

1 Life-cycle assessment of local feedstock supply scenarios to compare
2 candidate biomass sources

3 Running Title : LCA of local biomass feedstocks

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14 **KEYWORDS**

15 Biomass supply, Supply area, Inventory methods, Agricultural LCA, Emission Models, Soil
16 carbon dynamic, Local data

17
18 **TYPE OF PAPER**

19 Primary Research Article

ABSTRACT

The use of Life Cycle Assessment (LCA) as a comprehensive tool to assess environmental impacts of bioenergies is recommended. Nevertheless, several methodological points remain at stake, particularly regarding feedstock production step which is a key stage of bioenergy chains. The present work precisely focus on field emissions during feedstock production, improving assessment methods by the use of various process-based models. To do so, a real bioenergy chain case is studied, namely local feedstock supply for a boiler located in Northern France. The LCA compares flax shives, which is the reference, with four other biomass sources: miscanthus, cereal straw, linseed straw and triticale as a whole plant. Six feedstock supply scenarios were also compared. The objectives of this study was to perform and test a new LCA methodology for agricultural chains by integrating local characteristics and using models to estimate field fluxes as pesticide emissions and soil organic carbon (SOC) variation. Results showed flax shives and linseed straw had the lowest impacts, except for global warming, as a consequence, supply scenarios with the largest share of flax shives had the lowest impacts. For all selected impact categories, transportation and fertilization were the main contributors. Soil organic carbon dynamic led to high carbon sequestration level (e.g. miscanthus) or to high CO₂ emission level (e.g. flax shives), thus significantly mitigating or enhancing global warming impact. A sensitivity analysis showed the large influence of allocation procedure (economic or mass-based). This study demonstrated the relevance of model integration using local data in agricultural LCA and especially for SOC dynamic and pesticide field emissions. Moreover, this work brought scientific elements to both support the choice of flax shives as the main biomass feedstock, and to help decision making to choose the best alternative biomass supply for the boiler.

46 1 INTRODUCTION

47 In a global context of greenhouse gases emission reduction and energy independency, The
48 European Union recently set a 20 % target share of renewable energy consumption by 2020
49 (European Commission, 2009). Among the various renewable energy sources, biomass plays
50 a key role. It contributed 10.2 % of the global primary energy supply in 2008 (Chum *et al.*,
51 2011), and is expected to develop two- to six-fold within 2050. From the environmental point
52 of view, bioenergy pathways need to be assessed with a comprehensive approach, accounting
53 for all the impacts generated throughout their life-cycle life cycle of energy, and making it
54 possible to identify pollution trade-offs between life cycle steps and ecological compartments.
55 Life Cycle Assessment (LCA) is a prime example of such tools, and its use has been
56 recommended by the European Union to provide sustainability criteria for biofuel certification
57 schemes (Commission of the European Communities, 2007). Nevertheless, several
58 methodological points need to be addressed in the application of LCA to bioenergy, in
59 particular regarding the estimation of field emissions during feedstock production in the
60 inventory phase, and the methods used for considering co-products (Chum *et al.*, 2011). The
61 choice of spatial and temporal scales for the studied system also requires special
62 consideration.

63 Feedstock production is a key stage in the LCA of bioenergy chains, particularly as
64 agriculture is the first anthropogenic source of nitrous oxide (N₂O) (Bessou *et al.*, 2010). The
65 methods used to estimate these emissions as well as that of other reactive nitrogen (Nr) fluxes
66 in the field are therefore critical. Even when following the ISO 14040 LCA standards (Guinée
67 *et al.*, 2002), there are several possible choices, many possible methods to assess emissions

from the studied system. When assessing the life cycle environmental impacts of biomass production and the whole bioenergy chains, one of the key stage is the agricultural production step. The simplest method relies on emission factors, or fixed coefficients expressing the ratio of Nr fluxes to fertilizer N input rates. They may account for the main direct and indirect emissions of N₂O based on a N balance at the field scale. This is the case of the IPCC Tier 1 approach for greenhouse gas (GHG) inventories (Bessou *et al.*, 2010; Institut de l'élevage *et al.*, 2010). For the other types of Nr fluxes, methods relying on N balances and experimental data may also be used. For NH₃, emission factors usually depend on nitrogen fertilizer chemical form as it is one of the main drivers of volatilization (ADEME, 2010; Nemecek & Kägi, 2007). Nitrate leaching fluxes may be assessed by relating a pool of soil nitrate potentially available for leaching with the duration of the deep water drainage period (Nemecek & Kägi, 2007). Emission factors, or even flat emission rates are used for NO emissions from soils (Nemecek & Kägi, 2007).

The major shortcoming of the default emission factors lies in their ignoring the complexity of the biophysical processes responsible for Nr emissions, and the effects of soil characteristics, climate, crop management and land use which entail a strong variability across space and time (Smith *et al.*, 1998).

A promising alternative to overcome this limitations consists of using process-based models that can simultaneously evaluate the emissions of Nr (NO₃⁻ leaching, NH₃ volatilization, NO and N₂O emissions occurring in field (Durandeu *et al.*, 2010)). For pesticides, which are widely applied on crops, assessment methods usually evaluate fluxes to soil, air, water and into the crops themselves following fixed proportions of the pesticide application rate (Audsley *et al.*, 2003; Nemecek & Kägi, 2007). Because the proportions emitted to air, water and crops are minute according to the latter authors, the majority or even the totality of pesticide residues are assumed to end up in agricultural soils. This contradicts

the fact that biodegradation within a few months to years after application is the major pathway in the fate of pesticides (Barriuso *et al.*, 1996), and that volatilization may play a significant role for some compounds (Bedos *et al.*, 2002). Regarding phosphorus losses emissions, Nemecek and Kägi use emission factors to estimate the P fluxes transported in runoff water, deep drainage and eroded soil particles. Evaluation of the latter component requires experimental datasets or the adaptation of soil erosion models that account for soil, climate and crop characteristics. This requires local data as inputs to one of the various existing erosion models (Merritt *et al.*, 2003). Finally, the GHG balance of biomass rarely includes the effect of feedstock production on soil carbon dynamics (whether leading to soil C sequestration or release) (Brandão *et al.*, 2011), although it is important, especially for perennial crops compared to annuals (Clifton-Brown *et al.*, 2007; Frank *et al.*, 2004; Kahle *et al.*, 2001) cited by Bessou *et al.* (2010). For instance, the conversion of cropland to switchgrass (a perennial grass) was estimated to lead to a 400 kg C ha⁻¹ yr⁻¹ increase of soil C in the US (Anderson-Teixeira *et al.*, 2009). Arrouays *et al.*, 2002 cited by Mila I Canals *et al.*, 2007b also provide reference values for soil C evolution for various types of crop practices for annual crops, and land-use changes between perennial (pasture, forest) and annual crops in France. LCA studies also use average rates of carbon sequestration provided for main French annual crops (Arrouays *et al.*, 2002), cited by Institut de l'élevage, IFIP *et al.*, 2010. Soil C dynamics are actually controlled by local drivers (soil clay content, pH, air temperature and rainfall patterns) along with management practices (tillage, crop types and sequences), which questions the relevance of this simple approach, similarly to the use of fixed emission factors for Nr fluxes.

Aside from field emissions, another pitfall in the LCA inventory lies in the source of technical data for crop management to plant operation. Actual data from existing chains may be far different from optimized, blueprint data (for biomass logistics or conversion plants), or

general statistics (for crop management and yields). For example, the energy balances calculated by Silalertruksa and Gheewala (Silalertruksa & Gheewala, 2009) for cassava ethanol plant in Thailand concluded to a much lower energy efficiency of the actual plant compared to the initial plant design figures. Moreover, the large variability recorded in the literature on the energy and GHG balances of biofuels are due to the agricultural production stage, and more specifically to the nitrogen fertilization practices and crop yields. The latter are highly variable between studies, and not always connected because they may be estimated from independent sources (Bureau *et al.*, 2010). This calls for the consideration of locally recorded crop management and logistics practices, as well as energy and industrial input values coming from real case study data. Moreover, the definition of logistics scenarios and supply area needs to be paid special attention since they entail specific crop management practices, in relation to local soil and weather conditions. Since these characteristics strongly affect the LCA results, a geographically explicit approach should be favored (Gabrielle *et al.*, 2009; Gold & Seuring, 2011).

The objective of this work was thus two-fold. First, we sought to improve the LCA methodology for agricultural chains by integrating local characteristics in the inventory step, in particular by using various process-based models. The second objective was to test and apply these methodological developments to a real bioenergy chain case-study, by comparing the impacts of possible feedstocks, some of which being little documented in the literature. As a consequence, this work also provided LCA references on the latter types of biomass feedstocks. The chosen case study was a multi-fuelled boiler that burns rather uncommon biomasses: flax shives, linseed straw, and triticale as a whole plant, as well as more common ones: cereal straw and miscanthus. While several LCA studies have been published for the latter (Bessou, 2009; Gabrielle & Gagnaire, 2008), very few to none exist for the first

biomass types (Turunen & van der Werf, 2006; van der Werf & Turunen, 2008). We thus had to adapt our methodology to the specificities of these biomass types.

2 MATERIALS AND METHODS

2.1 LCA goal and scope and system description

2.1.1 Objectives, functional unit and system boundaries

The general objective of the study was to compare the environmental impacts of various feedstock supply chains for a multi-fuelled boiler located in Northern France (Picardy region). This area produces flax and major arable crops, with only few forests or hedges and currently provides flax shives as the main biomass supply for the boiler. The comparison includes scenarios combining various local feedstock types, ruling out wood chips, performed to prevent from flax shives supply deficiencies.

The main function of the boiler is to provide heat year-round to various public facilities and housing. Hence the annual heat production of the boiler (10,000 MWh) was chosen as the functional unit. The associated amount of biomass varies across biomass types as a function of their Lower Heating Values (LHVs), reported in Table 1. Heating losses during combustion were accounted for (5% for flax shives and 20% for other biomass feedstocks). The boiler is designed to be supplied with various biomass types provided they are within an acceptable humidity range of 10% to 20%, the optimum level being 15%. Biomass is fed into the boiler as bales via a conveyor belt or in bulk via an endless screw.

For each biomass type, the system included the following steps: field production, harvesting, storage and logistics for delivery to the boiler location site. For flax shives, on site storage of unscutched flax bales and flax fiber transformation were also included. Conversely, the spreading of combustion ashes in the field, the boiler manufacturing and combustion step

were not included in the studied system. This was justified since the ash production rate (e.g. 46 t/year for flax shives) was too low to modify nutrient inputs to the crops present in the cropland surrounding the boiler site (e. g. about 1100 ha for flax grown area). As the boiler construction is the same for all biomass types, it was not necessary to take it into account in the comparison. Finally, biomass combustion and its emissions are not negligible, but we preferred not to assess it, as we did not have the homogenous and reliable enough data to account for them: no data were available for flax shives and several other studies reveal variable emission ranges for the other studied biomasses (Biomass Energy Center *et al.*, 2009; IFEU, 2000; LCA food, 2004). Each step of the life cycle and the corresponding hypothesis are detailed in the following subsections.

2.1.2 Choice of studied biomasses and scenarios

To meet the humidity constraints of the boiler, only dry biomasses could be supplied as an alternative to flax shives. Also, these raw materials needed to be produced locally to minimize transport costs. Hence, linseed and cereal straws, miscanthus and triticale as a whole plant were studied. Supply scenarios resulting from various biomass combinations were elaborated with the company operating the boiler to meet their needs and constraints. They are presented Table 2, with a decreasing proportion of flax shives relative to alternative feedstocks.

2.1.3 Supply area characteristics

To reduce logistics costs, the feedstock supply area was determined as the surrounding “Small Natural Regions” (SNR) within a radius of 50 km maximum around the boiler location. These geographical and administrative entities areas are characterized by homogeneous soil and weather characteristics. Farm types in these areas were subsequently determined from 2007 Common Agricultural Policy (CAP) farm database. They represent agricultural production

systems and are characterized by their crop rotations, their soil constraints, and the area share between crops. Integrating both agronomic (soil and technical) and economic (economic crop value) characteristics of crops already grown on farms, we were able to determine the area that could be sown to biomass crops. Depending on the farm type, we were also able to estimate the potential quantities of straw that could be exported without jeopardizing soil organic carbon content. Overall, the supply area determined above appears to always produce a sufficient amount of biomass for all the supply scenarios, for each biomass type.

2.1.4 Crop management assumptions

Crop management data are summarized in Table 3. Data were recorded by the supply and storage company Lin 2000, completed by regional agricultural experts and regional statistics (AGRESTE, 2006).

2.1.5 Biomass logistics

Various mobilization processes were defined with the supply and storage company experts for each biomass type, depending on the distances between the agricultural fields, the intermediate storage sites (farm storage or collective storage) and the boiler site (Figure 2). Flax was the only biomass with 100% of production stored on farm. For the other crops, only 50% is stored on-farm and the rest in a collective storage. For on-farm storage, a loader follows the tractor and trailer system for each trip to load and unload the trailer, whereas for the collective storage one loader loads biomass on the field and another loader unloads it at the storage site.

2.1.6 Choice of indicators

In this study we considered the following mid-point impact categories and reference substances: abiotic depletion (kg Sb-eq), acidification (kg SO₂-eq), eutrophication (kg PO₄-

eq), global warming (kg CO₂-eq), ozone depletion (kg CFC-11-eq), terrestrial ecotoxicity (kg 1,4-DB-eq), photochemical oxidation (kg C₂H₄-eq) and energy consumption (corresponding to renewable and non-renewable energy consumption, MJ). As toxicity and ecotoxicity are usually impact categories less reliable than others in LCA studies (Reap *et al.*, 2008), we preferred to work only with one of those categories (terrestrial ecotoxicity), instead of artificially give more weight to toxicity categories by including many of them in the present study. Thus we considered terrestrial ecotoxicity as the representative impact category for all the toxicity impact categories.

2.1.7 Allocation assumptions

The various studied systems contain several processes yielding more than one product or by-product (for example flax straw produces flax shives, long fibers, tows and seeds). Thus, it was necessary to allocate environmental impacts and to choose the appropriate allocation assumptions. Flax shives, the reference biomass, has a much lower economic value than flax long fibers. We thus considered that flax scutching is only performed to produce long fibers or tows and not flax shives, implying that flax fibers should carry the main part of the environmental burden of scutching. For this reason, environmental impacts were shared according to the economic value of the products and by-products. The main LCA studies already published on flax use the same argument and also selected economic allocation (González-García *et al.*, 2010a; González-García *et al.*, 2009; González-García *et al.*, 2010c; van der Werf & Turunen, 2008). The prices considered for flax fibres were given by the Lin 2000 company, and correspond to market prices of the year 2010. For the co-products of wheat grains and linseed seeds, we also chose economic allocation and considered a mean of market prices over the year 2010 for grains and prices are given by the regional agricultural cooperative federation (FRCA) for straw. They correspond to the P, K and organic matters

content of straws plus a producer margin. The price of flax shives were given by Lin 2000 and correspond to the price paid to flax producers. A sensitivity analysis involving a mass-based allocation was also performed to test the effect of our allocation hypothesis on LCA results.

2.2 Life-cycle Inventory:

2.2.1 Inventory methods:

Nitrogen and phosphorus emissions due to crop production (fertilizer uses, land use type, crop type) were calculated according to the following methods:

- Two types of N₂O emissions are distinguished depending on the time after fertilizer N application and their location in the N cascade (IPCC, 2006). Direct emissions are due to the effects of mineral and organic fertilizer inputs on soil nitrification and denitrification activities, and were determined from ADEME (2010) with a 1% emission factor. The latter also applied to the fate of crop residues. Indirect N₂O emissions are related to N losses from the cultivated field through leaching and NO emissions, and were calculated from IPCC (2006).
- The emissions of NH₃ to air were obtained by using emission factors from Institut de l'élevage (2010) amounting to 2% of N inputs from ammonium nitrate and 8% of N inputs from urea ammonium nitrate. For this study we did not consider other forms of nitrogen fertilizers.
- For the emission of NO₃⁻ to water we used a nitrogen balance method adapted from IFEU (2000) and integrating emission factors from IPCC (2006). The N balance depended on crop rotation and soil type.

- The emissions factor for NO emissions was derived from ADEME (2010) from field studies in Northern France. The factor was set at 1% for winter crops and 2% for spring crops.
- Phosphorus emissions in water by leaching, run-off and erosion were estimated according to Nemecek and Kägi (2007). Phosphorus emission by erosion requires an estimation of the amounts of eroded soil, which was determined by the Universal Soil Loss Equation (Weischmeier & Smith, 1965). The equation was parameterized with local soil, climate and crop data. Each combination of soil, crop, and crop rotation type occurring within the studied area was weighed by its own area.

Unfortunately, none of the above methods were suitable for miscanthus. We thus used emission factors from a modeling approach by Bessou (2009) in the same region, which are thus specific to miscanthus and to our study area.

In order to estimate pesticides emissions and carbon sequestration, the biophysical models AMG and Pest-LCI were used. Their respective inputs and outputs are summarized in Table 4. Pest-LCI simulates the fate of pesticide to assess emissions during application and post-application, from soils and crop leaves. To do so, it models the fate of each fraction of pesticide reaching a compartment of the simulated system (air, crop, soil surface, water drainage system and groundwater). It accounts for the soil and climate conditions as well as bio-physico-chemical properties of the pesticide molecule, and simulates the following processes: volatilization from crop leaves and topsoil, wind drift loss, runoff, soil biodegradation and pesticide leaching in soil (through the characterized horizons). AMG is a dynamic model that simulates the evolution of the humified organic matter over time, accounting for inputs from crop residues and their humification and mineralization rates. The initial soil organic carbon content, which is a very sensitive parameter, was determined after

typical pairs of crop rotation and soil types from a regional project using plot measures on the evolution of organic matter in arable soils (Duparque *et al.*, 2011). The variations in soil carbon content over 20 years were thus simulated. To have parallel and coherent simulations between pesticide emissions and soil C sequestration, the same soil types were distinguished for the two simulations sets. Pest-LCI model was run with a mean weather year from 20-year serie of past weather data (1988-2007).

The AMG and Pest-LCI models were again not adapted to miscanthus. We thus used a carbon sequestration value of $0.6 \text{ t C ha}^{-1} \text{ yr}^{-1}$ from (Clifton-Brown *et al.*, 2004), and the pesticide emission methods from (Audsley *et al.*, 2003).

Inventory data for the manufacturing and supply of fertilizers, pesticides, tractors, agricultural machinery, lorries, trailers, storage infrastructures, fuel and electricity came from Ecoinvent database, version 2.1 (Swiss Center for Life Cycle Inventories, 2009). Fuel consumption for in field production stage was determined according to the method described in (Institut de l'élevage *et al.*, 2010). Fuel consumption for transportation was estimated by expert from Regional Agricultural Cooperative Federation advices (Bertrand, 2010). Data on flax transformation (energy and building) were recorded by Lin 2000.

2.2.2 *Impact assessment*

Abiotic depletion, acidification, eutrophication, global warming, ozone depletion and photochemical oxidation were calculated using the CML 2 baseline v2.05 method (PRéConsultants, 2008). USES-LCA 2.0 method (van Zelm *et al.*, 2009) was chosen to assess terrestrial ecotoxicity. Energy consumption was calculated according to Cumulative Energy Demand (CED) method version 1.08. All impact calculations were done with Sima Pro 7.2 software (PRéConsultants, 2010).

3 RESULTS

3.1 Comparison of feedstocks

Figure 3 compares the LCA results of the 5 feedstocks, relative to those of flax shives which serve as a baseline. The field emissions calculated are summarized in Table 1. Except for global warming, flax shives and linseed straw had the lowest impacts while triticale has, by far, the highest values for all impact categories. For three categories out of eight, linseed straw had an impact value lower than flax shives. Differences of impact values between flax shives and the other feedstocks varied within a wide range, especially with triticale which had impact values 2-fold (for energy consumption) to almost six-fold (acidification) higher than flax shives. Miscanthus and cereal straw had a similar LCA profile except for global warming. Miscanthus was actually a net GHG sink -due to its high soil carbon sequestration rates, which more than offset the GHG emissions related to crop cultivation and biomass transport and storage. This only occurred with miscanthus, which was consistent with our hypothesis that miscanthus always replaces arable crop and never comes instead of a pasture or other perennial crop.

3.2 Contribution analysis

Overall, transportation and fertilization were the top 2 contributors for all impacts for all feedstocks, except for flax shives for which the scutching step was also a key step, contributing 26% of the photochemical oxidation (PO) impact (Figure 4 and Figure 5) with a similar profile across feedstocks . The respective contribution of transport and fertilization varied across feedstocks. Transport contributed more than 75% of the abiotic depletion (AD), ozone depletion (OD) and PO categories, whereas fertilization contributed more than 34% of

the same impacts for triticale. In fact, these impacts are strongly related to fossil energy consumption such as diesel fuel for transportation or natural gas for fertilizer production.

Fertilization was the major contributor to the acidification (AC) impact for flax shives (53%), miscanthus (62%), cereal straw (70%) and triticale (76%). This is mostly due to the emissions of ammonia occurring upon the application of fertilizer N, which was highest with triticale and lowest (by a factor of 20), for linseed straw (Table 5). As a result, linseed straw was the only feedstock for which transport predominated in the acidification impact, with a 64% share compared to 30% for fertilization. Acidification during transport arises from the consumption of diesel by farm machinery and trucks. Regarding eutrophication, fertilization contributed 77% (linseed straw) to 97% (flax shives) of the impacts, due to the emissions of reactive nitrogen and phosphorus emissions to water (via leaching, run-off, and erosion) and air (ammoniac) during biomass production. The contribution of other steps was negligible compared with fertilization. Similarly to ammonia, the emissions of nitrates and P were highest for triticale and lowest for linseed straw (Table 5).

Figure 6 breaks down the fertilization stage into fertilizer production and the field emissions arising from fertilizer application (thus not including the application stage per se). For global warming and acidification, the impacts are largely influenced by field emissions of reactive nitrogen (N_2O , NH_3 and NO_x). Fertilizer production plays a minimal role in the eutrophication, which mostly depends on phosphorus and NO_3^- emissions. The relative contribution of both compounds varies significantly according to biomass type, due to different fertilizer input rates and to the variability of soil characteristics between biomass production areas. Indeed, phosphorus emissions depend on soil erosion rates, which strongly vary between the different biomass production areas.

Fertilization is overall the top contributor to the global warming impact of cereal straw, triticale and miscanthus, with a share of 57% (miscanthus) up to 71% (cereal straw). For flax shives

and linseed straw, fertilization contributed from 29% to 32% of the impacts, and was second to transport for the latter. Fertilization impacts global warming through both fertilizer production and N₂O emissions after fertilizer application. The latter can make up more than 50% of the GHG emissions related to fertilization (Figure 5).

Soil carbon dynamics were very influential in the global warming (GW) impacts, whether mitigating (through soil C sequestration) or enhancing life-cycle GHG emissions. Simulations with the AMG model showed that 4 feedstocks resulted in a net release of CO₂ from soils (Table 5), with a contribution of 35% to the GW impact for flax shives, 10% for triticale and 16% for linseed straw. Conversely, soil carbon was stored for cereal straw and miscanthus, leading to a 13% reduction of GW for cereal straw and to a negative GW impact for miscanthus. For the latter, soil carbon sequestration made up 132% of the life-cycle GHG emissions related to production and mobilization.

Pesticide emissions (field emissions + pesticide production) made up 22% of the terrestrial ecotoxicity impact for flax shives and cereal straw, and 10% for triticale, mostly involving herbicides. Pesticide emissions were negligible for linseed straw and miscanthus.

Transportation was the top contributor to energy consumption for linseed straw (83%), cereal straw (66%) and miscanthus (50%). Scutching was responsible for 53% of the energy consumption of flax shives, and fertilization accounted for 39% of energy consumption by triticale. Overall, post-harvesting stages (transportation, storage, flax scutching) represented the greater share of the energy consumption of biomass supply. Figure 7 indicates the net energy yield (NEY) of each biomass, expressed as the quantity of heat produced by biomass combustion for each MWh of fossile energy consumed throughout the biomass life cycle. Linseed straw and flax shives had the highest NEY with respectively 26.5 and 25.6 MWh_p/MWh_c whereas triticale had the lowest NEY (8.5 MWh_p/MWh_c). These results are consistent with the energy consumption levels of biomasses: as the LHVs of biomasses are similar, biomass types with the lowest energy consumption level have the highest NEY.

3.3 Comparison of supply scenarios

Results of supply scenario comparison are presented in index base 100 where reference is S1. Figure 8 compares the 6 supply scenarios tested, using the first one (100% flax shives) as a baseline. Supply scenarios with the largest share of flax shives (S1 and S2) had the lowest impacts except for global warming. In accordance with the feedstock comparison results, supply scenarios containing miscanthus (S5 and S6) are the only ones with a global warming impact smaller than the baseline (-55% for S6 and -25% for S5). This is due to the inclusion of carbon sequestration under miscanthus. However, S5 and S6 to a larger extent present high impacts all other categories. Overall, scenario S6 bears the highest impacts for six impact categories out of eight.

3.4 Sensitivity analysis

Because three feedstocks assessed here are by-products with low cost, the allocation procedure may affect LCA results. Using mass-based allocation rather than economic allocation strongly influenced the comparison between feedstocks (**Erreur ! Source du renvoi introuvable.**). The impacts of miscanthus and triticale became lower than those of by-products, because allocation ratios changed drastically. For example, with mass-based allocation, the affectation between cereal grains and cereal straw is 87% to grains and 13% to straw, whereas with economic allocation this affectation is 98% to grains and 2% to straw.

395 4 DISCUSSION

396 4.1 Comparison with other LCA studies

397 Comparison with other studies was only possible for miscanthus and triticale. We found
398 LCAs dealing with flax shives (González-García *et al.*, 2009) and cereal straw (Gabrielle &
399 Gagnaire, 2008), but they used different system boundaries and functional units, hampering
400 the comparisons. Straw and flax shives were actually considered as intermediate products
401 while the main products were biofuels, and results were expressed per kilogram or MJ of
402 biofuel.

403 For miscanthus, our results were consistent with Brandão (2011), who indicated a
404 largely negative global warming impact, at $-1567 \text{ kg CO}_2\text{-eq ha}^{-1}$, compared to our -1010 kg
405 $\text{CO}_2\text{-eq ha}^{-1}$ estimate. Concerning other environmental impacts as eutrophication,
406 acidification and energy consumption, the levels we obtained were in the same range as the
407 studies by Brandão (2011) and IFEU (2000) (recalculated from the original spreadsheet data).
408 Nevertheless, the difference between energy output per ha and the energy consumption
409 (excluding transport) given by Brandão (2011) is higher than our value (338.7 GJ ha^{-1}
410 compared to 185.5 GJ ha^{-1} , respectively). This difference may originate from different LHVs
411 (5 Mwh t^{-1} for Brandão and 4.4 Mwh t^{-1} in the present study) combined with different biomass
412 yields (19.2 t ha^{-1} for Brandão versus 16.5 t ha^{-1} in our study).

413 Concerning triticale, our results are in the same order of magnitude as those from
414 IFEU (2006). However, global warming and acidification impacts were twice higher in our
415 study. This is due to the different N input rates considered (100 kg N ha^{-1} in IFEU and 140 kg
416 N ha^{-1} here) and also to the flux estimation and impact characterization methods. The IFEU
417 study dates from 2000 and uses methods which may be outdated. For example, it uses a global

warming potential of 320 kg CO₂-eq/kg N₂O for N₂O whereas in the updated characterization methods this value is 298 CO₂-eq/kg N₂O.

4.2 Modelling approach for direct emission assessments

4.2.1 Relevance of considering carbon sequestration and of using AMG

Here, we assessed the contribution of biomass crops to soil C variations by modelled Soil Organic Carbon (SOC) dynamics. It differs from the approach of Nemecek and Kägi (2007) which is based on the C content of the biomass exported from the field, considering it a sink for atmospheric CO₂. This implies that all the emissions of biogenic C should be tracked and accounted for in the life-cycle of the product derived from this biomass, which is better in terms of temporal effects but challenging (Rabl *et al.*, 2007). A widespread, more practical alternative consists of considering biomass C as climate-neutral, as Schmidt *et al.* (Schmidt *et al.*, 2004) did in their study of flax production. The latter two approaches actually disregard the effects of crop cultivation on soil C dynamics, let alone the effects of soil type, crop rotation or climate, which play a major role in the GHG balance of agricultural crops (Ceschia *et al.*, 2010). Using a soil C model such as AMG is a prime means of overcoming this limitation and accurately assesses soil carbon sequestration or release rates. As shown here, these rates may mitigate or conversely enhance the global warming impact of crops. AMG is particularly relevant in our case study, as its validity was tested over eight European experimental datasets. Nevertheless, no perennial crop, as miscanthus, is parameterized within the AMG model. As an alternative to detailed modelling studies, Arrouays *et al.* (2002) derived from the literature a single SOC sequestration rate for wheat straw in France, of 0.32 t C ha⁻¹ yr⁻¹. This value is close to the mean value estimated in our study with AMG model (0.23 t C ha⁻¹ yr⁻¹). However, Arrouays *et al.* (2002) did not report figures specific to triticale, flax and miscanthus. Moreover, using a single figure ignores the variability in crop production

conditions. For example, the SOC sequestration rates for wheat straw modelled with AMG ranged from $-0.03 \text{ t C ha}^{-1} \text{ yr}^{-1}$ to $0.42 \text{ t C ha}^{-1} \text{ yr}^{-1}$ according to soil type and/or crop rotation. Beyond global warming impact assessment, another interest of using AMG is that SOC is considered a relevant indicator of soil quality for LCA (Brandão *et al.*, 2011; Milà i Canals *et al.*, 2007a; Milà i Canals *et al.*, 2007b). Thus, accounting for soil quality in LCA could be facilitated by the use of SOC models such as AMG. Crop models are not adapted yet to estimate direct in field emissions of miscanthus or other perennial crops (e.g. switchgrass). So, in this study, carbon sequestration was derived from literature factor emission and not from modeling like the other biomass sources. Thus, this difference in calculation method that could cause an overvaluation or an undervaluation of the SOC sequestration compared to the SOC sequestrations of the other biomass estimated by AMG.

4.2.2 *Relevance of PestLCI use*

Regarding ecotoxicity indicators, current methods to estimate pesticide emissions in LCA (Audsley *et al.*, 2003; Webb *et al.*, 2009) rely on emission factors that neither integrate soil characteristics, molecule characteristics nor weather conditions. The PestLCI model makes it possible to account for different farming conditions as well as spraying methods for a specific pesticide product (application rate and above crop spaying or soil incorporation) (de Backer *et al.*, 2009). More detailed pesticide flux models exist, such as PRZM (Suarez, 2005), but their application to LCA is still marginal because of their complexity. On the contrary, PestLCI was especially designed for agricultural LCA studies and requires a limited number of parameters and input data while improving the estimation of field pesticide fluxes and ecotoxicity indicators (Berthoud *et al.*, 2011), even if parameterization is only currently validated for Danish cropping conditions.

4.2.3 Model integration in LCA

This study showed the relevance of using complex emission models in the life-cycle inventory, to account for the characteristics of the feedstock supply area instead of using default emission factors. This approach makes it possible to integrate diversity in supply areas in agricultural LCA, and to better take into account geographical aspects in decision-making. This was also emphasized by Gabrielle and Gagnaire (2008) for emissions of reactive N, who showed that the LCA of bioethanol from wheat grains was strongly influenced by soil types at regional level. Alder *et al.* (Adler *et al.*, 2007) used the Daycent ecosystem model to simulate the GHG balance of several bioenergy cropping systems in the US, and evidenced large differences in the emissions of N₂O compared to the IPCC Tier 1 emission factors, especially with N fixing crops. However they did not attempt at comparing Daycent with field data. Validation of models for the range of substances to be input into LCA certainly hampers to their use in LCAs, which has been limited so far. Another reason is the quantity of input data required by models such as AMG or PestLCI. Lastly, models may only be used over a limited study area. For larger areas, beyond the regional scale, with even wider ranges of crop practices and pedo-climatic conditions (e.g. LCA of wheat in France), a synthetic typology of different situations is required in order to limit the number of model runs.

4.3 Allocation procedure

Even if a recent report recommends the use of mass-based allocation for the LCA of bioproducts LCA (ISO, 2006), economic allocation for flax is very often chosen from a practical point of view, due to the large price difference between the main product and its by-products (González-García *et al.*, 2010a; González-García *et al.*, 2009; González-García *et al.*, 2010b; van der Werf & Turunen, 2008). Nevertheless, Schmidt *et al.*, (2004) chose the system expansion method to simultaneously account for the impacts of main products and by-products in flax production, at the European level. Commenting on their sensitivity analysis involving an economic allocation, the authors underline the relatively high uncertainty level

of both methods, especially at the European level. Contrary to the results obtained in our study, Schmidt *et al.* (2004) only found a minor role of the agricultural step in the assessment, and thus conclude that further data collection about agricultural practices or market prices would be irrelevant. The fact that their results contradict ours might stem from the scope of their study being far different from ours (the life cycle of an insulation product at European scale versus the supply of a local biomass boiler). Nevertheless, from the present study, one could also overturn Schmidt *et al.*'s conclusion by arguing that the low weight of the agricultural step might precisely be due to the poor quality of the agricultural data.

4.4 Practical use of LCA results for storage and supply company

Beyond methodological aspects, the present LCA study also produced specific results for biomass supply for the Lin 2000 company. Indeed, the flax shives which are the main product currently used as fuel for the boiler also proved least impacting on the environment compared to alternative biomass sources. For flax shives, the company was also able to identify the main routes to improvement it could develop, regarding mainly logistics and inner energy consumption during flax processing. Regarding more precisely flax production in the field, a special attention needs to be paid to fertilization which is the main contributor to most of impact categories. Nitrogen fertilization can be optimized in field thanks to precise N balances over years, and surveying tools during crop growth to trigger applications. The comparison of supply scenarios gives the company additional information to choose between biomass sources that could complement flax shives. Thus, depending on the impact category which appears as a priority, one scenario can be preferred to another. For example, if climate change is set as the most important one, scenarios with miscanthus and cereal straw (S6 and S5) are the least impacting. From a practical perspective, technico-economic criteria should also come into play when changing feedstock supply. Thus, information provided by LCA

should be combined with data on the availability of feedstocks and the associated risks. If the feedstock with the least GHG emissions, miscanthus, is not already grown within the supply area, cereal straw would certainly be preferred, as it is already available. For linseed straw, the available quantity within the studied area can only supply a limited fraction of the boiler's demand (about 10%, as specified in scenario S2). Moreover, linseed straw shows high harvesting and logistics costs (Savouré, 2011) making it a rather costly biomass source. Thus linseed straw, whose environmental impacts are very similar to flax shives, will only be a partial alternative to flax shives. Overall, the choice of biomass sources remains a multi-criteria issue, including not only environmental, but also technico-economic and sociological aspects (such as the willingness of farmers to grow new dedicated energy crops). Nevertheless, LCA results contribute to help decision making regarding the best choice of biomass sources for energy production.

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528 **5 ACKNOWLEDGEMENTS**

529 This study was supported by FranceAgrimer and Picardy Region. The authors wish to thank
530 all the partners of the OPTABIOM project, and more especially Fédération Régionale des
531 Coopératives Agricoles de Picardie (FRCA Picardie) and Lin 2000 cooperative for their
532 precious contribution.

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701 **Table 1** : Biomass LHVs and corresponding water content of raw material.

Biomass	Water content of Raw Material (RM) (%)	LHV (MWh/t RM)
Flax shives	12	4.4
Miscanthus	15	4.4
Triticale, whole plant	15	4
Cereal straw	15	4
Linseed straw	13	4.1

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705 **Table 2 :** Supply scenarios for the boiler

Supply scenario	Proportion of Flax shives	Proportions of Alternative biomass types
S1	100 %	0%
S2	80 %	10 % cereal straw + 10 % linseed straw
S3	50 %	50 % cereal straw
S4	50 %	25 % cereal straw + 25 % triticale
S5	50%	25% cereal straw + 25% miscanthus
S6	0 %	50 % miscanthus + 50 % cereal straw

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709 **Table 3 :** Summary of the inputs for each crop used

	Wheat	Flax	Linseed	Triticale, whole plant	Miscanthus, (for 1 average year)ⁱ
<i>Average annual yields</i>					
Grain yield (t FM/ha)	8.5	/	3	/	/
Straw yield (t FM/ha)	1.33 ⁱⁱ	6.5	1.5	/	/
Whole plant (t FM/ha)	/	/	/	16.5	16.5
<i>Crop management (for 1 year)</i>					
Ploughing (pass. nb.)	1	/	/	1	0.05
Stubble ploughing (pass. nb.)	1	2	2	1	0.15
Sub-soiling (pass. nb.)	/	/	/	/	0.05
Cultivator (pass. nb.)	/	1	1	/	0.05
Harrowing (pass. nb.)	1	1	1	1	0.05
Crushing (pass. nb.)	/	/	/	/	0.2
Seeding rate	110 (kg/ha)	135 (kg/ha)	44 (kg/ha)	90 (kg/ha)	900 (rhizomes/ha)
N fertiliser rate (kg N/ha)	180	30	90	140	45
P fertiliser (kg P₂O₅/ha)	/	20	60	/	20
K fertiliser (kg K₂O/ha)	/	40	80	/	10
Pesticide, active ingredient (kg a.i. /ha)	1.58	1.75	1.75	0.457	0.345
Type of harvesting machine	Complete harvester	Specific machinery	Complete harvester	Mower	Chopper
Pulling (pass. nb.)	/	1	/	/	/
Baling (pass. nb.)	1	1	1	1	0.9
Swathing (pass. nb.)	1	/	1	/	/
Flax turning (pass. nb.)	/	2	/	/	/

ⁱ Values are calculated for 1 average year over 20 yearsⁱⁱ In fact 4.5 t/ha but straw is harvested once every three year in order to preserve soil organic matter content

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712 **Table 4 :** Short description of models used to assess pesticide emissions and soil C sequestration

Model	Inputs	Outputs
Pest-LCI (Birkved & Hauschild, 2006)	<ul style="list-style-type: none"> • Soil characteristics per horizon (texture, density, organic matter content) • Active ingredient amount • Active ingredient bio-physic-chemical characteristics • Pesticide application management (application time, application machine and form) • Monthly water balance 	Active ingredient emissions to water, soil and air
AMG (Saffih-Hdadi & Mary, 2008)	<ul style="list-style-type: none"> • Crop rotation • Crop yields • Soil management • Soil type (texture, organic matter and calcareous content) • Average rainfall and irrigation 	Soil carbon sequestration for one crop rotation and/or one crop

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716 **Table 5 :** Inventory of field emissions for the 5 feedstocks, expressed in kg substance per functional unit (10,000
 717 MWh heat).

Substance emitted	Compartment	Flax shives	Miscanthus	Cereal straw	Triticale, whole plant	Linseed straw
N₂O (direct + indirect)	Air	1.16E+02	3.07E+02	3.83E+02	1.16E+03	8.21E+01
NO_x	Air	6.23E+01	1.55E+02	2.70E+02	9.03E+02	4.64E+01
NH₃	Air	7.74E+01	2.57E+02	4.82E+02	6.45E+02	3.09E+01
NO₃	Water	1.64E+03	6.74E+03	4.00E+03	1.84E+04	1.08E+03
Phosphorus	Water	2.37E+02	1.41E+02	1.75E+02	9.13E+02	1.45E+01
C from soil	Air	-1.38E+04	1.08E+05	1.02E+04	-2.27E+04	-5.56E+03
Pesticides (Active ingredients)	Air	2.62E+01	6.18E-04	3.06E+01	9.88E+01	8.81E+00
	Water	3.15E-02	6.18E-04	3.16E-02	3.78E-01	1.06E-02
	Soil	0.00E+00	5.46E-02	0.00E+00	0.00E+00	0.00E+00

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721 **7 FIGURE LEGENDS**

722 **Figure 1:** Boundaries of the studied system.

723 **Figure 2:** Mobilization hypothesis of biomasses

724 **Figure 3:** Biomass comparison in index base 100 with economic allocation

725 **Figure 4:** Contributions of dedicated energy crops to the category impacts

726 **Figure 5:** Contributions of crop by-products to the category impacts

727 **Figure 6:** Contribution analysis of the fertilization (fertilizer production + fertilizer field emissions).

728 **Figure 7:** Net energy yield of biomasses.

729 **Figure 8:** Scenario comparison in index base 100

730 **Figure 9:** Biomass comparison with mass allocation