HAL
open science

Environmental impact of the substitution of imported soybean meal with locally-produced rapeseed meal in dairy cow feed

Simon Lehuger, Benoit Gabrielle, Nathalie Gagnaire

To cite this version:

Simon Lehuger, Benoit Gabrielle, Nathalie Gagnaire. Environmental impact of the substitution of imported soybean meal with locally-produced rapeseed meal in dairy cow feed. Journal of Cleaner Production, 2009, 17 (6), pp.616-624. 10.1016/j.jclepro.2008.10.005 . hal-00087583v2

HAL Id: hal-00087583

https://hal.science/hal-00087583v2

Submitted on 20 Nov 2008

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

Environmental impact of the substitution of imported soybean meal with locally-produced rapeseed meal in dairy cow feed

Simon Lehuger*, Benoît Gabrielle, Nathalie Gagnaire

INRA, AgroParisTech, UMR 1091 Environment and Arable Crops, 78850 Thiverval-Grignon, France

*: corresponding author.

Tel.: +33 1 30 81 55 55; fax: +33 1 30 81 55 63.

E-mail address: Simon.Lehuger@grignon.inra.fr

ABSTRACT

Growing public concerns about the traceability, safety and environmental-friendliness of food products provide an incentive for shorter supply chains in agricultural production. Here, we assessed the environmental impacts of the substitution of imported soybean meal with locally-produced rapeseed meal in French dairy production systems, using a life-cycle approach. Two feeding rations based on either French-produced rapeseed meal or Brazilian-produced soy meal as concentrates, were compared for nine impact categories, including global warming, ecotoxicity and eutrophication. Crop production was the main contributor to most impacts, while overseas transport of soy meal only had a marginal effect. The "Soybean" ration appeared more environmentally efficient than the "Rapeseed" ration because it involved less intensive management practices, in particular regarding synthetic fertilizers consumption. However, land-use changes brought about by soybean cultivation should also be examined.

Keywords: Life Cycle Assessment; Soybean; Rapeseed; Concentrated feed; Environmental impact; Dairy

1. Introduction

In Europe, new practices for milk production reflecting increasing public concern for the traceability of supply chains, environmental protection and food safety are emerging. In particular, animal feeding is expected to meet these new demands. Today, conventional dairy farms usually incorporate imported soybean meal as a high-protein complement to maize silage in the ration of dairy cows. The soybean meal used in Europe is predominantly produced overseas. In France, 5 million tons of soybean meal was used for animal feeding in 2004, 90% were imported, mostly from Brazil [1]. It is thus difficult to control the entire supply chain from the field production to the consumption in French farms. On the other hand, the installation of local production networks of concentrates would improve the quality and traceability of dairy products. Locally-produced meals obtained from protein crops and oil crops may easily be substituted to imported soybean meal in animal feed [2]. In addition, the produced oil may be used to fuel the farm's machinery or heating systems. The current development of such forms of renewable energy to meet the targets set by recent EU directives is an additional driver for the production of oil crops, which is expected to generate a large quantity of meal as co-products available for livestock. However, a thorough assessment of both imported and local types of feeding systems is necessary to judge the ecological relevance of such a substitution. Since this assessment should encompass the range of environmental impacts generated by both alternative systems, the Life Cycle Assessment (LCA) methodology appears as the most relevant framework [3].

Examples of LCAs are increasingly frequent for products with an agricultural phase. In the case of biofuels or animal products (milk and meat), for instance, the cradle-to-farm-gate subsystem generally contributes more to environmental impacts than the transport or industrial phases [4-6]. For animal products, the protein source for concentrated feed is responsible for most of the environmental impacts and energy use within this subsystem [7-

14]. In the literature, it is usually mentioned that using imported soybean meal, which is a predominant source of protein concentrates in Europe, is not a good option compared to locally-produced feed ingredients in terms of environmental performance. Cederberg [7], Cederberg and Flysjö [9] and Casey and Holden [10] recommended locally-produced concentrates for dairy cow feeding. Similarly, for pig meat production, the auto-production of protein concentrates at local or regional scale was mentioned to induce less impact on the environment than feed imported from overseas [11]. Road and sea transports, combined with a relatively higher reliance on pesticides and P fertilizers are the main reasons reported in the literature to make imported concentrates more harmful to the environment than their local counterparts. The following impact categories: climate change, acidification, abiotic depletion and pesticide use were especially reported to present higher impacts [9,10-14]. However, these LCAs were done at the whole production system level [7,8,14], which made it difficult to single out the contribution of concentrates. Here, we chose to focus on this particular item in the overall environmental performance of milk production systems that were otherwise very similar in terms of livestock density and milk productivity. This paper thus takes part in the scientific debate on the rational localisation of food chains to minimize environmental impacts. While it is generally assumed that “local food is best” from an environmental end ethical standpoint, this assertion warrants careful tests and should not be generalized [15].

The present study focuses on the environmental performance of locally-produced rapeseed meal, as part of a research and extension program aiming at developing the use of rapeseed in French dairy farms and improving their self-sufficiency. It responds to a willingness of dairy farmers to develop local or regional supply chains in order to substitute soybean meal with local products. In this shortened supply chain, the farmers would grow oil crops and extract the oil in small-scale crushing units.

The objective of this work was thus to compare dairy cows' rations based either on locally-produced rapeseed meal or on imported soybean meal. The evaluation was done using LCA for French region where farmers are showing increasing interest in such alternatives.

2. Materials and methods

2.1. Life Cycle Assessment methodology and objectives

Life cycle assessment (LCA) is a methodology based on a global approach of the production system ("cradle-to-grave") and on a multicriteria approach of environmental impacts. Its principle consists of quantifying the resources consumed and the emissions to the environment at all stages of the life cycle of the product [3]. The fluxes are subsequently interpreted in terms of impacts on the environment, for a range of categories (global warming, eutrophication, etc…). The impacts were calculated by aggregating the life cycle impacts of the various ingredients of the rations. We followed the standardized LCA methodology [16], involving the following steps:

1) Goal and scope,

2) Life cycle inventory for each ingredient,

3) Characterization of impacts,

4) Comparison of the scenarios "Soybean" versus "Rapeseed".

2.2. Functional unit and system boundaries

The system's function was to produce rations for dairy cows, including the crop production processes and on-farm delivery of the rations' ingredients. We defined the functional unit (FU) as 1000 kg of feed, designed with identical protein and energy content. System boundaries are shown on Fig. 1: they include the production of the agricultural inputs up to the production of the ration on the farm. All the ingredients of the rations are included, i.e. maize silage, minerals, and concentrates. The transport and transformation of soybean or rapeseed grains

are also taken into account. The crushing process of soybean and rapeseed produces two co-products: oil and meal.

2.3. Life cycle inventory

Life cycle inventories of inputs and emissions were implemented for the production of each of the ration ingredients, encompassing the production of agricultural inputs, the cultivation of crops, the crushing of seeds, and the various transports involved between these steps. The inventory was conducted in the particular context of the “Pays de La Loire” region, an administrative entity of about 32 000 km^2 in western France, with a high density of dairy farms. We assumed soybean meal to be imported from Brazil, which contributes 75% of the soybean meal used in this region.

We used a mass-based allocation ratio to split the inventory data between the co-products, based on the extraction yields of the two possible oil extraction technologies: industrial or small-scale crushing units. Soybean meal was produced in industrial plants in Brazil while rapeseed grains were locally crushed in small-scale units. Industrial crushing of 100 kg soybean was assumed to yield 17 kg oil and 80 kg meal [7], and small-scale crushing of 100 kg rapeseed to yield 37.7 kg oil and 56.1 kg meal [17]. Economic allocation was not possible because it is difficult to set a market value for the rapeseed meal produced on farms, in the absence of such market as yet (Sourie J.C., pers. comm., 2005). A second alternative, system expansion, was tested and is further discussed in the Discussion and interpretation section.

Based on prior results from animal feeding trials, we assumed that 1.5 kg of rapeseed meal was needed to provide the same amount of proteins as 1 kg of soybean meal [2]. This substitution rate affects on the level of other ingredients in the rations to maintain the equivalence in terms of protein and energy content. Maize silage made up 82.5% of the total dry matter (DM) of the "Soybean" ration, and the soybean meal 13.8%. In the "Rapeseed" ration, maize silage was equivalent to 73.7% of the total DM and rapeseed meal to 22.3%

(Table S1 in supporting materials, [18]). These rations were complemented with heat-treated soybean meal, minerals and vitamins. Feeding trials were conducted with the two types of rations in an experimental farm [17] where two pools of 21 cows were fed *ad libitum* with both rations. The daily intake of DM was significantly different between the two rations, being 20.1 kg DM and 22.3 for the "Soybean" and "Rapeseed" rations, respectively. This was probably due to a difference in palatability between the rations, and resulted in different milk production rates and compositions across scenarios. The effect of the differences of ingestion and milk production are discussed in supporting information [18].

2.3.1. Crop production

Agricultural inputs comprise fertilizers, pesticides, seeds and agricultural machinery. The data were taken from the Ecoinvent database [19], and combined to compute the impacts associated with elementary management operations such as soil tillage, fertilisation, sowing, plant protection, harvest and transport. For each operation, machinery and energy inputs, and emissions to the environment were inventoried. The direct emissions occurring in the arable field as a result of crop cultivation are discussed in a separate section.

Crop management data specific to the study areas in France and Brazil were sought from relevant agricultural extension services. The crops' life cycle was assumed to start upon harvest of the preceding crop, and to stop upon harvest of the crop considered.

Maize silage management data was based on data collected in an experimental farm ("La Jaillière") representative of dairy farms in the region of interest, Pays de la Loire (Gillet J.P., Pers. Comm., 2005). All the maize silage consumed by the herd was assumed to be home-produced on the farm. The maize crops were fertilized with 30 t of manure produced by the herd, supplemented with 30 kg N as ammonium nitrate.

Rapeseed was assumed to be produced in the same region, according to two cropping systems. One is typical of dairy farming systems, in that fertilization is mostly organic with an

input of 40 m^3 ha^{-1} plus 70 kg N as ammonium nitrate. Technical data were collected from the experimental farm "La Jaillière" (Gillet J.P., Pers. Comm., 2005). The other cropping system was representative of a farm specialized in cereals, which relies exclusively on mineral fertilizers. The data for those farms were based on regional surveys carried out by a local extension service (Charbonnaud J. and Arjauré G., Pers. Comm., 2005). The amount of cow manure and slurry used to fertilize the maize and rapeseed crops on the farm were assumed to correspond with the manure produced by the animals fed with the "Rapeseed" or the "Soybean" rations. Soybean was assumed to be produced in Brazil in the Cerrados region (Centre-West), and managed under a direct drill, mulch-based cropping system with sorghum as possible cover crops. Management data were taken from a range of studies on these systems [9, 20-21]. Table S2 [18] recapitulates the characteristics of the different cropping systems.

2.3.2. Crushing of soybean and rapeseed grains

Small-scale crushing processes of rapeseed were inventoried thanks to data from the French Technical Centre for Oilseed Crops (CETIOM). The crushing technology is meant to be set up by farmers' cooperatives in order to share installation costs.

The selected process which was a succession of crushing-heating-pressing steps may crush 8700 tons of rapeseed grains per annum, and involves the participation of about 85 farmers in the project. The crushing technology does not rely on chemical solvent to improve feed quality, and produces meal under the 10% fat limit required for cow feeding. However, the heat treatment reduces the degradability of the proteins in the rumen by producing "bypass proteins".

A regional survey on the availability of arable land for the growth of rapeseed in both types of farms was carried out in the region [17]. Dairy farms had 37 cows on average, and their needs in soybean meal were estimated at 22 t yr^{-1} per farm. Should this amount be fully substituted

with rapeseed meal, the needs would become 38t yr^{-1} which corresponds to c. 20 ha of oilseed rape. Because only half of this acreage is usually available on the dairy farms for this crop, we considered that and the other half was grown on cereal farms [17]

Industrial crushing of soybean was located in Brazil, in the vicinity of the seaports from which soybean meal is shipped to Europe. The plant was modelled based on data from an industrial crushing plant in Bordeaux (France; Carré P., pers. comm., 2005), and from the Ecoinvent database for a generic chemical plant [22] (Table S3, [18]). The model crushing plant was dimensioned to produce 440 000 tons of oil and meal per year, a fairly standard output rate for this kind of unit (Carré P., pers. comm., 2005). A specificity of industrial oil extraction lies in the use of hexane, a toxic solvent with potential hazards for the environment. The emissions into air were set at 13 10^{-5} kg hexane per kg of seeds processed [19].

2.3.3. Transport

Soy beans are transported from the farm to the crushing plant by truck over a distance of 25 km [9]. Soy meal is transported to commercial seaports by road (60%), by train (33%), and by water (7%) over a total distance of 900 km [23]. Then, soy meal is transported by cargo from Brazil to France over a distance of 9500 km and finally over a distance of 200 km by truck up to the farm. Rapeseed grain is collected on dairy farms (50%) or cereal farms (50%), and transported by trucks to the small-scale crushing unit. Rape meal is subsequently delivered by trucks up to dairy farms. Overall, rapeseed grains and rape meal cover a maximum distance of 200 km (Sourie J.C., Pers. Comm., 2005).

2.3.4. Direct field emissions

These emissions occur during crop growth in the field and include gaseous losses of N in the form of nitrous oxide (N_2O), ammonia (NH_3) and nitrogen oxides (NO_x), and losses of nitrates and phosphorus (P) via leaching and runoff. We estimated these emissions using simple

models, as proposed by Ecoinvent [19] for rapeseed, soybean, cover crop and maize silage and detailed in [18]. Table 1 recapitulates the field emissions for the main crops included in the cows' rations. The variations of soil carbon stocks were not included and are dealt with in the Discussion section.

2.4. Impact categories and characterisation factors

Life Cycle Impact Assessment is the phase in which the inventory data are interpreted in terms of environmental impacts. We selected the following impact categories: *abiotic depletion, climate change, stratospheric ozone depletion, human toxicity, freshwater aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity, photo-oxidant formation, acidification, eutrophication, and land use*. We used the CML 2 baseline characterisation method (version 2.1 – [3]) which uses an array of characterisation models specific to each impact category and is described in [18].

3. Results

Based on our assumptions and system boundaries, the "imported Soybean" scenario appeared more environmental-friendly than the "domestic Rapeseed" one, with the exception of the photochemical oxidation (Table 2). The latter was higher for soybean because of the hexane emissions occurring during the industrial crushing process. The environmental performance of the rapeseed crops was lower than that of the soybean crops, due to the higher reliance on synthetic fertilizer N and the lower yields of the former. Lastly, transatlantic transport of soy meal from Brazil to France contributed for a minor part to the environmental impacts compared to the rest of the life cycle, even for categories usually affected by transport processes like climate change. A similar trend was mentioned in other studies [4, 12].

The results presented below are dependent on the assumptions detailed in the Material and Methods section, regarding system boundaries, allocation procedures and flux estimation

methods. While this is a standard issue in LCA, the fact that it is here applied to agricultural systems entails additional uncertainties. Gaseous and leaching losses from agro-ecosystems are highly variable over time and space, implying that our results may not be readily extrapolated outside the particular regions of France and Brazil selected in our analysis. In particular, they should not be generalized to all agricultural systems using soybean imported from Brazil.

3.1. Impact results

3.1.1. Abiotic depletion

The abiotic depletion impact was 1.06 kg Sb-eq per FU for the "Soybean" scenario and 1.15 Sb-eq per FU for the "Rapeseed" scenario (Table 2). Oil depletion contributed more than the other resources used to the impact indicators, with a share of 64% for the "Soybean" scenario and 56% for the "Rapeseed" scenario. Other LCAs of complete milk production system encompassing crop and animal production systems [14, 24] reported a similar share, although their results cannot be compared with ours since they were expressed as energy use in MJ per kilogram of milk end-product.

3.1.2. Climate change

Per 1000 kg of feed, the global warming impact amounted to 391 kg CO_2-eq for the "Soybean" scenario and to 471 kg CO_2-eq for the "Rapeseed" scenario (Table 2). Greenhouse gas (GHG) emissions mainly occurred in the arable fields (Figure 2a), predominantly as N_2O, which contributed 65% and 69% of the total emissions for the "Soybean" and "Rapeseed" scenarios, respectively. Direct field emissions had a higher impact compared to that of agricultural machinery (including the production of machinery and the emissions during cropping operations), fertilizer production and transport. In particular, the transatlantic shipping of soy meal, which is one of the main differences between the two rations,

contributed only 3% of the overall impacts and did not offset the higher GHG emissions due to synthetic N fertilizer production with the rapeseed ration. The global warming impact of the fertilizer production phase was indeed four times higher for the "Rapeseed" scenario than for the "Soybean" one, amounting to 91.8 and 23.6 kg CO_2-eq per functional unit, respectively.

It should be noted that our results, expressed in the form of rations, are much lower than those of LCAs encompassing the animal production subsystem. The latter made up circa 20% of the global warming potential estimated by Casey and Holden [10] for a complete Irish dairy system, and c. 30% of that estimated by Cederberg and Mattsson [14]. In these studies, methane from enteric fermentation contributed 50% of the final indicators, while concentrates contributed 13% in the Casey and Holden system [10].

3.1.3. Ozone layer depletion

The production of 1000 kg of feed induced a potential of ozone layer depletion of 1.92 10^{-5} CFC11-eq for the "Soybean" scenario and 2.04 10^{-5} CFC11-eq for the "Rapeseed" scenario (Table 2). For both scenarios, two halon gases (bromotrifluoromethane and bromodifluoromethane) contributed 98% of the impacts, being emitted during the extraction, treatment and transport of oil and natural gas.

3.1.4. Human toxicity

The human toxicity potential measures the impacts of toxic chemical molecules on human health. The calculated potentials were 203 kg 1.4-DCB-eq for the "Soybean" scenario and 229 kg 1.4-DCB-eq for the "Rapeseed" scenario (Table 2). Emissions occurred mainly during the production of metals (iron, copper) for the farm and industrial machines. The emissions of dioxins contributed approximately 40% of the impacts for both scenarios.

3.1.5. Ecotoxicity

The CML2 method calculates mid-point impacts on a range of ecosystems, based on an estimation of the concentrations in contaminants induced by the system under study, and their predicted no-effect concentration [3]. The ecosystems include: freshwater, marine aquatic, and terrestrial ecosystems. In other LCAs of milk production, the ecotoxicity impacts were more simply assessed as pesticide use expressed in g of active substance per FU [4, 14] .Eide [4] excluded all the pesticides used in the agricultural phase to estimate the ecotoxicity indicator. Here, the pesticides were included but the emissions of heavy metals contained in fertilizers were analysed separately.

Per FU, fresh water aquatic ecotoxicity was 19.0 kg 1.4-DCB-eq for the "Soybean" scenario, and 38.6 kg 1.4-DCB-eq for the "Rapeseed" scenario (Table 2). For the "Rapeseed" scenario, the direct field emissions of fungicides contributed 75% of the total impact result. For the "Soybean" scenario, direct field emissions of pesticides contributed only a minor part to this impact whereas a major part of it was due to the disposal of the various metals involved in the upstream chains (nickel, steel, copper).

The marine aquatic ecotoxicity impact was 4.84 10^{+4} kg 1.4-DCB-eq for the "Soybean" scenario, and 5.83 10^{+4} kg 1.4-DCB-eq for the "Rapeseed" scenario (Table 2). The differences between both scenarios stemmed from differences in fertilizer use. The production of P fertilizers largely contributed to this impact, with a share of 28% and 36% for the "Soybean" and "Rapeseed" scenarios, respectively. The use of P-fertilizers was 2.1 kg per FU for the "Soybean" scenario and 3.2 kg per FU for the "Rapeseed" scenario.

Per 1000 kg of feed, terrestrial toxicity was 1.82 kg 1.4-DCB-eq for the "Soybean" scenario and 6.43 kg 1.4-DCB-eq for the "Rapeseed" scenario (Table 2). Terrestrial ecotoxicity was largely explained by pesticide emissions. However, this impact is likely to be under-estimated

because the emissions of heavy metals were ignored. Their potential contribution is analysed in [18].

3.1.6. Photochemical oxidation

Photochemical oxidation potential was higher for the "Soybean" scenario than for the "Rapeseed" scenario, amounting to 5.13 10^{-2} vs. 3.79 10^{-2} kg C_2H_2-eq per FU, respectively (Table 2). The "Soybean" scenario has a higher impact indicator because of the sea transport and industrial crushing phases. The emissions of SO_2 explained approximately 40% of the impacts in both scenarios. Hexane losses from industrial crushing contributed 21% of the "Soybean" scenario impact.

3.1.7. Acidification

Per FU, the acidification potentials were 2.13 kg SO_2-eq and 3.39 kg SO_2-eq for the "Soybean" and "Rapeseed" scenarios, respectively (Table 2). The higher impact for the "Rapeseed" scenario can be explained by the emissions of ammonia upon spreading of slurry as a fertilizer for rapeseed production on the dairy farms. Moreover, the production of synthetic fertilizers used on the cereal farms caused further emissions of acidifying gases, with a larger share again for the “Rapeseed” ration because of the higher N requirements of rapeseed (Figure 2b). Ammonia losses upon slurry spreading may be strongly mitigated by appropriate application techniques, which are addressed in the Discussion section.

Haas et al. [24] reported acidification potentials of 119 kg SO_2-eq and 136 kg SO_2-eq per ha for an extensified dairy farming system and an intensive one, respectively, considering ammonia emissions to occur only during the storage and application of manure. Using an area-based FU favoured the extensified system, which had a lower stocking rate (livestock unit per ha). Expressing these results per 1000 kg of milk yields acidification potentials of 13 and 14 kg SO_2-eq for the intensive and extensified systems, respectively. The values we

obtained here only represent a fraction of these indicators because the impacts were not assessed on a whole-farm level, as were those of Haas et al. The production of 1000 kg of our "Rapeseed" ration corresponds to 26% of their intensive systems (expressed as 1000 kg of milk), and the production of 1000 kg of "Soybean" ration a share of 16%.

3.1.8. Eutrophication

The "Rapeseed" and "Soybean" scenarios have a similar eutrophication potential: 3.59 kg PO_4-eq vs. 3.89 kg PO_4-eq, respectively (Table 2). Direct field emissions per FU were slightly higher for rapeseed production than for soybean production (Figure 2c).

Hospido et al. [5] estimated the eutrophication potential of 1000 litres of packaged liquid milk to 5.31 kg PO_4-eq of which 73% originated from dairy feed production. Since the production of 1000 L of raw milk required the ingestion of 1303 kg of feed, their eutrophication potential was equivalent to 3.0 kg PO_4-eq per 1000 kg of feed, which is a similar potential of our values. In the same way, Cederberg and Mattsson [14] estimated the eutrophication potential to 6.05 kg PO_4-eq for the production of 1000 kg energy corrected milk at the farm gate. Their result, expressed per 1000 kg of feed, was equal to 4.2 kg PO_4-eq. Thus, our indicators are similar to other literature estimates, even if the latter did not consider differences in feed composition.

3.1.9. Land use

Land use for the production of 1000 kg of ration was 677.4 m^2 $year^{-1}$ with the "Soybean" scenario, and 1056.3 m^2 $year^{-1}$ with the "Rapeseed" scenario (Table 2). As could be expected, land use from agricultural activities was markedly higher than industrial activities. Two facts may explain the differences in land use between the two feeding scenarios. On the one hand, the proportion of concentrated feed differs between the two rations: 15.8% for the "Soybean" ration vs. 25.3% for the "Rapeseed" ration. On the other hand, the amount of meal potentially

produced by one ha of arable land is 2000 kg for soy meal vs. 1680 kg for rapeseed meal. Soy meal production is thus more efficient from a land use point of view, being more productive per unit area and also per unit of feeding ration.

3.2. Contributions of feed ingredients

For a given ration, it is possible to analyse the contribution of each ingredient to the various impacts. For the "Rapeseed" ration, the concentrates contributed more than 50% of all the impact indicators (Figure 3a), although, the proportion of concentrates in the rations is much less than that of maize silage, on a mass basis. For the "Soybean" Ration, the concentrates contributed also more than 50% of all the impact indicators except for climate change, acidification and eutrophication (Figure 3b). The contribution of concentrates to the impacts was somewhat higher for the "Rapeseed" ration because their share is larger than in the "Soybean" ration: 223 kg rapeseed meal plus 30 kg heat-treated soy meal vs. 138 kg soybean meal plus 20 kg heat-treated soybean meal for 1000 kg of feed. The other ingredients had much less importance in the impact results.

4. Discussion and interpretation

4.1. The advantages of legumes crops

The "Soybean" scenario was more environmentally-efficient because of i/ its low consumption of agricultural inputs, most notably fertilizer N, ii/ lower direct emissions in the field, and iii/ reduced crop management operations. Most of the traits behind the soybean's superiority are related to its being a legume crop. The use of leguminous protein crops grown in France may thus arguably appear as the optimal strategy to reduce the environmental load of feed production, compared to using meal from an oil crop such as rapeseed. Protein crops such as lupine and peas would have several advantages: higher protein content, lower N fertilisation requirements due to symbiotic fixation of atmospheric N, and integration in low-

input agricultural systems. However this option was not investigated here because of technical problems in processing lupine and pea grains to produce meal adequate for cow feed. In fact, the extrusion phase and the heat treatment of the lupine grains resulted in proteins that were too soluble in the cows' rumen. The heat treatment did not produce sufficient protein protection against rumen degradation [17]. This lack of a domestic legume fodder crop alternative to soybean explains why our results contradict that of other studies comparing home-grown and imported animal feed.

4.2. Comparison with other similar studies

As mentioned in the introduction, published LCAs of animal products are mostly favourable to local fodder production systems. Such examples include pig meat in France and Sweden, and milk in Sweden and Ireland, and are discussed below in connection with our results. In pig production, the choice of feed ingredients largely determines the environmental impact of meat production. In three LCAs of pig meat production, Cederberg and Flysjö [8], Eriksson et al. [11], and van der Werf et al. [12] recommended using locally-produced feed ingredients as an alternative to soybean meal, by including pea crops in the on-farm crop rotations. The use of pea crops generated similar environmental benefits to those described above for the soybean, making it possible to reduce the use of mineral fertilizers on the farm in comparison with crop rotations exclusively based on cereals [11]. However, pea crops seemed to induce higher N leaching losses than rapeseed or soybean crops [12]. In comparison with soybean, the incorporation of fodder pea into the diet increases the output of protein per hectare, and reduces overseas transport of protein concentrates and pesticide use. Additionally, peas may be incorporated in the final feed without any pressing or processing stage. In previous LCAs of pig systems, soybean was not compared with rapeseed, but was globally not a better alternative in comparison with local, high-protein legume crops.

Regarding dairy production, there has not been yet, to our knowledge, any study comparing cow rations with similar production targets and forage systems, as carried out here. Published LCAs of milk production focused on the comparison of farming systems whether intensive, extensified or organic [14, 24]. In these cases, the differences between scenarios comprised the stocking rate, the milk productivity per cow, their pasture intake, and the rate of purchased concentrates. The combination of these factors makes it difficult to single out the contribution of concentrates in the overall environmental performance of the production systems, and to compare them with our results. As a consequence, no clear conclusion may be drawn regarding the substitution of imported concentrates with locally-produced rapeseed meal. On the one hand, Cederberg and Mattsson [14], in a comparison of conventional and organic milk production in Sweden, recommended using domestic or regionally-produced rapeseed meal as opposed to ingredients imported from other continents to reduce the use of energy and pesticides – but with little quantitative data to support this claim. On the other hand, Casey and Holden [10] assessed the greenhouse gas balance from an average Irish milk production system and advised to replace rapeseed in the diet with ingredients of equal nutritional value but lower global warming potential.

Our results may seem to contradict the above-mentioned studies, but it should be stressed that the original question we addressed was: which alternative crop may improve the environmental performance of French dairy farms in comparison with imported soybean? In that respect, the production of rapeseed meal did not emerge as a good option. If technically possible, the use of rapeseed oil as biofuel on the farm provides another option to improve the environmental balance of rapeseed meal, as discussed in section 4.5.

4.3. Carbon sequestration in soil

Crop management (in particular residue management and tillage practices) may influence the organic matter content of arable soils, and result in either net release or sequestration of CO_2.

Variations in soil C stocks therefore affect the climate change indicator, depending on whether the soil is a net sink or source of atmospheric CO_2. In the above results, soil C variations were not taken into consideration. In this paragraph, we examine ways of better accounting for soil C variations in the LCA.

For the soybean production in Brazil, a change of tillage practice from conventional tillage to no-till, mulch-based cropping systems has been shown to induce a sequestration of C in the agricultural soils of the Cerrados. These oxisols may therefore function as a sink for atmospheric CO_2. Bayer et al. [25] estimate that no-till practices resulted in an additional sequestration rate of 0.35 Mg C ha^{-1} $year^{-1}$ compared to conventional tillage on native Cerrados soils' C content. Metay et al. [26] reported the same rate when comparing no-till under cover crops and disc tillage on Cerrados soils. We accounted for such a rate of C sequestration in our system study by translating it as an equivalent decrease of atmospheric CO_2 for the soybean production chain. As a result, the global warming indicator for the "Soybean" scenario decreased from 391to 307 kg CO_2-eq per FU.

Regarding the "Rapeseed" scenario, the potential of C sequestration associated with the introduction of rapeseed crops into current cropping systems in France seemed negligible. Arrouays et al. [27] reviewed the existing literature on the effect of a range of management practices on soil C dynamics, and concluded that the substitution of maize or winter wheat for rapeseed would have no impact on soil C stocks. In addition, spreading of the slurry generated by the cattle-shed on the rapeseed crops has negligible potential as a C sink: slurry application was reported to have virtually no effect on total soil organic carbon [28]. Even when repeated over successive years, slurry application only results in rapid and transient increases of the soil microbial biomass [28]. Unlike soybean crops grown in Brazil, the introduction of rapeseed in the French cropping systems is expected to have no significant impact on soils' C content, and hence on the climate change indicator of the "Rapeseed" scenario.

4.4. Sensitivity to cropping techniques for rapeseed

In the baseline scenarios, rapeseed was produced with two possible managements, whether grown on dairy or cereal farms, with an equal share between the two. In order to test the sensitivity of our assessment to crop management, we calculated the impact indicators assuming 100% of the rapeseed to come either from dairy farms or from cereal farms (Table 2). Under the first hypothesis, most impacts were reduced compared to the baseline situation. The use of organic fertilizers produced on the farm decreases the reliance on mineral fertilizers, which production is costly in energy use and environmental emissions. However, acidification increased because of the volatilization of ammonia upon slurry spreading. Conversely, assuming all rapeseed to be grown in cereal farms increased most impact indicators. Two points are unfavourable to these production systems: heavy reliance on mineral fertilizers, and more frequent cropping operations. This sensitivity analysis suggests potential improvements for the rapeseed crops: using farmyard manure instead of mineral fertilizers, and reducing the number of cropping operations. However, with organic fertilizers, the emissions of compounds involved in acidification and eutrophication impacts must be controlled. A range of mitigation options are available to the farmer, including cover crops to prevent nitrate leaching, and application techniques or optimal timing with respect to meteorological conditions to reduce ammonia volatilization [29]. In our study, the rapeseed crops were fertilized with slurry applied on the stubbles of the preceeding cereal. Rapid incorporation of this slurry into the soil may mitigate the ensuing ammonia volatilization by 70% [29]. Using this abatement rate decreased ammonia emissions from 15.7 to 5.7 kg NH_3-N ha^{-1} yr^{-1}, and the acidification impact from 3.39 to 2.61 kg SO_2-eq per FU for the "Rapeseed" scenario. This result was still higher than the acidification impact of the "Soybean" scenario (2.13 kg SO_2-eq per 1000 kg of feed).

4.5. Sensitivity to direct field emissions and system expansion

In the interpretation of LCA results, the ISO standard 14 043 recommends a sensitivity check, to assess the reliability of the final results and conclusions. It consists of a comparison of the reference results with results obtained using different assumptions on the allocation methods and on the inventory data [30].

In the following paragraphs, we address two items that showed a significant contribution to the final results: the values of direct field emissions during crop growth, and the allocation between meal and oil products.

The sensitivity to the field emissions of NO_3^-, NH_3 and N_2O of the impact categories most likely to be affected (climate change, acidification and eutrophication) was simply assessed by varying the emissions by ± 10%. Fig. 4 shows that for the climate change and acidification indicators the minimum values of the "Rapeseed" scenario were higher than the maximum values of the "Soybean" scenario, whereas, for the eutrophication indicator the range of variation were similar. Our above conclusions thus appear robust to changes in direct emissions, and highlight the margin by which rapeseed emissions should be reduced to outperform the soybean scenario.

The ISO standard 14 041 [31] advises that "*allocation should be avoided by expanding the product system to include the additional functions related to the co-products*". In response to the recent increase in fuel costs, the farmers may elect to use the rapeseed oil resulting from meal production to replace fossil fuels for farm machinery. As a consequence, the co-production of oil from the crushing of rapeseed grains can be integrated within the scope of our system by expanding it to include all vegetable oil (from rapeseed and soybean) and by including the avoided burden of consumption of diesel fuel (Fig.1) (using the fuel life cycle data of Jungbluth et al. [32] for diesel). We estimated the substitution of “Soybean” scenario with “Rapeseed” scenario to generate a net surplus of 123 kg of oil per 1000 kg of feed, an

amount equivalent to 107 kg of diesel fuel. We considered that only this net surplus was used as fuel for farm machinery, the rest was considered as keeping the same initial function. Using the larger system boundaries made it possible to avoid mass allocation between products, and resulted in differentiated environmental impacts (Table S4, [18]). When making the assumption that the C plant uptake and the C emissions during combustion of vegetable oil were balanced and substituting vegetable oil for diesel fuel results in CO_2 emissions saving, which reduces the climate change impact of the "Rapeseed" scenario by 155 kg CO_2-eq. On the one hand, the ecotoxicity impacts increased 30 to 574% relative to the baseline "scenario", along with the acidification (+93%) and eutrophication (+34%) impacts. On the other hand, there was a large decrease in the abiotic depletion (-193%), climate change (-40%), ozone layer depletion (-214%), and photochemical oxidation (-77%) impacts. These results are in sharp contrast with the baseline ones based on a mass allocation strategy, but mostly evidence the benefits of substituting the renewable co-product (oil) for fossil fuel – provided it is technically possible.

5. Conclusion

The primary objective of this work was to try and answer the following question: "Does on-farm production of rapeseed meal in lieu of soybean meal imports provide opportunities for reducing the environmental impact of milk production?". We addressed it by using LCA methodology and came to the conclusion that, based on the specifics of the French production region and systems considered, continued reliance on imported soybean meal seemed to be more environmental-friendly than on locally-produced rapeseed meal. Soy meal transport from Brazil to France did not come out as a significant source of impacts, even regarding climate change. Moreover, soybean crops are part of low-input cropping systems, with extensive use of pesticides and N-fertilizers, and no-till techniques that reduce the needs in agricultural machinery and increase the C stock of soils. However, the environmental

performance of rapeseed crops may be improved by reducing inputs, and particularly synthetic fertilizer N, as was shown in a sensitivity analysis.

From a methodological standpoint, the robustness of our conclusions should be further tested by improving the estimation of direct field emissions using biophysical soil-crop models. Expanding the system to embed the oil co-product was tested against a mass-based allocation strategy, and gave contrasting results in terms of impacts. One impact category out of 10 was improved when substituting the “Soybean” ration with the “Rapeseed” ration and using the mass-based allocation strategy, whereas 4 out of 10 were improved with system expansion.

Lastly, the fact that either production system may induce land use changes has not been dealt with in this study, although it may cause major environmental impacts, essentially in Brazil due to the expansion of soy area and forest clearing [33]. Similarly in the rapeseed scenario, the consequences of farmers diverting their cereal land to grow rapeseed should be considered, since it implies either an increase of cereal yields on the remaining cropland or a gain of new arable land over natural ecosystems. Fully tackling the issue of land use changes (both direct and indirect) and of spatial context therefore appears a major challenge in future application of LCA methodology to agricultural chains.

Acknowledgements

The authors would like to thank A. Quinsac (CETIOM), P. Carré (CETIOM), P. Brunschwig (Institut de l’élevage), J.C. Sourie (INRA Economie publique).

M. Corbeels (CIRAD), J.P. Gillet (Arvalis), G. Arjauré and J. Charbonnaud (CETIOM) are acknowledged for help in the data inventory of soybean and rapeseed crops and we wish to thank A. Aveline (ESA Angers) for useful advice and discussions.

Financial support from ACTA (Association de Coordination Technique Agricole) is acknowledged.

References

[1] ISTA, Internationale Statistische Agrarinformationen. Oil World Annual 2004. Hamburg, Germany: Mielke GmbH, 2004.

[2] Brunschwig P, Cadot M, Lemarié J. Le point sur le tourteau de colza pour les bovins (Overview on rapeseed meal for cattle feed) [in French]: Proléa, CETIOM, Institut de l'Elevage, 1996.

[3] Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, et al. Handbook on life cycle assessment. Operationnal Guide to the ISO standards. Leiden, Pays-bas: Centre of environmental Science, Université de Leiden, 2002.

[4] Eide MH. Life Cycle Assessment (LCA) of industrial milk production. International Journal of Life Cycle Assessment 2002;7(2):115-26.

[5] Hospido A, Moreira MT, Feijoo G. Simplified life cycle assessment of galician milk production. International Dairy Journal 2003;13(10):783-96.

[6] Bernesson S, Nilsson D, Hansson PA. A limited LCA comparing large- and small-scale production of rape methyl ester (RME) under Swedish conditions. Biomass & Bioenergy 2004;26(6):545-59.

[7] Cederberg C. Life Cycle Assessment of milk production, a comparison of conventional and organic milk. SIK report Nr 643.; 1998.

[8] Cederberg C, Flysjö A. Environmental assessment of future pig farming systems. SIK report Nr 723-2004. 2004.

[9] Cederberg C, Flysjö A. Life Cycle Inventory of 23 Dairy Farms in South-Western Sweden. SIK report Nr 728. 2004.

[10] Casey JW, Holden NM. Analysis of greenhouse gas emissions from the average Irish milk production system. Agricultural Systems 2005;86(1):97-114.

[11] Eriksson IS, Elmquist H, Stern S, Nybrant T. Environmental systems analysis of pig production - The impact of feed choice. International Journal of Life Cycle Assessment 2005;10(2):143-54.

[12] Van der Werf HMG, Petit J, Sanders J. The environmental impacts of the production of concentrated feed: the case of pig feed in Bretagne. Agricultural Systems 2005;83(2):153-77.

[13] Van der Werf HMG. The evaluation of the environmental impacts of protein sources for concentrated feed. In: Proceedings workshop “Grain legumes and the environment: how to assess benefits and impacts”; 2004; Zürich, Switserland: European Association for grain legume research, Paris, France; 2004. p. 27-33.

[14] Cederberg C, Mattsson B. Life cycle assessment of milk production - a comparison of conventional and organic farming. Journal of Cleaner Production 2000;8(1):49-60.

[15] Edwards-Jones G, Milà i Canals L, Hounsome N, Truninger M, Koerber G, Housome B, et al. Testing the assertion that “local food is best”: the challenges of an evidence-based approach. Trends in Food Science & Technology 2008; 19: 265-274.

[16] ISO International Standard 14 040. Environnemental management – Life cycle assessment – Principles and Framework. Geneva, Switzerland: International Organisation for standardisation (ISO); 1997.

[17] Quinsac A, Brunschwig P, Sourie JC, Carré P, Lehuger S, Loisin JP, et al. Substitution, en élevage bovin laitier des oléo-protéagineux domestiques au soja importé (Subsitution of imported soybean with domestic rapeseed in dairy cow feed) [in French]. Final report ACTA 03/11; 2006.

[18] Lehuger S, Gabrielle B, Gagnaire N. Supporting Online Materials are available on: http://www-egc.grignon.inra.fr/pages-fr/ressources/Supporting_Information.pdf

[19] Nemecek T, Heil A, Huguenin O, Erzinger S, Blaser S, Dux D, et al. Life cycle inventories of agricultural production systems. Final report Ecoinvent 2000 No 15. Dübendorf, CH: FAL Reckenholz, FAT Tänikon, Swiss Centre For Life Cycle Inventories, 2003.

[20] Scopel E, Da Silva FAM, Corbeels M, Affholder FO, Maraux F. Modelling crop residue mulching effects on water use and production of maize under semi-arid and humid tropical conditions. Agronomie 2004;24(6-7):383-95.

[21] Corbeels M, Scopel E, Cardoso A, Bernoux M, Douzet JM, Neto MS. Soil carbon storage potential of direct seeding mulch-based cropping systems in the Cerrados of Brazil. Global Change Biology 2006;12(9):1773-87.

[22] Althaus HJ, Chudacoff M, Hellweg S, Hischier R, Jungbluth N, Osses M, et al. Life cycle inventories of chemicals. Dubendorf: Swiss Centre for Life Cycle Inventories, 2003.

[23] Verdonk R. Brazil oilseeds and products annual 2003. Sao Paulo: USDA Foreign Agriculture Service; 2003.

[24] Haas G, Wetterich F, Köpke U. Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. Agriculture, Ecosystems and Environment 2001;83(2001):43-53.

[25] Bayer C, Martin-Neto L, Mielniczuk J, Pavinato A, Dieckow J. Carbon sequestration in two Brazilian Cerrado soils under no-till. Soil & Tillage Research 2006;86(2):237-45.

[26] Metay A, Moreira JAA, Bernoux M, Boyer T, Douzet JM, Feigl B, et al. Storage and forms of organic carbon in a no-tillage under cover crops system on clayey Oxisol in dryland rice production (Cerrados, Brazil). Soil & Tillage Research 2007;94(1):122-32.

[27] Arrouays D, Balesdent J, Germon JC, Jayet PA, Soussana JF, Stengel P. Stocker du carbone dans les sols agricoles de France? (Sequestration of carbon in arable soils in France) [in French]. Expertise Scientifique Collective. Paris: INRA; 2002.

[28] Rochette P, Angers DA, Cote D. Soil carbon and nitrogen dynamics following application of pig slurry for the 19th consecutive year: I. Carbon dioxide fluxes and microbial biomass carbon. Soil Science Society of America Journal 2000;64(4):1389-95.

[29] Huijsmans JFM, de Mol RM. A model for ammonia volatilization after surface application and subsequent incorporation of manure on arable land. Journal of Agricultural Engineering Research 1999;74(1):73-82.

[30] ISO International Standard 14 043. Environmental management – Life cycle assessment – Interpretation. Geneva, Switzerland: International Organisation for standardisation (ISO), 2000b.

[31] ISO International Standard 14 041. Environmental management – Life cycle assessment – Goal and scope definition and inventory analysis. Geneva, Switzerland: International Organisation for standardisation (ISO); 1998.

[32] Jungbluth N. Erdöl. Sachbilanzen von Energiesystemen. Final report No. 6 ecoinvent 2000. Dübendorf and Villigen, Switzerland.; 2003.

[33] Bickel U, Dros JM. The Impacts of Soybean Cultivation on Brazilian Ecosystems. Three case studies. Amsterdam: WWF Forest Conversion Initiative; 2003.

[34] Wilcke W, Lilienfein J. Nutrient leaching in oxisols under native and managed vegetation in Brazil. Soil Science Society of America Journal 2005;69(4):1152-61.

Figure captions

Figure 1. Flow diagram of the system life cycle with initial system boundaries and with system expansion integrating vegetable oil co-production from crushing processes and its substitute, the diesel fuel.

Figure 2. Calculated environmental impacts for 1000 kg of feed: a) Climate change (kg CO_2-eq), b) Acidification (kg SO_2-eq), c) Eutrophication (kg PO_4-eq), and contributions of crop production, fertilizer production, agricultural machinery operation and sea and road transports to the indicators.

Figure 3. Relative contributions of the feed ingredients to the impact indicators for the "Rapeseed" (a) and "Soybean" (b) rations.

Figure 4. Sensitivity of the *climate change*, *acidification* and *eutrophication* impacts to direct field emissions of NH_3, N_2O and NO_3^-, for 1000 kg of feed produced with the "Soybean" and "Rapeseed" scenarios. The segments depict the minimum, maximum, and baseline value, for field emissions varying from -10% to +10% relative to their baseline value for the maize, rapeseed and soybean crops. Units: climate change, kg CO_2-eq× 10^{+2}; acidification, kg SO_2-eq; eutrophication, kg PO_4-eq.

Figure 1.

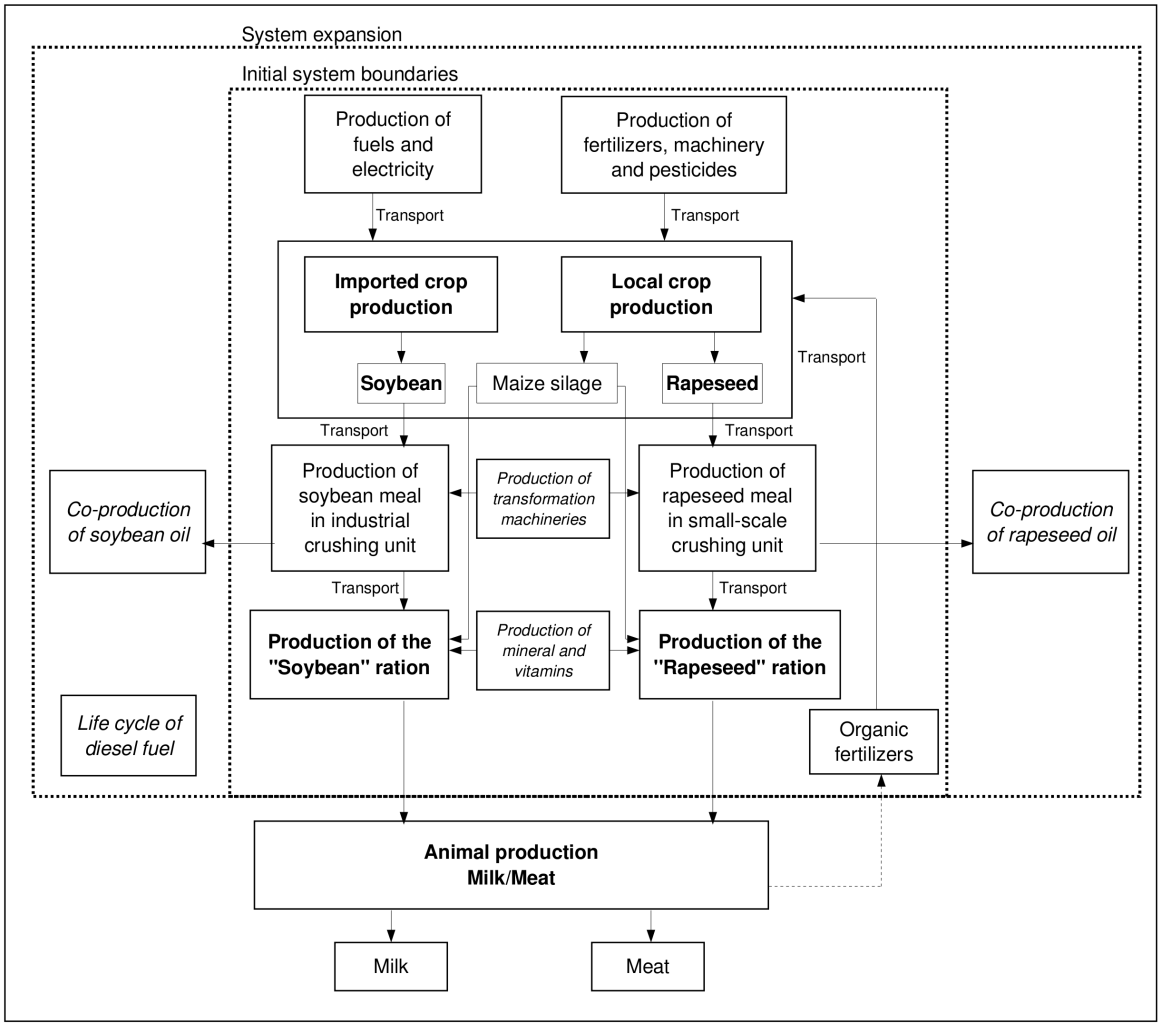

System expansion
Initial system boundaries
Production of fuels and electricity
Production of fertilizers, machinery and pesticides
Transport
Transport
Imported crop production
Local crop production
Transport
Soybean
Maize silage
Rapeseed
Transport
Transport
Co-production of soybean oil
Production of soybean meal in industrial crushing unit
Production of transformation machineries
Production of rapeseed meal in small-scale crushing unit
Co-production of rapeseed oil
Transport
Transport
Production of the "Soybean" ration
Production of mineral and vitamins
Production of the "Rapeseed" ration
Life cycle of diesel fuel
Organic fertilizers
Animal production Milk/Meat
Milk
Meat

Figure 2.

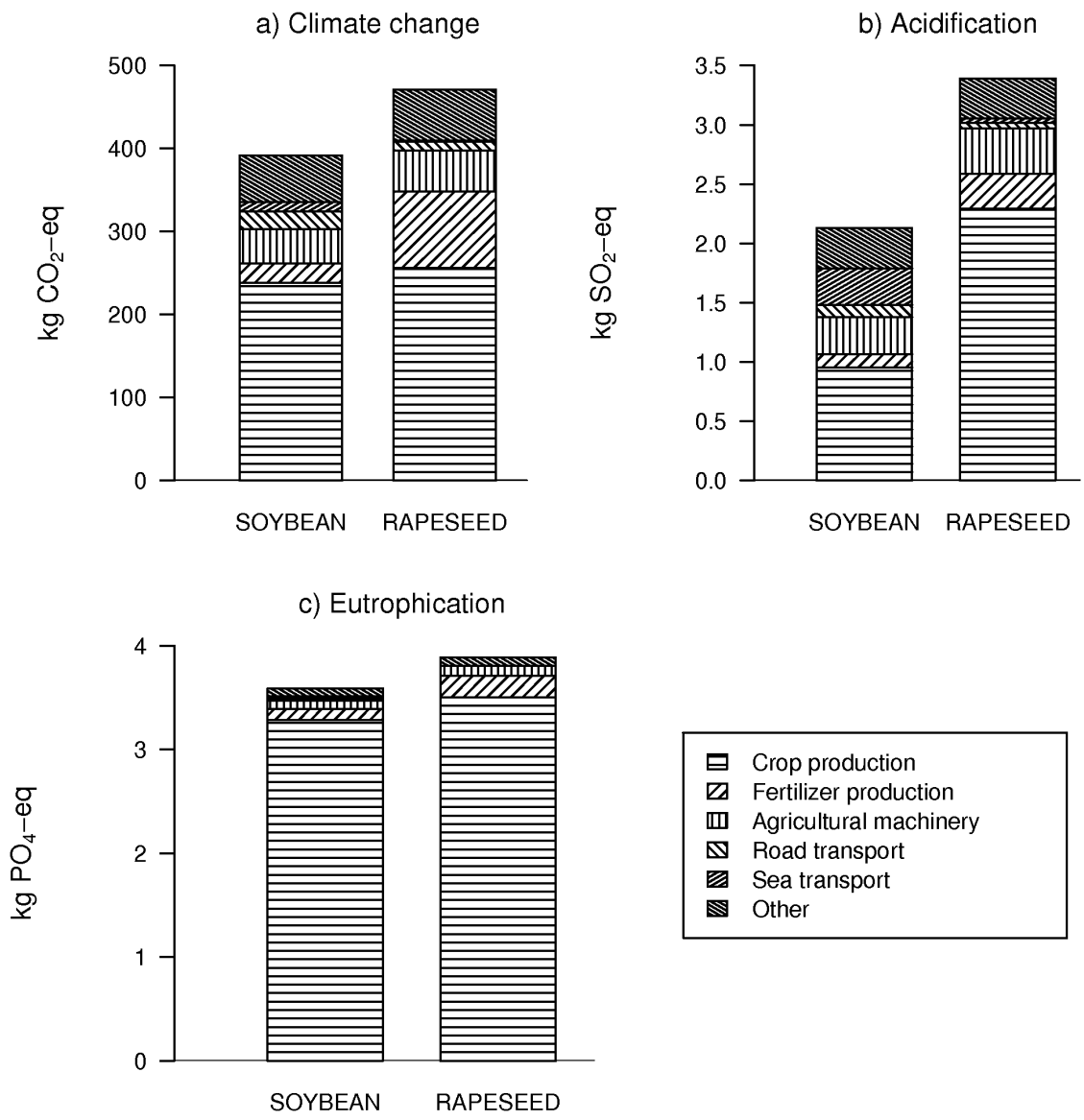
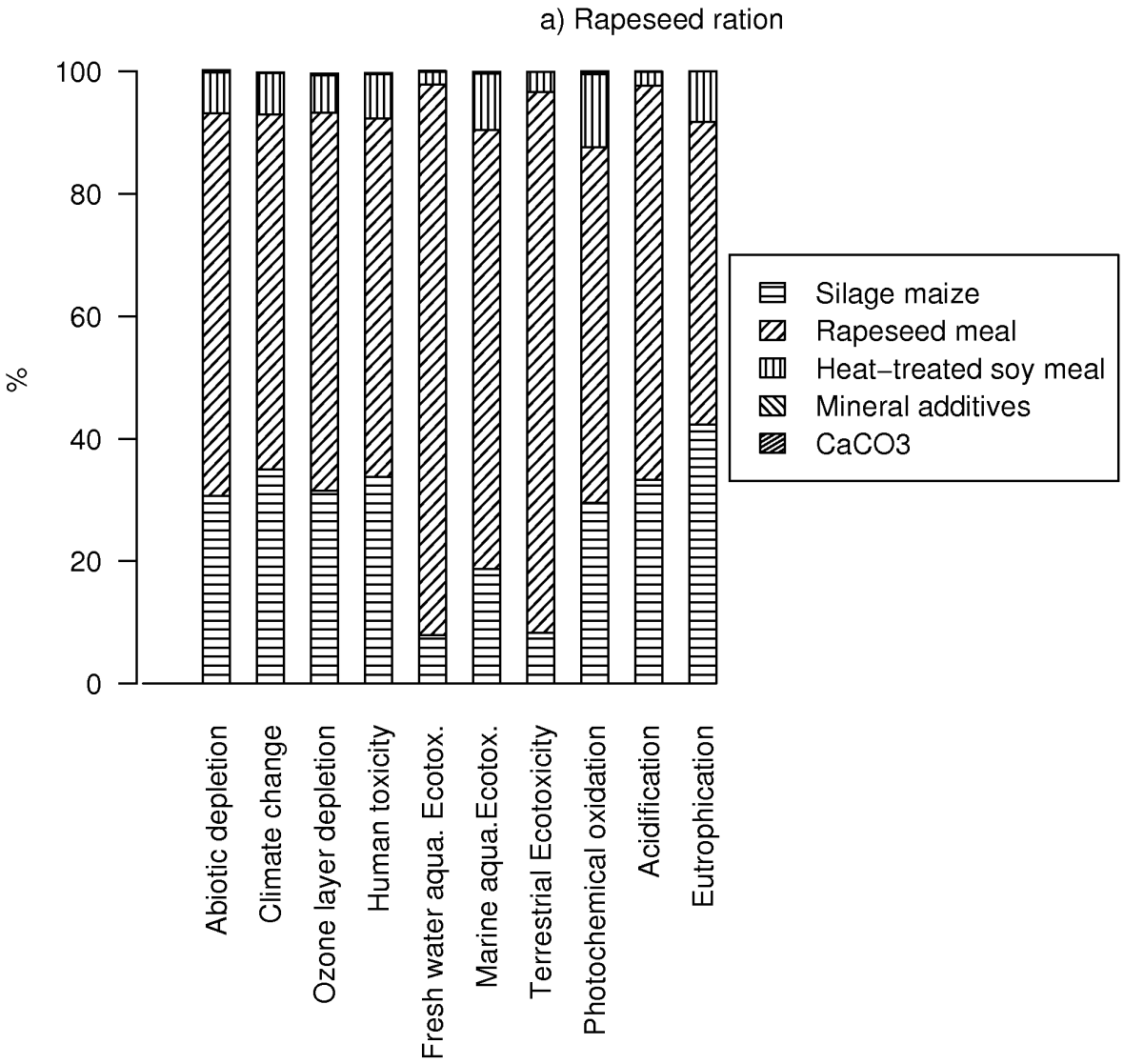

a) Climate change
kg CO_2−eq
500
400
300
200
100
0
SOYBEAN
RAPESEED
b) Acidification
kg SO_2−eq
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SOYBEAN
RAPESEED
c) Eutrophication
kg PO_4−eq
4
3
2
1
0
SOYBEAN
RAPESEED
Crop production
Fertilizer production
Agricultural machinery
Road transport
Sea transport
Other

Figure 3.


a) Rapeseed ration
100
80
60
40
20
0
%
Silage maize
Rapeseed meal
Heat−treated soy meal
Mineral additives
CaCO3
Abiotic depletion
Climate change
Ozone layer depletion
Human toxicity
Fresh water aqua. Ecotox.
Marine aqua.Ecotox.
Terrestrial Ecotoxicity
Photochemical oxidation
Acidification
Eutrophication

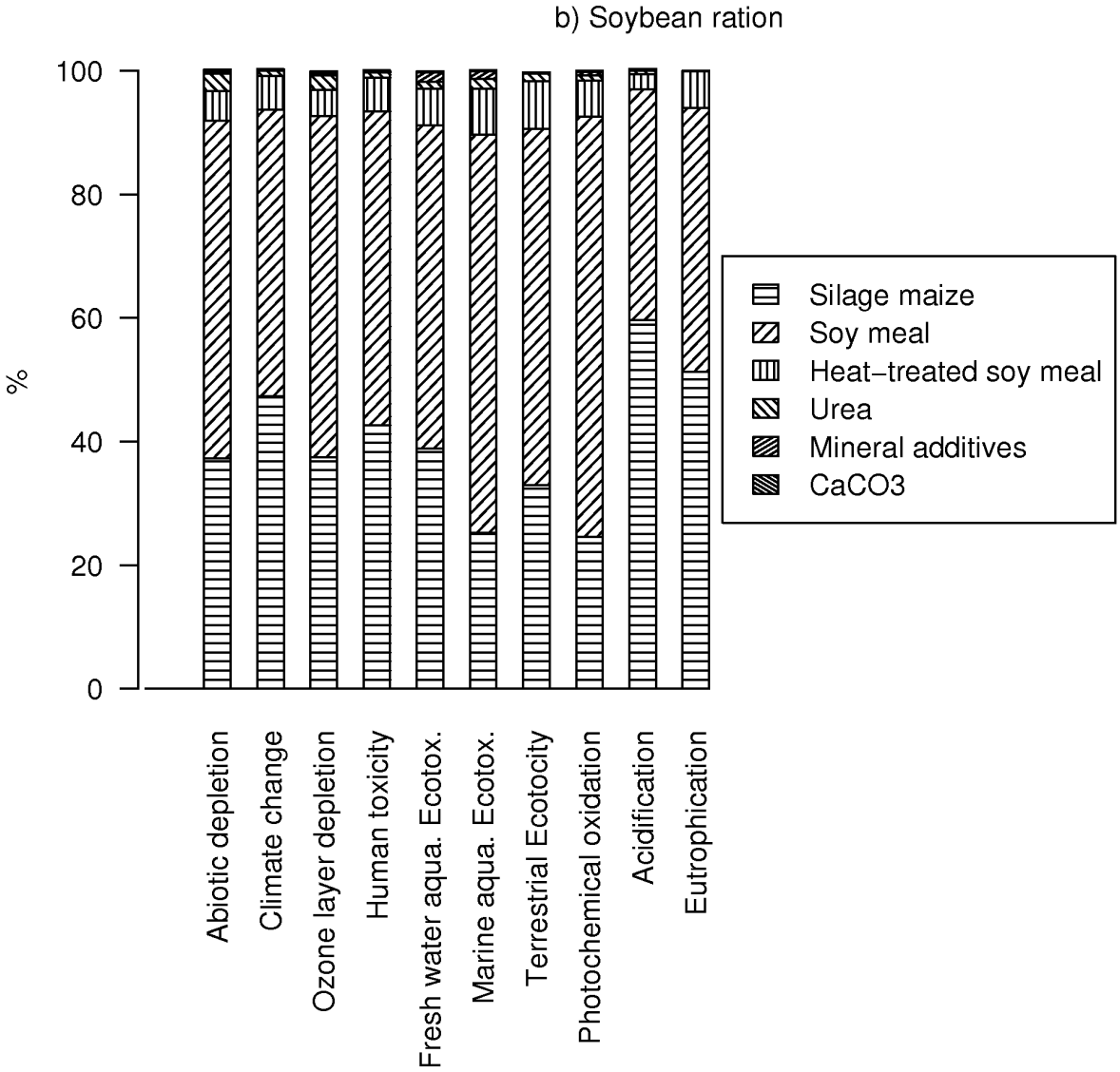

b) Soybean ration
%
100
80
60
40
20
0
Abiotic depletion
Climate change
Ozone layer depletion
Human toxicity
Fresh water aqua. Ecotox.
Marine aqua. Ecotox.
Terrestrial Ecotocity
Photochemical oxidation
Acidification
Eutrophication
Silage maize
Soy meal
Heat−treated soy meal
Urea
Mineral additives
CaCO3

Figure 4.

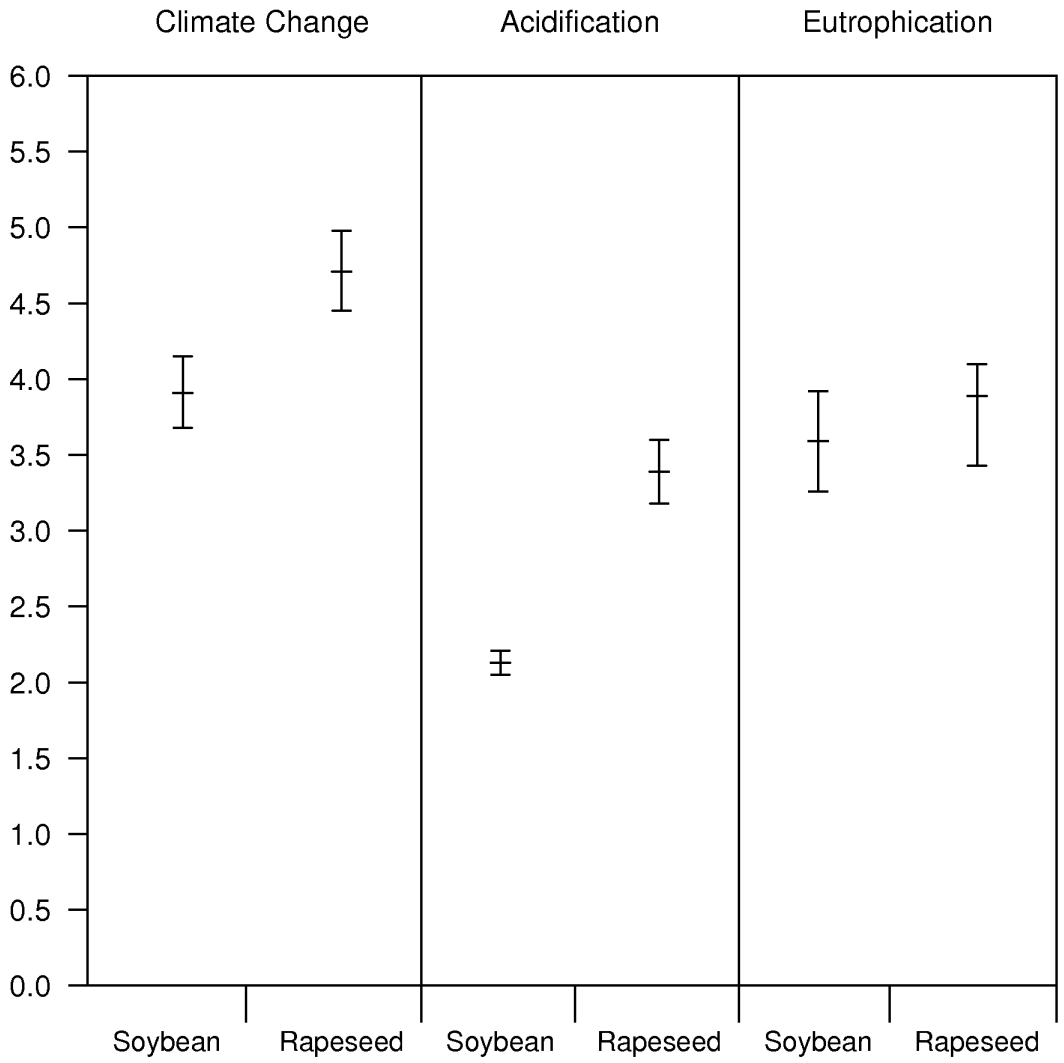

Climate Change
Acidification
Eutrophication
6.0
5.5
5.0
4.5
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
Soybean
Rapeseed
Soybean
Rapeseed
Soybean
Rapeseed

Table 1

Field emissions estimated with the Ecoinvent method, in kg N ha^{-1} or kg P ha^{-1}.

	Rapeseed from dairy farm	**Rapeseed from cereal farm**	**Soybean**	**Maize Silage**
N_2O	3.1	3.6	3.6	3.4
NO_x	0.5	0.6	0.5	0.5
NH_3	15.7	3.0	0.2	5.8
NO_3^-	37	35	30[a]	83
PO_4^{3-}	0.04	0.05	1.2[a]	0.2

a: NO_3^- losses for soybean are from Wilcke and Lilienfein [34] and PO_4^{3-} losses for soybean are from Cederberg and Flysjö [13].

Table 2

Environmental impacts of rations' production expressed per 1000 kg of feed for the reference "Soybean" and "Rapeseed" scenarios. Two additional scenarios show the sensitivity of the results to rapeseed production in either a cereal farm (with mineral fertilisation) or a dairy farm (including manure application).

Impact category	Unit	**SOYBEAN scenario**	**RAPESEED scenario**	**100% rapeseed from dairy farm**	**100% rapeseed from cereal farm**
Abiotic depletion	kg Sb-eq	1.06	1.15	$9.49\ 10^{-1}$	1.35
Climate change	kg CO_2-eq	$3.91\ 10^{+2}$	$4.71\ 10^{+2}$	$4.11\ 10^{+2}$	$5.29\ 10^{+2}$
Ozone layer depletion	kg CFC-11-eq	$1.92\ 10^{-5}$	$2.04\ 10^{-5}$	$1.70\ 10^{-5}$	$2.37\ 10^{-5}$
Human toxicity	kg 1.4-DCB-eq	$2.03\ 10^{+2}$	$2.29\ 10^{+2}$	$2.03\ 10^{+2}$	$2.54\ 10^{+2}$
Fresh water aquatic ecotoxicity	kg 1.4-DCB-eq	$1.90\ 10^{+1}$	$3.86\ 10^{+1}$	$1.01\ 10^{+1}$	$6.66\ 10^{+1}$
Marine aquatic ecotoxicity	kg 1.4-DCB-eq	$4.84\ 10^{+4}$	$5.83\ 10^{+4}$	$3.16\ 10^{+4}$	$8.49\ 10^{+4}$
Terrestrial ecotoxocity	kg 1.4-DCB-eq	1.82	6.43	6.70	6.15
Photochemical oxidation	kg C_2H_2-eq	$5.13\ 10^{-2}$	$3.79\ 10^{-2}$	$3.04\ 10^{-2}$	$4.55\ 10^{-2}$
Acidification	kg SO_2-eq	2.13	3.39	4.11	2.67
Eutrophication	kg PO_4-eq	3.59	3.89	3.98	3.80
Land use	$m^2.year^{-1}$	$6.774\ 10^{+2}$	$1.056\ 10^{+3}$	$1.056\ 10^{+3}$	$1.056\ 10^{+3}$

Supporting Material for:

Environmental impact of the substitution of imported soybean meal with locally-produced rapeseed meal in dairy cow feed

Simon Lehuger*, Benoît Gabrielle, Nathalie Gagnaire

INRA, AgroParisTech, UMR 1091 Environment and Arable Crops, 78850 Thiverval-Grignon, France

*To whom the correspondence should be addressed. E-mail: Simon.Lehuger@grignon.inra.fr

This pdf file includes:

Materials and methods

Discussion

References

Tables S1 to S5

Figure S1

Supporting Material

Supporting materials and methods

Methods to estimate the direct field emissions

Direct field emissions were assessed using methods developed by Ecoinvent [1].

NH3 emissions were calculated from the amount of manure and slurry spread on the field with a simple empirical model from Menzi et al. [2]. The emission factor for NH_3 losses from mineral fertilizer (ammonium nitrate) was 0.02 kg of NH_3-N per kg N applied.

Nitrate leaching to ground water and P losses were estimated with simple mass balances of N and P. For nitrate leaching computations, the mass balance of N included the following items: N-mineralization from the soil organic matter, N-fertilization, crop N-uptake and ammonia losses upon fertilizer spreading. Nitrogen mineralization was estimated on a monthly basis, leading to a total of 160 kg N ha^{-1} $year^{-1}$ for soil of the study area in France. A risk factor of leaching from fertilizers was applied to the mass-balance estimate of nitrate leaching. The risk of N leaching is a fraction of potentially leachable N of the N applied through fertilizers and depends on crop type, on crops' potential rooting depth, and on the month in which fertilizers are applied. A leaching risk appears when fertilizer is applied at sub-optimal times, as quantified by Nemecek et al. [1]. Since the Ecoinvent models for P and N losses had not been adapted and tested for the pedoclimatic conditions of Brazil, we used references from the literature for those data.

Calculations of N_2O emissions are based on the IPCC Tier 1 method [3]. Direct emissions induced by organic or mineral fertilizers were calculated as a function of available N, with an emission factor of 1.25%. Available N corresponds to the fertilizer N applied minus the losses of NH_3. Indirect emissions occurring through NH_3 volatilization and NO_3^- leaching were calculated with emission factors of 1% and 2.5%, respectively according to Mosier et al. [4].

The amounts of N fixed by crops and returned to soils as crop residues were estimated with IPCC references [3]. NO_x emissions were simply estimated as 21% of N_2O emissions [1].

Characterisation models of the CML 2 baseline method

The impact category *Abiotic Depletion* relates to the extraction of minerals and fossil fuels necessary to the system [5]. The abiotic depletion factor is based for each mineral or fossil fuel extracted on concentration reserves and rate of deaccumulation. The equivalent substance unit is kg antimony (Sb) per kg of extraction.

The characterisation model for *Climate Change* impact category is based on the IPCC model [6], which defines the Global Warming Potential of the major greenhouse gases. The characterisation factor is global warming potential for a 100-year time horizon (GWP100) for each greenhouse gas emission (in kg CO_2 equivalent/kg emission).

The characterisation model for the *Stratospheric Ozone Depletion impact* defines the ozone depletion potential of different gases (kg CFC-11 equivalent/kg emission).

The characterisation factors for the *Human Toxicity* impact category are expressed as Human Toxicity Potentials and calculated with the multimedia fate and exposure model USES-LCA, expressed in kg 1.4-dichlorobenzene equivalent (1.4 DCB) /kg emission [7].

The *Ecotoxicity* impact categories refer to the impacts of toxic substances on freshwater aquatic ecosystems, marine aquatic ecosystems and terrestrial ecosystems. Characterisation factors are expressed as Ecotoxicity Potential for each emission of a toxic substance (kg 1.4-dichlorobenzene equivalent/kg emission) [7].

The characterisation factors for the *Photo-oxidant Ozone formation* impact category are based on the photochemical ozone creation potential for precursor gases [8]. Characterisation factor is expressed in kg ethylene (C_2H_2) equivalent/kg emission.

In the absence of regional references for the *eutrophication* potentials, we used the generic factors recommended by Guinée et al. [5]. They are expressed in kg PO_4^{3-} equivalent/kg

emission, with the following values: NH_3: 0.35, NO_3^-: 0.1, NO_x: 0.13, PO_4^{3-}: 1. Although *acidification* is an impact category in which local conditions are very significant, we also used the average European characterisation factors recommended by Guinée et al., [5], with the following values: N_2O: 0.5, NO_x: 0.5, SO_2: 1.2, NH_3: 1.6 kg SO_2-eq.

The SimaPro 6 software package (PRé Consultants, Amersfoort, NL) was used to model the life cycles and to assess the impacts.

Supporting Discussion

Influence of functional unit and system boundaries

Although the rations were calculated to have the same energy and protein contents, they were given *ad libitum* to the animals, and finally the daily intake of DM was 10% higher with the "Rapeseed" ration compared to the “Soybean” one. The daily intake of maize silage was nearly identical for the two rations, ranging from 16.4 kg to 16.6 kg DM, but the intake of concentrates was twice larger for the "Rapeseed" ration (5 kg rapeseed meal vs. 2.8 kg soy meal day^{-1} for one cow). As a result, the cows' milk productivity was 14% higher with rapeseed meal than with soybean meal. To take this difference (in favour of the rapeseed ration) into account, we tested here taking milk as a functional unit (FU), in lieu of the rations. Thus, we used “one kilogram of energy corrected milk” (ECM) as FU, following the standardization proposed by Sjaunja et al. [9] to correct for variations in fat and protein contents in raw milk. The feeding trial provided the milk production rates and compositions for the two rations, and made it possible to calculate their overall yield: the "Soybean" ration produced 29.6 kg ECM day^{-1}, and the "Rapeseed" ration, 33.7 kg ECM day^{-1}. The subsystem of animal production was excluded from the alternative system boundaries based on the assumption that there were negligible differences in terms of emissions between the two scenarios downstream of the rations’ production. In addition, all the impacts generated by the

cows herd were allocated to milk production, without considering concomitant meat production through the selling of cull cows and surplus calves. Thus, the values resulting from this alternative system encompass the effect of differentiated daily intake and milk productivity between both scenarios. The impact indicators (Fig. S1) reveal similar trends as those obtained with 1000 kg of feed as FU. They are all higher with the "Rapeseed" scenario than with the "Soybean" one, except for the photochemical oxidation indicator. Even though the productivity of cows increased by 14% with the rape meal ration, this higher milk output could not compensate for the higher environmental impacts associated with the agricultural production of rapeseed grains, combined with the two-fold greater daily intake of rapeseed meal compared to soy meal.

Heavy metal accumulation in agricultural soils

The results did not take into account the presence of heavy metals (HM) in agricultural inputs, although some fertilizers are known to contain significant amounts of HM [1]. This paragraph explores the sensitivity of our results to the integration of heavy metals in the assessment (Table S5). It is based on the Ecoinvent methodology for calculating emissions of HM to agricultural soil [1], which is a simple input-output balance sheet. Soil inputs of HM from agricultural inputs are inventoried, and outputs (as plant uptake) are calculated based on expected concentrations in plant tissues and plant biomass at harvest. The use of the default concentrations in plants produced erroneous results, with plant uptake exceeding the amounts of HM present in the soil. Consequently, the impact indicators of human toxicity or ecotoxicity were negative. In order to test the effect of HM accumulation in soil as well, we hypothesized that either 100% of heavy metal from inputs accumulated in the soil (in the absence of plant uptake), or that no HM were input to the soil (which corresponds to the baseline results). The concentrations of HM in the inputs were taken from Nemecek et al. [1], and include mineral fertilizers, farmyard manure and seeds. The impacts results under both

hypothesises are compared in Table S1. The hypothesis of 100% HM accumulation in soils resulted in significantly greater impacts than without the consideration of HM. Under hypothesis of no accumulation of HM, the indicators of human toxicity, fresh water aquatic ecotoxicity, marine aquatic ecotoxicity and terrestrial ecotoxicity were higher for the "Rapeseed" scenario, while under the alternative hypothesis (100% accumulation in soil), the results were approximately similar between the two scenarios.

Improvement of the life cycle inventory

Direct field emissions associated with crop production appeared critical for the climate change, acidification, and eutrophication impacts. Therefore, a better estimation of field emissions of NO_3^-, NH_3, N_2O, and NO_x has to be sought. This calls for the use of process-based models simulating the fate of agricultural inputs as a function of local soil and climate characteristics, which are paramount to determining direct field emissions with improved accuracy. There lies a perspective for this study, consisting of using the CERES-EGC model for estimating direct field pollutions associated with rapeseed in France and soybean in Brazil. CERES-EGC is a mechanistic model simulating the dynamics of water, carbon, and nitrogen in the soil-crop systems [10] and for which different modules were added to predict N_2O emissions [11], NH_3 volatilization [12] and NO_x emissions [13, 14].

Supporting references

[1] Nemecek T, Heil A, Huguenin O, Erzinger S, Blaser S, Dux D, et al. Life cycle inventories of agricultural production systems. Final report Ecoinvent 2000 No 15. Dübendorf, CH: FAL Reckenholz, FAT Tänikon, Swiss Centre For Life Cycle Inventories, 2003.

[2] Menzi H, Katz PE, Fahrni M, Neftel A, Frick R. A simple empirical model based on regression analysis to estimate ammonia emissions after manure application. Atmospheric Environment 1998;32(3):301-07.

[3] IPCC, OECD, IEA. Reference manual. Revised 1996. IPCC Guidelines for national greenhouse gas inventories. Bracknell: UK Meteorological Office, 1996.

[4] Mosier A, Kroeze C, Nevison C, Oenema O, Seitzinger S, Cleemput Ov. Closing the global N2O budget: nitrous oxide emissions through the agricultural nitrogen cycle: OECD/IPCC/IEA phase II development of IPCC guidelines for national greenhouse gas inventory methodology. Nutrient Cycling in Agroecosystems 1998;52(2/3):225-48.

[5] Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, et al. Handbook on life cycle assessment. Operationnal Guide to the ISO standards. Leiden, Pays-bas: Centre of environmental Science, Université de Leiden, 2002.

[6] Forster P, Ramaswamy V, Artaxo P, Berntsen T, Betts R, Fahey DW, et al. Changes in Atmospheric Constituents and in Radiative Forcing. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

[7] Huijbregts MA, Thissen U, Guinée JB, Jager T, Van de Meent D, Ragas AMJ, et al. Priority assessment of toxic substances in the life cycle assessment, I: Calculation of toxicity potentials for 181 substances with the nested multi-media fate exposure and effects model USES-LCA. Chemosphere 2000;41(4):575-88.

[8] Derwent RG, Jenkin ME, Saunders SM, Pilling MJ. Photochemical ozone creation potentials for organic compounds in northwest Europe calculated with a master chemical mechanism. Atmospheric Environment 1998;32(14-15):2429-41.

[9] Sjaunja LO, Baevre L, Junkkarinen L, Pedersen J, Setala J. A Nordic proposal for an energy corrected milk (ECM) formula. In: 27th session of the International Comission for Breeding and Productivity of Milk Animals; 1990; Paris; 1990.

[10] Jones CA, Kiniry JR. CERES-N Maize: a simulation model of maize growth and development. Temple, TX: Texas A&M University Press, College Station, 1986.

[11] Gabrielle B, Laville P, Henault C, Nicoullaud B, Germon JC. Simulation of nitrous oxide emissions from wheat-cropped soils using CERES. Nutrient Cycling in Agroecosystems 2006;74(2):133-46.

[12] Génermont S, Cellier P. A mechanistic model for estimating ammonia volatilization from slurry applied to bare soil. Agricultural And Forest Meteorology 1997;88:145-67.

[13] Laville P, Henault C, Gabrielle B, Serca D. Measurement and modelling of NO fluxes on maize and wheat crops during their growing seasons: effect of crop management. Nutrient Cycling in Agroecosystems 2005;72(2):159-71.

[14] Rolland MN, Gabrielle B, Laville P, Serca D, Cortinovis J, Larmanou E, et al. Modeling of nitric oxide emissions from temperate agricultural soils. Nutrient Cycling in Agroecosystems 2007;DOI 10.1007/s10705-007-9122-6.

Supporting Table S1

Composition (in percents of dry matter) of the rations supplemented with either soybean meal ("Soybean" ration) or rapeseed meal ("Rapeseed" ration).

	Soybean ration	**Rapeseed ration**
	(% DM)	(% DM)
Maize silage	82.5	73.7
Soybean meal	13.8	-
Rapeseed meal	-	22.3
Heat-treated Soybean meal	2	3
Urea	0.1	-
Mineral additives 7/21/5[a]	1	-
Mineral additives 0/28/3[a]	-	0.8
$CaCO_3$	0.1	0.2

a: %P/%Ca/%Mg.

Supporting Table S2

Crop yields, agricultural inputs, number of cropping operations and pesticide use for the major crops used in the rations. N_{min} and N_{org} denote mineral and organic (slurry and manure) forms of fertilizer N.

	Maize from dairy farm	**Rapeseed from dairy farm**	**Rapeseed from cereal farm**	**Soybean from Brazil**
Yield (kg DM ha^{-1})	11 000	3000	3000	2500
Fertilisation	150 kg N_{org} (30 t manure) 30 kg N_{min}	120 kg N_{org} (40 m^3 slurry) 70 kg N_{min}	150 kg N_{min} 70 kg P 100 kg K	8 kg N_{min} 31 kg P 57 kg K 50 kg lime
Cropping operations	9	11	13	7
Pesticide (active ingredient in kg ha^{-1})	3	2.4	2.4	1.6
Previous crop	Wheat	Wheat	Wheat	Soybean

Supporting Table S3

Energy and input consumption of the small-scale and industrial crushing processes, per 1000 kg of seeds processed.

	Small-scale crushing process	**Industrial crushing process**
Energy		
Natural gas (kWh)	136.7	270
Electricity (kWh)	76.7	46
Inputs		
Water (L)	166.7	280
Hexane (kg)	-	0.4
Machinery (kg)	0.37	0.021
Building area (m^2)	0.00034	0.034

Supporting Table S4

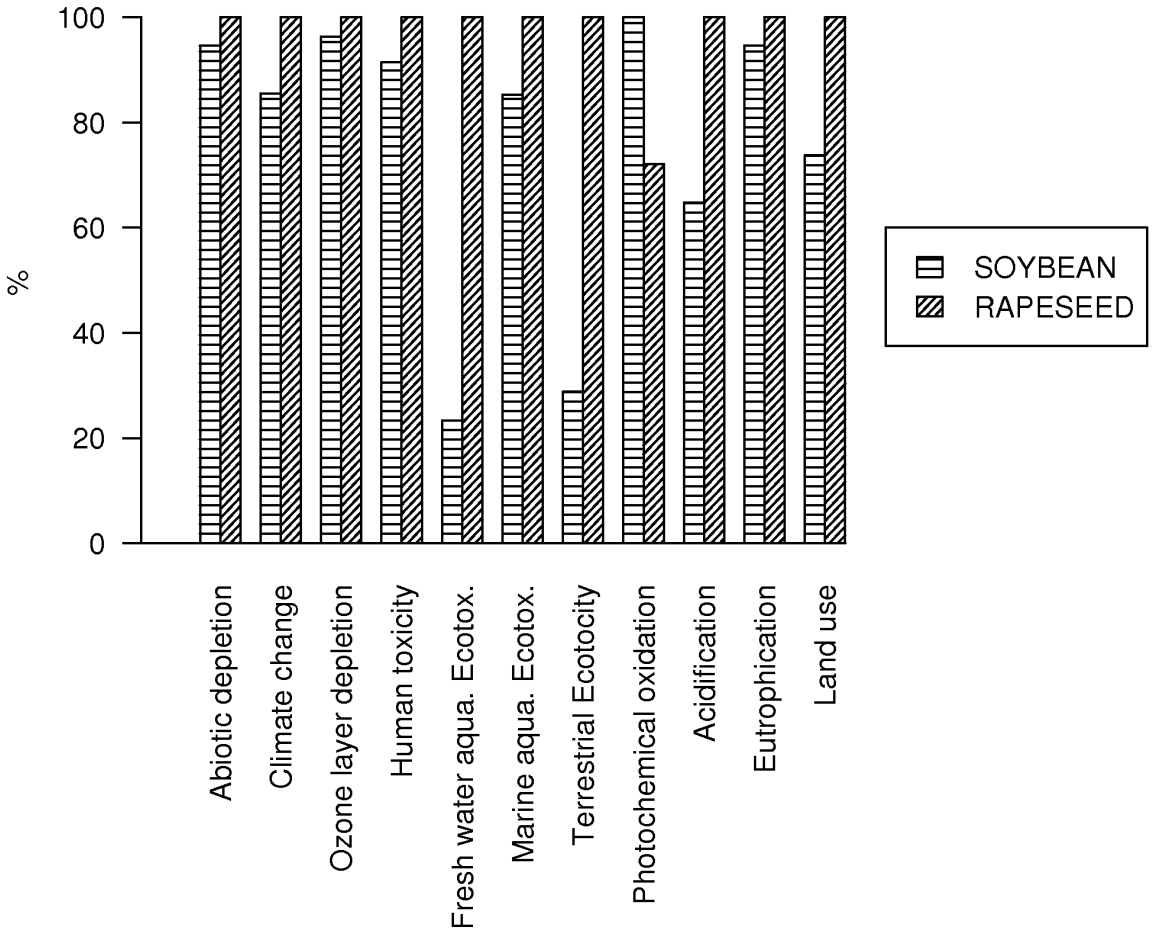
Environmental impacts resulting from the substitution of the "Soybean" scenario with the "Rapeseed" scenario according to the baseline system boundaries and with system expansion. System expansion encompassed the net surplus of oil co-production resulting from the crushing processes and its substitution to diesel fuel for farm machinery.

Impact category	Unit	Substitution with mass allocation	Substitution with system expansion
Abiotic depletion	kg Sb-eq	$9.0\ 10^{-2}$	-2.05
Climate change	kg CO_2-eq	$8.0\ 10^{+1}$	$-1.55\ 10^{+2}$
Ozone layer depletion	kg CFC-11-eq	$1.2\ 10^{-6}$	$-4.11\ 10^{-5}$
Human toxicity	kg 1.4-DCB-eq	$2.6\ 10^{+1}$	$6.0\ 10^{+1}$
Fresh water aquatic ecotoxicity	kg 1.4-DCB-eq	$6.40\ 10^{+1}$	$1.09\ 10^{+2}$
Marine aquatic ecotoxicity	kg 1.4-DCB-eq	$9.9\ 10^{+3}$	$1.57\ 10^{+4}$
Terrestrial ecotoxocity	kg 1.4-DCB-eq	4.61	8.00
Photochemical oxidation	kg C_2H_2-eq	$-1.34\ 10^{-2}$	$-3.96\ 10^{-2}$
Acidification	kg SO_2-eq	1.26	1.99
Eutrophication	kg PO_4-eq	0.30	1.23

Supporting Table S5

Human toxicity and ecotoxicity indicators (kg 1.4-DCB-eq) according to two hypotheses: accumulation of 100% of the heavy metal (HM) inputs in soils (a) vs. no consideration of HM (b).

	a) 100% of HM inputs		b) 0% of HM inputs	
	SOYBEAN	RAPESEED	SOYBEAN	RAPESEED
Human toxicity	$5.83\ 10^{+2}$	$5.74\ 10^{+2}$	$2.05\ 10^{+2}$	$2.30\ 10^{+2}$
Fresh water aquatic ecotoxicity	$2.50\ 10^{+2}$	$2.92\ 10^{+2}$	$1.90\ 10^{+1}$	$8.37\ 10^{+1}$
Marine aquatic ecotoxicity	$1.90\ 10^{+5}$	$1.84\ 10^{+5}$	$4.84\ 10^{+4}$	$5.83\ 10^{+4}$
Terrestrial ecotoxicity	$3.96\ 10^{+1}$	$4.05\ 10^{+1}$	1.8	6.4

Supporting Figure S1

Comparison of environmental impact indicators for the "Rapeseed" and "Soybean" scenarios in the case of the alternative functional unit "1 kg of energy corrected milk". The indicators are expressed as a percentage of the highest value.