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

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ABSTRACT

In Europe, dairy cow feed is made up of a large part of forage and a minor part of concentrates that balance the ration and stimulate milk production. In conventional dairy production systems, the concentrates usually consist of soybean meal imported from America with limited traceability. In France, emerging demands for higher standards of traceability, food safety, and environmental protection are prompting the development of alternative milk production systems, involving among others the substitution of imported soybean meal with locally-produced rapeseed meal in dairy cow feed.

This paper focuses on the environmental assessment of such a substitution, using the Life Cycle Assessment (LCA) methodology. Two rations were analyzed, based on either locally-produced rapeseed meal or Brazilian-produced soy meal as concentrates, and on silage maize as forage. The system under study included the production and the transport steps of all the ingredients of these two rations, taking into account the specificities of French and Brazilian cropping systems for crop production. Rapeseed meal was supposed to be crushed locally in small-scale units managed by farmers while soybean was crushed in Brazil in industrial plants prior shipping to France. The environmental impacts were expressed with respect to one kg of "energy corrected" milk for both scenarios ("Soybean" and "Rapeseed"), and involve nine impacts categories.

Crop production was the main contributor to the climate change, ecotoxicity, acidification, eutrophication and land use impact categories, while overseas transport of soy meal by cargo ships had only a marginal effect. As a result, the "Rapeseed" scenario produced higher impacts than the "Soybean" scenario for all the afore-mentioned categories. Overall, the "Soybean" scenario appeared more environmentally-efficient than the "Rapeseed" scenario because of a lower consumption of agricultural inputs, lower direct field emissions, and a reduced number of crop management operations. However, the environmental performance of

the rapeseed crop may be improved by reducing agricultural inputs and particularly synthetic fertiliser N, using the manure-produced on the farm. Our conclusions should also be mitigated by the fact we did not take into account the land use changes likely to result from an increase demand for soybean in Brazil. The expansion of soy-cropped area, at the expense of primary forest areas, appears as a cause of major environmental damages, that has no equivalent in Europe. Taking into account land-use changes in both production systems is therefore a major prospect for future research on the environmental assessment of such feed substitutions.

Keywords: Life Cycle Assessment; Soybean; Rapeseed; Concentrated feed, Dairy cow ration; Milk production; Direct field emissions; Environmental impact; Substitution

1. Introduction

In Europe, new practices for milk production are emerging that reflect increasing public concern for the traceability of the supply chains, environmental protection and food safety. In particular, animal feeding is expected to satisfy these new demands. Today, conventional dairy farms usually incorporate imported soybean meal as a concentrate in the ration of dairy cows as a high-protein complement to silage maize. The soybean meal used in Europe is predominantly produced overseas. In France, for example, 5 million tons of soybean meal were used for animal feeding in 2004, of which 90% was imported, mostly from Brazil (ISTA, 2004). 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 (peas, beans, lupines...) and oil crops (rapeseed, sunflower, flax...) may easily be substituted to imported soybean meal in animal feed (Brunschwig *et al.*, 1996). In addition, the oil produced with oil crops 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 several 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 that must be used by 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 Analysis (LCA) methodology appears as a most relevant framework.

Examples of environmental assessments based on LCA are increasingly frequent for products with an agricultural phase. In the case of industrial biofuels or animal products (milk and meat), for instance, the agricultural phase generally contributes more to impact indicators than

1 transport or industrial phases (Reinhardt, 2000; ADEME *et al.*, 2002; Berlin, 2002; Eide,
2 2002; Hospido *et al.*, 2003; Bernesson *et al.*, 2004). For animal products, within the
3 agricultural phases, it is the protein source for concentrated feed that is responsible for most
4 of environmental impacts and use of energy (Cederberg, 1998; Cederberg and Flysjö, 2004a;
5 Cederberg and Flysjö, 2004b; Casey and Holden, 2005; Van der Werf *et al.*, 2005; Van der
6 Werf, 2005). Using imported soybean meal, which is a predominant source of protein
7 concentrates in Europe, is usually not a good option to improve the environmental
8 performance of animal products, compared to locally-produced feed ingredients. Cederberg
9 (1998); Cederberg and Flysjö (2004a); Casey and Holden (2004) recommend locally-
10 produced concentrates for dairy cow feeding. Similarly, for pig meat production, the auto-
11 production of protein concentrates at local or regional scale has less impact on the
12 environment than feed imported from overseas. Road and sea transports, combined with a
13 relatively higher reliance on pesticides and P fertilizers make imported concentrates more
14 harmful to the environment than their local counterparts. This especially true for the following
15 impact categories: climate change, acidification, abiotic depletion and pesticide use
16 (Cederberg and Mattsson, 2000; Cederberg and Flysjö, 2004a; Van der Werf *et al.*, 2005; Van
17 der Werf, 2005).

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

24 The main objective of this work was thus to provide a comparison of milk production from a
25 ration based with locally-produced rapeseed meal or on imported soybean meal, in terms of

LCA results, for a region in France in which farmers are showing increasing interest in such alternatives.

2. Materials and methods

2.1. Life Cycle Analysis 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 the environmental impacts. The principle is to quantify the resources consumed and the emissions to the environment at all stages of the life cycle of the product (Guinée *et al.*, 2002). The fluxes are subsequently interpreted in terms of impacts on the environment, for a range of categories (global warming, eutrophication of ecosystems, etc..) The impacts are expressed relative to an unit of end-product or service, termed the functional unit. Here, we used one kilogram of milk as a functional unit, and the impacts were calculated by aggregating the life cycle impacts of the various ingredients of the rations. We followed the standardized LCA methodology (ISO 14 040, 1997), involving the following steps:

- 1) Definition of system boundaries and functional unit
- 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

We defined the functional unit (FU) as “one kilogram of energy corrected milk” (ECM), following the standardization proposed by Sjaunja *et al.* (1990) to correct for variations in fat and protein contents in raw milk. ECM is calculated as:

$$ECM = M ((38.3 F) + 24.2 P + 783.2) / 3140$$

where M is the mass of milk (kg), F is the fat content (g kg⁻¹) and P is the protein content (g kg⁻¹). Application of this formula to the data of Table 2 giving the milk productions and compositions for the two rations makes it possible to calculate their overall yield: for the "Soybean" ration produces 29.6 kg ECM day⁻¹, and for the "Rapeseed" ration, 33.7 kg ECM day⁻¹.

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. silage maize, minerals, and concentrates and the transport and transformation of soybean or rapeseed grains are also taken into account. The subsystem of animal production is excluded from the system boundaries based on the assumption that there were no differences in terms of emissions between the two scenarios after the production of the rations. From a qualitative standpoint, the impacts generated by animal waste are essentially a function of cattle diet (Smits *et al.*, 1995). In our case, there were no differences between both rations in terms of protein digestibility and energy content, on a dry weight basis. The quantities of N excreted by the cows essentially depend on their N intake (Vérité and Delaby, 2000), which were the same with the two rations, as was the objective of milk production (~30 kg milk cow⁻¹ day⁻¹). Thus, a priori, the excretion of N by the cows and the resulting environmental impacts could be expected to be similar across the two diets.

The crushing process of soybean and rapeseed produces two co-products: oil and meal, making it necessary to split the impacts between the co-products. We use a mass-based allocation ratio, as recommended by the ISO norms (1998). The ratios were determined from the extraction yields of the two possible oil extraction technologies: industrial or small-scale crushing units. Industrial crushing of 100 kg soybean was assumed to yield 17 kg oil and 80 kg meal (Cederberg, 1998), and small-scale crushing systems to yield 37.7 kg oil and 56.1 kg meal. Economic allocation was not possible because it is difficult to set a market value for the

rapeseed meal produced on farm, in the absence of such market yet (Sourie J.C., Pers. Comm., 2005).

Milk production leads to concomitant production of meat through the selling of cull cows and surplus calves. In fact, the daily ration is overall converted into milk, meat or calf. Because our study focused on milk production, we did not allocate between the milk and meat produced. All the impacts were thus allocated to milk production.

2.3 Life cycle inventory

Life cycle inventory of inputs and emissions were implemented for the production of each of the ration ingredients and encompassed 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 km² 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 cake used in this region. The substitution of the concentrates in the rations has an effect on the level of ingestion of the other ingredients by the cows, and also on their productivity. We thus took these differences into account, based on actual feeding trials conducted with the two types of rations in an experimental farm (Lehuger, 2005). The daily ingestions for both rations are presented in Table 1, while Table 2 presents the results on milk production and energy content. Based on these results, it was possible to determine the types and quantity of ingredients necessary for the production of one litre of milk end-product, for the soybean- and rapeseed-based rations.

2.3.1. Agricultural production of crops

Agricultural inputs comprise fertilizers, pesticides, seeds and agricultural machinery. The data were taken from the Ecoinvent database (Nemecek *et al.*, 2003), and made it possible to

1 compute the impacts associated with elementary management operations including soil
2 tillage, fertilisation, sowing, plant protection, harvest and transport. For each operation,
3 machinery and energy inputs, and emissions to the environment are inventoried. The direct
4 emissions occurring in the arable field as a result of crop cultivation are discussed in a
5 separate section.

6 Data on crop management was set according to the specificities to the study areas in France
7 and Brazil. The crops' life cycle was assumed to start upon harvest of the preceding crop, and
8 to stop upon harvest of the crop considered.

9 Silage maize was produced according to cropping practices from a conventional dairy farm,
10 based on data collected in an experimental farm ("La Jaillière") located within the region of
11 interest, Pays de la Loire (Gillet J.P., Pers. Comm., 2005).

12 Rapeseed was assumed to be produced in the same region, according to two cropping
13 systems. One is typical of dairy farming systems, in that fertilization is mostly organic.
14 Technical data were collected from the experimental farm "La Jaillière" (Gillet J.P., Pers.
15 Comm., 2005). The other cropping system was representative of a farm specialized in cereals,
16 which relies exclusively on mineral fertilizers. The data for those farms were based on
17 regional surveys carried out by an extension service (Charbonnaud J. and Arjauré G., Pers.
18 Comm., 2005).

19 Soybean was assumed to be produced in Brazil in the Cerrados region (Centre-West), and
20 managed under a direct drill, mulch-based cropping system. Millet or sorghum may be used
21 as cover crops. Management data were taken from a range of studies on these systems (Seguy
22 *et al.*, 2002; Scopel *et al.*, 2004; Cederberg and Flysjö, 2004a; Corbeels *et al.*, 2005). Table 3
23 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 Technical Centre for Oilseed Crops (CETIOM). The crushing technology is meant to be set up by farmers' cooperatives in order to share the installation costs. Two crushing processes were experimented: a succession of crushing-heating-pressing steps and a succession of cold pressure-extrusion-pressing steps. Only the first one was analysed in this paper, the second one having proved economically unprofitable because of its low output per annum. The selected process 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 meets the requirement that the meal should have a fat content under 10%. The heat treatment reduces the degradability of the proteins in the rumen by producing "bypass proteins". Table 5 lists the inputs and emissions to the environment per ton of grains processed. Emissions to air are related to the combustion of fossil fuels. Fifty percent of the grains are assumed to originate from dairy farms and 50% from cereal farms. This hypothesis was based on a survey of the arable land available for the growth of rapeseed in both types of farms in the region (Lehuger, 2005).

Industrial crushing of soybean was supposed to occur in Brazil, in the vicinity of the seaports from where the soybean cake 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 (Althaus *et al.*, 2003; see Table 5). 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 units (Carré P., Pers. Comm., 2005). A specificity of industrial oil extraction is the use of hexane, a toxic solvent with potential hazards for the environment. The emissions into air were set at $13 \cdot 10^{-5}$ kg hexane/kg seeds processed (Nemecek *et al.*, 2003).

2.3.3. *Transport*

Soy beans are transported from the farm to the crushing plant by truck and the soy meal is transported by cargo from Brazil to France over a distance of 9500 km. Soy meal is transported to commercial seaports by road (60%), by train (33%), and by water (7%) (Verdonk, 2003).

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 production 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 from soil of nitrates and phosphorus. The method proposed by Ecoinvent (Nemecek *et al.*, 2003) was used to assess these emissions. Briefly, the method calculates the emissions using simple models derived from the literature. The NH_3 emissions are calculated from the amount of manure and slurry spread on the field. Nitrate leaching to ground water and losses of phosphorous are estimated with simple models based on balances of soil inputs and outputs. Calculations of N_2O emissions are based on a model adapted of the IPCC method (IPCC *et al.*, 1996). NO_x emissions are simply estimated as 21% of N_2O emissions.

Two types of emissions were not included and are dealt with in the Discussion section: the fate of heavy metals associated with mineral fertilizers in soil, and the variations of soil carbon stocks.

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. These are broken down into broad categories, which have to be defined (Guinée *et al.*, 2002). In this study, 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*. Next, for each impact category, a global indicator is calculated by multiplying the results of the inventory analysis (aggregated resources and aggregated emissions) with a characterisation factor for each substance. In this LCA, we used the baseline characterisation method CML 2 baseline 2000 V2.1 developed by Guinée *et al.* (2002). This method uses an array of characterisation models specific to each impact category.

The impact category *Abiotic Depletion* is related to extraction of minerals and fossil fuels necessary to the system (Guinée *et al.*, 2002). 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)/kg extraction.

The characterisation model for *Climate Change* impact category is based on the IPCC model (Houghton *et al.*, 1996), which defines the Global Warming Potential of different greenhouse gases. Characterisation factor is global warming potential for a 100-year time horizon (GWP100) for each greenhouse gas emission (in kg CO₂ equivalent/kg emission).

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

The characterisation factors of *Human Toxicity* impact category are expressed as Human Toxicity Potentials and calculated with USES-LCA model. They are expressed for each

emission in kg 1,4-dichlorobenzene equivalent (1,4 DCB) /kg emission (Huijbregts *et al.*, 2000).

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; Huijbregts *et al.*, 2000).

The characterisation factors of the *Photo-oxidant formation* impact category are based on photochemical ozone creation potential for each emission of gases (Derwent *et al.*, 1998).

Characterisation factor is expressed in kg ethylene (C₂H₂) equivalent/kg emission.

For the *Acidification* impact, Acidification Potential is calculated with the average European Acidification factors expressed in kg SO₂ equivalent/kg emission.

For *Eutrophication*, generic eutrophication factors are used and are expressed in kg PO₄³⁻ equivalent/kg emission.

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

3. Results

3.1. Contribution of feed ingredients

For a given ration, it is possible to analyse the contribution of each ingredient to the various impacts. For the both rations, the concentrates contributes more than 50% for all the impact indicators (Figures 2a and 2b), although, the proportion of concentrates in the rations is much less than that of silage maize, on a mass basis. The amount of silage maize daily ingested for the rations "Rapeseed" and "Soybean" is similar, amounting to 16.4 kg and 16.6 kg dry matter, respectively. However, the contribution of concentrates to the impact indicators is relatively higher for the "Rapeseed" ration because the share of concentrate ingested is larger

than in the "Soybean" ration: 5 kg rapeseed meal vs. 2.8 kg soy meal day⁻¹ for one cow. The other ingredients have much less importance in the impact results.

Although the rations were calculated to be with the same energy and protein contents, the proportion of concentrate and forage was different for both rations and the cows milk productivity was higher with the rape meal (+14%) than with soybean meal. As the functional unit is one kg of milk, this gives an advantage to the "Rapeseed" scenario.

3.2. Impact results

3.2.1. Abiotic depletion

Abiotic depletion is $7.2 \cdot 10^{-4}$ kg Sb-eq per kg of energy-corrected milk (ECM) for the "Soybean" scenario and $7.6 \cdot 10^{-4}$ kg Sb-eq per kg ECM for the "Rapeseed" scenario (Table 6). Oil depletion contributes more than the other resources used to the impact indicators: 63% for the "Soybean" scenario and 55% for the "Rapeseed" scenario.

3.2.2. Climate change

Per kg of ECM, the global warming impact amounts to $3.0 \cdot 10^{-1}$ kg CO₂-eq for the "Soybean" scenario and $3.4 \cdot 10^{-1}$ kg CO₂-eq for the "Rapeseed" scenario (Table 6). Greenhouse gas emissions are mainly from direct field emissions (Figure 3a). Among the greenhouse gases, N₂O was the predominant form, contributing 69.7% and 72.4% 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. The differences between both scenarios were mainly due to fertilizer production. In fact, the rapeseed crops required more fertilizer N than the soybean crops. As a consequence, the global warming impact of the fertilizer production phase was three times higher for the "Rapeseed" scenario than for the "Soybean" one, with emissions of 0.06 vs. 0.02 kg CO₂-eq per functional unit.

3.2.3. Ozone layer depletion

According to the system definition, the production of one kg ECM induced a potential of ozone layer depletion of $1.30 \cdot 10^{-8}$ kg CFC11-eq for both scenarios (Table 6). Two halon gases: bromotrifluoromethane and bromodifluoromethane contributed 98% of the impacts during the extraction, the treatment and the transport of oil and natural gas.

3.2.4. Human toxicity

The human toxicity potential measures the impacts of toxic chemical molecules on human health. The calculated potentials were 1.39 kg 1,4-DCB-eq for the "Soybean" scenario and 1.52 kg 1,4-DCB-eq for the "Rapeseed" scenario (Table 6). Emissions occurred mainly during the production of metals (iron, copper). The emissions of dioxins contributed approximately 40% of the impacts for both scenarios.

3.2.5. Ecotoxicity

The CML2 method calculates 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 (Guinée *et al.*, 2002). The ecosystems include: freshwater, marine aquatic, and terrestrial ecosystems.

Per kg ECM, fresh water aquatic ecotoxicity was $1.29 \cdot 10^{-1}$ kg 1,4-DCB-eq for the "Soybean" scenario, and $5.54 \cdot 10^{-1}$ kg 1,4-DCB-eq for the "Rapeseed" scenario (Table 6). 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 chains (nickel, steel, copper...).

The marine aquatic ecotoxicity impact was very high for both scenarios: 32.9 kg 1,4-DCB-eq for the "Soybean" scenario, and 38.6 kg 1,4-DCB-eq for the "Rapeseed" scenario (Table 6).

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 19% and 24% for the "Soybean" and "Rapeseed" scenarios, respectively. The use of P-Fertilizers was $1.42 \cdot 10^{-3}$ kg/FU for the "Soybean" scenario and $2.12 \cdot 10^{-3}$ kg/FU for the "Rapeseed" scenario.

Per kg ECM, terrestrial toxicity was $1.23 \cdot 10^{-2}$ kg 1,4-DCB-eq for the "Soybean" scenario and $4.26 \cdot 10^{-2}$ kg 1,4-DCB-eq for the "Rapeseed" scenario (Table 6). Terrestrial ecotoxicity was largely explained by pesticides emissions. This impact is likely under-estimated because the emissions of heavy metals were not included in these results. Their potential contribution is analysed in the Discussion section.

3.2.6. Photochemical oxidation

Photochemical oxidation potential was higher for the "Soybean" scenario than for the "Rapeseed" scenario, amounting to $3.48 \cdot 10^{-5}$ vs. $2.51 \cdot 10^{-5}$ kg C₂H₂-eq/FU, respectively (Table 6). The "Soybean" scenario has a higher impact indicator because of the sea transport and industrial crushing phases. The emissions of SO₂ explained approximately 40% of the impacts in both scenarios. Hexane losses from industrial crushing contribute to 21% of the "Soybean" scenario impact, and only 6% for the "Rapeseed" scenario (due to the use of heat-treated soy meal).

3.2.7. Acidification

Per kg ECM, the acidification potential were 1.47 kg SO₂-eq and 2.29 kg SO₂-eq for the "Soybean" and "Rapeseed" scenarios, respectively (Table 6). The highest impact for the "Rapeseed" scenario may be explained by the emissions of ammonia upon the spreading of slurry as a fertilizer for rapeseed on the dairy farms. Moreover, the production of synthetic fertilizers on the cereal farms causes further emissions of acidifying gases. As a result, the

acidification potential from crop fertilization was much higher for the "Rapeseed" scenario (Figure 3b).

3.2.8. *Eutrophication*

The "Rapeseed" scenario has a higher eutrophication potential than the "Soybean" one: $2.9 \cdot 10^{-3}$ kg PO₄-eq vs. $3.5 \cdot 10^{-3}$ kg PO₄-eq, respectively (Table 6). Direct field emissions explained most of this difference. In fact, N₂O, NH₃ and NO₃⁻ emissions per FU were higher for rapeseed production than for soybean production (Figure 3c).

3.2.9. *Land use*

Land use for the production of 1 kg ECM was 0.46 m² year with the "Soybean" scenario, and 0.63 m² year with the "Rapeseed" scenario (Table 6). As could be expected, land use from agricultural activities was markedly higher compared to industrial activities. Two facts may explain the differences in land use between the two feeding scenarios. On the one hand, the intake of concentrated feed differs between the two rations: 2.8 kg day⁻¹ soy meal vs. 5.0 kg day⁻¹ rapeseed meal. 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, and lower quantities are required in the feeding ration.

4. Discussion

4.1. *Comparison of the soybean and rapeseed scenarios*

Based on the assumptions we made and the system limits we used, the "Soybean" scenario appears more environmental-friendly than the "Rapeseed" scenario, with the exception of the photochemical oxidation impact (Figure 4). Even though the productivity of cows increased 14% with the rape meal ration compared to the soy meal ration, the higher milk output could

not compensate for the higher environmental impacts associated with the agricultural production of rapeseed grains. The environmental performance of the rapeseed crop was lower than that of the soybean crops, due to reliance on synthetic fertilizer N and lower yields per ha. Also, the increase in milk productivity came at the cost of higher ingestion of concentrates in the "Rapeseed" scenario. Lastly, transatlantic transport of soy meal from Brazil to France only caused marginally higher impacts compared to the rest of the chains, even for categories usually affected by transport processes like climate change. This result was also mentioned in other studies (Eide, 2002; Van der Werf *et al.*, 2005).

On the other hand, the "Soybean" scenario seemed 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 these traits are related to soybean being a legume crop. The use of leguminous protein crops grown in France may arguably appear as the optimal strategy to reduce the environmental load of milk production, compared to using meal from an oil crop like 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. In fact, the extrusion phase and the heat treatment of the lupine grains were not satisfying in terms of protein insolubilisation in the rumen. The heat treatment did not produce sufficient protein protection against rumen degradation (Lehuger, 2005).

The fact that the milk productivity of cows increased by 14% (from 29.6 kg to 33.7 kg ECM day⁻¹) with the rapeseed scenario compared to the soybean one may also be expected to produce beneficial effects on the overall performance of the farm. The latter could indeed meet the same production target with a reduced number of cattle, which would reduce some

1 maintenance and operating costs on the farm (cattle housing manure and feed handling, etc...).
 2 Since these operations were outside our system, they were not accounted for. It might also be
 3 argued that the higher productivity of the "Rapeseed" scenario resulted in lower rates of
 4 excretion per kg ECM. This is actually contradicted by the fact that the daily intake of the
 5 "Rapeseed" rations (in kg DM.cow⁻¹ day⁻¹) was also 11% higher than that of the "Soybean"
 6 ration (being 20.1 kg and 22.3 kg DM cow⁻¹ day⁻¹ for the soybean and rapeseed ration, resp.)
 7 Consequently, the N intakes for one kg ECM were very similar: 14.6 gN kg⁻¹ ECM with the
 8 "Rapeseed" scenario and 14.8 gN kg⁻¹ ECM with the "Soybean" scenario. The resulting
 9 emissions from animal subsystem for one kg ECM may thus still expected to be the same for
 10 both scenarios, in accordance with our prior hypothesis that the animal production subsystem
 11 could be excluded from the scope of our study. To further investigate this hypothesis, the
 12 analysis of the urea concentrations in cow urine or in the milk could provide good indicators
 13 to predict N emissions into the environment, and to differentiate the environmental
 14 performances of both rations after ingestion (Vérité and Delaby, 2000).

15 4.2. Heavy metal accumulation in agricultural soils

16 The results presented here did not take into account the presence of heavy metals (HM) in
 17 agricultural inputs, although some fertilizers are known to contain significant amounts of HM
 18 (Nemecek *et al.*, 2003). This paragraph explores the sensitivity of our results to this
 19 hypothesis. The Ecoinvent methodology for calculating emissions of (HM) to agricultural soil
 20 (Nemecek *et al.*, 2003) is a simple input-output balance sheet. Soil inputs of HM from
 21 agricultural inputs are inventoried, and outputs are calculated based on expected
 22 concentrations in plant tissues and plant biomass at harvest. The use of the default
 23 concentrations in plants produced erroneous results, with plant uptake exceeding the amounts
 24 of HM present in the soil. Consequently, the impact indicators of human toxicity or
 25 ecotoxicity are 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.* (2003), and include mineral fertilisers, farmyard manure and seeds. The impacts results under both hypotheses are compared in Table 7. The hypothesis of 100% HM accumulation in soils results in significantly greater impacts than without the consideration of HM. Under the latter assumption, the indicators of human toxicity, fresh water aquatic ecotoxicity, marine aquatic ecotoxicity and terrestrial ecotoxicity were lower with the "Soybean" scenario. Taking into account HM accumulation in soil shifts the comparison in favour of the "Rapeseed" scenario for two impact categories: human toxicity ($3.80 \cdot 10^{-1}$ vs. $3.96 \cdot 10^{-1}$ kg 1,4-DCB-eq) and marine aquatic ecotoxicity (129 vs. 122 kg 1,4 DCB-eq), and narrowed the gap for the fresh water and terrestrial ecotoxicity impacts (Table 7).

4.3. Carbon sequestration in soil

Crop management (in particular residue management and tillage practises) may influence the organic matter content of arable soils, and result in either net release or sequestration of CO₂. The variations in soil C stocks therefore affects the climate change indicator, depending on whether the soil is a net sink or source of atmospheric CO₂. In the above results, soil C variations were not taking 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 practise from conventional tillage to no-till, mulch-based cropping system 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 CO₂. Bayer *et al.* (2006) estimate that no-till practises resulted in an additional sequestration rate of $0.35 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ compared to conventional tillage practises or native Cerrado soil C content. Metay *et al.* (2006) reported the same rate ($0.35 \text{ Mg C ha}^{-1}$

year⁻¹) when comparing no-till under cover crops and disc tillage for Cerrados soils. We accounted for such a rate of C sequestration in our system study by translating it as an equivalent decrease of atmospheric CO₂ for the soybean production chain. As a result, the global warming indicator for the "Soybean" scenario decreased from 3.01 10⁻¹ to 2.43 10⁻¹ kg CO₂-eq per kg ECM.

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.*, 2002). The latter authors reviewed the existing literature on the effect of a range of management practises 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 amendment generated by the cattle-shed, as hypothesized in the dairy farms' cropping systems, on the rapeseed crops involves a potential C sink that also seemed negligible. The literature reports that slurry application has no effect on total soil organic carbon (Arrouays *et al.*, 2002; Bol *et al.*, 2003). Even when repeated over successive years, slurry application only results in rapid and transient increases of the soil microbial biomass (Rochette *et al.*, 2000). Unlike soybean crops grown in Brazil, the introduction of rapeseed in the French cropping systems is expected to have no significant impact on the climate change indicator of the "Rapeseed" scenario.

4.4. Sensitivity to cropping techniques for rapeseed

In the baseline scenarios, rapeseed was supposed to be produced with two sets of management rules, whether it was grown in dairy farms or cereal farms, with an equal share between the two types of farms. In order to test the sensitivity of the LCA to crop management, the impact indicators were calculated assuming the origin of the rapeseed: 100% from dairy farm vs. 100% from cereal farm (Figure 5). Under the first hypothesis, most impacts were reduced compared to the baseline situation. The use of organic fertilizers produced on the farm

1 decreases the reliance on mineral fertilizers, whose production is costly in energy and
2 environmental emissions. However, acidification increased because of the volatilization of
3 ammonia due to slurry spreading. Conversely, assuming rapeseed to be grown at 100% in
4 cereal farms worsened the results of many indicators. Several points are unfavourable to these
5 production systems: heavy reliance on mineral fertilisers and an important number of
6 cropping operations. This sensitivity analysis points at potential improvements to improve the
7 environmental performance of rapeseed crops: the use of farmyard manure vs. mineral
8 fertilisers and the reduction of cropping operations. However, with organic fertilizers, the
9 emissions of compounds involved in acidification and eutrophication impacts must be
10 controlled. A range of mitigation options are available to the farmer, including application
11 techniques or timing with respect to meteorological conditions (Theobald *et al.*, 2005).

12 4.5. *Improvement of the life cycle inventory*

13 As shown in the previous paragraphs, the quality of the inventory for the agricultural
14 production phase should be improved by taking into account heavy metal accumulation and C
15 stock variations in soil.

16 Direct field emissions associated with crop production appeared as decisive for the climate
17 change, acidification, and eutrophication impacts. Therefore, a better estimation of field
18 emissions of NO_3^- , NH_3 , N_2O , and NO_x has to be sought. This calls for the use of process-
19 based models simulating the fate of agricultural inputs as a function of local soil and climate
20 characteristics, which are paramount to determining direct field emissions with improved
21 accuracy. There lies a perspective for this study, consisting of using the CERES model for
22 estimating direct field pollutions associated with rapeseed in France and soybean in Brazil.
23 CERES is a mechanistic model simulating the dynamics of water, carbon, and nitrogen in the
24 soil-crop systems (Jones and Kiniry, 1986) and for which different modules were added to

1 predict N₂O emissions (Gabrielle *et al.*, 2006), NH₃ volatilization (Genermont and Cellier,
2 1997) and NO_x emissions (Laville *et al.*, 2005).

3 **5. Conclusion**

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

14 However, the environmental performance of rapeseed crops may be improved by reducing
15 inputs, and particularly synthetic fertiliser N, as was shown in a sensitivity analysis.

16 From a methodological standpoint, the robustness of our conclusions should be further tested
17 by improving the estimation of direct field emissions using soil-crop models. Another
18 perspective would be to incorporate the oil co-produced back to the scope of our system, by
19 investigating its possible use on the farm as a biofuel. Lastly, the fact that either production
20 system may induce land use changes has not been dealt with in this study, although it may
21 cause major environmental impacts, essentially in Brazil due to the expansion of soy area and
22 forest clearing (Bickel and Dros, 2003). Fully tackling the issue of land use change therefore
23 appears as a major challenge in future application of LCA methodology to agricultural chains.

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8 **References**

9 ADEME, DIREM, Ecobilan, 2002. Bilan énergétiques et gaz à effet de serre des filières de
10 production de biocarburants. Technical report, ADEME, Paris.

11 Althaus, H.J., Chudacoff, M., Hellweg, S., Hirsch, R., Jungbluth, N., Osses, M., Primas, A.,
12 2003. Life cycle inventories of chemicals. Swiss Centre for Life Cycle Inventories,
13 Dübendorf, Switzerland.

14 Arrouays, D., Balesdent, J., Germon, J.C., Jayet, P.A., Soussana, J.F., Stengel, P., 2002.
15 Stocker du carbone dans les sols agricoles de France?. Expertise Scientifique
16 Collective. INRA, Paris, p. 330.

17 Bayer, C., Martin-Neto, L., Mielniczuk, J., Pavinato, A., Dieckow, J., 2006. Carbon
18 sequestration in two Brazilian Cerrado soils under no-till. Soil Till. Res. 86, 237-245.

19 Berlin, J., 2002. Environmental life cycle assessment (LCA) of Swedish semi-hard cheese.
20 Int. Dairy J. 12, 939-953.

21 Bernesson, S., Nilsson, D., Hansson, P.A., 2004. A limited LCA comparing large- and small-
22 scale production of rape methyl ester (RME) under Swedish conditions. Biomass
23 Bioenerg. 26, 545-559.

- 1 Bickel, U., Dros, J.M., 2003. The Impacts of Soybean Cultivation on Brazilian Ecosystems.
2 Three case studies. WWF Forest Conversion Initiative, Amsterdam, p. 33.
3 <http://www.panda.org/downloads/forests/impactsofsoybean.pdf>.
- 4 Bol, R., Kandeler, E., Amelung, W., Glaser, B., Marx, M.C., Preedy, N., Lorenz, K., 2003.
5 Short-term effects of dairy slurry amendment on carbon sequestration and enzyme
6 activities in a temperate grassland. *Soil Biol. Biochem.* 35, 1411-1421.
- 7 Brunschwig, P., Cadot, M., Lemarié, J., 1996. Le point sur le tourteau de colza pour les
8 bovins. CETIOM, Institut de l'Elevage. Paris, France, p. 56.
- 9 Casey, J.W., Holden, N.M., 2005. Analysis of greenhouse gas emissions from the average
10 Irish milk production system. *Agr. Syst.* 86, 97-114.
- 11 Cederberg, C., 1998. Life Cycle Assessment of milk production, a comparison of
12 conventional and organic milk. SIK report Nr 643, Gothenburg, Sweden
- 13 Cederberg, C., Flysjö, A., 2004a. Life Cycle Inventory of 23 Dairy Farms in South-Western
14 Sweden. SIK report Nr 728, Gothenburg, Sweden.
- 15 Cederberg, C., Flysjö, A., 2004b. Environmental assessment of future pig farming systems.
16 SIK report Nr 723-2004, Gothenburg, Sweden
- 17 Cederberg, C., Mattsson, B., 2000. Life cycle assessment of milk production - a comparison
18 of conventional and organic farming. *J. Clean. Prod.* 8, 49-60.
- 19 Corbeels, M., Scopel, E., Cardoso, A., Bernoux, M., Douzet, J.M., Siqueira Neto M., 2006.
20 Soil carbon storage potential of direct seeding mulch-based cropping systems in the
21 Cerrados of Brazil. *Glob. Change Biol.* 2006 (In press).
- 22 Derwent, R.G., Jenkin, M.E., Saunders, J.M., Pilling, M.J., 1998. Photochemical ozone
23 creation potentials for organic compounds in Northwest Europe calculated with a
24 master chemical mechanism. *Atmos. Environ.* 32, 2429-2441.

- 1 Gabrielle, B., Laville, P., Hénault, C., Nicoullaud, B., Germon, J.C., 2006. Simulation of
2 nitrous oxide emissions from wheat-cropped soils using CERES, *Nutr. Cycl.*
3 *Agroecos.* 74, 133-146.
- 4 Genermont, S., Cellier, P., 1997. A mechanistic model for estimating ammonia volatilization
5 from slurry applied to bare soil. *Agr. and Forest Meteorol.* 88, 145-167.
- 6 Guinée, J.B., Gorée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L.,
7 Wegener Sleeswijk, A., Suh, S., Udo de Haes, *et al.*, 2002. Handbook on life cycle
8 assessment. Operational Guide to the ISO standards. Centre of environmental Science,
9 Leiden University, Leiden, The Netherlands.
- 10 Eide, MH., 2002. Life cycle assessment (LCA) of industrial milk production. *Int. J. LCA* 7,
11 115-126.
- 12 Hospido, H., Moreira, M.T., Feijoo, G., 2003. Simplified LCA of galician milk production.
13 *Int. Dairy J.* 13, 783-796.
- 14 Houghton, J.T., Meira Filho, L.G., Callander, B.A., Harris, N., Kattenberg, A., Maskell, K.,
15 1996. *Climate change 1995: the science of climate change.* Cambridge University
16 Press, Cambridge.
- 17 Huijbregts, M.A., Thissen, U., Guinée, J.B., Jager, T., Van de Meent, D., Ragas, A.M.J.,
18 Wegener, S.A., Reijnders, L., 2000. Priority assessment of toxic substances in the life
19 cycle assessment, I: Calculation of toxicity potentials for 181 substances with the
20 nested multi-media fate exposure and effects model USES-LCA. *Chemosphere* 41,
21 575-588.
- 22 IPCC, OECD, IEA, 1996. Reference Manual. Revised 1996. IPCC Guidelines for national
23 greenhouse gas inventories, Vol.3. [http://www.ipcc-](http://www.ipcc-nggip.iges.or.jp/public/gl/invs1.htm)
24 [nggip.iges.or.jp/public/gl/invs1.htm](http://www.ipcc-nggip.iges.or.jp/public/gl/invs1.htm).

- 1 ISO International Standard 14 040, 1997. Environmental management – Life cycle
2 assessment – principle and framework. International Organisation for standardisation
3 (ISO), Geneva, Switzerland, p. 12.
- 4 ISTA, 2004. Oil World Annual 2004. Mielke GmbH, Hamburg, Germany.
- 5 Jones, C.A., Kiniry, J.R., 1986. CERES-N Maize: a simulation model of maize growth and
6 development. Texas A&M University Press, College Station, Temple, TX.
- 7 Laville, P., Henault, C., Gabrielle, B., Serca, D., 2005. Measurement and modelling of NO
8 fluxes on maize and wheat crops during their growing seasons: effect of crop
9 management. *Nutr. Cycl. Agroecosys.* 72, 159-171.
- 10 Lehuger, S., 2005. Evaluation environnementale de la substitution, en élevage bovin laitier,
11 du tourteau de soja importé par du tourteau de colza produit localement – Application
12 de l'Analyse de Cycle de Vie. Mémoire de fin d'études d'ingénieur. Ecole Supérieure
13 d'Agriculture d'Angers, INRA, Angers, France.
- 14 Metay, A., 2006. Storage and forms of organic carbon in a no-tillage under cover crops
15 system on clayey Oxisol in dryland rice production (Cerrados, Brazil). *Soil Till. Res.*
16 2006 (In press).
- 17 Nemecek, T., Heil, A., Huguenin, O., Erzinger, S., Blaser, S., Dux, D., Zimmerman, A., 2003.
18 Life Cycle Inventories of Production systems. Final report Ecoinvent 2000 No 15.
19 FAL Reckenholz, FAT Tänikon, Swiss Centre For Life Cycle Inventories, Dübendorf,
20 Switzerland, p. 284.
- 21 Reinhardt, G.A., 2000. Bioenergy for Europe: Which ones fit best? – A Comparative Analysis
22 for the Community. Final project report. IFEU, Heildeberg, Germany.
- 23 Rochette, P., Angers, D.A., Cote, D., 2000. Soil carbon and nitrogen dynamics following
24 application of pig slurry for the 19th consecutive year: I. Carbon dioxide fluxes and
25 microbial biomass carbon. *Soil Sci. Soc. Am. J.* 64, 1389-1395.

- 1 Scopel, E., Fernando, A.M.D.S., Corbeels, M., Affholder, F., Maraun, F., 2004. Modelling
2 crop residue mulching effects on water use and production of maize under semi-arid
3 and humid tropical conditions. *Agronomie* 24, 384-395.
- 4 Seguy, L., Bouzinac, S., Maronezzi, A.C., 2002. Un dossier du semis direct, Systèmes de
5 culture et dynamique de la matière organique. CIRAD-CA/SCV, Montpellier, p.
6 <http://agroecologie.cirad.fr>.
- 7 Sjaunja, L.O., Baevre, L., Junkkarinen, L., Pedersen, J., Setälä, J., 1990. A Nordic proposal
8 for an energy corrected milk (ECM) formula. 27th session of the International
9 Commission for Breeding and Productivity of Milk Animals, Paris.
- 10 Smits, M.C.J., Valk, H., Elzing, A., Keen, A., 1995. Effect of Protein Nutrition on Ammonia
11 Emission from a Cubicle House for Dairy-Cattle. *Livest. Prod. Sci.* 44, 147-156.
- 12 Theobald, M.R., Williams, A. G., Rosnoblet, J., Campbell, C., Cumby, T. R., Demmers, T. G.
13 M., Loubet, B., Parsons, D. J., Gabrielle, B., Nemitz, E., *et al.*, 2004. An assessment of
14 how process modelling can be used to estimate agricultural ammonia emissions and
15 the efficacy of abatement techniques, Final report, DEFRA project AM0130, Centre
16 for Ecology and Hydrology (CEH), Penicuik, UK.
- 17 Van der Werf, H M G, 2006. The evaluation of the environmental impacts of protein sources
18 for concentrated feed. In: Proceedings workshop “Grain legumes and the environment:
19 how to assess benefits and impacts”. November 18-19, 2004, Zürich, Switzerland.
20 AEP, European Association for grain legume research, Paris, France. p 27-33.
- 21 Van der Werf, H.M.G., Petit, J., Sanders, J., 2005. The environmental impacts of the
22 production of concentrated feed: the case of pig feed in Bretagne. *Agr. Syst.* 83, 53-
23 177.
- 24 Verdonk, R., 2003. Brazil oilseeds and products annual 2003. Gain report BR3003. USDA
25 Foreign Agriculture Service, Sao Paulo, Brazil, p. 45.

- 1 Vérité, R., Delaby, L., 2000. Relation between nutrition, performances and nitrogen excretion
- 2 in dairy cows. *Ann. Zootech.* 49, 217-230.
- 3

1 Table 1

2 Daily ingestion of the ration supplemented with soybean meal (Soybean ration) and the ration
3 supplemented with rapeseed meal (Rapeseed ration).

	Soybean ration	Rapeseed ration
	(kg DM)	(kg DM)
Silage maize	16.58	16.44
Soybean meal	2.77	-
Rapeseed meal	-	4.97
Heat-treated	0.40	0.67
Soybean meal		
Urea	0.02	-
Mineral additives	0.20	-
7/21/5*		
Mineral additives	-	0.18
0/28/3*		
CaCO ₃	0.12	0.04
Total	20.10	22.30

4 *: %P / %Ca / %Mg.

5

1 Table 2

2 Daily milk performances of the rations and calculation of the energy corrected milk (ECM).

	Raw milk	Fat content	Protein content	ECM
	(kg.day ⁻¹)	(g.kg ⁻¹)	(g.kg ⁻¹)	(kg.day ⁻¹)
Soybean ration	30.9	39.9	28.6	29.6
Rapeseed ration	34.6	40.8	29.3	33.7

3

1 Table 3

2 Yield, inputs and number of cropping operations for the major crops used in the rations. N_{\min} 3 and N_{org} denote mineral and organic (slurry and manure) N.

	Maize from dairy farm	Rapeseed from dairy farm	Rapeseed from cereal farm	Soybean from Brazil
Yield (kg/ha)	11 000	3000	3000	2500
Fertilisation	150 kg N_{org}	120 kg N_{org}	150 kg N_{\min}	8 kg N_{\min}
	30 kg N_{\min}	70 kg N_{\min}	70 kg P	31 kg P
			100 kg K	57 kg K
				50 kg lime
Cropping operations	9	11	13	7
Previous crop	Wheat	Wheat	Wheat	Soybean

1 Table 4

2 Field emissions estimated with the Ecoinvent method (Nemecek *et al.*, 2003), in kg N ha⁻¹ or
3 kg P ha⁻¹.

	Rapeseed from dairy farm	Rapeseed from cereal farm	Soybean	Silage maize
N ₂ O	1.6	1.8	1.8	2.7
NO _x	0.5	0.6	0.5	0.8
NH ₃	16.3	3.0	0.2	6.3
NO ₃ ⁻	62.7	60.3	36*	57.0
PO ₄ ³⁻	0.04	0.05	1.2*	0.2

4 *: NO₃⁻ and PO₄³⁻ losses for soybean are from Cederberg and Flysjö (2004a) because
5 Ecoinvent is not adapted for Brazilian cropping conditions.

6

- 1 Table 5
- 2 Energy and input consumption of the small-scale and industrial crushing processes, per 1000
- 3 kg 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 (m²)	0.00034	0.034

- 1 Table 6
- 2 The environmental impacts of milk production expressed per kg ECM for the "Soybean"
- 3 "Rapeseed" scenarios.

Impact category	Unit	SOYBEAN	RAPESEED
		scenario	scenario
Abiotic depletion	kg Sb-eq	$7.22 \cdot 10^{-4}$	$7.63 \cdot 10^{-4}$
Climate change	kg CO ₂ -eq	$3.01 \cdot 10^{-1}$	$3.44 \cdot 10^{-1}$
Ozone layer depletion	kg CFC-11-eq	$1.30 \cdot 10^{-8}$	$1.35 \cdot 10^{-8}$
Human toxicity	kg 1,4-DCB-eq	$1.39 \cdot 10^{-1}$	$1.52 \cdot 10^{-1}$
Fresh water aquatic	kg 1,4-DCB-eq	$1.29 \cdot 10^{-2}$	$5.54 \cdot 10^{-2}$
Ecotoxicity			
Marine aquatic	kg 1,4-DCB-eq	32.9	38.6
Ecotoxicity			
Terrestrial Ecotoxicity	kg 1,4-DCB-eq	$1.23 \cdot 10^{-3}$	$4.26 \cdot 10^{-3}$
Photochemical	kg C ₂ H ₂ -eq	$3.48 \cdot 10^{-5}$	$2.51 \cdot 10^{-5}$
oxidation			
Acidification	kg SO ₂ -eq	$1.47 \cdot 10^{-3}$	$2.29 \cdot 10^{-3}$
Eutrophication	kg PO ₄ -eq	$2.89 \cdot 10^{-3}$	$3.48 \cdot 10^{-3}$
Land use	m ² .year	0.46	0.63

- 1 Table 7
- 2 Human toxicity and ecotoxicity indicators (kg 1.4-DCB-eq) according to two hypotheses:
- 3 accumulation of 100% of the heavy metal (HM) inputs in soils (a) vs. no consideration of
- 4 heavy metals (b).

	a) 100% of HM inputs		b) 0% of HM inputs	
	SOYBEAN	RAPeseed	SOYBEAN	RAPeseed
Human toxicity	$3.96 \cdot 10^{-1}$	$3.80 \cdot 10^{-1}$	$1.39 \cdot 10^{-1}$	$1.52 \cdot 10^{-1}$
Fresh water aquatic	$1.70 \cdot 10^{-1}$	$1.93 \cdot 10^{-1}$	$1.29 \cdot 10^{-2}$	$5.54 \cdot 10^{-2}$
ecotoxicity				
Marine aquatic	$1.29 \cdot 10^{+2}$	$1.22 \cdot 10^{+2}$	$3.29 \cdot 10^{+1}$	$3.86 \cdot 10^{+1}$
ecotoxicity				
Terrestrial ecotoxicity	$2.69 \cdot 10^{-2}$	$2.68 \cdot 10^{-2}$	$1.23 \cdot 10^{-3}$	$4.26 \cdot 10^{-3}$

5

1 Figure captions

2 **Figure 1.** Flow diagram of the system life cycle.

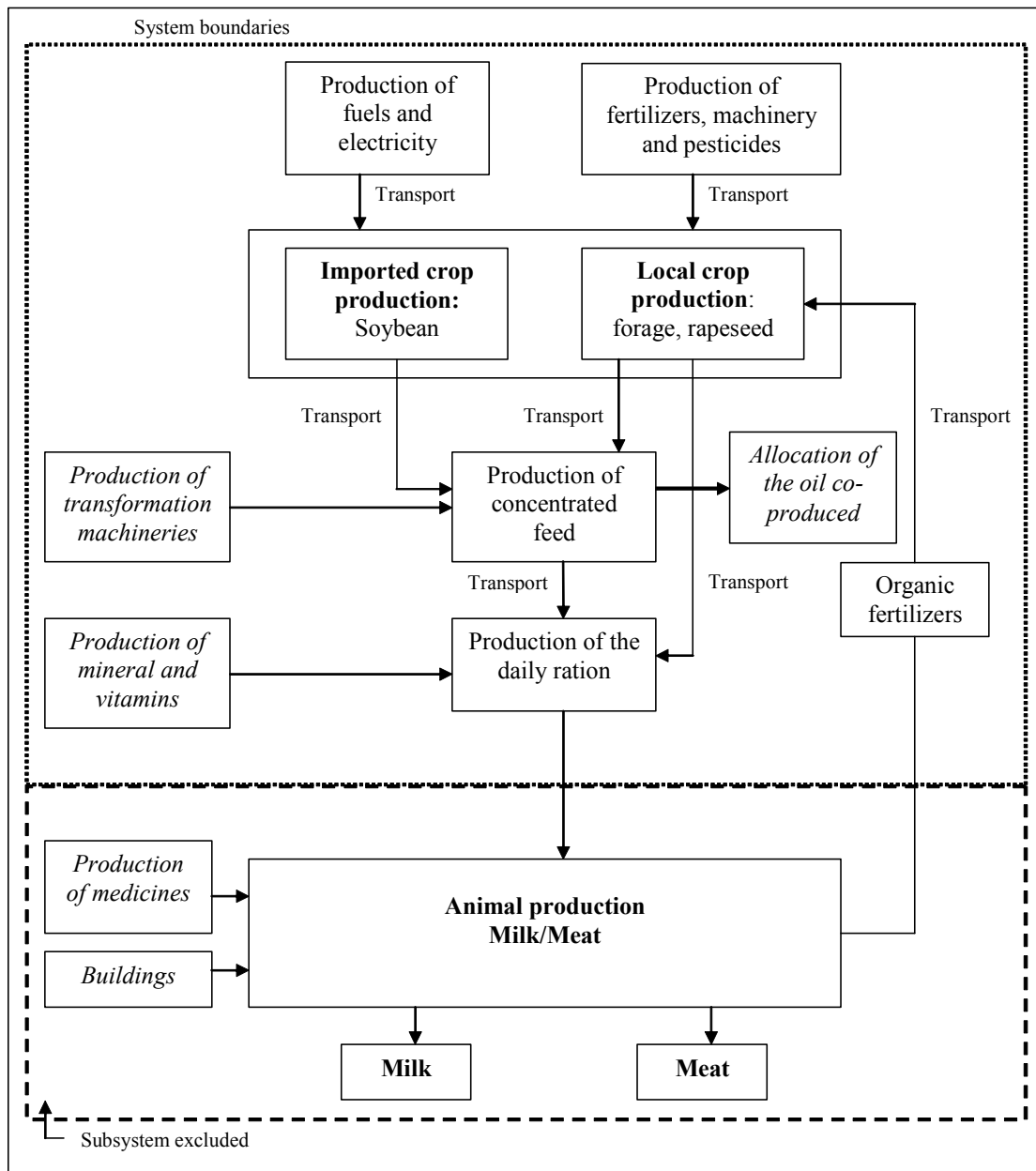
3 **Figure 2.** Relative contributions of the different ingredients to the impact indicators for the
4 "Rapeseed" (a) and "Soybean" (b) rations.

5 **Figure 3.** Calculated environmental impacts for 1 kg of energy corrected milk: a) Climate
6 change (kg CO₂-eq), b) Acidification (kg SO₂-eq), c) Eutrophication (kg PO₄-eq), and
7 contributions of crop production, fertiliser production, agricultural machinery operation and
8 sea and road transports to the indicators.

9 **Figure 4.** Comparison of environmental impact indicators for the "Rapeseed" and "Soybean"
10 scenarios. The indicators are expressed as a percentage of the highest value.

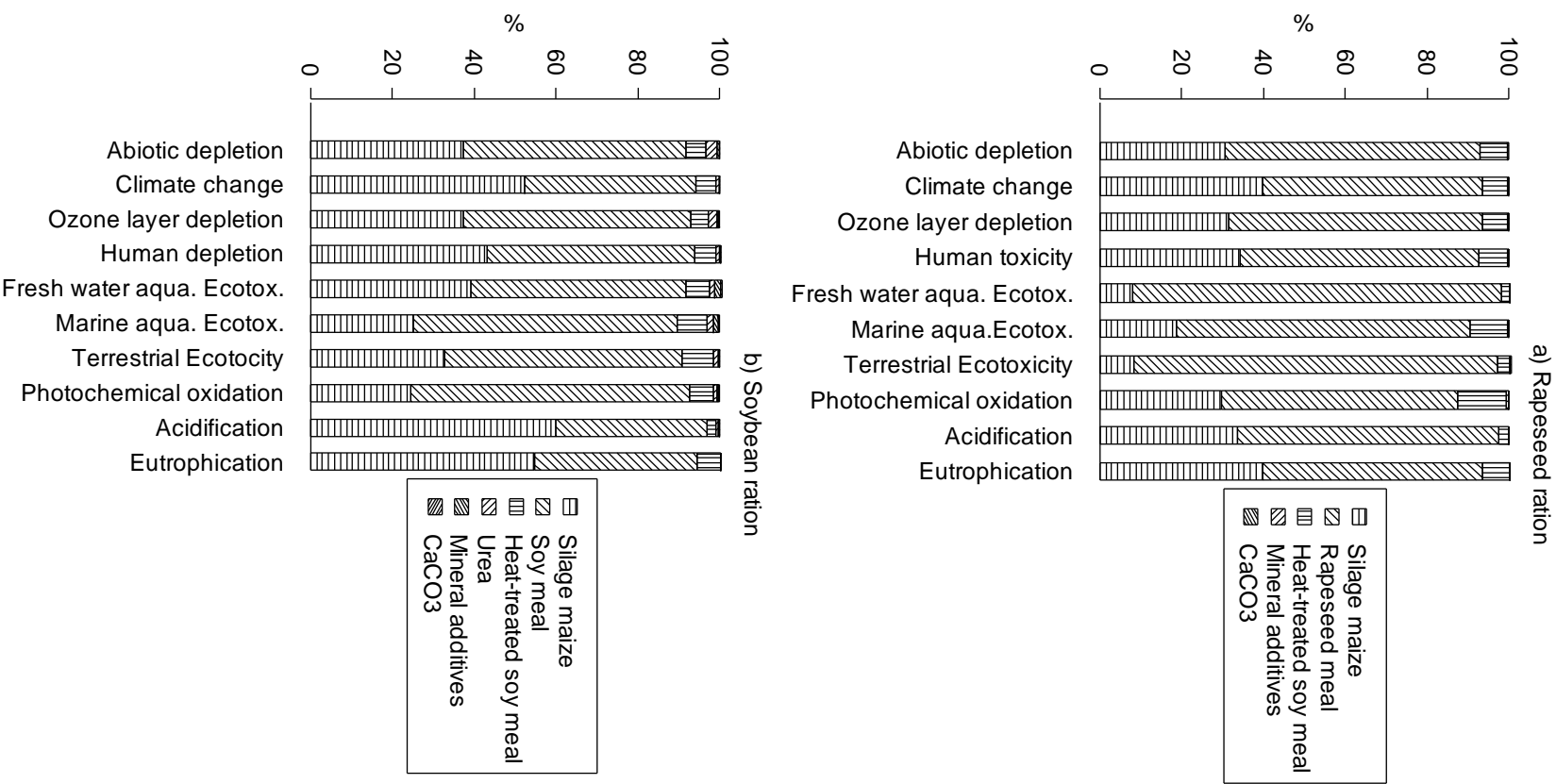
11 **Figure 5.** Effect of crop management on the performance of the "Rapeseed" ration. The
12 indicators are expressed as a percentage of the highest value (CF: Cereal Farm; DF: Dairy
13 Farm).

1



2

3 **Figure 1.** Flow diagram of the system life cycle.



1

2 **Figure 2.** Relative contributions of the different ingredients to the impact indicators for the

3 "Rapeseed" (a) and "Soybean" (b) rations.

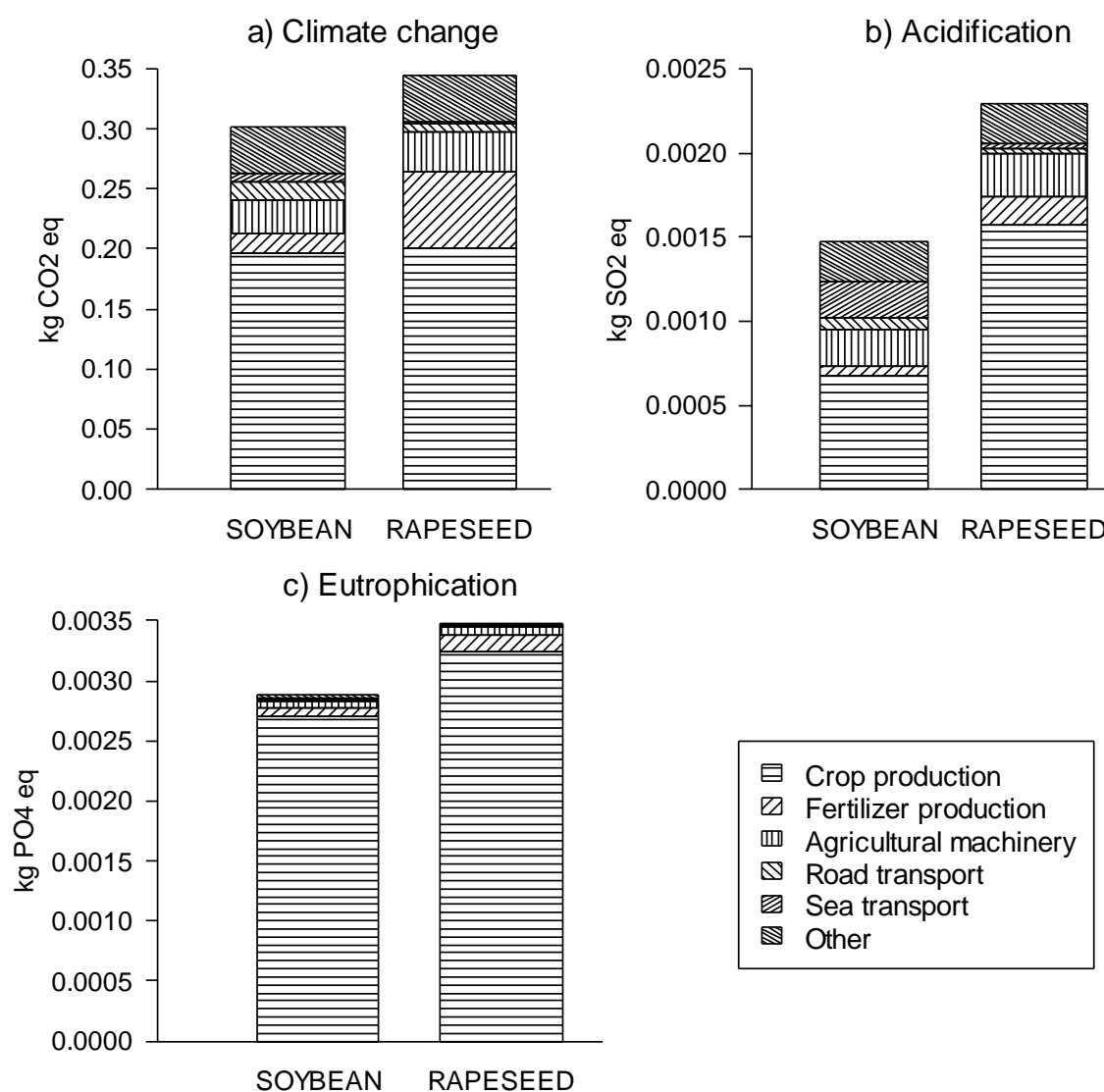


Figure 3. Calculated environmental impacts for 1 kg of energy corrected milk: a) Climate change (kg CO₂-eq), b) Acidification (kg SO₂-eq), c) Eutrophication (kg PO₄-eq), and contributions of crop production, fertiliser production, agricultural machinery operation and sea and road transports to the indicators.

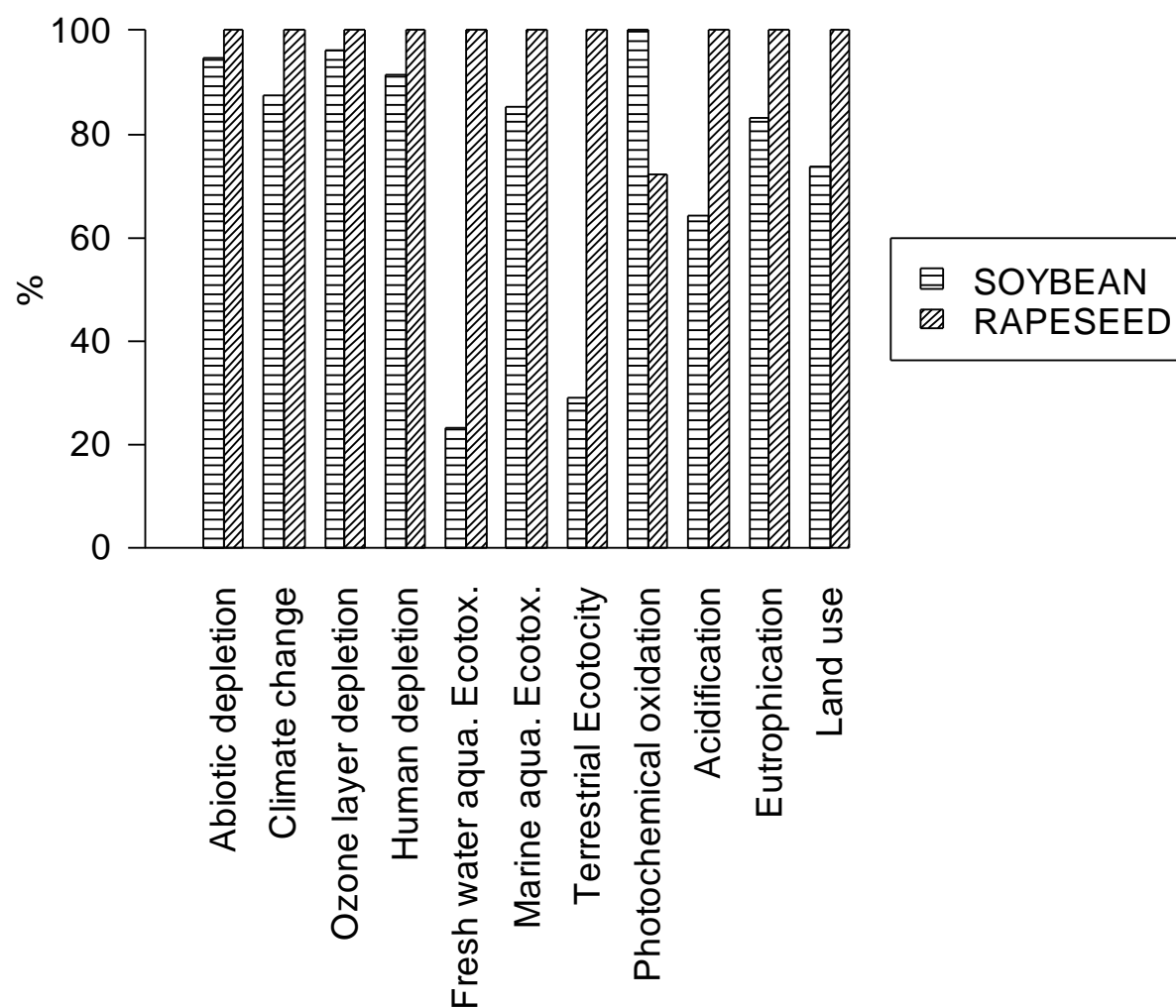


Figure 4. Comparison of environmental impact indicators for the "Rapeseed" and "Soybean" scenarios. The indicators are expressed as a percentage of the highest value.

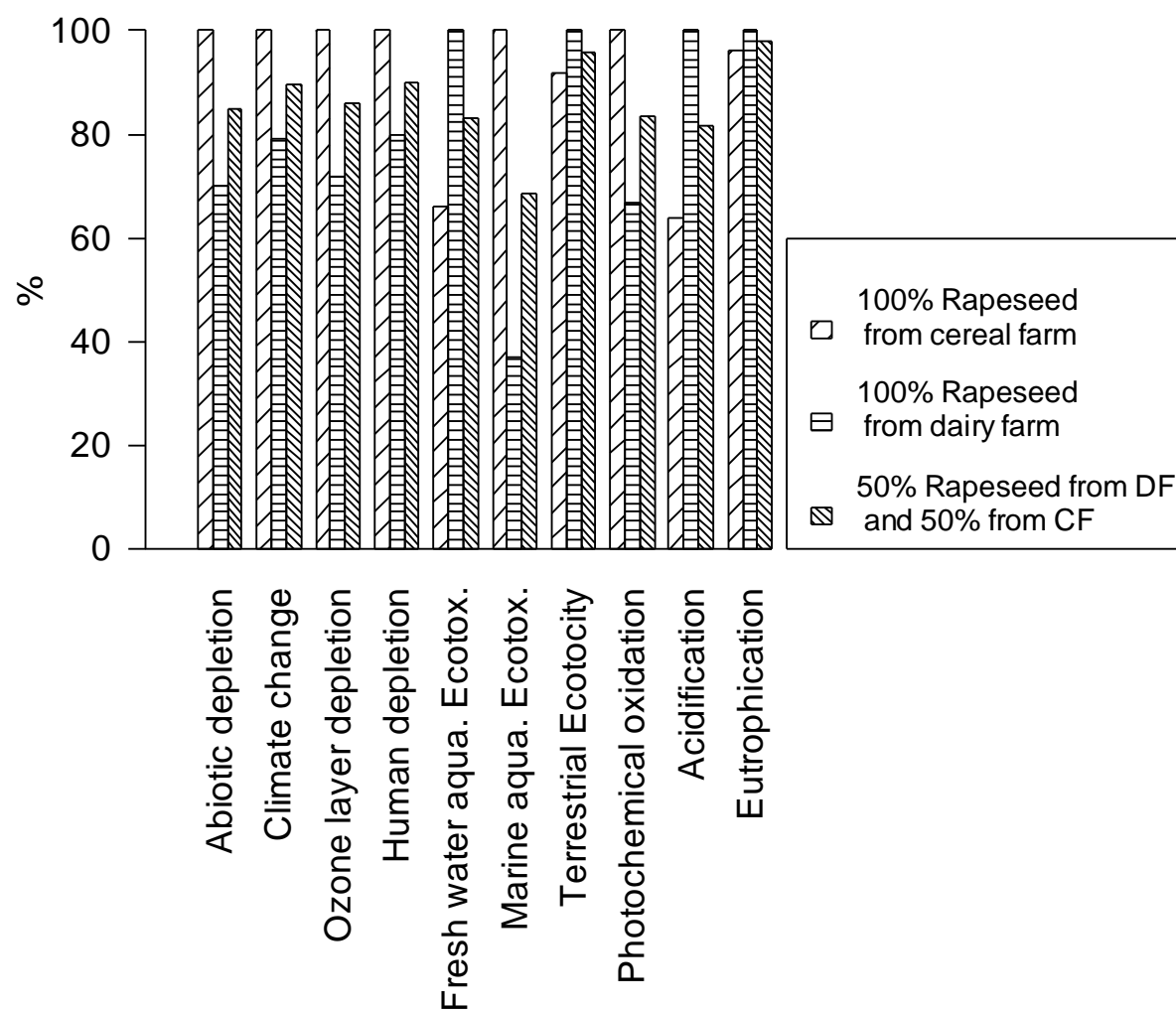


Figure 5. Effect of crop management on the performance of the "Rapeseed" ration. The indicators are expressed as a percentage of the highest value (CF: Cereal Farm; DF: Dairy Farm).