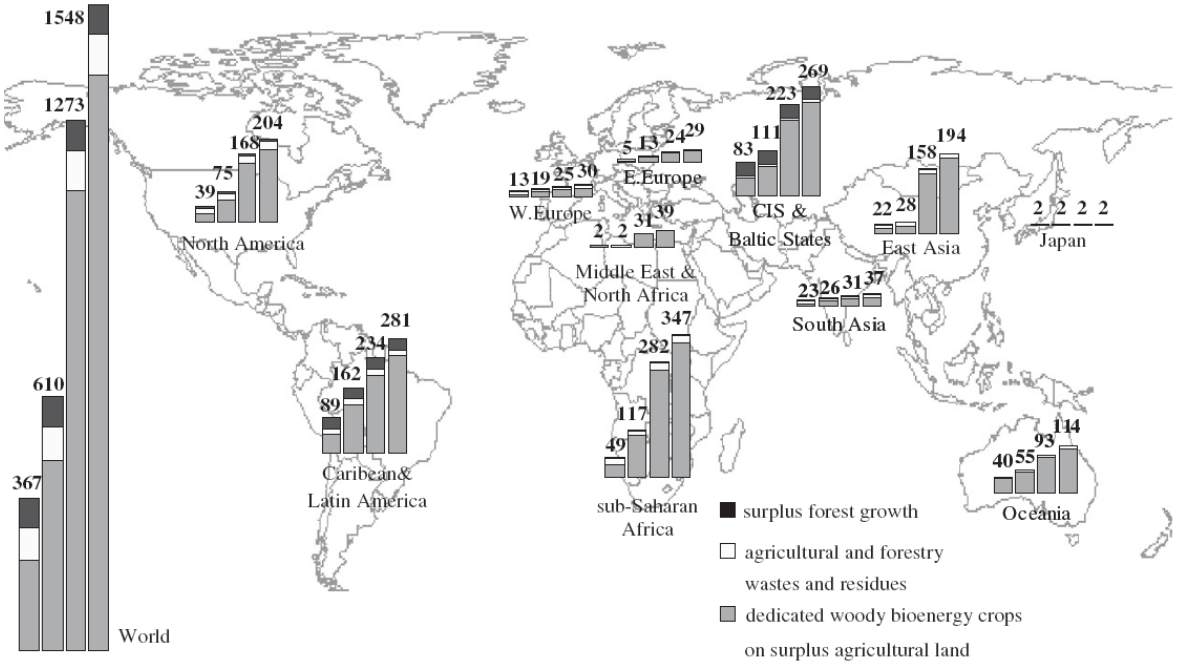



**Figure 22:** Total technical bioenergy production potential in 2050 based on systems 1–4 (EJ yr-1; the left bar is system 1, the right bar is system 4) (Smeets *et al.*, 2007)

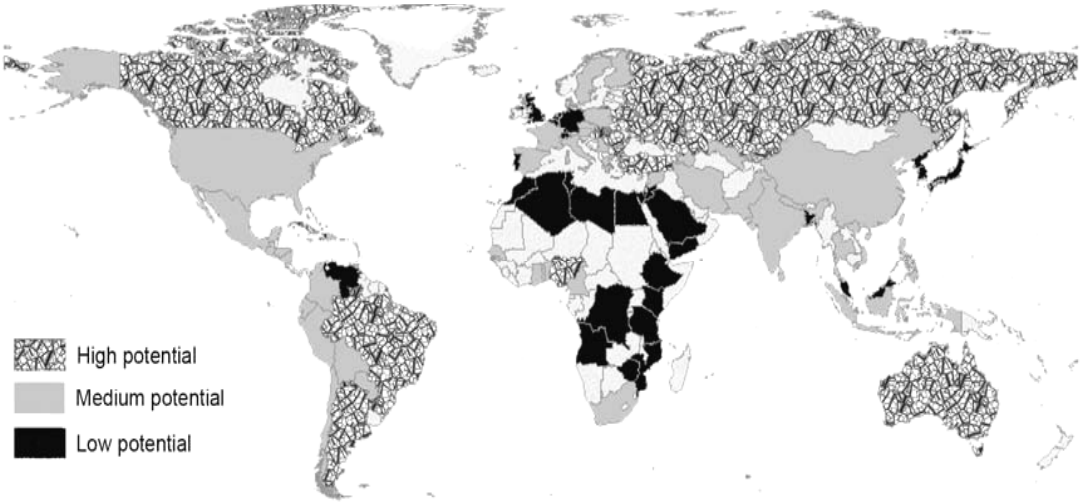



**Figure 23:** Biofuel production potential (Von Braun, 2007). Sources: Data on food insecurity are from FAO 2006. The land availability index is derived by the author based on data from FAO 2007, IEA 2007a,b and USDA 2006. Data on water availability are from WRI 2007.

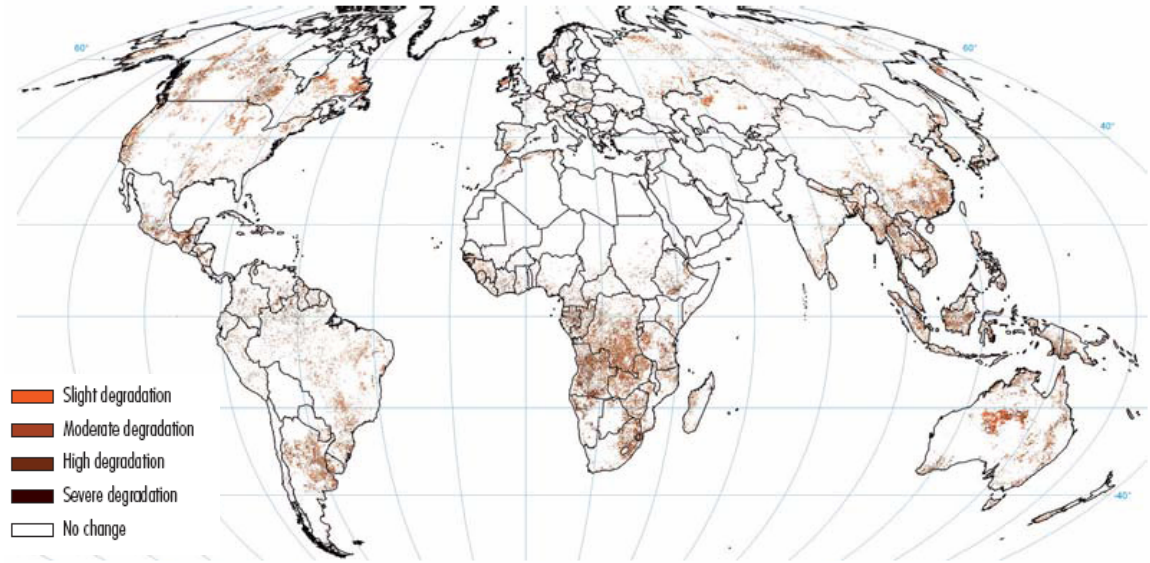



**Figure 24:** Global land degradation using biomass production and rain-use efficiency trends between 1981-2003, (Bai *et al.*, 2007 online: http://www.fao.org/nr/lada/images/maps/ndvi_preliminar.jpg)

Oceania is the least land-stressed region, whereas Japan is the most land-stressed country. The Middle East and North Africa, South and East Asia are relatively scarce of 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 require less drastic changes that 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 other. Regions with highest potentials could turn to bioenergy exporters. In developing countries, bioenergy may provide new incentives for investments in agricultural research and, by providing new income, make a modernization 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, like Sub-Saharan Africa or Latin America, but which also suffer from soil erosion or nutrient depletion. In sub-Saharan, some 95 Mha of land are 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 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 (Table 9).

**Table 9**: 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 |

As shown on Fig 24, 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 would be in state of more or less severe degradation (Bai *et al.*, 2008). The most degraded areas are mainly associated with forest degradation, although precise historic of land degradation processes has to be investigated at 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 to the atmosphere of one or 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 (Table 10, 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 mean to recover parts of the lost former sequestered carbon. Although yields on these lands will not be optimum, some species with low needs such as *Jatropha*, could be used in agroforestry systems (see picture) 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, 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 vegetations 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 at 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*/chili
Belize, February 2003 bessou©INRA

**Table 10:** Tropical degraded lands with potential for plantation establishment (Mha), 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 | 110 | 222 |
| **Total** | 203 | 87 | 330 | 620 |

To summarize, 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 3,500 EJ $yr^{-1}$(Hoogwijk *et al.*, 2005); 4,435 EJ $yr^{-1}$ (Smeets *et al.*, 2007) traduces 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 shall translate political orientation choices. In the case study of woody biomass for energy production as a mean of greenhouse gas reduction, Dornburg *et al.* (2007) showed that, in Poland, biomass potential is larger than the amount that could make it 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 even a 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 markets' 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 is 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). As well, 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 uptake lots of water and nutrients from the soils. Large-scale energy crop implementation would in some countries, like Poland or South Africa, lead to further enhancement of already stressed water situation (Berndes 2002 *In* 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) significant amount of forest is cut down, 45% of South America forest area could disappear within 100 years (Hoogwijk *et al.*, 2005). Part of the abandoned agricultural land areas should then 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 utilised agricultural area dedicated to *environmentally-oriented farming EOF*[66], 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 greenhouse gas emissions 1990 level that would make the price for carbon emission permit 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

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

2030, i.e. 17% of European current global energy consumption (EEA, 2006b). This *conservative estimate*[67] 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 take over it in the long-term (up to around 5.9 EJ $yr^{-1}$≈140 Mtoe $yr^{-1}$). *Environmentally-compatible* bioenergy potential[68] 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 current European food self-sufficiency level was set 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 competition effect between bioenergy and food production was restricted to France and Germany, the only member states that 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. Based on 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 25, taking into account the LHV for the conversion of the harvested dry biomass to an energy potential (EEA, 2006b).

Figure 25 shows that high share 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 anymore. 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[69] 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.

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

[68] *"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 to a development without increased bioenergy production (EEA, 2006b).

[69] 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.

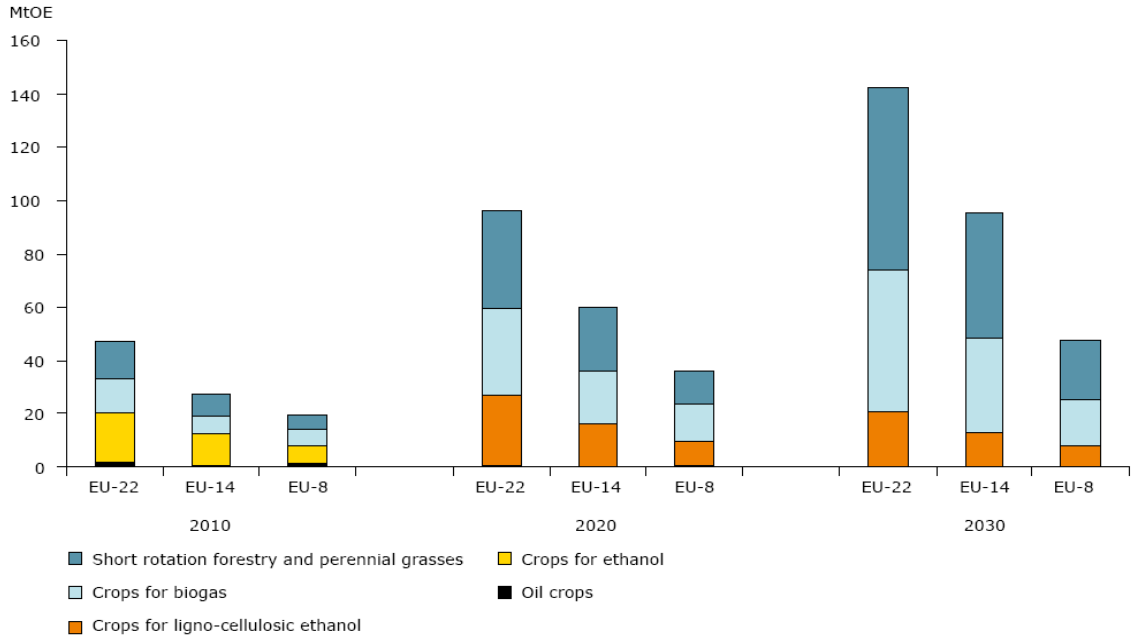



**Figure 25:** Environmentally-compatible agricultural bioenergy potential (EEA, 2006b)
Note: No data available for Cyprus, Luxembourg and Malta. 'Oil crops' comprise rapeseed and sunflower. 'Crops for ethanol' include the potential of grains from maize, wheat, barley/triticale. 'Crops for ligno-cellulosic 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.

## 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. 26). Given the numerous types of feedstock and processes that determine biofuels yields per unit of biomass input, there would be multiple scenarios for future global biofuel potentials and as much possible studies. Global assessments therefore extrapolate current biofuels trends and implemented target policies.

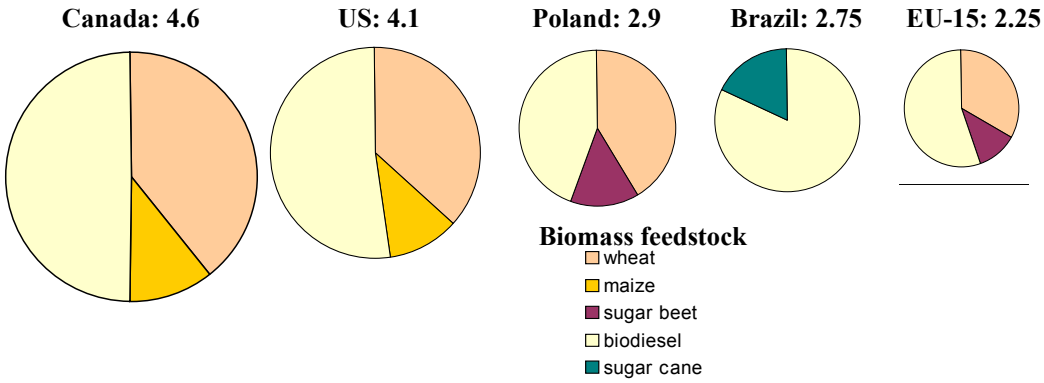



**Figure 26**: Land areas (ha) needed to produce 3,000 l of fossil fuel equivalent on the basis of the biofuel energy content; 1,000 l per type of biomass feedstock except for Brazil 2,000 l of ethanol from sugar cane. Drawn from data *In* OECD, 2006.

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 energy basis) in world transport fuel consumption within the major biofuel producing regions (Brazil, US, EU[70], Canada) (OECD, 2006). Except in Brazil and Poland, such a share would require excessive amounts of cropland areas (Fig. 27). In Brazil, the combination of high sugar cane yields and low transport fuel consumptions 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 share of biofuels produced in Europe and consumes much land areas, about five times more than ethanol from sugar beet (OECD, 2006).

As an extrapolation of land areas allocated to biofuels 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 biofuels production to reach targets of 5% in 2010 and 10% in 2020 displacement of both road transport gasoline and diesel future consumptions (on 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 produced 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 enlighten that it may make sense to focus more on ethanol blending rather than on biodiesel, as in the medium term substantial amounts of biofuels can only be achieved if the feedstock mix is

[70] In both studies here (OECD and Fulton *et al.*) data concern 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.

adjusted in favour of commodities with a higher biofuel output per hectare. This may also include imported feedstock and biofuels (OECD, 2006).

Within 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 sector 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 to 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).

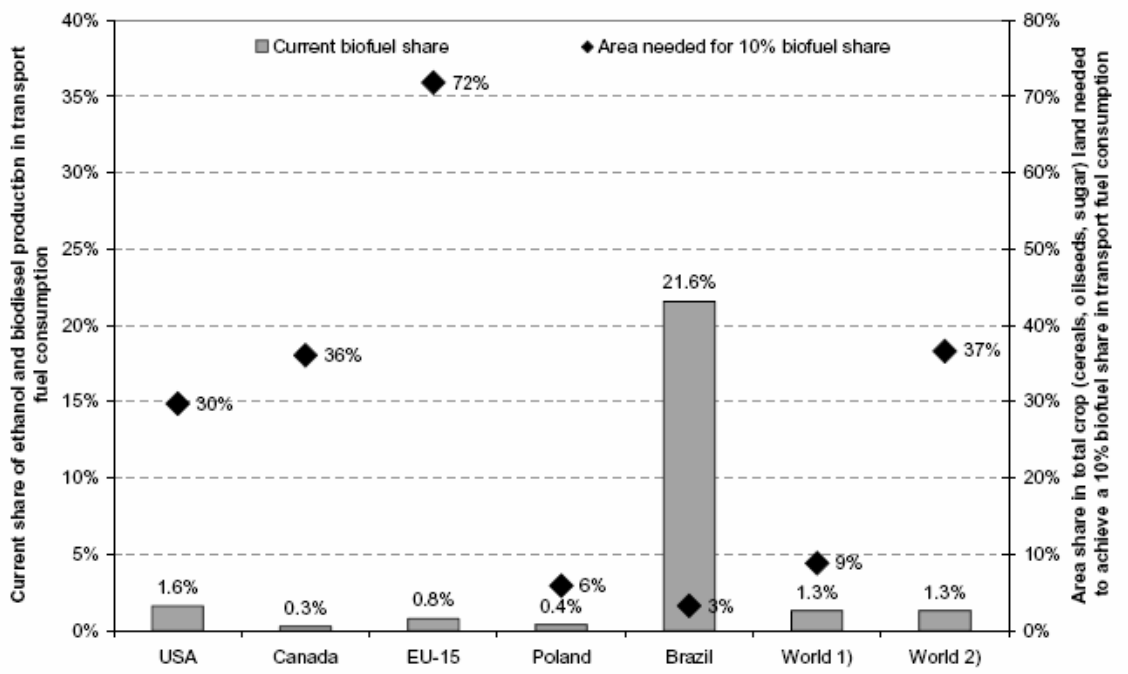



**Figure 27:** Biofuel shares in transport fuel consumption and land requirements for 10% biofuel shares in major biofuel producing regions (OECD, 2006)

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

The development of 2nd generation biofuels produced from residues and dedicated crops appear crucial to lower the pressure on cropland areas. Compared to 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.

By 2030, a 15 to 25% biofuels 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 (Walsh *et al.*, 2000 *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 on early stages. Finally, cooking oils and other municipal wastes could contribute to a lower extent to biofuel production, poor economics of scale being compensated 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 productions and consumptions

In the reference scenario of the IEA with 2004 as baseline and an 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 28 gives an overview of past and projected ethanol production showing that global trends follow the predicted scenario, the EU still produces less that expected compensated by the US where it is the contrary. In 2008, world biofuel production reached 46 Mtoe, 65 Gl of ethanol (beyond prediction on Fig. 28) and 16 Gl of biodiesel (ENERS, 2009).

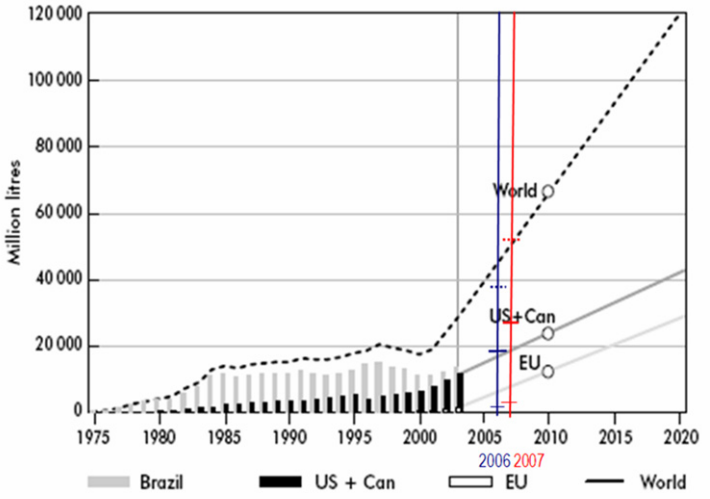



**Figure 28**: Fuel ethanol, production and projections to 2020 (Fulton *et al.* 2004) added production data for 2006 and 2007 from F.O. Licht.

The IEA Alternative Policy Scenario incorporates 1,400 different policies and measures that aim at enhancing energy security and mitigating $CO_2$ emissions. Measures in 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, biofuels production could raise up 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 (Fig. 29 & 30). If cellulosic ethanol were to be largely available by 2030, a larger share of biofuels (10%) could be possible with only little extra land area (+ 5.7 Mha).

Ethanol is expected to account for most of the biofuels 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 biofuels use in both countries. In both scenarios, biggest increase in biofuels production and consumption occur in Europe and the US may become sizeable net importers 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 biofuels demand imports. The shift from domestic production to import from abroad would be rather important in most European countries for ethanol, from around 95% of domestic production to an average 50% share in 2020 across member states. Trades 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 renewables and 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 to current levels, i.e. saving these 8 Gt, would mean to increase 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 transport sector thanks to more efficient and cleaner vehicles, notably using twice as much biofuels as in the Alternative Policy scenario (IEA, 2006).

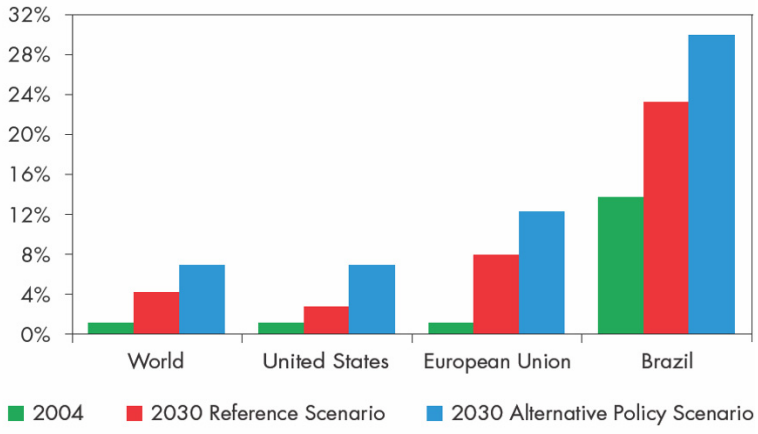



**Figure 29**: Share of biofuels in road-transport fuel consumption in energy terms (IEA, 2006)

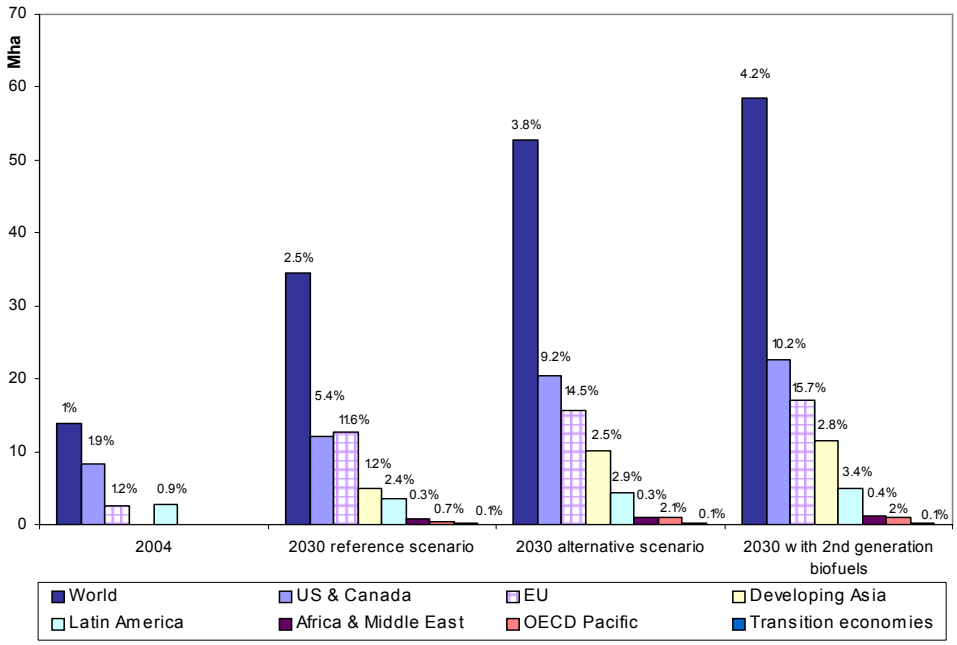



**Figure 30:** Land requirements in Mha and percent of respective agricultural land areas for biofuels production in the different scenarios from the IEA. Drawn from data *In* IEA, 2006

## 6.5 Impacts of biofuels on agricultural commodity prices

As background assumptions 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 be then allocated to energy crops, leading to an overall decrease in bioenergy potential in most cases due to the need of more agricultural land to produce enough food and feed on less productive areas. On the contrary, a global geographical optimization 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 demand. Commodity prices hence raised 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 escalate 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 increased by 4% and 7%, respectively, since 2000, the use of cereals for industrial purposes, such as biofuel production, 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 oilseeds price in the past five years was 0.2[71], compared to 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 biofuels expansion on projected food prices is not yet well established (Von Braun, 2007). Furthermore, due to competitive land uses, biofuels may also affect 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 grow preferentially maize over these same crops. Therefore, prices of rice and wheat increased. Comparing 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 commodities prices until 2014 were modelled with three biofuels development scenarios compared to a baseline scenario, i.e. *status quo* agricultural policies, normal weather patterns, and biofuels' growth in the US and Brazil only: the constant biofuels scenario with no growth in biofuels production, the policy-target scenario with projected biofuels' growth in line with officially stated goals (in 2004), and a third scenario considering the policy-target scenario and higher oil prices (Fig. 31).

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

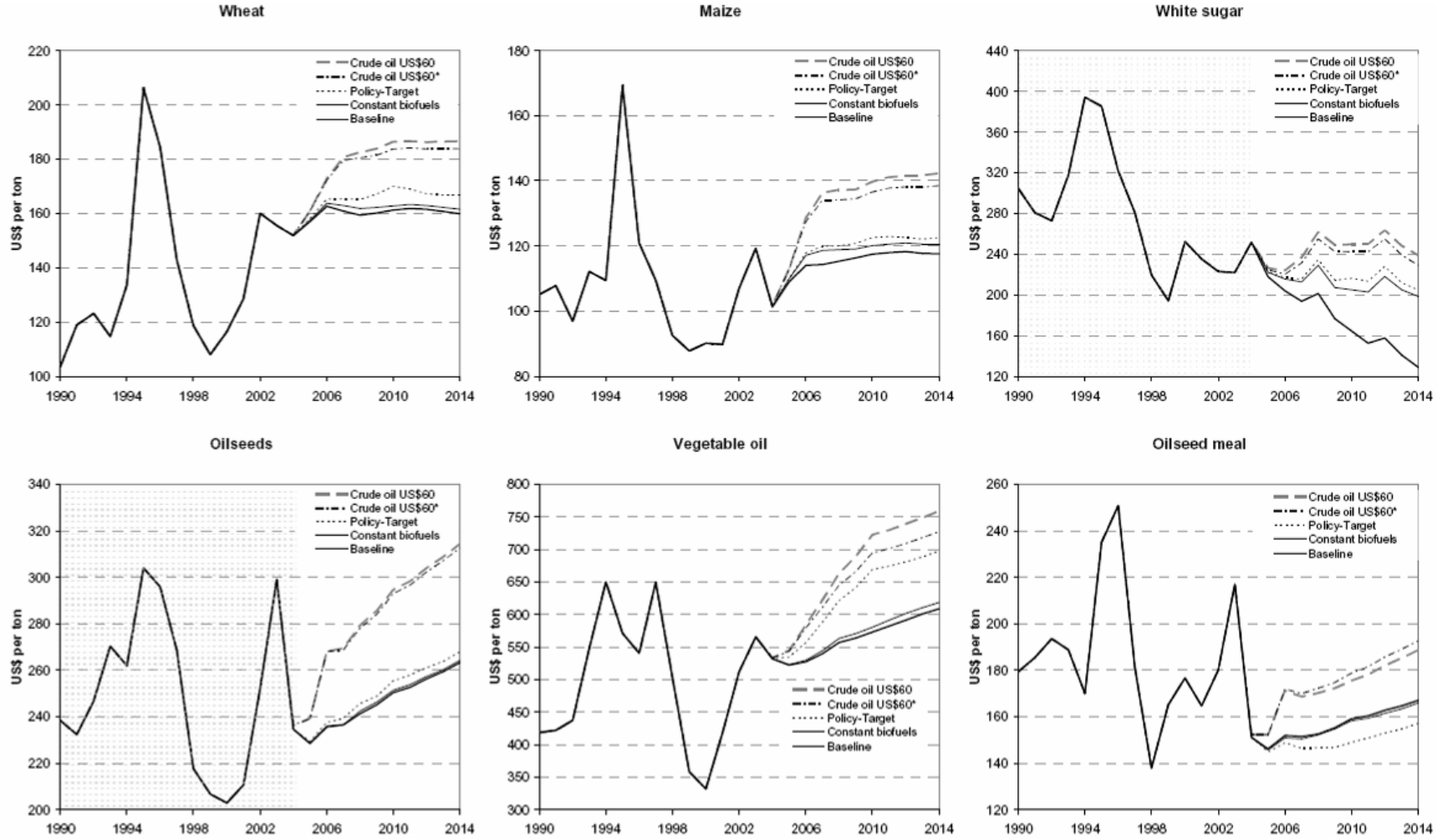



**Figure 31:** World crop market prices under alternative scenario assumptions (OECD, 2006)
Note: "Crude oil US$60*" denotes a scenario assuming higher crude oil prices, but unchanged petrol-based fuel prices (and hence unchanged biofuel prices) relative to the policy-target scenario.

The biofuels' growth in the baseline scenario has a relatively small impact on coarse grains, wheat and vegetable oil prices, increases of 2.5%, 1% and 1.5%, respectively, compared to the constant biofuels 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 was much larger, a 37% price increase up to 60% in the policy-target scenario.

The policy-target scenario will especially require enhanced biofuel productions in the EU and Canada added to increased productions in the US and Brazil. Therefore, substantial growth in feedstock needs implies significant trade patterns changes. EU imports of vegetable oils would increase three-fold, while wheat exports would fall by 41%, Canadian wheat and coarse grains exports would decrease by 34% in 2014. Overall world prices for most commodities increase substantially compared to the constant biofuel scenario, by 4% for wheat, 15% for vegetable oils also inducing an increase of butter price of 3% as a substitute for oil (OECD, 2006). Despite more important decrease of oilseed meal price (-6%), meat market prices increase with and without growth in biofuels 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, commodities prices first increase due to the high production costs. Biofuels costs also increase, but compared to 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 to 5.5% in the policy-target scenario. The increase of biofuels production would lead to a further increase in commodities prices (OECD, 2006). In comparison, IFPRI modelled that in a 2007-baseline biofuels production, maize and oils prices would be higher by 6%, wheat, and sugar by 4% in 2014 compared to a constant biofuels 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 biofuels 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 others factors. In Boeters *et al.* (2008), the impacts on food prices of a 10%-biofuel target turn out to be negligible at the European Union scale compared to 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, biofuel international trades including the future role of developing countries, and the development of 2$^{nd}$ 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 frame 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 between 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 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$ 1,190), 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).

# Appendix 2

# Characteristics of sugar beet

**Table A2: Sugar beet characteristics**

(OECD, 2001; ITB, 2003; ADEME/DIREM, 2002; Tzilivakis *et al.,* 2005)

| **Sugar beet** | **Characteristics** |
|---|---|
| Latin name | *Beta vulgaris* L. ssp. *Vulgaris* var. a*ltissima* |
| Family | *Chenopodiaceae* |
| Genetics | Basic chromosome number: n=9<br>Self-incompatible<br>Double sterility for the triploïds: chromosome imbalance and cytoplasmically inherited male sterility<br>Mostly triploid hybrids since 1970s but in France the diploid varieties still represent 50%. |
| Photosynthetic pathway | **$C_3$** |
| Soils | Mostly loamy soils but also clay or calcareous soils.<br>Generally good deep soils as sugar beet is an exigent crop. |
| pH | The most acid-sensitive of the common arable species/slightly tolerant of acidic conditions<br>Suitable soil pH: 7-8 |
| Water supply | Requires high amounts of moisture and fields are irrigated in regions with low precipitation |
| Day length | Long day plant |
| Dormancy | Long-lived dormant seeds |
| Lifetime | Biennial requiring a period of vernalisation at the end of the first year before it can flower. However sugar beet is harvested after about 7 months (at least 150-170 growing days, up to 200 days), whilst the vegetation cycle is not finished i.e. before the stem elongation starts. |
| Yields in tonnes (fresh beet) per ha per year | France: 66-**80** (17.7%-18% sugar)<br>England: **34**-60 (16% sugar)<br>FAO: 40-60 (15% sugar)<br>up to 70-80 under certain conditions |
| Moisture content at harvest | 70-80% |
| Sugar content (% of FM) | 15, 16, 18-20 |

Advantages (+) and drawbacks (-) within a crop rotation

| | |
|---|---|
| + Deep roots exploiting deep water and restructuring deep soil layers | - Water use and erosion risk after the harvest |
| + Low leaching risk after the harvest | - Leaching risk before the sowing |
| + No re-growth | - Weed control complex during the crop |
| + Break crop that interrupt the cycles of cereals' parasites. | - Can favour the development of the wheat bulb fly |

# Appendix 3

# Characteristics of *Miscanthus*

**Table A3:** Miscanthus characteristics (age of stand: 3-5 years)
(Lewandowski, 2000; Lewandowski and Heinz, 2003; ADEME/ITCF, 1998; Venendaal *et al*., 1997)

| **Miscanthus** | **Characteristics** |
|---|---|
| Latin name | *Miscanthus ssp.* |
| Other names | Elephant Grass, E-Grass, Chinese Silvergrass |
| Family | Subfamily *Andropogoneae* of the *Poaceae* family |
| Genetics | Basic chromosome number: 19<br>Self-incompatible |
| Triploid (x *giganteus,* x *sinensis)* | 3n=57 |
| Photosynthetic pathway | **$C_4$** |
| Soils | Wide range (including soils polluted by heavy metals which are not up taken by the plant) |
| pH | Tolerant: 5.5-7.5 |
| Water supply | Not tolerant to stagnant water and prolonged drought periods, no soil compaction |
| Day length | Long day plant |
| Lifetime | 15-25 years (the oldest plantation in Europe was 18 year old in 2003 in Denmark) |
| Maximum height | **2-4** m |
| Yields in tonnes DM.ha$^{-1}$.a$^{-1}$ | Austria: 22<br>Britain: 10-15<br>Denmark: 5-15<br>France: 10-22<br>Germany: **4**-30<br>Greece: 26-**44**<br>Italy: 30-32<br>Switzerland: 13-19<br>Spain: 14-34<br>Turkey: 38 |
| Moisture content at harvest | 16-62 |
| Cellulose (% of DM) | 43 |
| Hemicellulose (% of DM) | 26.7-27.8 |
| Lignin (% of DM) | 24 |
| Ash total (% of DM) | 1.6-4.2 |
| Compounds (% of DM) | N: 0.19-0.67<br>P: 0.04-0.11<br>K: 0.31-1.28<br>Ca: 0.08-0.14<br>Cl: 0.10-0.50<br>S: 0.04-0.19<br>Mg: 0.03-0.07 |
| C (% of DM) | 47.8-49.7 |
| H (% of DM) | 5.46-5.92 |
| O (% of DM) | 5.46-5.92 |
| LHV / HHV (MJ.Kg$^{-1}$ DM) | 17.05/19.2 |
| Ash fusion temperature | 1,020°C |

**Table A3 follow-up:** Compared advantages of early and late harvests of *Miscanthus* (Venendaal *et al*., 1997; Clifton-Brown *et al.,* 2002; Heaton *et al.,* 2004; Lewandowski, 2000; Lewandowski and Heinz, 2003; Lewandowski *et al.,* 2003; Jørgensen pers. com. 2009)

| **Advantages(+) Drawbacks (-)** | **Cut before January** | **Cut after January** |
|---|---|---|
| *Total biomass yield* | + More biomass | - Less biomass<br>14 to 15% losses between December and February + further 13% losses between February and March<br>20 to 30% losses in biomass (up to 50% in Greece) |
| | mean: 17.6 DMt $ha^{-1}$ $yr^{-1}$<br>mean upper values of yield ranges by study: 21.5 DMt $ha^{-1}$ $yr^{-1}$<br>mean (with January): 15.7 DMt $ha^{-1}$ $yr^{-1}$ | mean: 19.3 DMt $ha^{-1}$ $yr^{-1}$<br>mean upper values of yield ranges by study: 26.8 DMt $ha^{-1}$ $yr^{-1}$<br>mean (with January): 15.3 DMt $ha^{-1}$ $yr^{-1}$ |
| *Water content* | - Higher<br>In autumn up to 66%<br>44% in December (Portugal)<br>More water implies drying or low calorific value in the case of direct combustion (50% of water will reduce the energy content by 2.45 MJ/kg): overall energy balance?<br>More complicate to store | + Lower<br>40% in February-March in (Austria)<br>In spring 15-25% (extreme 50%)<br>Storage without further drying<br><br>Easier to store |
| *Ash content high levels of K, Cl, Na= problem of particule emission, corrosion or deposit during combustion* | - Higher ash content | + Lower ash content<br>Rain leaching of alkali metals (Na, K)<br>Significant decrease of chloride and sulphur contents |
| *Crop management* | + Longer harvest window | - Short harvest window one month<br>+ Can be conditioned by bales which is convenient for transport |
| *Environmental impacts* | + Less soil compaction<br>Bare soil emissions? | + Less nitrogen in the biomass: more nutrients (N, K) return to the soil<br>+ Less $SO_2$ equivalents emissions |

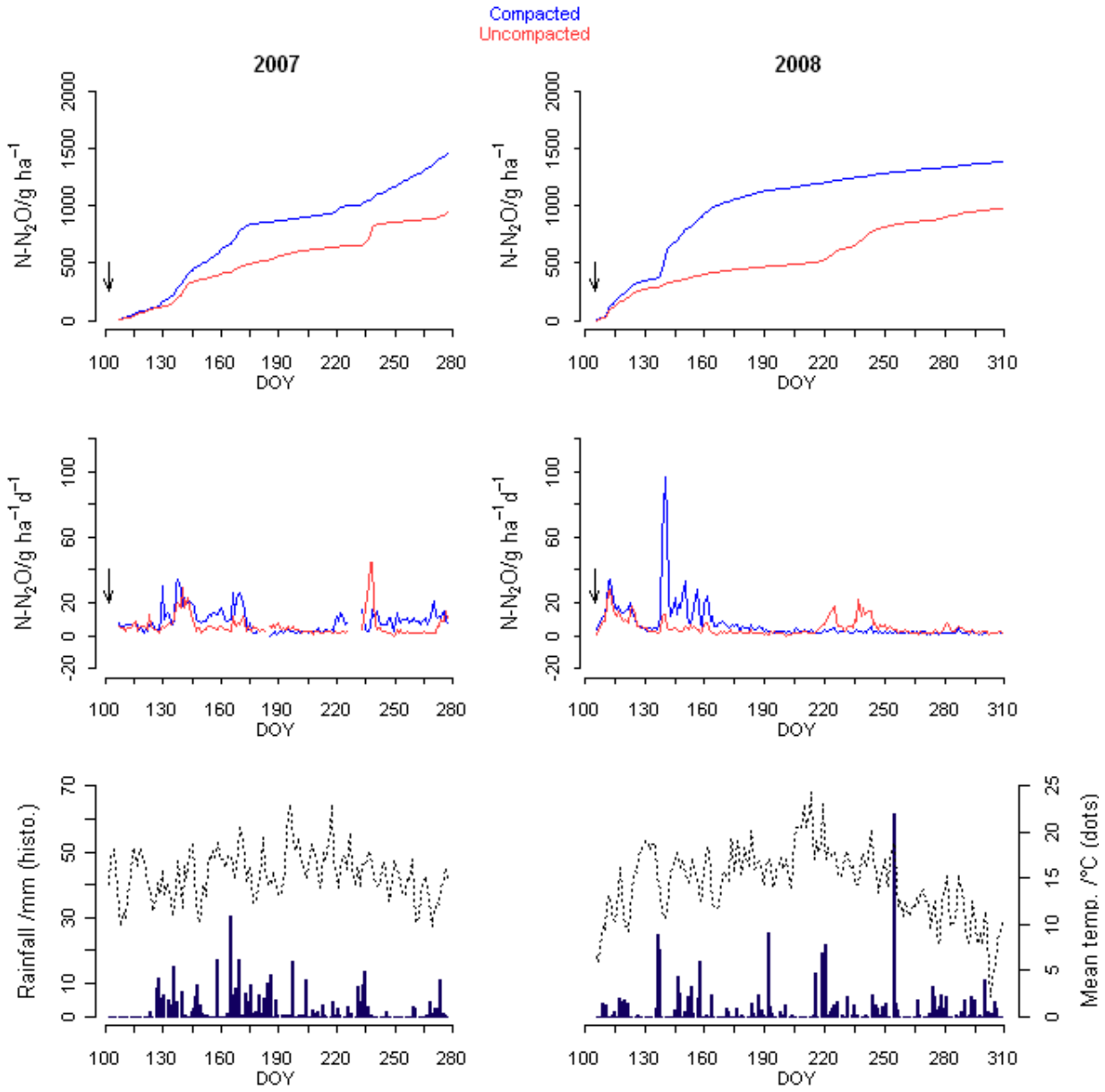
# Appendix 4

## Observed $N_2O$ fluxes in sugar beet plots 2007-2008

When referring to these measurements, please quote as following:

*Mary B., Gréhan E., Roussel M., Bessou C. (2007-2008) In Bessou C. Ph.D. thesis*

Arrows indicate the date of fertilization, i.e. 12/04/2007 & 14/04/2008

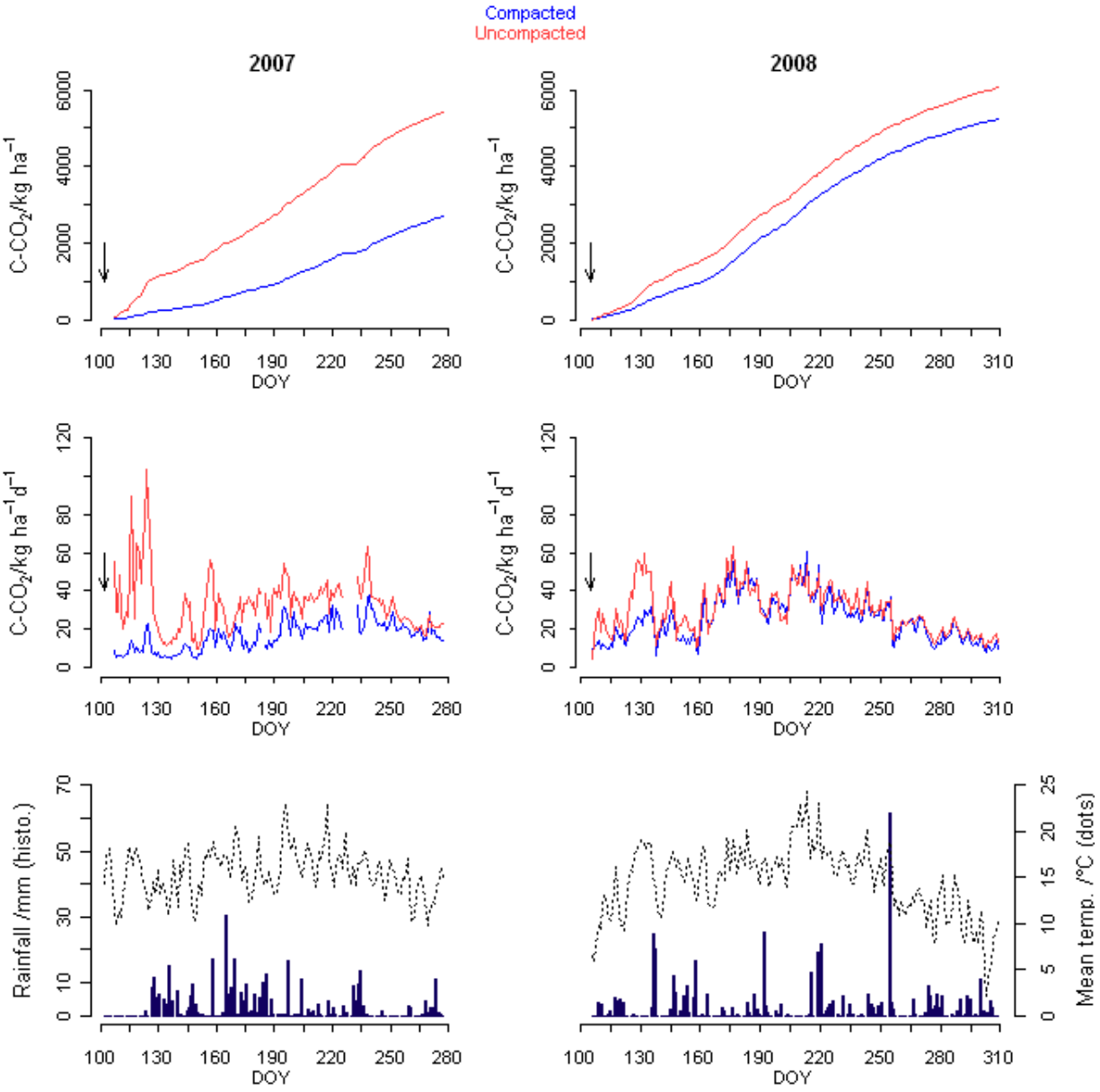


# Appendix 5

# Observed $CO_2$ fluxes in sugar beet plots 2007-2008

When referring to these measurements, please quote as following:

*Mary B., Gréhan E., Roussel M., Bessou C. (2007-2008) In Bessou C. Ph.D. thesis*

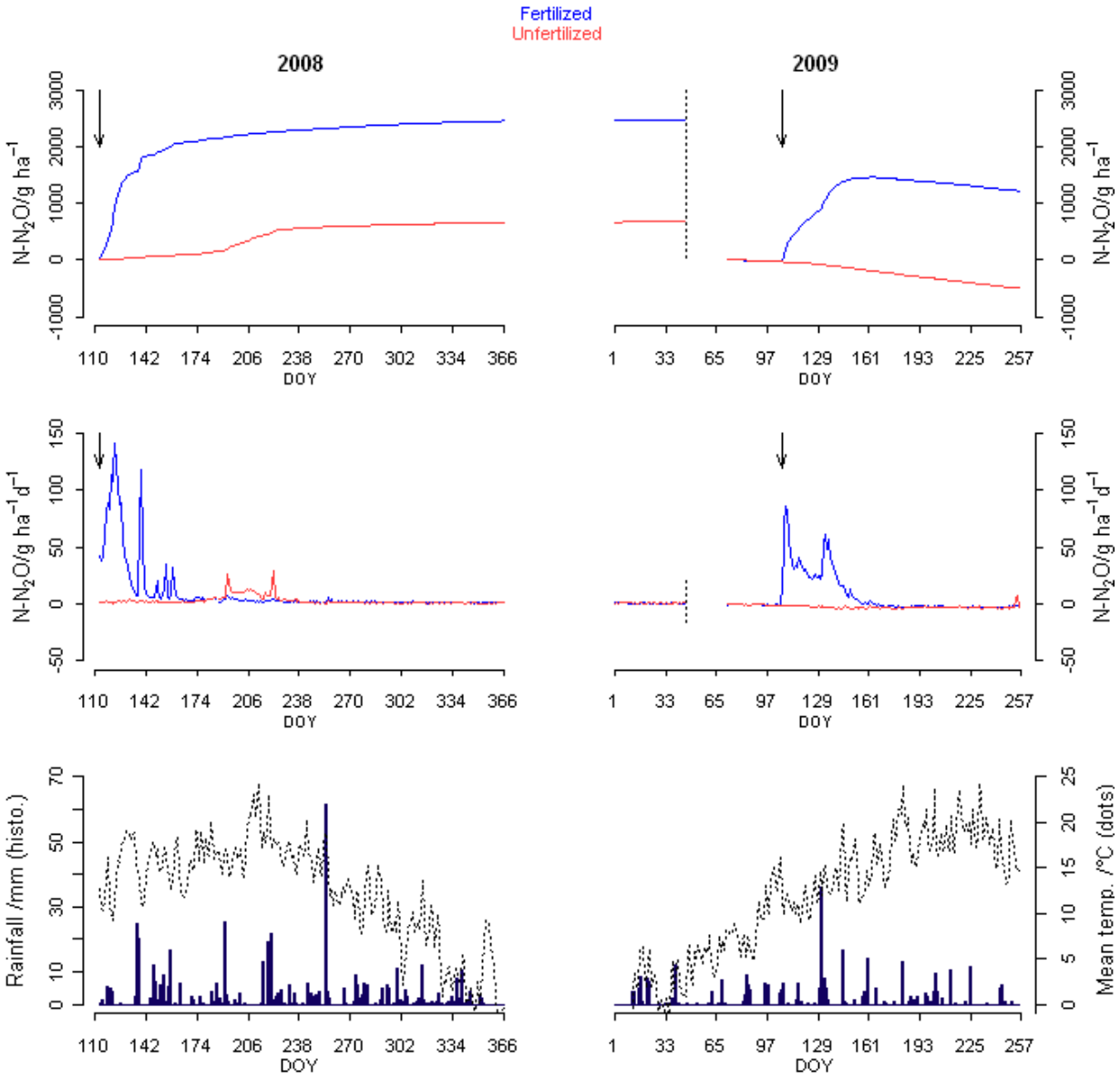


# Appendix 6

# Observed $N_2O$ fluxes in *Miscanthus* plots 2008-2009

When referring to these measurements, please quote as following:

*Mary B., Gréhan E., Roussel M., Bessou C. (2007-2008) In Bessou C. Ph.D. thesis*

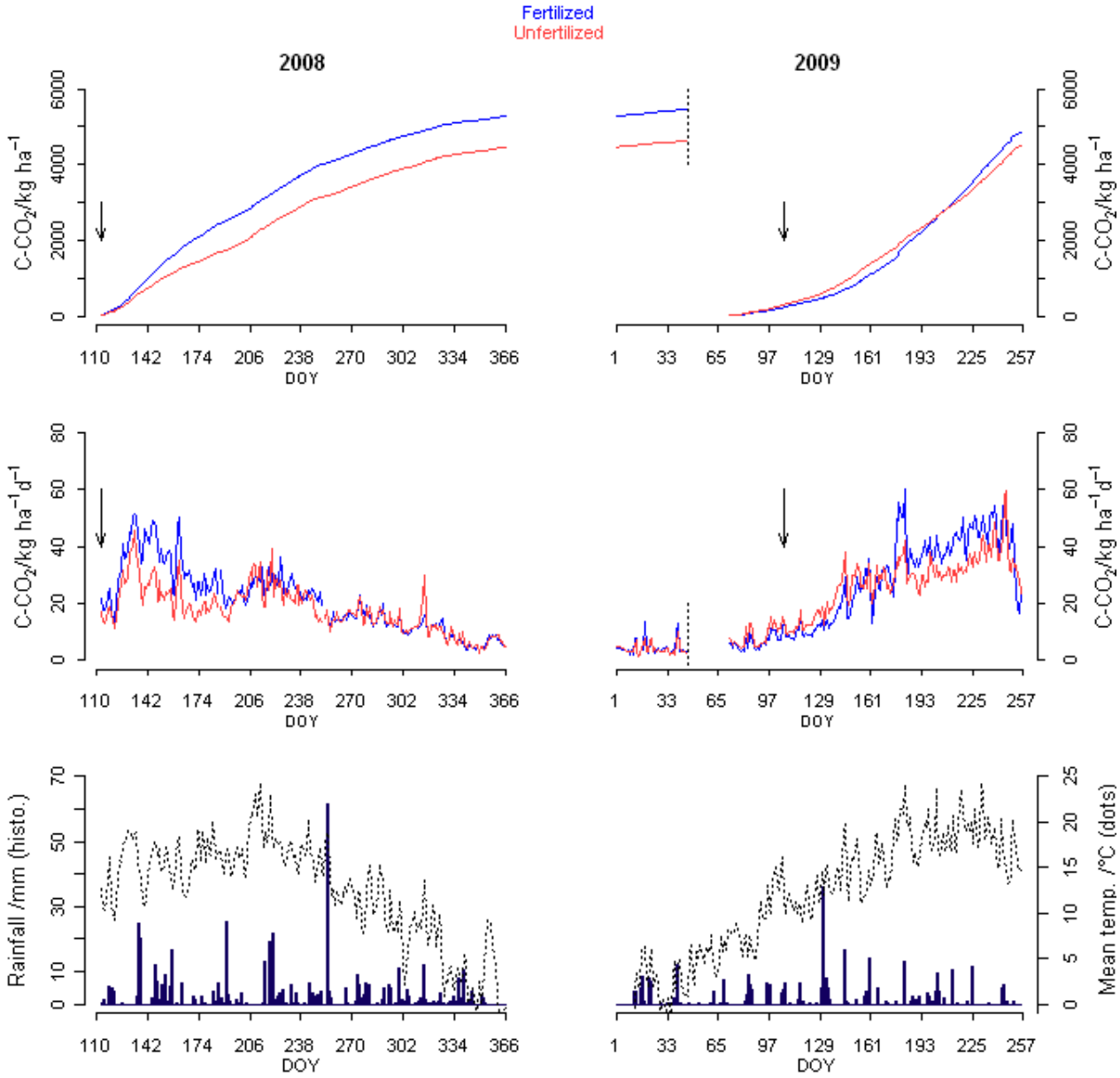


# Appendix 7

# Observed $CO_2$ fluxes in *Miscanthus* plots 2008-2009

When referring to these measurements, please quote as following:

*Mary B., Gréhan E., Roussel M., Bessou C. (2007-2008) In Bessou C. Ph.D. thesis*



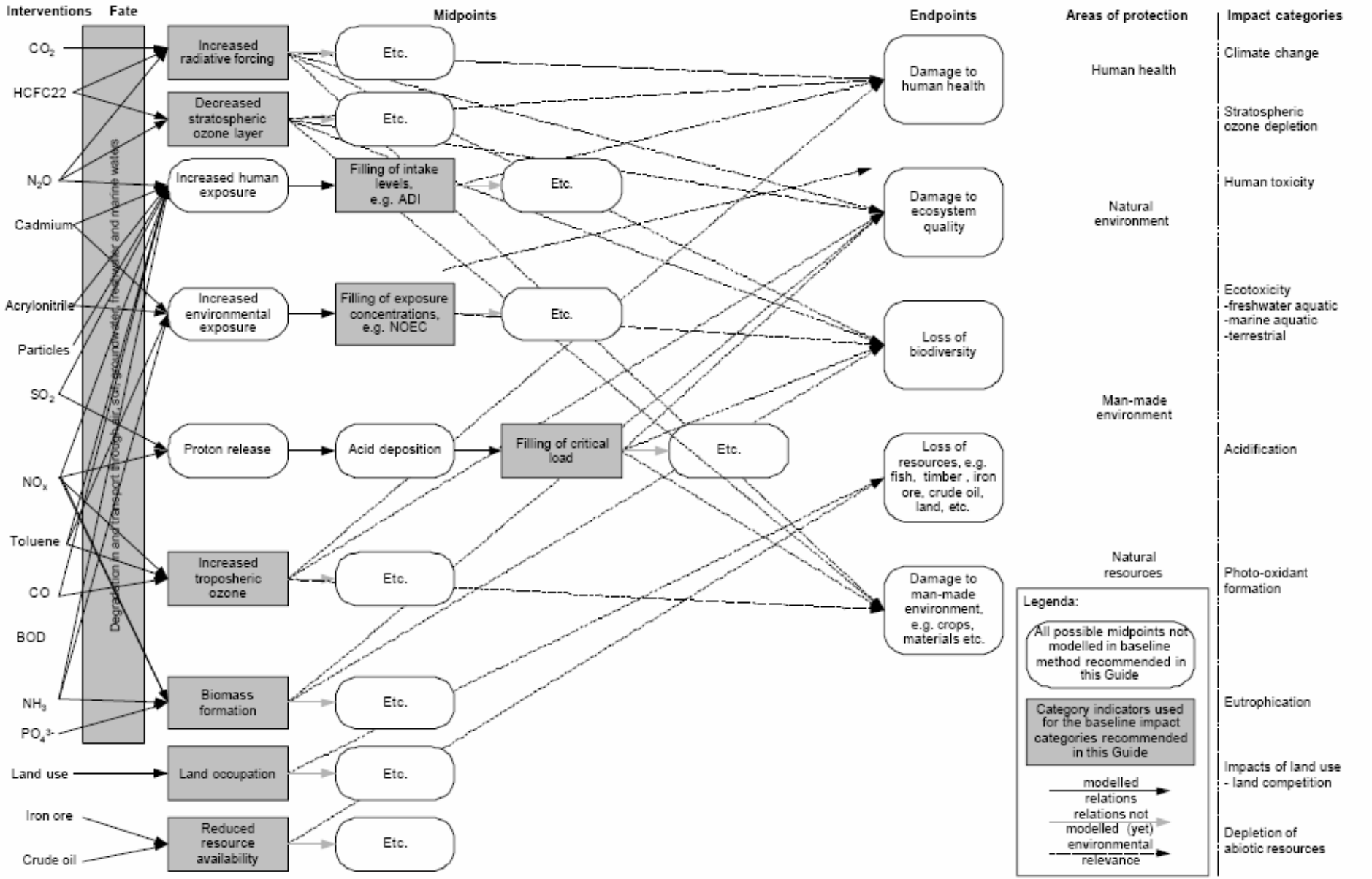



**Figure A8:** Currently modelled (midpoint) category indicators for the baseline impact categories recommended in the CML LCA Guide with an indication of their environmentel relevance (Guinée *et al.,* 2002)

Appendix 8

# Appendix 9

# Comparison of NOE & NOE2
# Bayesian calibration results (Lehuger S.)

We implemented NOE2 within CERES, which otherwise simulates $N_2O$ emissions by nitrification and denitrification with the NOE sub-model. These modifications consisted in adding the function of water content for the nitrification rate and the functions defined for the ratios of emitted $N_2O$ over total nitrified and denitrified nitrogen. The discretization of ammonium distribution and soil water and temperature within fine soil layers could not be implemented, since it also requires changes in other routines of the CERES code. In the current CERES version, N-fertilizer is distributed over the first 30 cm and nitrification and denitrification calculated over the first 20 cm with average temperature and water content for two sub-layers across these 20 cm. This finer discretization may be implemented in CERES through future model developments.

Using data sets of our continuous field emission monitoring in sugar beet plots, we proceeded to a Bayesian calibration to define the best set of parameter values. This calibration was done for all parameters of both NOE and NOE2 sub-models by adjusting CERES outputs for the sugar beet uncompacted plot (2007) with observed fluxes (Fig. A8-1 & A8-2). CERES performances with the best sets of parameter values for NOE or NOE2 were then tested against the other sets of observed fluxes and compared. Results are shown in Fig. A8-3, A8-4, A8-5, A8-6. Given the global better performances of CERES-NOE2 compared to CERES-NOE (RMSEP), we decided to use NOE2 to produce input data for our local LCAs.

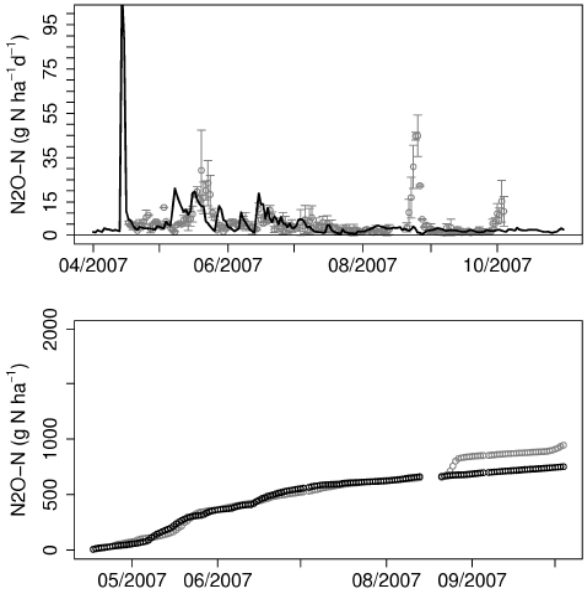



**Figure A8-1:** CERES-NOE (best set of parameter values)
Simulated fluxes in black, observed in grey
Uncompacted sugar beet plot in 2007
RMSE ($N_2O$ daily emissions): 7.3 g N-$N_2O$ $ha^{-1}$ $d^{-1}$
RMSE (cumulated $N_2O$ emissions): 79.08 g N-$N_2O$ $ha^{-1}$

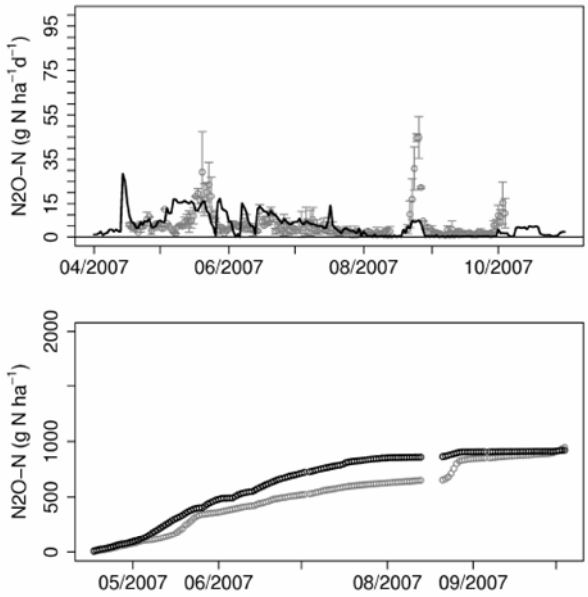



**Figure A8-2:** CERES-NOE2 (best set of parameter values)
Simulated fluxes in black, observed in grey
Uncompacted sugar beet plot in 2007
RMSE ($N_2O$ daily emissions): 6.97 g N-$N_2O$ $ha^{-1}$ $d^{-1}$
RMSE (cumulated $N_2O$ emissions): 138.89 g N-$N_2O$ $ha^{-1}$

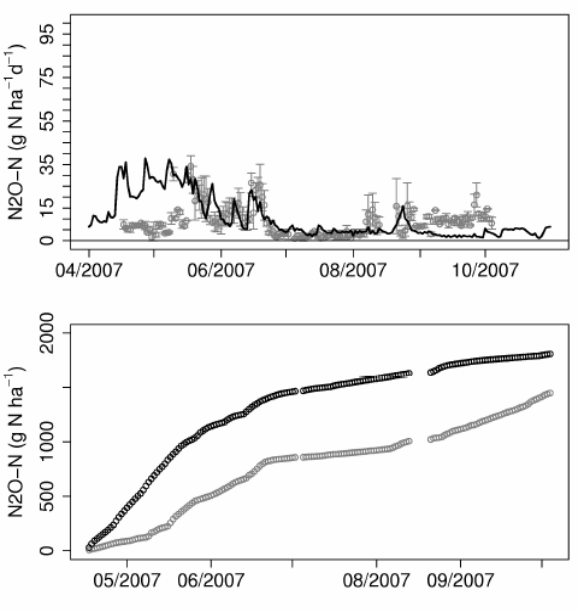



**Figure A8-3:** CERES-NOE (best set of parameter values)
Simulated fluxes in black, observed in grey
Compacted sugar beet plot in 2007
RMSEP ($N_2O$ daily emissions): 10.83 g N-$N_2O$ ha$^{-1}$ d$^{-1}$
RMSEP (cumulated emissions): 557.39 g N-$N_2O$ ha$^{-1}$

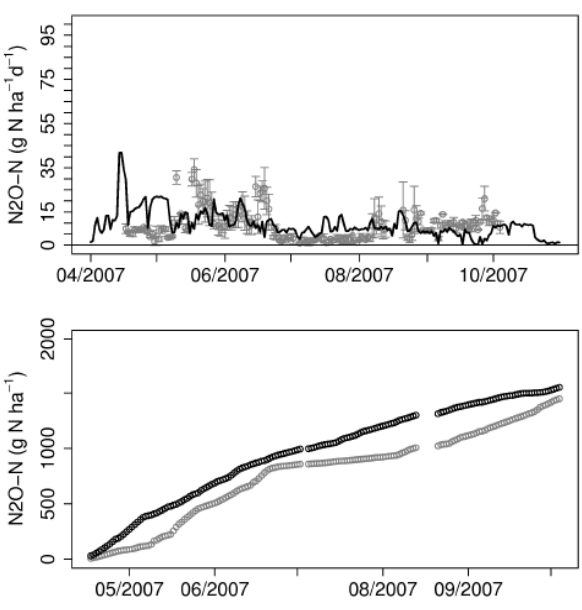



**Figure A8-4:** CERES-NOE2 (best set of parameter values)
Simulated fluxes in black, observed in grey
Compacted sugar beet plot in 2007
RMSEP ($N_2O$ daily emissions): 8.31 g N-$N_2O$ ha$^{-1}$ d$^{-1}$
RMSEP (cumulated $N_2O$ emissions): 207.91 g N-$N_2O$ ha$^{-1}$

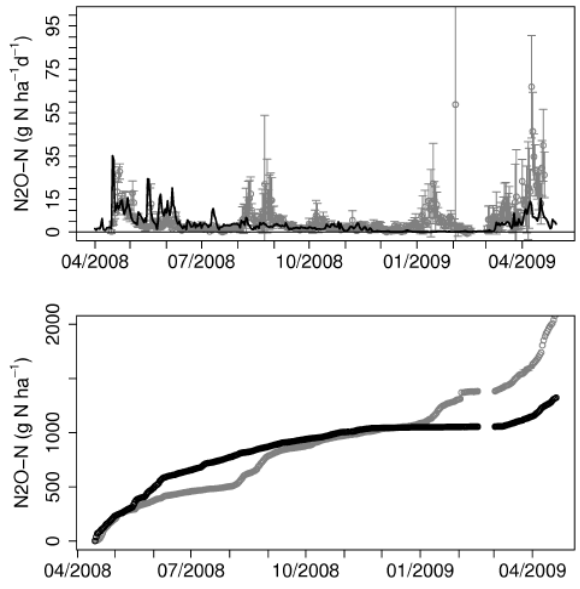



**Figure A8-5:** CERES-NOE (best set of parameter values)
Simulated fluxes in black, observed in grey
Uncompacted sugar beet plot in 2008
RMSEP ($N_2O$ daily emissions): 8.12 g N-$N_2O$ ha$^{-1}$ d$^{-1}$
RMSEP (cumulated emissions): 232.02 g N-$N_2O$ ha$^{-1}$

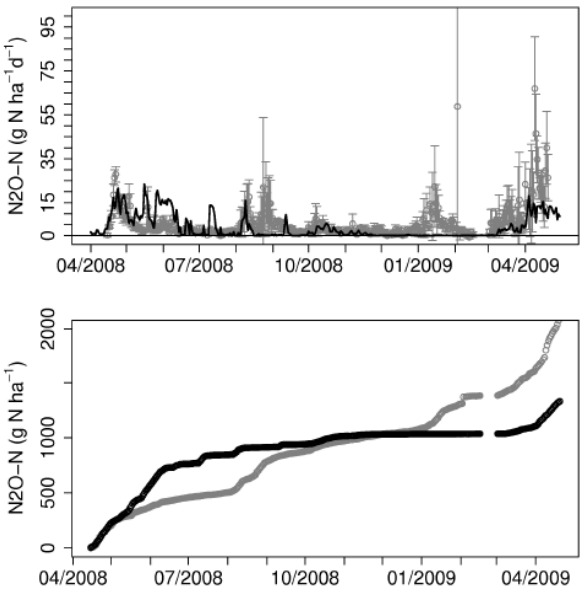
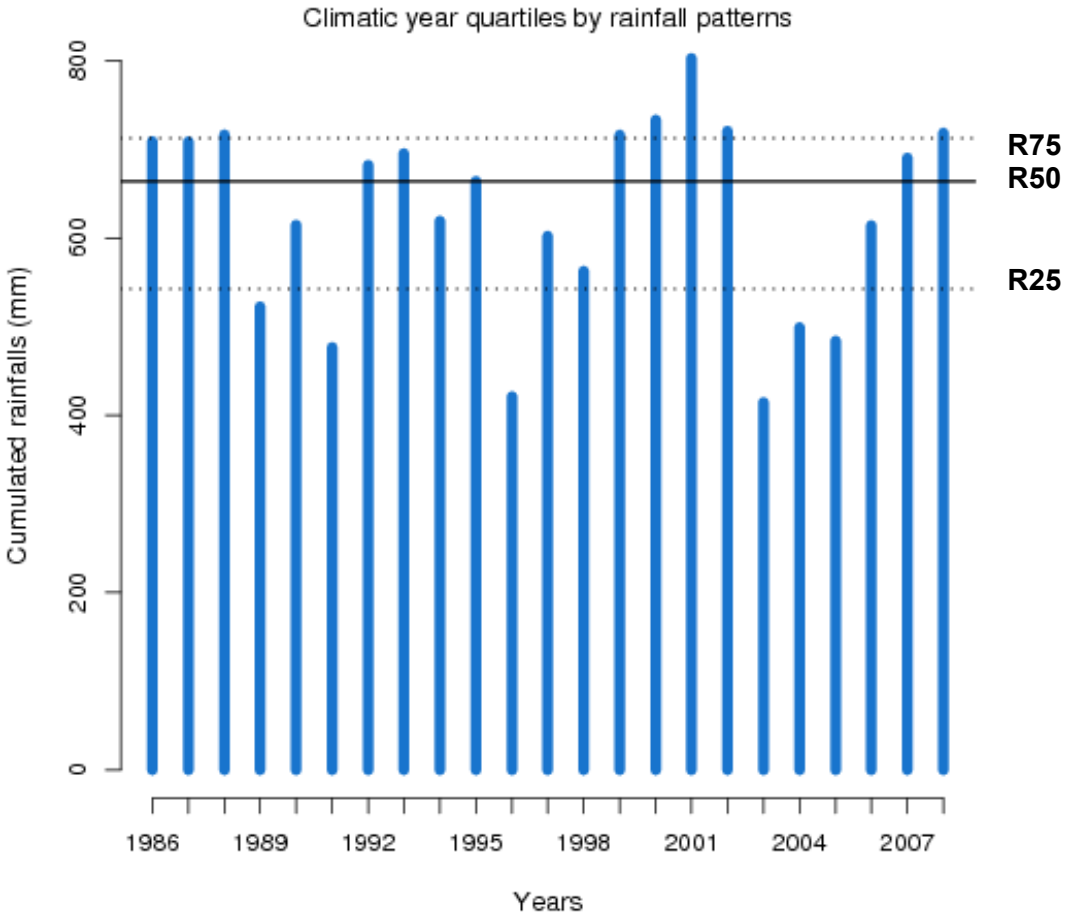



**Figure A8-6:** CERES-NOE2 (best set of parameter values)
Simulated fluxes in black, observed in grey
Compacted sugar beet plot in 2007
RMSE ($N_2O$ daily emissions): 8.04 g N-$N_2O$ ha$^{-1}$ d$^{-1}$
RMSE (cumulated $N_2O$ emissions): 266.78 g N-$N_2O$ ha$^{-1}$

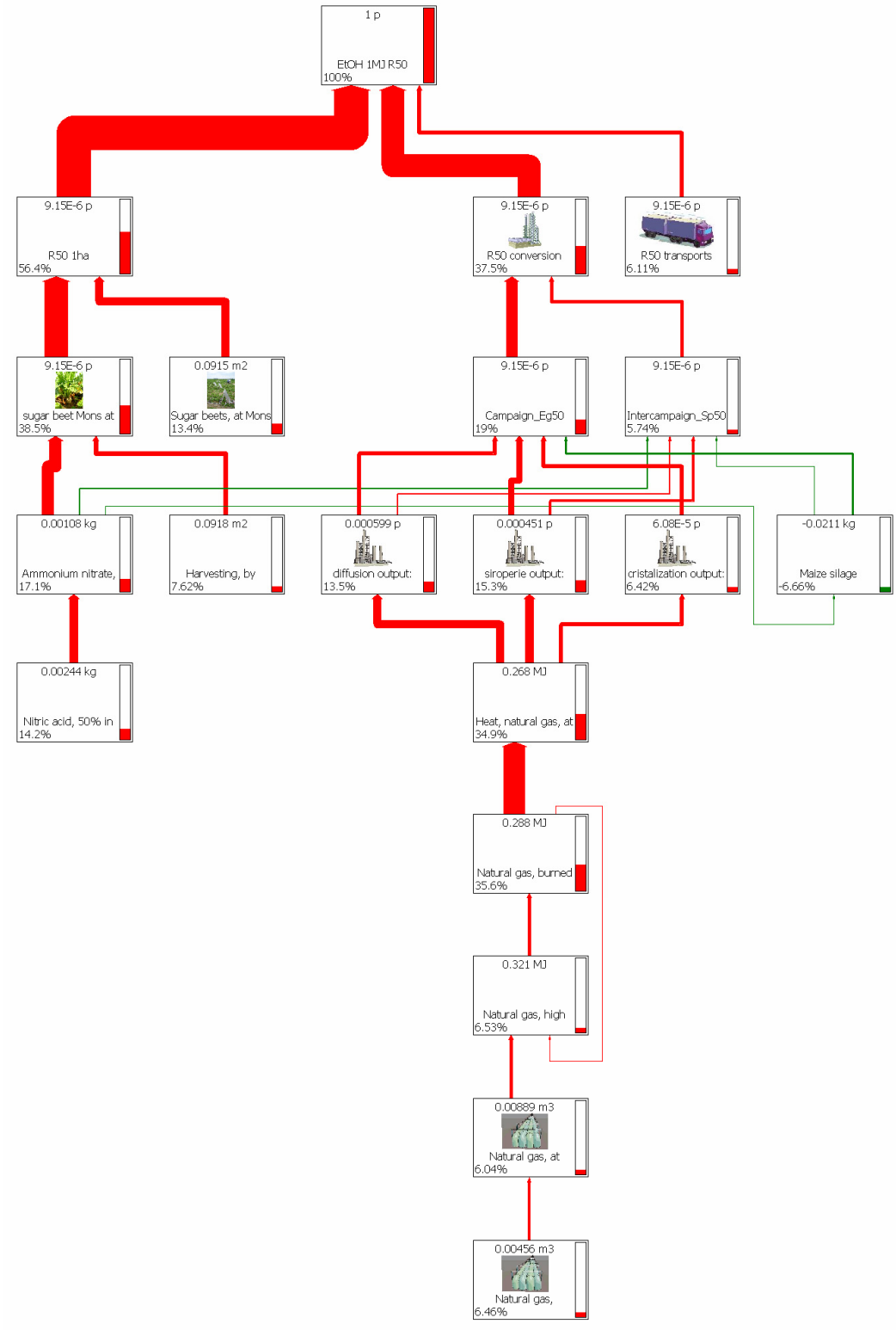
Appendix 10

# Climatic years in Estrées-Mons 1989-2009: rainfall patterns



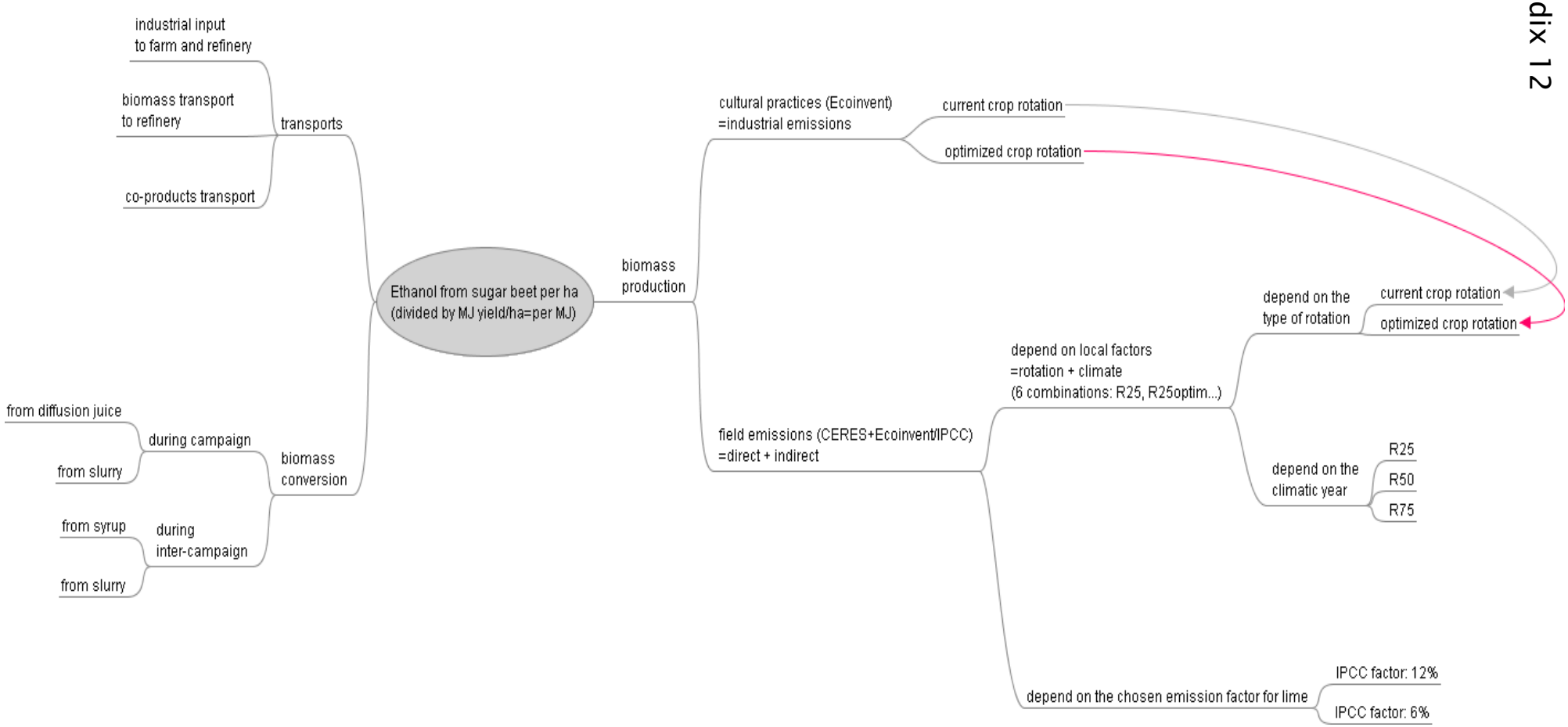
Appendix 11

# Ethanol from sugar beet R50: global warming characterization network Simapro, 5% shortcut



# Detailed inventory data for sugar beet ethanol

## 1 / Overview



# 2/ Cultural practices: detailed inventory data

## 2a/ Current crop rotation (per ha)

| | | | |
|---|---|---|---|
| Process | | | |
| Type | Assembly | | |
| Time frame | 2005-2009 | | |
| Geography | Europe, occidental | | |
| Technology | Best available technology | | |
| Name | | | |
| sugar beet Mons at farm gate | | | |
| | per ha | | |
| Materiial/assembly | | | |
| Sugar beet seed IP, at regional storehouse/CH U no pesticide | 10 | kg | 1.3 units/ha 1 unit=7.5kg |
| [thio]carbamate-compounds, at regional storehouse/RER U | 7.8 | g | Thiram with seeds NORDIKA 6g/unit |
| Pyretroid-compounds, at regional storehouse/RER U | 15.6 | g | Tefluthrin with seeds 12g/unit |
| Pesticide unspecified, at regional storehouse/RER U | 143 | g | Imidaclopride 90g/unit & Hymexazol 20g/unit with seeds |
| Carbofuran, at regional storehouse/RER U | 600 | g | insecticide at sowing 5% 12kg/ha (prohibited in 2008?) |
| Potassium chloride, as K2O, at regional storehouse/RER U | 110 | kg | |
| Triple superphosphate, as P2O5, at regional storehouse/RER U | 70 | kg | |
| Ammonium nitrate, as N, at regional storehouse/RER U | 140 | kg | |
| Borax, anhydrous, powder, at plant/RER U | 600 | g | 4L: 150g/L |
| Magnesium oxide, at plant/RER U | 165 | g | NUTRIB 3L: 1L=140g sulfur+55g magnesie+70g manganese |
| Manganese, at regional storage/RER U | 210 | g | |
| Sulphite, at plant/RER U | 420 | g | NUTRIB 3L: 1L=140g sulfur+55g magnesie+70g manganese |
| [thio]carbamate-compounds, at regional storehouse/RER U | 131 | g | KARATE K 1.25L ( lambda-cyhalothrin + pyrimicarbe) 105g/L |
| [sulfonyl]urea-compounds, at regional storehouse/RER U | 20 | g | SAFARI 20g/ha |
| [thio]carbamate-compounds, at regional storehouse/RER U | 236 | g | 4 x herbicides (mix) |
| | | | BETANAL 157g/L et 1.5L: 236g |
| Triazine-compounds, at regional storehouse/RER U | 400 | g | GOLTIX 300g/ha + VENZAR 100g/ha + (oil 0.4L + solvent 0.1L below) |
| Solvents, organic, unspecified, at plant/GLO U | 2 | kg | oil 0.4L+ solvant 0.1L: approximation 1L=1kg |
| Tap water, at user/RER U | 1600 | kg | 100L/ha/passage x 2 pr for cleaning: 2 ferti+4herbi+1insec+1nutri |
| Lime, from carbonation, at regional storehouse/CH U | 0.6 | kg | |
| | | | |
| Processus | | | |
| Tillage, ploughing/CH U | 1 | ha | incorporation of residus by ploughing |
| Fertilising, by broadcaster/CH U | 2 | ha | ferti |
| Tillage, harrowing, by spring tine harrow/CH U | 1 | ha | seedbed: crosskill+vibroculteur |
| Sowing/CH U | 1 | ha | |
| Application of plant protection products, by field sprayer/CH U | 6 | ha | phyto + Bore=6 passages |
| Harvesting, by complete harvester, beets/CH U | 1 | ha | |
| Solid manure loading and spreading, by hydraulic loader and spreader/CH U | 600 | kg | lime application |

# 2b/ Optimized crop rotation (per ha)

| | |
|---|---|
| Process | |
| Type | Assembly |
| Time frame | 2005-2009 |
| Geography | Europe, occidental |
| Technology | Best available technology |

| Name | | | |
|---|---|---|---|
| sugar beet Mons at farm gate_optimized rotation | per ha | | |
| | | | |
| | | | |
| Matériaux/assemblages | | | |
| Sugar beet seed IP, at regional storehouse/CH U no pesticide | 10 | kg | 1.3 units/ha 1 unit=7.5kg |
| [thio]carbamate-compounds, at regional storehouse/RER U | 7.8 | g | Thiram with seeds NORDIKA 6g/unit |
| Pyretroid-compounds, at regional storehouse/RER U | 15.6 | g | Tefluthrin with seeds 12g/unit |
| Pesticide unspecified, at regional storehouse/RER U | 143 | g | Imidaclopride 90g/unit & Hymexazol 20g/unit with seeds |
| Carbofuran, at regional storehouse/RER U | 600 | g | insecticide at sowing 5% 12kg/ha (prohibited in 2008?) |
| Potassium chloride, as K2O, at regional storehouse/RER U | 110 | kg | |
| Triple superphosphate, as P2O5, at regional storehouse/RER U | 70 | kg | |
| Ammonium nitrate, as N, at regional storehouse/RER U | 125 | kg | |
| Borax, anhydrous, powder, at plant/RER U | 600 | g | 4L: 150g/L |
| Magnesium oxide, at plant/RER U | 165 | g | NUTRIB 3L: 1L=140g sulfur+55g magnesie+70g manganese |
| Manganese, at regional storage/RER U | 210 | g | NUTRIB 3L: 1L=140g sulfur+55g magnesie+70g manganese |
| Sulphite, at plant/RER U | 420 | g | NUTRIB 3L: 1L=140g sulfur+55g magnesie+70g manganese |
| [thio]carbamate-compounds, at regional storehouse/RER U | 131 | g | KARATE K 1.25L ( lambda-cyhalothrin + pyrimicarbe) 105g/L |
| [sulfonyl]urea-compounds, at regional storehouse/RER U | 20 | g | SAFARI 20g/ha |
| [thio]carbamate-compounds, at regional storehouse/RER U | 236 | g | 4 x herbicide mix |
| | | | BETANAL 157g/L et 1.5L: 236g |
| Triazine-compounds, at regional storehouse/RER U | 400 | g | GOLTIX 300g/ha + VENZAR 100g/ha + (oil 0.4L + solvent 0.1L below) |
| Solvents, organic, unspecified, at plant/GLO U | 2 | kg | oil 0.4L+ solvent 0.1L: approximation 1L=1kg |
| Tap water, at user/RER U | 1400 | kg | 100L/ha/passage x 2 for washing: 2 ferti+4herbi+1insec+1nutri |
| Rape seed IP, at regional storehouse/CH U | 1.5 | kg | cover crop 3kg |
| Lime, from carbonation, at regional storehouse/CH U | 0.6 | kg | 3t every 5 yrs (pers com. DAMAY, CHAUCHARD) |
| | | | |
| Processus | | | |
| Tillage, ploughing/CH U | 1 | ha | incorporation of residues by ploughing |
| Fertilising, by broadcaster/CH U | 1 | ha | ferti |
| Tillage, harrowing, by spring tine harrow/CH U | 1 | ha | seedbed: crosskill+vibroculteur |
| Sowing/CH U | 1 | ha | |
| Application of plant protection products, by field sprayer/CH U | 6 | ha | phyto + Bore=6 passages |
| Harvesting, by complete harvester, beets/CH U | 1 | ha | |
| Solid manure loading and spreading, by hydraulic loader and spreader/CH U | 600 | kg | lime application |
| Mulching/CH U | 0.5 | ha | covercrop |
| Fertilising, by broadcaster/CH U | 0.5 | ha | covercrop sowing |

# 3/ Field emissions: detailed inventory data

| | |
|---|---|
| Type | Unit process |
| Time frame | 2005-2009 |
| Geographie | Europe, occidental |
| Technology | Best available technology |
| Products | |

| Sugar beets, at Mons per ha | sub-compartment | units (molecular weight basis) | R25 | R25 optim | R50 | R50 optim | R75 | R75 optim |
|---|---|---|---|---|---|---|---|---|
| Emissions to air | | | | | | | | |
| Ammonia | low. pop. | kg | 6.55 | 4.23 | 5.91 | 3.99 | 8.87 | 5.89 |
| Nitrogen oxides | low. pop. | g | 1820.00 | 1516.00 | 1723.00 | 1393.00 | 1737.00 | 1407.00 |
| Dinitrogen monoxide indirect | low. pop., long-term | g | 160.00 | 1528.00 | 2110.00 | 1765.00 | 1926.00 | 1817.00 |
| Dinitrogen monoxide | low. pop. | g | 1693.00 | 163.00 | 604.00 | 243.00 | 617.00 | 338.00 |
| | | | | | | | | |
| Emissions to water | | | | | | | | |
| Nitrate | groundwater | kg | 23.00 | 36.00 | 193.00 | 68.00 | 184.00 | 94.00 |
| Phosphate | river | kg | 0.63 | 1.89 | 0.21 | 1.89 | 1.89 | 0.63 |
| Phosphate | river | kg | 1.89 | 0.63 | 1.89 | 0.63 | 0.63 | 1.89 |
| Phosphate | groundwater | kg | 0.21 | 0.21 | 0.63 | 0.21 | 0.21 | 0.21 |
| | | | | | | | | |
| Emissions to soil | | | | | | | | |
| Lambda-cyhalothrin | agricultural | g | 131.00 | 131.00 | 131.00 | 131.00 | 131.00 | 131.00 |
| Thiram | agricultural | g | 7.80 | 7.80 | 7.80 | 7.80 | 7.80 | 7.80 |
| Tefluthrin | agricultural | g | 15.60 | 15.60 | 15.60 | 15.60 | 15.60 | 15.60 |
| Triflusulfuron-methyl | agricultural | g | 10.00 | 10.00 | 10.00 | 10.00 | 10.00 | 10.00 |
| Metamitron | agricultural | g | 210.00 | 210.00 | 210.00 | 210.00 | 210.00 | 210.00 |
| Phenmedipham | agricultural | g | 236.00 | 236.00 | 236.00 | 236.00 | 236.00 | 236.00 |
| Cadmium | agricultural | kg | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 |
| Chromium | agricultural | kg | 0.23 | 0.23 | 0.23 | 0.23 | 0.23 | 0.23 |
| Copper | agricultural | kg | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 |
| Lead | agricultural | kg | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Nickel | agricultural | kg | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 |
| Zinc | agricultural | kg | 0.08 | 0.08 | 0.08 | 0.08 | 0.08 | 0.08 |

$CO_2$ field emissions following lime application (kg/ha molecular weight basis):

high IPCC emission factor (12%): 264 kg/ha

low IPCC emission factor (6%): 132 kg/ha

# 4/ Conversion to ethanol: detailed inventory data (based on ADEME/DIREM 2002)

| | Diffusion | | | R25 | R25 optim | R50 | R50 optim | R75 | R75 optim |
|---|---|---|---|---|---|---|---|---|---|
| | | yields t/ha (16.24% sugar) | | 75.91 | 64.04 | 94.53 | 79.02 | 96.42 | 79.69 |
| | | brut (17% sugar; 16% ) | | 85 | 72 | 106 | 89 | 108 | 89 |
| Products | | yields t/ha (17% sugar, less stones & sand) | | 71 | 60 | 89 | 74 | 91 | 75 |
| Brut beets (input) | 1000.00 | kg | x 1000 | 85058 | 71756 | 105917 | 88542 | 108032 | 89284 |
| Natural gas | 166.75 | MJ | | 14183 | 11965 | 17661 | 14764 | 18014 | 14888 |
| Soude 50% | 0.31 | kg | | 27 | 23 | 33 | 28 | 34 | 28 |
| Sulfuric acid | 0.37 | kg | | 32 | 27 | 40 | 33 | 40 | 33 |
| N ammonitrate | 0.14 | kg N | | 12 | 10 | 15 | 12 | 15 | 13 |
| P TSP | 0.11 | kg P2O5 | | 9 | 8 | 12 | 10 | 12 | 10 |
| K Potasse | 0.32 | kg K2O | | 27 | 23 | 34 | 28 | 35 | 29 |
| Stones + sand | 21.45 | kg | | 1824 | 1539 | 2272 | 1899 | 2317 | 1915 |
| Over-pressed pulp | 140.65 | kg | | 11964 | 10093 | 14898 | 12454 | 15195 | 12558 |
| **Diffusion juice (output)** | 1068.16 | kg | | 90855 | 76647 | 113136 | 94577 | 115395 | 95369 |
| | juice converted to syrup | | 83% | 75083 | 63341 | 93496 | 78159 | 95363 | 78813 |
| | juice converted directly to ethanol | | 17% | 15772 | 13306 | 19640 | 16419 | 20033 | 16556 |
| | | | tkm (beet) | 653 | 556 | 806 | 679 | 822 | 684 |
| | | | total Raw Material /kg | 59 | 49 | 73 | 61 | 74 | 62 |
| | | | tkm (pulp) | 117 | 103 | 139 | 121 | 141 | 121 |
| | **Fermentation during campaign** | | | | | | | | |
| | | | | 15772 | 13306 | 19640 | 16419 | 20033 | 16556 |
| Diffusion juice (input) | 1000.00 | kg | : 1000 | 16 | 13 | 20 | 16 | 20 | 17 |
| Slurry | 58.08 | kg | | 916 | 773 | 1141 | 954 | 1163 | 962 |
| Electricity | 4.41 | MJ elec | | 70 | 59 | 87 | 72 | 88 | 73 |
| Natural gas | 18.38 | MJ | | 290 | 245 | 361 | 302 | 368 | 304 |
| Sulfuric acid | 0.04 | kg | | 0.7 | 0.6 | 0.9 | 0.7 | 0.9 | 0.7 |
| **Vine (output)** | 103.18 | litre | | 1627 | 1373 | 2026 | 1694 | 2067 | 1708 |
| | | | total Raw Material /kg | 0.7 | 0.6 | 0.9 | 0.7 | 0.9 | 0.7 |
| | **Distillation during campaign** | | | | | | | | |
| | | | | 1627 | 1373 | 2026 | 1694 | 2067 | 1708 |
| Vine (input) | 1000 | litre | : 1000 | 1.6 | 1.4 | 2.0 | 1.7 | 2.1 | 1.7 |
| Electricity | 32.67 | MJ elec | | 53 | 45 | 66 | 55 | 68 | 56 |
| Natural gas | 3067.299 | MJ | | 4992 | 4211 | 6216 | 5196 | 6340 | 5240 |
| **Alcool (output)** | 794 | kg | | 1292 | 1090 | 1609 | 1345 | 1641 | 1356 |
| | **Conversion to syrup** | | | | | | | | |
| Diffusion juice (input) | 1000 | kg | : 1000 | 75 | 63 | 93 | 78 | 95 | 79 |
| Natural gas | 242.74 | MJ | | 18225 | 15375 | 22695 | 18972 | 23148 | 19131 |
| Limestone | 23.26 | kg | | 1747 | 1473 | 2175 | 1818 | 2218 | 1833 |
| Coke | 1.92 | kg | | 144 | 122 | 180 | 150 | 183 | 151 |
| N ammonitrate | 0.21 | kg N | | 16 | 14 | 20 | 17 | 20 | 17 |
| P TSP | 0.62 | kg P2O5 | | 46 | 39 | 58 | 48 | 59 | 49 |
| K Potasse | 0.04 | kg K2O | | 3 | 2 | 3 | 3 | 3 | 3 |
| **Syrup (ouput)** | 208.26 | kg | | 15637 | 13192 | 19472 | 16278 | 19860 | 16414 |
| | | | | 16 | 13 | 19 | 16 | 20 | 16 |
| | | stored | 20% | 3 | 3 | 4 | 3 | 4 | 3 |
| | via crystallization | | 80% | 13 | 11 | 16 | 13 | 16 | 13 |
| | stored & fermented during inter-campaign | | 73% | 2.3 | 1.9 | 2.8 | 2.4 | 2.9 | 2.4 |
| | stored then crystallized | | 28% | 1.6 | 1.4 | 2.0 | 1.7 | 2.1 | 1.7 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | **Crystallization during campaign** | | | | | | | | |
| Siyrup (input) | 1000.00 | kg | | 13 | 11 | 16 | 13 | 16 | 13 |
| Natural gas | 918.43 | MJ | | 10133 | 8548 | 12618 | 10548 | 12870 | 10636 |
| **Sugar (output)** | 440.13 | kg | | 5506 | 4645 | 6856 | 5731 | 6993 | 5779 |
| **Slurry (output)** | 347.32 | kg | | 4345 | 3665 | 5410 | 4523 | 5518 | 4561 |
| | **Crystallization during inter-campaign** | | | | | | | | |
| Siyrup (input) | 1000.00 | kg | | 1.6 | 1.4 | 2.0 | 1.7 | 2.1 | 1.7 |
| Natural gas | 918.43 | MJ | | 1332 | 1123 | 1658 | 1386 | 1691 | 1398 |
| **Sugar (output)** | 215.96 | kg | | 355 | 299 | 442 | 370 | 451 | 373 |
| **Slurry (output)** | 571.49 | kg | | 939 | 793 | 1170 | 978 | 1193 | 986 |
| | **Fermentation during inter-campaign** | | | | | | | | |
| Diffusion juice (input) | 1000.00 | kg | | 2.27 | 1.91 | 2.82 | 2.36 | 2.88 | 2.38 |
| Slurry | 1752.82 | kg | | 3974 | 3353 | 4949 | 4137 | 5048 | 4172 |
| Electricity | 130.36 | MJ elec | | 296 | 249 | 368 | 308 | 375 | 310 |
| Natural gas | 87.18 | MJ | | 198 | 167 | 246 | 206 | 251 | 207 |
| Phosphoric acid 52% | 6.22 | kg | | 14.1 | 11.9 | 17.6 | 14.7 | 17.9 | 14.8 |
| Chlorhydric acid 100% | 9.66 | kg | | 21.9 | 18.5 | 27.3 | 22.8 | 27.8 | 23.0 |
| **Vine (output)** | 1012.20 | litre | | 2295 | 1936 | 2858 | 2389 | 2915 | 2409 |
| | | | total Raw Material /kg | 36 | 30 | 45 | 37 | 46 | 38 |
| | **Distillation during inter-campaign** | | | | | | | | |
| Vine (input) | 1000.00 | litre | : 1000 | 2.30 | 1.94 | 2.86 | 2.39 | 2.91 | 2.41 |
| Electricity | 421.19 | MJ elec | | 967 | 815 | 1204 | 1006 | 1228 | 1015 |
| Natural gas | 2689.65 | MJ | | 6173 | 5207 | 7687 | 6426 | 7840 | 6479 |
| N ammonitrate | 9.75 | kg N | | 22 | 19 | 28 | 23 | 28 | 23 |
| P TSP | 2.72 | kg P2O5 | | 6 | 5 | 8 | 7 | 8 | 7 |
| K Potasse | 28.21 | kg K2O | | 65 | 55 | 81 | 67 | 82 | 68 |
| Vinasses | 302.38 | kg | | 694 | 585 | 864 | 722 | 881 | 728 |
| **Alcool (output)** | 794 | kg | | 1822 | 1537 | 2269 | 1897 | 2314 | 1913 |
| | **Dehydration** | | | | | | | | |
| Alcool (input) | 1000.00 | kg | | 3.11 | 2.63 | 3.88 | 3.24 | 3.96 | 3.27 |
| Electricity | 18.59 | MJ elec | | 58 | 49 | 72 | 60 | 74 | 61 |
| Natural gas | 1369.50 | MJ | | 4265 | 3598 | 5311 | 4440 | 5417 | 4477 |
| **Ethanol (output)** | 1003.01 | kg | | 3124 | 2635 | 3890 | 3252 | 3967 | 3279 |

# Appendix 13

# Detailed inventory data for *Miscanthus* ethanol

## 1/ Detailed inventory data for the production of 1t of rhizome (1 ha provides 10t of useful rhizomes)

| | |
|---|---|
| Process | |
| Type | Unit process |
| Time frame | 2005-2009 |
| Geography | Europe, occidental |
| Technology | Best available technology |

| | | | | |
|---|---|---|---|---|
| Products | | | | |
| Miscanthus rhizome U/FR ss co2 | 1 | ton | 1 ha yields 10t of useful rhizomes | |
| | | | | |
| Avoided products | | | | |
| | | | | |
| Resources | | | | |
| Occupation, arable | 0.3 | ha | | |
| | | | | |
| Materials/fuels | | | | |
| Glyphosate, at regional storehouse/RER U | 0.072 | kg | 0.720kg per ha (Lehuger), 1 yr | |
| Ammonium nitrate, as N, at regional storehouse/RER U | 10 | kg | 1/10 of input 2nd yr only | |
| Triple superphosphate, as P2O5, at regional storehouse/RER U | 4 | kg | 1/10 of input 1st yr only | |
| Potassium chloride, as K2O, at regional storehouse/RER U | 6 | kg | 1/10 of input 1st yr only | |
| Lime, from carbonation, at regional storehouse/CH U | 0.06 | ton | 1/10 (3t/15 yrs) DEFRA,2002 for 20yrs | |
| | | | | |
| Electricity/heat | | | | |
| Tillage, ploughing/CH U | 0.1 | ha | | |
| Tillage, cultivating, chiselling/CH U | 0.1 | ha | | |
| Tillage, harrowing, by rotary harrow/CH U | 0.3 | ha | | |
| Solid manure loading and spreading, by hydraulic loader and spreader/CH U | 0.06 | ton | lime | |
| Fertilising, by broadcaster/CH U | 0.2 | ha | N; P+K | |
| Potato planting/CH U | 0.1 | ha | | |
| Irrigating/ha/CH U | 0.1 | ha | | |
| Application of plant protection products, by field sprayer/CH U | 0.1 | ha | weed control | |
| Hoeing/CH U | 0.1 | ha | weed control | |
| Mulching/CH U | 0.2 | ha | | |
| Mowing, by rotary mower/CH U | 0.1 | ha | | |
| Tillage, rotary cultivator/CH U | 0.1 | ha | cutting rhizomes | |
| Tillage, hoeing and earthing-up, potatoes/CH U | 0.1 | ha | lifting rhizome | |
| Harvesting, by complete harvester, potatoes/CH U | 0.1 | ha | | |
| Potato grading/CH U | 0.1 | kg | | |
| Transport, lorry >16t, fleet average/RER U | 88.4 | tkm | inputs to farm: hypothesis 65km | |
| | | | | |
| Emissions to air | | | | |
| Carbon dioxide, fossil | 26.4 | kg | low. pop. | lime: 3t pr 15yr |
| Dinitrogen monoxide | 0.04020079 | kg | low. pop. | |
| Ammonia, as N | -0.0367443 | kg | low. pop. | |
| Nitrogen oxides | 0.00547986 | kg | low. pop. | |
| Dinitrogen monoxide | 0.00979708 | kg | low. pop., long-term | indirect |
| | | | | |
| Emissions to water | | | | |
| Nitrate | 0.95719143 | kg | groundwater | |
| Phosphate | 0.05516129 | kg | groundwater | |
| Phosphate | 0.15169355 | kg | river | |
| | | | | |
| Emissions to soil | | | | |
| Glyphosate | 0.072 | kg | agricultural | |
| Cadmium | 0.00046176 | kg | agricultural | |
| Zinc | 0.00535822 | kg | agricultural | |
| Copper | 0.0009358 | kg | agricultural | |
| Nickel | 0.0016083 | kg | agricultural | |
| Chromium | 0.02127504 | kg | agricultural | |
| Lead | 0.00037056 | kg | agricultural | |

## 2/ Detailed inventory data for Miscanthus production (50 N-kg ): emissions from1 ha

annual mean production over 15 yrs: 11.18tDM/yr, emissions from lime not shown

| | |
|---|---|
| Process | |
| Type | Unit process |
| Time frame | 2005-2009 |
| Geography | Europe, occidental |
| Technology | Best available technology |

| | | | |
|---|---|---|---|
| Products | | | |
| Miscanthus 15 yrs N50 U/FR ss co2 | 1 ha N50: annual mean over 15 yrs: 11.18tDM | | |
| | | | |
| Avoided products | | | |
| | | | |
| Resources | | | |
| Occupation, arable | land | 1 | ha |
| | | | |
| Materials/fuels | | | |
| Miscanthus lime emission, high 1 ha yr-1 | | 1 | ha |
| | | | |
| Electricity/heat | | | |
| | | | |
| Emissions to air | | | |
| Ammonia | low. pop. | -1.3111857 | kg |
| Nitrogen oxides | low. pop. | 0.2652594 | kg |
| Dinitrogen monoxide | low. pop. | 1.769629 | kg |
| Dinitrogen monoxide | low. pop. | 0.1295311 | kg |
| | | | |
| Emissions to water | | | |
| Nitrate | groundwater | 41.56539 | kg |
| Phosphate | groundwater | 0.183871 | kg |
| Phosphate | river | 0.5056452 | kg |
| | | | |
| Emissions to soil | | | |
| Glyphosate | agricultural | 0.288 | kg |
| Cadmium | agricultural | 0.0003342 | kg |
| Copper | agricultural | 0.0022819 | kg |
| Lead | agricultural | 0.0010994 | kg |
| Nickel | agricultural | 0.004915 | kg |
| Zinc | agricultural | 0.0121249 | kg |
| Chromium | agricultural | 0.0650054 | kg |

## 3/ Detailed inventory data for Miscanthus production (120 N-kg ): emissions from1 ha

annual mean production over 15 yrs: 12.85tDM/yr, emissions from lime not shown

| Process | |
|---|---|
| Type | Unit process |
| Time frame | 2005-2009 |
| Geography | Europe, occidental |
| Technology | Best available technology |

| Products | | | |
|---|---|---|---|
| Miscanthus 15 yrs N120U/FR ss co2 | 1 ha N50: annual mean over 15 yrs: 12.85tDM | | |
| | | | |
| Avoided products | | | |
| | | | |
| Resources | | | |
| Occupation, arable | land | 1 | ha |
| | | | |
| Materials/fuels | | | |
| Miscanthus lime emission, high 1 ha yr-1 | | 1 | ha |
| | | | |
| Electricity/heat | | | |
| | | | |
| Emissions to air | | | |
| Ammonia | low. pop. | 1.6926333 | kg |
| Nitrogen oxides | low. pop. | 0.8213836 | kg |
| Dinitrogen monoxide | low. pop. | 2.741328 | kg |
| Dinitrogen monoxide | low. pop., long-term | 0.3387669 | kg |
| | | | |
| Emissions to water | | | |
| Nitrate | groundwater | 116.80003 | kg |
| Phosphate | groundwater | 0.183871 | kg |
| Phosphate | river | 0.5056452 | kg |
| | | | |
| Emissions to soil | | | |
| Glyphosate | agricultural | 0.288 | kg |
| Cadmium | agricultural | 0.000346 | kg |
| Copper | agricultural | 0.0039446 | kg |
| Lead | agricultural | 0.0015509 | kg |
| Zinc | agricultural | 0.0240038 | kg |
| Chromium | agricultural | 0.065956 | kg |
| Nickel | agricultural | 0.0080033 | kg |

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# Émissions de gaz à effet de serre des biocarburants, Amélioration des Analyses de Cycle de Vie par la prise en compte des facteurs de production locaux

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Thèse de doctorat - 16/11/2009 - Résumé

Les biocarburants sont produits à partir de biomasse qui peut être renouvelée à des échelles de temps beaucoup plus courtes que les carburants fossiles; ils sont ainsi une source secondaire d'énergie renouvelable. Parmi les énergies renouvelables, l'accent politique a notamment été porté sur l'utilisation des biocarburants dans les transports car ce domaine constitue l'une des sources majeures croissantes de gaz à effet de serre à l'échelle mondiale. Comparés aux carburants fossiles, les biocarburants émettent moins de gaz à effet de serre, à condition que les économies de $CO_2$ grâce à la fixation photosynthétique des plantes ne soient pas enrayées par les émissions de gaz à effet de serre tout au long du cycle de vie du carburant. Une Analyse de Cycle de Vie (ACV) est donc nécessaire pour comptabiliser l'ensemble des émissions depuis la production jusqu'à la combustion du biocarburant. Cependant les facteurs d'émissions utilisés dans les ACVs ne permettent d'obtenir que des estimations moyennes des émissions ne prenant en compte aucune spécificité locale. Il est pourtant crucial d'estimer avec plus de précision les émissions agricoles et notamment celles de $N_2O$. En effet ce gaz a un pouvoir de réchauffement 298 fois supérieur à celui du $CO_2$ et l'agriculture est la principale source de $N_2O$ anthropique. Notre compréhension du déterminisme de ces émissions au champ reste cependant limitée, notamment concernant les impacts du changement de structure du sol sur ces émissions. Dans un premier temps, notre objectif fut d'approfondir notre connaissance des effets du tassement du sol sur la dynamique des émissions de $N_2O$ et de mieux caractériser ces effets en combinant expérimentation au champ et modélisation. À l'aide de chambres automatiques en Picardie sur des parcelles tassées et non tassées de betterave, nous avons mesuré en 2007-2008 des émissions cumulées de 944-977 g N-$N_2O$ $ha^{-1}$ en non tassé et de 1,448-1,382 g N-$N_2O$ $ha^{-1}$ en tassé. Les émissions plus importantes dans les parcelles tassées furent liées en grande partie à des conditions anoxiques propices à la dénitrification. Ces émissions ne varièrent que peu d'une année sur l'autre malgré des doses d'engrais azoté très différentes (100 et 150 kg N $ha^{-1}$). Nous avons modifié le modèle d'émissions de $N_2O$ (NOE Hénault *et al., 2*005) grâce à des données publiées pour mieux simuler la nitrification et tenir compte des variations des fractions de $N_2O$ sur les produits totaux de la dénitrification et de la nitrification en fonction des teneurs en eau et nitrates du sol. D'autre part le modèle modifié (NOE2) permit de simuler l'influence des facteurs locaux pedo climatiques et la distribution de l'engrais selon une discrétisation fine dans le profil vertical du sol (1 cm). Malgré une amélioration par rapport à NOE dans la prédiction du taux de nitrification et des flux après l'apport d'engrais, NOE2 sous-estima les émissions totales. La sévère sous-estimation des pics d'émission en été a illustré que les cycles sec-humides du sol peuvent entraîner de fortes émissions dont le déterminisme doit encore être élucidé et modélisé. Dans un deuxième temps, nous avons alors utilisé un modèle d'agro-écosystème (CERES-EGC) couplé à NOE2 pour produire les données d'inventaire locales pour nos ACV comparées d'éthanol de betterave et de *Miscanthus.* Les ACVs indiquent que l'éthanol de *Miscanthus* produit en Picardie entraîne potentiellement beaucoup moins d'impacts sur l'environnement comparé à celui de betterave. Comparés à l'essence, l'éthanol de *Miscanthus* permettrait une réduction de gaz à effet de serre de 82-85% par MJ, celui de betterave une réduction de 28-42% par MJ. Ce dernier éventail de réduction illustre combien les résultats de l'ACV s'avérèrent sensibles aux variations des facteurs de production locaux. Ce fut aussi déterminant pour les catégories d'impact acidification ou eutrophication, auxquelles les émissions au champ contribuent pour une grande part. L'identification de systèmes de production optimum nécessite de trouver les facteurs de production locaux aboutissant au difficile compromis entre de forts rendements à l'hectare et des faibles impacts environnementaux. Une analyse qualitative des impacts sur le sol en tant que ressource devrait compléter ces ACVs locales pour servir d'outil d'aide à la décision. Enfin, de plus amples modifications du modèle CERES sont nécessaires pour mieux simuler la production de *Miscanthus* et mieux reproduire les multiples aspects des diverses pratiques culturales.