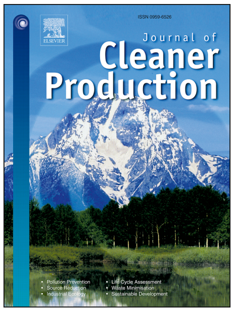

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Would transitioning from conventional to organic oat grains production reduce environmental impacts? A LCA case study in North-East Canada

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Abstract

Oat is mainly produced in northern latitude countries such as Canada, which is world's second largest producer. Hence it is necessary to study the environmental impacts of alternative farming systems to conventional oat production in order to improve its environmental performance. A life cycle assessment (LCA) was conducted to compare the environmental impacts and potential improvement opportunities of conventional and organic oat grain production using three different functional units (FU); per mass (t^{-1} of grains), surface area (ha^{-1}) and gross sales income (1000 CAD^{-1}). Most data were collected from a major farm located in Quebec, Canada. Our analysis shows that organic production is globally preferable based on the area and monetary FUs, whereas conventional production has lower impacts for 11 midpoint impact categories per ton of grain produced. In addition, organic production has higher damages on human health and ecosystem quality, both per ton of grain and per hectare. The relevance of each FU is discussed in the article. The hotspots of conventional oat production are the production and use of synthetic fertilizers, while those of organic oat production are the use of organic fertilizers and manure transportation. A sensitivity analysis showed that the choice of the N_2O emission factor and phosphorus emission model considerably affects the magnitude of the impacts on climate change and freshwater eutrophication. Also, the implementation of green manure in the rotation system and the reduction of manure transport distances would significantly affect the results. In conclusion, our LCA case study does not support the idea that converting conventional oat grain production to organic agriculture would systematically decrease environmental impacts in eastern Canada. The environmental benefits largely depend on specific farmer practices and regional context.

Keywords: Life Cycle Assessment, oat grains, organic agriculture, conventional agriculture, N_2O and P emission models, functional unit.

1. Introduction

Intensive agriculture is known to be a major contributor to climate change, nitrogen and phosphorus cycles disruption, and loss of biodiversity (Willett et al., 2019). For example, the food supply chain is estimated to be responsible for up to one third of anthropogenic greenhouse gas (GHG) emissions worldwide (Crippa et al., 2021). One of the questions that arise in the current context is how to adapt our agricultural practices so that we can feed a population of 9 billion people by the middle of the century, while meeting the imperatives of development, sustainability and climate agreements. Replacing conventional farming systems with organic farming, which does not use synthetic fertilizers and pesticides, is often cited as a solution. In addition, organic products are perceived as healthier and more respectful of farmers and animals (Essoussi and Zahaf, 2008). However, the most important challenge for organic farms is to increase crop yields with minimal damage to the environment (Tuomisto et al., 2012).

Critics of organic farming argue, among other things, that it is an inefficient system because it requires the expansion of agricultural land to deliver the same amount of food as conventional production models (Meemken and Qaim, 2018; Tal, 2018; Trewavas, 2001). The control of parasites, diseases, weeds and soil fertility are also major challenges in organic agriculture. The non-utilisation of synthetic pesticides can be offset by alternative techniques, including increased seeding rates, using biopesticides to control pests, and intensified tillage for mechanical weed control (Snyder and Spaner, 2010). All these additional operations bear economic and environmental costs that must be estimated to evaluate the relevance of any transition from a conventional to an organic production system. The environmental benefits of such a transition also depend on the specific site and cultural characteristics, such as pedoclimatic conditions and agricultural crops and yields (Adewale et al., 2019), the type and amount of organic fertilizers applied, and the intensity of field operations (Seufert and Ramankutty, 2017).

Farms, whether conventional or organic, increasingly rely on other economic sectors, due to the use of various farming inputs, extensive mechanization, transport, processing, and distribution of products. Therefore, assessing the environmental impacts of a transition from conventional to organic production requires many economic flows specific to each of the production systems be accounted for. Life Cycle Assessment (LCA) is a biophysical accounting method that follows a product during its life cycle in order to inventory the inputs (resources and energy) and outputs (emissions) flows between the technosphere (the place where humans transform resources into products) and the ecosphere (the environment) (van der Werf et al., 2020). This holistic and multi-criteria method has been used since the 90s in the field of agriculture and is now well established and widely used to quantify the environmental impacts caused by agricultural production (Meier et al., 2015). However, LCA studies can yield different results depending on methodological choices, particularly the choice of functional unit (FU) (Meier et al., 2015). The Product Category Rules (PCR) for assessing the environmental performance of arable and vegetable crops propose the use of a mass unit when conducting an LCA (Environdec, 2020). However, other types of FU can be used in the same study to gain some additional insights (van der Werf et al., 2020), such as impacts related to farm land management and farmer income maximization (Nemecek et al., 2011). The desire to consider alternative FUs in an agricultural LCA is explained by the inability to exhaustively assess the environmental impacts through of multifunctional systems (Ponsioen and van der Werf, 2017; van der Werf et al., 2020). Beyond biomass production, agricultural systems also provide other services, such as income for farmers, protein, calories, ecosystem services, etc.

Each FU responds to a specific research question, and the choice of a specific FU reflects the way we envision agriculture rather than an attempt to approximate the ultimate function of agricultural production (Cerutti et al., 2013; Nemecek et al., 2011). As a general rule, LCA studies show that organic farming has a greater environmental impact than conventional farming per unit of mass of products, and a lower impact per unit of area, with strong regional variations depending on pedoclimatic conditions (Meier et al., 2015; Smith et al., 2020; Tuomisto et al., 2012).

Numerous LCA studies comparing conventional and organic production of cereal crops have been published, but none for oat (Lee and Choe, 2019; Miksa et al., 2020; Pelletier et al., 2008; Tricase et al., 2018). Smith et al. (2019) showed that organic oat production emits more GHGs than its conventional counterpart per unit of mass, but the authors did not assess other environmental impacts. Yet, oat is one of the most cultivated cereals worldwide for both animal and human consumptions. Oat protein is also a potential alternative to reduce the carbon footprint of common protein concentrates, based on whey (Heusala et al., 2020). Oat production is particularly important in countries located in Northern latitudes such as Canada, which is the world's second largest oat producer, with an estimated production of 4.5 Mt yr^{-1} (Statistics Canada, 2020). Given the importance of oat to Canadian agriculture, the comparison of the environmental impacts of conventional and organic oat production systems in Canada is an important issue.

In this study, we present the results of an LCA comparing the environmental performance of organic and conventional oat grain production in the province of Quebec, Canada, under similar pedoclimatic conditions. The goal of this analysis was to identify the main environmental impacts and the hotspots of both production models, and to assess the sensitivity of the results to certain parameters and agricultural practices, in order to examine the environmental relevance of transitioning from conventional to organic oat grain production under northern pedoclimatic conditions. The results of this study will provide solid information to farmers on various important environmental impacts and hotspots of both conventional and organic farming, to help them improve their farming practices.

2. Materials and methods

2.1 Goal and scope definition

2.1.1 Goal

This study aims to (i) evaluate and identify key strategies for mitigating the environmental impacts of oat grain production; and (ii) assess the influence of methodological choices on LCA results, specifically the choice of functional units (mass, area and monetary) as well as nitrogen emission factors and phosphorus emission model. We conducted an attributional LCA, meaning that macroeconomic issues and the conversion of agricultural systems at the regional scale are beyond the scope of this study.

2.1.2 Farm selection

The study farm selection was based on the availability and reliability of data as well as the presence of oat parcels grown in both conventional and organic production practices under the same pedoclimatic conditions, and under best management practices available (e.g., no-till for conventional, precision agriculture, etc.). The Olofée farm (48°38'06.3"N; 72°29'37.5" W), located at Saint-Félicien (Quebec, Canada), met all the pre-established criteria, with about 160 and 80 ha of oat produced annually under conventional and organic models, respectively. The average pedoclimatic characteristics of the farm can be consulted in the supplementary materials (**Table S1**).

2.1.3 Functional units

The main function of the systems compared it to produce oat grains. The FU for this study is therefore based on the mass of grain produced: the production of one ton of oat grains (14.5% moisture content, and without packaging). Results have also been calculated for two other FU. One based on the surface cultivated: management of one hectare devoted to the production of oat grains (14.5% humidity, and without packaging), and the other based on income: procurement of 1,000 Canadian dollars (CAD) in gross income from the sale of the oat grains (14.5% moisture content, and without packaging).

2.1.4 System boundaries

All processes, from extracting the raw materials necessary for agricultural production to the grain storage, are accounted for in the analysis (Cradle-to-farmgate approach, see **Figure 1**). The inventory structure was divided into two stages: field operations and operations that take place in the farm infrastructure. The following methodological choices were made for establishing the system boundaries and quantifying the environmental impacts:

- No impact was attributed to crop rotation practices. The impacts related to the agricultural phase were attributed to a single year of oat cultivation.

- No allocation of environmental burdens between the livestock and the organic farming system was applied. We only accounted for manure transport;

- No impact allocation between grain and straw was implemented, as the straw is left in the fields after harvest;

- The fixation of biogenic carbon (sequestered by plants) was not accounted for. It was therefore not included in the environmental impacts;

- Changes in the organic carbon content of the soils were not included in the inventory.

- The impacts of infrastructures used for sorting, drying and storage have been excluded. Only energy data have been included for each of these operations.

- *Ecoinvent*'s "allocation, cut-off by classification" database, version V3.4, was used (Wernet et al., 2016) to model the background processes.

2.1.5 Reference flows

Conventional oat. The average yield of conventional oats at Olofée Farm during 2015-2020 period has been 4.2 t ha^{-1}. The conventional oat kernel price paid to Quebec producers averaged 230 CAD t^{-1} during the 2014-2020 period (**Table S2**). Therefore, the farmer must produce 4.348 kg of conventional grains in order to have a gross income of 1,000 CAD. The quantity of grain produced represents the reference flow of monetary FU.

Organic oat. The average yield of organic oats at Olofée Farm during 2015-2020 period has been 3 t ha^{-1}. The organic oat kernel price paid to Quebec producers averaged 407 CAD t^{-1} during the 2014-2020 period (**Table S2**). Therefore, the farmer must produce 2,457 kg of organic grain to yield a gross income of 1,000 CAD.

Processing grain oat at farm mill. After harvest, the grains are pre-cleaned, sorted, dried, transported to the warehouse and stored. According to estimates by the farm's agricultural production managers, these steps generate an average loss of 9% on a mass basis. Losses are recorded in the FU's reference flows.

2.1.6 Life cycle impact assessment methodology and software

In order to quantify the main impacts of the two agricultural systems, we used the characterization factors proposed by ReCiPe 2016 midpoint hierarchist (problem-oriented approach) and endpoint hierarchist (damage-oriented approach) (Huijbregts et al., 2017). SimaPro 8 software (Pré-Consultans) was used for the compilation of the life cycle inventory and its characterization into life cycle impacts.

2.2 Life cycle inventory data and assumptions

Primary data on transport and amounts of agricultural inputs (seeds, fertilizers, fuels and pesticides) were collected using a questionnaire and discussions with the farm operations manager at Olofée farm. Data were collected in 2020 and the inventory was constructed with average data over the 2015-2020 period of conventional and organic oat grain production (yields, farm operations, amount of fertilizer and grain prices). The inputs and agricultural operations required for the production of conventional grains are presented in **Tables 1**, **2 and 3**. It should be noted that the inventories presented in these tables do not take losses into account; a loss coefficient of 1.0989 (amount needed to compensate for a 9% loss) was applied a posteriori to the quantities shown in the inventories to model the environmental impacts.

The *ecoinvent V.3.4* database was used to build the inventories of transport processes, seed production, fuel, synthetic fertilizers and pesticides. Furthermore, chicken manure is applied at a rate of 2.5 t ha^{-1}. The fertilizer load values proposed by Côté et al. (2009) for chicken manure marketed in Quebec were used: 27.06 kg N t^{-1}, 34.03 kg P_2O_5 t^{-1} and 21 kg K_2O t^{1} (wet basis). With respect to transport, chicken manure was collected 340 km away from the study site (680 km round trip).

The number of tractors and other agricultural machinery was calculated based on their use (number of hours) for each operation and their economic life span. Also, thanks to a global positioning system (GPS) installed in the agricultural machines, it was possible to know the precise fuel consumption of each agricultural machine to perform a specific operation. The inventory of operations was based on the recommendations of Nemecek and Kagi (2007). The datasets used and the modifications made in *ecoinvent* datasets are provided in the supplementary materials (**Tables S3, S4** and **S5**).

2.2.1 Emissions related to nitrogen, phosphorus and pesticides

We chose the nitrogen emission factor and phosphorus emission models because of their ability to identify the nature of the applied fertilizers and to select pedoclimatic parameters that best reflect the field reality. For example, direct nitrous oxide (N_2O) emissions were calculated using the emission factors proposed by Rochette et al. (2018) for the North-East Canadian agricultural context. Phosphorus emissions were calculated using the agricultural management tool *PEDT* (phosphorus export diagnostic tool) adapted to Quebec's agricultural context (Michaud et al., 2008). With respect to pesticides emissions, we followed the guidelines proposed by Audsley et al. (2003) to inventory the distribution of pesticide emissions in the three environmental compartments (air, water and soil). **Table S6** shows the data sources and values used for each pollutant and **Table S7** shows the retained values for nitrogen, phosphorus and pesticides emissions.

2.3 Sensitivity analyses

Knowing that the methodological choices can strongly influence the final results of an LCA, we conducted a sensitivity analysis on nitrogen and phosphorus emission factors, as well as manure transport distance and the integration of green manures in the rotation of both organic and conventional systems.

Nitrogen and phosphorus emissions. There is currently no standard procedure for inventorying nitrogen emissions resulting from the application of synthetic and organic fertilizers in agricultural LCA (Goglio et al., 2018). Most studies use the approaches proposed by the IPCC (Schmidt Rivera et al., 2017). We compared the results obtained in the baseline scenario with those using the updated IPCC emission factors for NH_3, N_2O and NO_3 (Hergoualc'h et al., 2019). Regarding phosphorus emissions, we compared the results obtained in the baseline scenario with the most widely used phosphorus emissions model in LCA, namely, *SALCA-P*. **Table S8** shows values used for nitrogen and phosphorus emissions and **Table S9** shows the emissions considered in the sensitivity analysis.

Manure transport. As mentioned above, chicken manure is transported over a distance of 680 km (round trip). Given that transport is generally very important for some impact categories, such as global warming, ecotoxicity and fossil resource scarcity, we carried out a sensitivity analysis on this parameter by using an average distance of 100 km (round trip).

Green manure. Dependence on exogenous nutrient sources is one of the major issues on both organic and conventional farms (Snyder and Spaner, 2010). To mitigate this issue, green manures can be incorporated into the rotation of organic and conventional systems to improve soil fertility (Knudsen et al., 2014). We tested the effect of this complementary method of nitrogen supply on our results by introducing dry pea crop as a green manure. According to the data adapted for Quebec, the plant residues (roots and stems) of dry pea would contribute 25 kg N ha^{-1} (Parent and Gagné, 2013). This nitrogen input reduces the amount of synthetic and organic nitrogen application in the two systems evaluated: the urea and monoammonium phosphate requirements in the conventional system drops to 71.46 kg ha^{-1} and 35.73 kg ha^{-1}, respectively, whereas the chicken manure demand drops to 1.6 t ha^{-1} in the organic system. It should be noted that the additional farming operations required to cultivate the green manure are not considered, because they are identical for both systems. Also, we assume that yields will not be affected by this change in fertilization technique, which may differ in practice.

3. Results

In this section, we present the general results for the midpoint and endpoint impact categories as well as the sensitivity analyses. The detailed analysis for each of the 18 midpoint categories is presented in **Table S10**. The hotspot analysis of midpoint impact category is presented in **Figure 2,** and **Figure 3** shows the midpoint impact categories that contribute the most to the endpoint impacts.

3.1 Midpoint impact categories

Among the 54 comparisons made with midpoint indicators (18 categories × 3 FUs), organic oat production showed a better environmental performance in 59% of cases (32/54), while conventional production was better in 40% of cases (22/54) (**Table 4**). When considering the mass FU (the production of 1 t of grain), the organic farming system has stronger environmental impacts than the conventional system for 11 out of 18 categories. The better environmental performance of conventional production is due in particular to a significantly higher yield per hectare (+28%), fewer hours of agricultural operations (-55%) and less transport. The ranking of several impact categories was reversed when using the area and the monetary FUs. Per unit of area, the organic production showed better environmental performance for 10 impact categories out of 18. The conventional system is outperformed because of its high intensity of agricultural inputs (e.g., synthetic fertilizer, herbicides) per unit area. However, since the organic system has lower yield, the oat grain production will have to increase somewhere else, leading to additional potential environmental impacts. With respect to the monetary FU, the organic farming system showed the best environmental performance for 16 of 18 impact categories examined. In fact, a ton of organic oat grains yields on average 79% higher gross income than conventional grains (**Table S2**). Environmental categories having their ranking reversed are mainly influenced by agricultural yield and grain price.

For three impact categories (global warming, stratospheric ozone depletion and human carcinogenic toxicity), conventional production showed the best environmental performance per ton of grains. However, the ranking is reversed for all categories in favor of organic production when the area and economic FUs are considered (**Table 4**). For six impact categories (marine eutrophication, freshwater eutrophication, ozone formation/human health, ozone formation/terrestrial ecosystems, terrestrial ecotoxicity and land use), conventional production showed the best performance for both the mass and area FUs. This is due to its lower nitrate (marine eutrophication) and phosphorus (freshwater eutrophication) emissions relative to organic production, even on a per hectare basis. The need for more manure transport in the organic production system increases the emissions related to terrestrial ecotoxicity, such as copper. Also, additional field operations increase NO_x emissions from fuel combustion (ozone formation). Finally, organic production uses almost 15% more seeds per hectare, which results in a higher use of agricultural land for seed production. However, the ranking is reversed for all six categories in favor of organic production when we consider the impacts for 1,000 CAD of income (**Table 4**).

Our analysis indicates that organic grain production releases more fine particles and contributes more to terrestrial acidification due to NH_3 emissions from manure applications, regardless of the considered FU (**Table 4**). Conventional production, conversely, has a greater impact than organic production for seven impact categories (scarcity of mineral and fossil resources, water consumption, non-carcinogenic human toxicity, ecotoxicity in freshwater and marine environments and production of ionizing radiation) regardless of the considered FU. This is explained by the major role of synthetic fertilizers production on all of these categories, contributing at least 50% of the impact for each of them.

3.2 Endpoint damage categories

Among the nine comparisons made with endpoint indicators (3 categories × 3 FUs), conventional production rates better than organic production in five cases (**Table 5**). In terms of tons of grain, conventional production has a better environmental performance for all three damage categories. Organic production has higher impacts related to human health and ecosystem quality both per ton of grain and per hectare. Organic production is better rated only in terms of monetary FU.

With respect to human health, only the categories of fine particle formation and global warming contribute significantly to the total impact of this category (**Figure 3**). Given that organic production releases more fine particles than conventional production, for both the mass and area FUs, conventional production has a lower impact on human health. Ammonia emission is the main factor affecting human health (55% organic and 40% conventional) (**Table S11**). In both production models, the majority of ammonia is released during the application of organic and synthetic fertilizers (98% organic and 96% conventional). Per hectare, organic production emits 52% more ammonia than conventional production.

Ecosystem quality is particularly affected by land use, terrestrial acidification and, to a lesser extent, by global warming (**Figure 3**). As observed for the human health category, organic oat production performance is lower on both mass and area FU. The main reason for this is that organic production has a lower yield than conventional production, which results in higher use of agricultural land per ton of grain produced. When evaluating oat production per hectare, both systems required the same amount of land. For this reason, the impact difference is small (3.7%; **Table 5**). Also, as we have seen for human health, ammonia emission remains higher in the organic system compared to conventional.

The resource depletion category is affected by the use of fossil and mineral resources (**Figure 3**). For both production systems, fossil resource use largely dominates the impacts related to the resource depletion category. Crude oil (91% organic and 67% conventional) and natural gas (7% organic and 30% conventional) are the main resources consumed over the entire life cycle of both production systems (**Table S11**). Farming operations (39%) and manure transport (34%) dominate the impacts for organic production, while conventional production is primarily affected by fertilizer production (47%), particularly urea production (33%), and farming operations (22%). Conventional production performed better on a mass basis due to higher yields. However, conventional production performed worse on both area and monetary bases. The main reason for this is the very high use of fossil energy to produce synthetic fertilizers, especially urea.

3.3 Sensitivity analysis

Nitrogen and phosphorus emissions. We identified six midpoint impact categories and two endpoint damage categories directly affected by nitrogen and phosphorus emissions (**Tables S12 and S13**). This analysis shows that the choice of nitrogen emission factors has a strong influence on midpoint impact categories. Choosing IPCC emission factors (Hergoualc'h et al., 2019): i) reverses the ranking between conventional and organic production for climate change and stratospheric ozone depletion categories. The ranking was not reversed for the marine eutrophication, fine particle matter formation and terrestrial acidification categories; ii) reduces the impact of organic production on global warming relative to conventional production with all FUs; iii) increases the impact on eutrophication of marine waters by approximately 65% for both systems; and iv) barely affects the categories of fine particle formation and terrestrial acidification (by less than 5%).

Using the SALCA-P model reduces the total impacts on the eutrophication of freshwater by 70% and 79% for conventional and organic production systems, respectively. While the impact of organic production per hectare was slightly higher (+10%) than conventional production in the baseline scenario, the use of the SALCA-P model reverses this, due to the increased contribution of P emissions from fertilizer production following the decrease in the contribution of emissions to the field for both production systems.

Regarding the endpoint impact categories, the use of IPCC nitrogen emission factors slightly lowers the human health (7% for organic and 5% for conventional) and ecosystem quality (2.4% for organic and 1.6% for conventional) impacts compared to the baseline scenario (**Table S13**). Since phosphorus emissions contributed only to the freshwater eutrophication category, they had no impact on human health and a marginal influence on ecosystem quality (**Figure 3**).

Manure transport. When chicken manure is collected closer to the farm (100 km vs. 680 km round trip), organic production becomes preferable than conventional agriculture for the categories of human carcinogenic toxicity and terrestrial ecotoxicity with all FUs (**Table S12**). It should be noted that even with a significant reduction in manure transport, the impact for the global warming category is still 12.3% higher for the organic than for the conventional system. When transport is reduced, both production systems have a better environmental performance for the same number of categories of impact (9 out of 18 midpoint categories) per unit of mass (the FU most favorable to conventional agriculture), whereas conventional agriculture performs better in 11 categories in the baseline scenario. This highlights the significant weight of transportation when comparing the two farming systems.

With regard to the endpoint categories, the reduction in transport does not have a substantial impact on the human health and ecosystem quality categories (**Table S13**). Conversely, transport reduction markedly decreases the impact for the resource depletion category. In this case, organic production shows better performances with all FUs.

Green manure. Nitrogen inputs from green manure substantially improves the environmental performance of both farming systems on all studied midpoint impact categories (**Table S12**). This scenario is more favorable to conventional farming because of the high contribution of synthetic fertilizer production and use. Overall, the introduction of green manure increases the environmental benefit of the conventional relative to the organic production system for the categories favorable to conventional production, and decreases the benefit gap of organic production for the categories that are unfavorable to conventional agriculture. Under this scenario, conventional production becomes favorable for the categories of fossil resource scarcity (mass only) and stratospheric ozone depletion (mass and area). The results for the remaining 16 impact categories were not reversed.

The use of green manure does not reverse any endpoint impact categories relative to the baseline scenario (**Table S13**). However, green manure substantially improves the environmental performance of both farming systems for the categories of human health and resource depletion. This is due to the considerable reduction in ammonia emissions (36% for organic and 40% for conventional) as a result of a decrease in the application of organic and synthetic fertilizers. In this case, the performance of conventional agriculture increases compared to organic production both per hectare and per ton of grain. Ecosystem quality is the only category for which the use of green manure has almost no effect because the amount of land for the oat cropping phase remains the same.

Green manure and manure transport. The analysis shows that the environmental midpoint impacts of both production systems are significantly reduced when both green manure and reduced manure transport are implemented (**Table S12**). This scenario is more favorable to organic production as it further reduces the transportation requirements and lowers emissions from manure application. When we evaluate the effect of these two parameters together, conventional production shows better performance: i) for 9 out of 18 impact categories (50%) per ton of grain; ii) for 6 out of 18 categories (33%) per unit of area; and iii) for 2 out of 18 categories (11%) on a monetary FU basis (fine particulate matter formation and terrestrial acidification).

The combination of green manure use and reduced transport does not significantly affect the endpoint categories relative to the use of green manure only, except for the resource depletion category. In this case, organic production is preferable regardless of the FU.

4. Discussion

4.1 Identification of hotspots

Pre-cleaning, sorting, drying and storage operations contribute very little to the impacts of conventional and organic grain production. For conventional production, most impact categories are strongly affected by the production and use of synthetic nitrogen fertilizers. This important contribution comes mainly from the production of ammonia. Indeed, the synthesis of ammonia, which is the precursor of almost all nitrogenous mineral fertilizers (e.g. urea and MAP), is very intensive in energy (Haber-Bosch process), water and infrastructure (Hasler et al., 2015). This is mainly due to the use of natural gas as an energy source and as a substrate (hydrogen). As shown by our results, the use of production techniques aimed to reduce the use of nitrogenous fertilizers, such as green manures, represents excellent ways to improve environmental performances of conventional and organic agriculture.

The main hotspots in organic grain production are the emissions in the field and the transport of organic fertilizers. Besides introducing green manure into the crop rotation, other techniques need to be implemented to reduce field emissions, especially N_2O and NH_3 which have a strong influence on several impact categories. To reduce the volatilization and leaching of nutrients, several recommendations can be followed, such as avoiding manure application before rains

(Michaud and Laverdière, 2004), introducing more grasses in crop rotations using cover crops and avoiding soil compaction (Schoumans et al., 2014), incorporating organic fertilizers quickly (Rochette et al., 2001), etc. Knowing that the implementation of these practices increases production costs, the potential decrease in farmers' income can be a barrier to the generalization of these practices (Oenema et al., 2009). Lastly, the reduction of manure transport is a key condition to increase the environmental competitiveness of organic production relative to conventional production. As discussed in Oelofse et al. (2013), it's necessary to encourage farmers to rethink their strategy for maintaining soil fertility in order to promote the emergence of a resilient agricultural system.

4.2 Influence of field emission models

The alternative emission models have little influence on the endpoint categories. Conversely, the sensitivity analysis shows a significant change for the midpoint categories of climate change and marine and freshwater eutrophication. Regarding the global warming category, the use of IPCC nitrogen emission factors favors organic farming because it reduces its impacts per ton of products to the level of conventional agriculture, while organic agriculture has a 25% higher impact with the emission factors used in the baseline analysis. According to Rochette et al. (2018) (baseline analysis from Eastern Canada), the application of synthetic fertilizers emits 19.2% more N_2O than the application of organic fertilizer (0.0211 vs. 0.0177 kg N_2O-N kg N^{-1}) while this difference increases by almost 167% when using IPCC emission factors (0.016 vs. 0.006 kg N_2O-N kg N^{-1}). In our case, even if the application of organic fertilizer is higher than the application of synthetic fertilizers, the N_2O emissions from organic production are lower when using IPCC emission factors. This highlights the importance of the N_2O emission factors when comparing organic and conventional farming systems. When available, we recommend using emission factors from local systems.

When IPCC's NO_3 emission factors are used, the eutrophication potential of marine waters significantly increases for both types of agricultural production. Of the seven nitrate emission factors compared in Henryson et al. (2020), IPCC's resulted in the higher emissions, indicating that the use of these emission factors may overestimate nitrate leaching. According to Henryson et al. (2020), the IPCC emission factors offer the possibility of spatial differentiation, but the resolution remains coarse.

The use of the SALCA-P model significantly reduces the impacts of the eutrophication potential of freshwater for both types of agricultural production. One of the main differences is the consideration of phosphorus emissions due to runoff. According to the SALCA-P model, there is no phosphorus runoff when slopes are < 3%, while the PEDT model includes slopes < 3%. As the average slope on the study farm is 2%, the SALCA-P model did not account for the effect of runoff. In addition, the SALCA-P model only partially accounts for the difference between organic and conventional agriculture. Only the effects of the application of organic and synthetic fertilizers are included, although the erosion and sediment transport processes involved in phosphorus emissions are more dependent on pedoclimatic characteristics and tillage than on the contribution of a type of fertilizer (Michaud et al., 2008; Michaud and Laverdière, 2004).

4.3 Influence of the functional unit

The FU is a strong driver of environmental impacts since it determines the ranking of the two production systems for half of midpoint impact categories and all endpoint impact categories. Despite the interest of using alternative to the mass-based FU to gaining additional insights, a number of considerations must be accounted for about the limits of each of them. The mass FU is the most commonly used in agricultural LCAs (Meier et al., 2015) and indicates the most environmentally efficient production system (Cerutti et al., 2013). However, focusing on the environmental efficiency of a production can increase impacts at the scale of an agricultural region (Salou et al., 2017). Mass FU is indeed a measure of environmental efficiency (impact/product ratio) rather than a measure of the environmental impact of an agricultural system (van der Werf et al., 2007). For example, a certain agricultural region might have nitrate and phosphorus reduction targets in order to decrease eutrophication of surface water. In this case, it is the total emissions of the system that is of interest and not its environmental efficiency. This shows the limitations of using the mass FU alone to assess local impacts.

Using one hectare of cultivated area as a FU allows to quantify the total impacts of an agricultural system and to choose agricultural productions that exert less environmental pressure at the regional level (van der Werf et al., 2007). However, depending on the agricultural context, the transition to low input production systems can require the use of additional land in order to produce an amount of products equivalent to that of high productivity systems, which can generate additional impacts related to land use change (Kirchmann, 2019). Indeed, alternative surface- and income-based FUs lead to non-functionally equivalent systems, and the additional impact of producing grains elsewhere must be kept in mind while interoperating the results. In order to account for the potential side effects of a transition to organic farming, one must widen the boundaries of the system. As discussed in van der Werf et al. (2020), it is necessary to analyze indirect effects, which include not only land use changes, but many other parameters allowing a better understanding of the consequences of transitioning to organic production, including changes in diets, reduction of waste in the food value chain, changes in price levels as well as causal chains of land use. However, an attributional LCA includes, within the system boundaries, only the processes and material flows directly used in the production, consumption and disposal of a product; therefore indirect effects are not included (Brander et al., 2008). To include indirect impacts, it is necessary to use consequential LCA.

Regarding monetary FU, few studies have used the monetary FU to assess the impacts of agricultural production (e.g., Acosta-Alba et al., 2020; Cerutti et al., 2013; Mouron et al., 2006; Van Der Werf and Salou, 2015). This approach has the merit of grouping several characteristics of agricultural production, such as food quality and yields, into a single FU, and of avoiding potential rebound effects (consumers would have less money to spend elsewhere) (Van Der Werf and Salou, 2015). However, as discussed in Repar et al. (2017), the use of the monetary FU poses a number of theoretical and practical problems. In our case, the use of this FU turned out to be less relevant than mass and area FUs for two main reasons. First, like most studies that have used this FU, we have used gross income instead of net income. Here as elsewhere (Crowder and Reganold, 2015; Smith et al., 2020), the difference in grain price more than compensates for the lower yields of the organic system. However, as the production costs (CAD kg^{-1} or CAD ha^{-1}) are generally higher for organic than for conventional farming, the benefits (income minus costs) of organic and conventional systems are not as different, and even, in many cases, lower for organic production models (Froehlich et al., 2018; Uematsu and Mishra, 2012). Second, using monetary FU (net or gross) for a single product and for a single year of cultivation does not provide a realistic estimation of a farm income. In fact, the overall incomes of organic and conventional agriculture depend, among other things, on the prices associated with all the crops included in the rotation system (Crowder and Reganold, 2015). The use of a monetary FU is therefore more relevant when the net income is accounted for over the entire rotation.

4.4 Comparative analysis with other studies

To our knowledge, this is the first comprehensive LCA comparing organic and conventional oat production systems. Korsaeth et al. (2012) conducted an LCA for conventional oat production in Norway, but the authors only used midpoint indicators. The other three studies limited their analysis to the carbon footprint and used a single mass FU. **Table S14** summarizes the main characteristics of these studies.

Smith et al. (2019) found that conventional and organic oat productions emit 350 and 370 kg CO_2 eq t^{-1}, respectively. In comparison, we find a larger difference between the conventional and organic productions (349 and 436 kg CO_2 eq t^{-1}, respectively). Nitrogen fluxes seem to be the main drivers of the discrepancy with Smith et al. (2019), who used the N_2O emission factors proposed by the IPCC (2006) (De Klein et al., 2006). When nitrogen emission factors proposed by the IPCC (2019) are used (sensitivity analysis), the CO_2 eq emissions amount 316 and 307 kg CO_2 eq t^{-1} for conventional and organic production systems, respectively (**Table S11**), which seems to indicate that IPCC's emission factors (old and new versions) reduce the difference of CO_2 eq emissions between the two production systems.

Desjardins et al. (2020) calculated the carbon footprint of conventional oats for all Canadian provinces. The authors found 222 and 920 kg CO_2 eq t^{-1} for the average production in Canada and the province of Quebec, respectively. Accounting for changes in the organic carbon content of the soil (SOC), seems to be one of the most important factors explaining this difference. This variable also played a major role in the analysis by Korsaeth et al. (2012), where SOC losses contributed 46% of total field emissions.

4.5. Limitations

The results presented in this article are insightful but should be interpreted with caution, at least for certain categories of impact. As reported in other studies, accounting for changes in soil organic carbon strongly impacts the climate change category. This variable plays an important role when comparing conventional and organic systems. Although organic production is more intensive in tillage, it returns organic matter back to the soil (e.g., organic fertilizer inputs), which can improve its carbon footprint in the long term. Moreover, the entire crop rotation must be considered for accurate estimates of soil carbon changes (Knudsen et al., 2014). Rotation is indeed a key strategy for controlling weeds and improving soil fertility, which highlights the relevance of evaluating farming systems over an entire rotation and over longer periods of time.

A number of methodological compromises were made due to a lack of data. First, we were unable to quantify indirect impacts (consequential LCA) due to lack of data related to, among other things, market studies on the production and use of animal organic fertilizers and the absence of causal models of agricultural land use. Second, we used gross rather than net income because of the unavailability of detailed economic data on the farm. Finally, the lack of data on the variability of agricultural yields for organic oats in the province of Quebec did not allow us to perform a sensitivity analysis for this parameter. However, the general trends observed remain valid regardless of yields, including the importance of choosing specific emission factors, the importance of introducing green manure in the rotation and, in the case of organic oats, the importance of reducing the transport of manure fertilizers.

van der Werf et al. (2020) discussed the methodological limitations of LCA for modelling certain impact categories. This also constitutes a limitation of our study. Indeed, ReCiPe does not differentiate between the land use impact category of an organic and conventional agricultural plot, which prevents a relevant analysis of these two systems for damage related to the quality of ecosystems. Knowing that land use represents more than 70% of the impacts on ecosystem quality, it is therefore necessary to improve the representation of the diversity of agricultural systems in life cycle impact assessment (LCIA) methodologies. Recently, Chaudhary and Brooks (2018) proposed characterization factors for species losses from five land use types (managed forests, plantations, pastures, cropland, urban) under three intensity levels (minimal, light, and intense use). Therefore, such methodology enables to assess the impact on land use of various management practices. The “Global LCIA guidance Phase 3” (GLAM), aimed at establishing a comprehensive, consistent and global LCIA method, has already identified the methodology proposed by Chaudhary and Brooks (2018) as a potential candidate for an updated recommendation. However, it is still needed to review this methodology by paying attention on uncertainties, coarse resolution of space and land use classes, and small differences between intensity levels (UNEP/SETAC Life Cycle Initiative, 2021).

5. Conclusions

This work is a first effort to understand the environmental impacts of conventional and organic oat grain production systems under homogeneous boundary conditions. Until now, most studies had only evaluated GHG emissions related to oat production, which is insufficient to understand the dynamics of impacts within an environmental system. The main conclusions of the present work are summarized below:

• The comparison of organic and conventional farming systems strongly depends on the selected FU. Mass FU allows the detection of more efficient systems at the risk of increasing regional impacts. The area FU makes it possible to detect systems with less pressure on the environment at the risk of displacing pollution outside the boundaries of the studied system. The indirect effects of the transition from conventional to organic agriculture must be accounted for to estimate potential increased regional impacts and displacement of pollution.

• When only the two most commonly used FUs in LCA studies (mass and area) are considered, conventional production has a higher impact for 7 midpoint impact categories vs. 8 for organic production in the baseline scenario, regardless of the FU considered. This shows that the environmental performance is also determined by the nature and the intensity of agricultural practices within each production system (e.g., increased need for transport, type and quantity of organic fertilizer applied, use of nitrogenous fertilizers, application of herbicides, etc.).

• The transition to organic farming would increase the damage to human health and the quality of ecosystems, both per tons of products and per hectare. However, the consideration of land use in LCA must be improved to be able to compare precisely the impacts of both agricultural systems on the quality of ecosystems.

• The choice of the N_2O emission factor and phosphorus emission model considerably affects the magnitude of the impacts on climate change and marine andfreshwater eutrophication.

• Green manures are an excellent way to improve the environmental performance of both production systems. However, the generalization of this practice requires a favorable economic context for farmers.

• Although the transition from conventional to organic agriculture is presented as a plausible solution to solve various environmental problems, the environmental relevance of this transition depends on specific regional characteristics and agricultural practices. The relevance of oat organic production mainly relies on the availability of organic fertilizer close to the farm and the improvement of fertilization techniques.

• In order to better quantify the environmental impacts of conventional and organic systems, future studies must make a number of methodological improvements. It is strongly recommended that the impacts of farming systems be evaluated over the entire rotation and over longer time periods. Also, it is important to evaluate the costs associated with farming techniques which reduce environmental impacts. This information will help farmers and governments to identify potential economic barriers to farming transition. Finally, increasing organic production is one of the targets of the Province of Quebec's bio-food policy. It is therefore necessary to conduct similar large-scale studies before implementing transition policies.

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6. References

Acosta-Alba, I., Boissy, J., Chia, E., Andrieu, N., 2020. Integrating diversity of smallholder coffee cropping systems in environmental analysis. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-019-01689-5

Adewale, C., Reganold, J.P., Higgins, S., Evans, R.D., Carpenter-Boggs, L., 2019. Agricultural carbon footprint is farm specific: Case study of two organic farms. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2019.04.253

Audsley, E., Alber, S., Clift, R., Cowell, S., Crettaz, P., Gaillard, G., Hausheer, J., Jolliett, O., Kleijn, R., Mortensen, B., Pearce, D., Roger, E., Teulon, H., Weidema, B., Zeijts, H. van, 2003. Harmonistion of environmental life cycle assessment for agriculture. Final report. Concert. action AIR3-CT94-2028.

Brander, M., Tipper, R., Hutchison, C., Davis, G., 2008. Consequential and attributional approaches to LCA: a Guide to policy makers with specific reference to greenhouse gas LCA of biofuels. Econom. Press 44, 1–14.

Cerutti, A.K., Bruun, S., Donno, D., Beccaro, G.L., Bounous, G., 2013. Environmental sustainability of traditional foods: The case of ancient apple cultivars in Northern Italy assessed by multifunctional LCA. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2013.03.029

Chaudhary, A., Brooks, T.M., 2018. Land Use Intensity-Specific Global Characterization Factors to Assess Product Biodiversity Footprints. Environ. Sci. Technol. https://doi.org/10.1021/acs.est.7b05570

Côté, D., Gasser, M.O., Poulin, D., 2009. Guide de conception des amas de fumier au champ II. L'Institut de recherche et de développement en agroenvironnement (IRDA).

Crippa, M., Solazzo, E., Guizzardi, D., Monforti-Ferrario, F., Tubiello, F.N., Leip, A., 2021. Food systems are responsible for a third of global anthropogenic GHG emissions. Nat. Food.

https://doi.org/10.1038/s43016-021-00225-9

Crowder, D.W., Reganold, J.P., 2015. Financial competitiveness of organic agriculture on a global scale. Proc. Natl. Acad. Sci. U. S. A. https://doi.org/10.1073/pnas.1423674112

De Klein, C., Novoa, R.S.., Ogle, S., SmithK.A, Rochette, P., Wirth, T.., McConkey, B.., Mosier, A., Rypdal, K., Walsh, M., Williams, S.., 2006. N2O emissions from managed soils, and CO2 emissions from lime and urea application. I: Gytarsky, M., Hiraishi, T., Irving, W., Krug, T. & Penman, J. (rEd.) 2006 IPCC Guidelines for National Greenhouse gas Inventories. Geneva: IPCC Intergovernmental Panel .

Desjardins, R.L., Worth, D.E., Dyer, J.A., Vergé, X.P.C., McConkey, B.G., 2020. The Carbon Footprints of Agricultural Products in Canada, in: Environmental Footprints and Eco-Design of Products and Processes. https://doi.org/10.1007/978-981-13-7916-1_1

Environdec, 2020. Product category rules (PCR) "arable and vegetable crops". Valid until: 2024-12-07. Version 1.0.

Essoussi, L.H., Zahaf, M., 2008. Decision making process of community organic food consumers: An exploratory study. J. Consum. Mark. https://doi.org/10.1108/07363760810858837

Froehlich, A.G., Melo, A.S.S.A., Sampaio, B., 2018. Comparing the Profitability of Organic and Conventional Production in Family Farming: Empirical Evidence From Brazil. Ecol. Econ. https://doi.org/10.1016/j.ecolecon.2018.04.022

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., Gao, X., Hanis, K., Tenuta, M., Campbell, C.A., McConkey, B.G., Nemecek, T., Burgess, P.J., Williams, A.G., 2018. A comparison of methods to quantify greenhouse gas emissions of cropping systems in LCA. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2017.03.133

Hasler, K., Bröring, S., Omta, S.W.F., Olfs, H.W., 2015. Life cycle assessment (LCA) of different fertilizer product types. Eur. J. Agron. https://doi.org/10.1016/j.eja.2015.06.001

Henryson, K., Kätterer, T., Tidåker, P., Sundberg, C., 2020. Soil N2O emissions, N leaching and marine eutrophication in life cycle assessment – A comparison of modelling approaches. Sci. Total Environ. https://doi.org/10.1016/j.scitotenv.2020.138332

Hergoualc'h, K., Akiyama, H., Bernoux, M., Chirinda, N., Prado, A. del, Kasimir, Å., MacDonald, D.M., Ogle, M., Regina, K., Weerden, T. van der, 2019. N2O emissions from managed soils, and CO2 emissions from lime and urea application. I. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse gas Inventories. IPCC Intergovernmental Panel on Climate Change, Geneva, pp. 11.1–11.48.

Heusala, H., Sinkko, T., Sözer, N., Hytönen, E., Mogensen, L., Knudsen, M.T., 2020. Carbon footprint and land use of oat and faba bean protein concentrates using a life cycle assessment approach. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2019.118376

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-016-1246-y

Kirchmann, H., 2019. Why organic farming is not the way forward. Outlook Agric.

https://doi.org/10.1177/0030727019831702

Knudsen, M.T., Meyer-Aurich, A., Olesen, J.E., Chirinda, N., Hermansen, J.E., 2014. Carbon footprints of crops from organic and conventional arable crop rotations - Using a life cycle assessment approach. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2013.07.009

Korsaeth, A., Jacobsen, A.Z., Roer, A.G., Henriksen, T.M., Sonesson, U., Bonesmo, H., Skjelvåg, A.O., Strømman, A.H., 2012. Environmental life cycle assessment of cereal and bread production in Norway. Acta Agric. Scand. A Anim. Sci. https://doi.org/10.1080/09064702.2013.783619

Lee, K.S., Choe, Y.C., 2019. Environmental performance of organic farming: Evidence from Korean small-holder soybean production. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2018.11.075

Meemken, E.M., Qaim, M., 2018. Organic Agriculture, Food Security, and the Environment. Annu. Rev. Resour. Econ. https://doi.org/10.1146/annurev-resource-100517-023252

Meier, M.S., Stoessel, F., Jungbluth, N., Juraske, R., Schader, C., Stolze, M., 2015. Environmental impacts of organic and conventional agricultural products - Are the differences captured by life cycle assessment? J. Environ. Manage. https://doi.org/10.1016/j.jenvman.2014.10.006

Michaud, A.., Giroux, M., Beaudin, I., Desjardins, J., Gagné, G., Duchemin, M., Deslandes, J., Landry, C., Beaudet, P., Lagacé, J., 2008. ODEP; un Outil de diagnostic des exportations de phosphore. Projet « Gestion du risque associé aux facteurs source et transport du phosphore des sols cultivés au Québec ».

Michaud, A.R., Laverdière, M.R., 2004. Cropping, soil type and manure application effects on phosphorus export and bioavailability. Can. J. Soil Sci. 84, 295–305. https://doi.org/10.4141/S03-014

Miksa, O., Chen, X., Baležentienė, L., Streimikiene, D., Balezentis, T., 2020. Ecological challenges in life cycle assessment and carbon budget of organic and conventional agroecosystems: A case from Lithuania. Sci. Total Environ. https://doi.org/10.1016/j.scitotenv.2020.136850

Mouron, P., Nemecek, T., Scholz, R.W., Weber, O., 2006. Management influence on environmental impacts in an apple production system on Swiss fruit farms: Combining life cycle assessment with statistical risk assessment. Agric. Ecosyst. Environ. https://doi.org/10.1016/j.agee.2005.11.020

Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G., 2011. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agric. Syst. https://doi.org/10.1016/j.agsy.2010.10.002

Nemecek, T., Kagi, T., 2007. Life cycle inventories of Agricultural Production Systems. ecoinvent report No. 15.

Oelofse, M., Jensen, L.S., Magid, J., 2013. The implications of phasing out conventional nutrient supply in organic agriculture: Denmark as a case. Org. Agric. https://doi.org/10.1007/s13165-013-0045-z

Oenema, O., Witzke, H.P., Klimont, Z., Lesschen, J.P., Velthof, G.L., 2009. Integrated assessment of promising measures to decrease nitrogen losses from agriculture in EU-27. Agric. Ecosyst.

Environ. https://doi.org/10.1016/j.agee.2009.04.025

Parent, L.E., and Gagné, G., 2013. Guide de référence en fertilisation, (2e édition). Centre de référence en agriculture et agroalimentaire du Québec [CRAAQ].

Pelletier, N., Arsenault, N., Tyedmers, P., 2008. Scenario modeling potential eco-efficiency gains from a transition to organic agriculture: Life cycle perspectives on Canadian canola, corn, soy, and wheat production. Environ. Manage. https://doi.org/10.1007/s00267-008-9155-x

Ponsioen, T.C., van der Werf, H.M.G., 2017. Five propositions to harmonize environmental footprints of food and beverages. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2017.01.131

Repar, N., Jan, P., Dux, D., Nemecek, T., Doluschitz, R., 2017. Implementing farm-level environmental sustainability in environmental performance indicators: A combined global-local approach. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2016.07.022

Rochette, P., Chantigny, M.H., Angers, D.A., Bertrand, N., Côté, D., 2001. Ammonia volatilization and soil nitrogen dynamics following fall application of pig slurry on canola crop residues. Can. J. Soil Sci. https://doi.org/10.4141/S00-044

Rochette, P., Liang, C., Pelster, D., Bergeron, O., Lemke, R., Kroebel, R., MacDonald, D., Yan, W., Flemming, C., 2018. Soil nitrous oxide emissions from agricultural soils in Canada: Exploring relationships with soil, crop and climatic variables. Agric. Ecosyst. Environ. https://doi.org/10.1016/j.agee.2017.10.021

Salou, T., Le Mouël, C., van der Werf, H.M.G., 2017. Environmental impacts of dairy system intensification: the functional unit matters! J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2016.05.019

Schmidt Rivera, X.C., Bacenetti, J., Fusi, A., Niero, M., 2017. The influence of fertiliser and pesticide emissions model on life cycle assessment of agricultural products: The case of Danish and Italian barley. Sci. Total Environ. https://doi.org/10.1016/j.scitotenv.2016.11.183

Schoumans, O.F., Chardon, W.J., Bechmann, M.E., Gascuel-Odoux, C., Hofman, G., Kronvang, B., Rubæk, G.H., Ulén, B., Dorioz, J.M., 2014. Mitigation options to reduce phosphorus losses from the agricultural sector and improve surface water quality: A review. Sci. Total Environ. https://doi.org/10.1016/j.scitotenv.2013.08.061

Seufert, V., Ramankutty, N., 2017. Many shades of gray—the context-dependent performance of organic agriculture. Sci. Adv. https://doi.org/10.1126/sciadv.1602638

Smith, L.G., Kirk, G.J.D., Jones, P.J., Williams, A.G., 2019. The greenhouse gas impacts of converting food production in England and Wales to organic methods. Nat. Commun. https://doi.org/10.1038/s41467-019-12622-7

Smith, O.M., Cohen, A.L., Reganold, J.P., Jones, M.S., Orpet, R.J., Taylor, J.M., Thurman, J.H., Cornell, K.A., Olsson, R.L., Ge, Y., Kennedy, C.M., Crowder, D.W., 2020. Landscape context affects the sustainability of organic farming systems. Proc. Natl. Acad. Sci. U. S. A. https://doi.org/10.1073/pnas.1906909117

Snyder, C., Spaner, D., 2010. The sustainability of organic grain production on the Canadian prairies - A review. Sustainability. https://doi.org/10.3390/su2041016

Statistics Canada, 2020. Estimated areas, yield, production, average farm price and total farm value of principal field crops, in metric and imperial units. Table 32-10-0359-01. https://doi.org/https://doi.org/10.25318/3210035901-eng

Tal, A., 2018. Making conventional agriculture environmentally friendly: Moving beyond the glorification of organic agriculture and the demonization of conventional agriculture. Sustain. https://doi.org/10.3390/su10041078

Trewavas, A., 2001. Urban myths of organic farming. Nature. https://doi.org/10.1038/35068639

Tricase, C., Lamonaca, E., Ingrao, C., Bacenetti, J., Lo Giudice, A., 2018. A comparative Life Cycle Assessment between organic and conventional barley cultivation for sustainable agriculture pathways. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2017.07.008

Tuomisto, H.L., Hodge, I.D., Riordan, P., Macdonald, D.W., 2012. Does organic farming reduce environmental impacts? - A meta-analysis of European research. J. Environ. Manage. https://doi.org/10.1016/j.jenvman.2012.08.018

Uematsu, H., Mishra, A.K., 2012. Organic farmers or conventional farmers: Where's the money? Ecol. Econ. https://doi.org/10.1016/j.ecolecon.2012.03.013

UNEP/SETAC Life Cycle Initiative, 2021. Global LCIA guidance phase 3 - creation of a global life cycle impact assessment method: scoping document. Paris.

van der Werf, H.M.G., Knudsen, M.T., Cederberg, C., 2020. Towards better representation of organic agriculture in life cycle assessment. Nat. Sustain. https://doi.org/10.1038/s41893-020-0489-6

Van Der Werf, H.M.G., Salou, T., 2015. Economic value as a functional unit for environmental labelling of food and other consumer products. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2015.01.077

van der Werf, H.M.G., Tzilivakis, J., Lewis, K., Basset-Mens, C., 2007. Environmental impacts of farm scenarios according to five assessment methods. Agric. Ecosyst. Environ. https://doi.org/10.1016/j.agee.2006.06.005

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-016-1087-8

Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., DeClerck, F., Wood, A., Jonell, M., Clark, M., Gordon, L.J., Fanzo, J., Hawkes, C., Zurayk, R., Rivera, J.A., De Vries, W., Majele Sibanda, L., Afshin, A., Chaudhary, A., Herrero, M., Agustina, R., Branca, F., Lartey, A., Fan, S., Crona, B., Fox, E., Bignet, V., Troell, M., Lindahl, T., Singh, S., Cornell, S.E., Srinath Reddy, K., Narain, S., Nishtar, S., Murray, C.J.L., 2019. Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. Lancet. https://doi.org/10.1016/S0140-6736(18)31788-4

Table 1

Agricultural products and transport required for oat production under organic and conventional management

Management scenarios	Products⁄Processes	Transport (km)	Inputs per functional unit		
			Area (Kg ha^{-1})	Mass (Kg t^{-1})	Income (kg 1.000 CAD^{-1})
Conventional	Seeds	500[5]	140	33.3	144.9
	Glyphosate (herbicide)[1]	500[5]	0.840	0.200	0.870
	Ag surf (herbicide)[2]	500[5]	0.230	0.054	0.240
	Refine SG (herbicide)[3]	500[5]	0.030	0.007	0.031
	MCPA ester 600 (herbicide)[4]	500[5]	0.346	0.082	0.358
	Urea (46-0-0)	500[5]	120	28.57	124.2
	Monoammonium phosphate (MAP) (11-52-0)	500[5]	60	14.29	62.11
	Potassium chloride (0-0-60)	500[5]	60	14.29	62.42
Organic	Seeds	500[5]	160	35.5	131.04
	Chicken manure	680	2500	833.3	2047.5

[1] 1.5 L ha^{-1} (540 g L^{-1}). [2] 0.25 L ha^{-1} à (920 g L^{-1}). [3] 30 g ha^{-1}. [4] 0.58 L ha^{-1} (600 g L^{-1})

[5] From the port of Montreal to farm

Table 2
Agricultural operations for oat production under organic and conventional management

Management scenarios	Processes	Tractor power (kW) & Weight (kg)	Agricultural machinery Weight (kg)	Diesel consumption (L h^{-1})	Operating rate (h ha^{-1})	Inputs per functional unit		
						Area (Kg ha^{-1})	Mass (Kg t^{-1})	Income (kg 1000 CAD^{-1})
Conventional	Plant protection application (glyphosate)	125 & 8000	6500	28	0.040	1.13	0.27	1.17
	Sowing and fertilizing (urea and MAP)	268 & 16000	8000	52	0.083	4.33	1.03	4.47
	Fertilizing (potassium chloride)	115 & 6800	4500	25	0.050	1.26	0.30	1.30
	Plant protection application (Refine SG and MCPA ester 600)	125 & 8000	6500	28	0.040	1.13	0.27	1.17
	Windrowing	105 & 5000	1800	23	0.200	4.62	1.10	4.78
	Harvesting	298 & 16800	-	70	0.200	14	3.33	14.5
Organic	Manure unloading (walking floor)	-	-	30	0.081	2.43	0.81	2
	Tillage	253 & 16000	4600	51	0.091	4.65	1.55	3.81
	Seedbed	253 & 16000	4600	48	0.083	3.99	1.33	3.27
	Organic fertilization	125 & 8000	3700	28	0.250	7	2.33	5.73
	Mechanical weed control and manure Incorporation	253 & 16000	3700	48	0.083	4	1.33	3.28
	Sowing	115 & 6800	4200	25	0.125	3.12	1.04	2.56
	Mechanical weed control	115 & 6800	2100	22	0.125	2.76	0.92	2.26
	Mechanical weed control	115 & 6800	2100	25	0.143	3.57	1.19	2.92
	Windrowing	105 & 5000	1800	23	0.200	4.59	1.53	3.76
	Harvesting	298 & 16800	-	70	0.200	14	4.67	11.4

Table 3
Mill farm operations for oat production under organic and conventional management

Management scénarios	Processus	Input	Unit	Inputs per functional unit		
				Area (Kg ha^{-1})	Mass (Kg t^{-1})	Income (kg 1.000 CAD^{-1})
Conventional	Grain transport (field to mill)	Diesel	L	10.5	2.5	10.87
	Grading and precleaning	Electricity	kWh	25.2	6	26.09
	Drying	Propane	L	33.6	8	34.78
	Transport to storage	Diesel	L	10.5	2.5	10.87
	Storage	Electricity	kWh	12.6	3	13.04
Organic	Grain transport (field to mill)	Diesel	L	7.5	2.5	6.14
	Grading and precleaning	Electricity	kWh	18	6	14.74
	Drying	Propane	L	24	8	19.66
	Transport to storage	Diesel	L	7.5	2.5	6.14
	Storage	Electricity	kWh	9	3	7.37

Table 4
Characterisation results using the mass-based, land-based and income-based functional units (FU) for midpoint impact categories.

Impact category	Functional unit (FU)	Conventional	Organic	Impact variation (%)	Sensitive to FU?
Global warming ($kg\ CO_2$ eq)	1 t^{-1}	349.0	436.3	25.0	YES
	1 ha^{-1}	1465.7	1310.1	-10.6	
	1000 CAD^{-1}	1517.4	1073.0	-29.3	
Stratospheric ozone depletion (kg CFC-11 eq)	1 t^{-1}	0.007	0.010	39.3	YES
	1 ha^{-1}	0.030	0.030	-0.5	
	1000 CAD^{-1}	0.031	0.024	-21.3	
Ionizing radiation (kBq Co-60 eq)	1 t^{-1}	3.9	3.2	-18.9	NO
	1 ha^{-1}	16.6	9.6	-41.9	
	1000 CAD^{-1}	17.1	7.9	-54.0	
Ozone formation. Human health ($kg\ NO_x$ eq)	1 t^{-1}	0.8	1.2	56.3	YES
	1 ha^{-1}	3.2	3.6	12.0	
	1000 CAD^{-1}	3.3	2.9	-11.4	
Fine particulate matter formation (kg PM2.5 eq)	1 t^{-1}	1.1	1.9	80.4	NO
	1 ha^{-1}	4.5	5.8	28.9	
	1000 CAD^{-1}	4.7	4.8	2.0	
Ozone formation. Terrestrial ecosystems ($kg\ NO_x$ eq)	1 t^{-1}	0.8	1.2	56.2	YES
	1 ha^{-1}	3.3	3.7	12.0	
	1000 CAD^{-1}	3.4	3.0	-11.4	
Terrestrial acidification ($kg\ SO_2$ eq)	1 t^{-1}	6.5	13.1	101.9	NO
	1 ha^{-1}	27.2	39.3	44.2	
	1000 CAD^{-1}	28.2	32.2	14.1	
Freshwater eutrophication (kg P eq)	1 t^{-1}	0.2	0.2	54.1	YES
	1 ha^{-1}	0.7	0.7	10.1	
	1000 CAD^{-1}	0.7	0.6	-12.9	
Marine eutrophication (kg N eq)	1 t^{-1}	0.8	1.2	56.3	YES
	1 ha^{-1}	3.2	3.6	11.7	
	1000 CAD^{-1}	3.4	3.0	-11.7	
Terrestrial ecotoxicity (kg 1.4-DCB)	1 t^{-1}	699.8	983.9	40.6	YES
	1 ha^{-1}	2939.6	2955.2	0.5	
	1000 CAD^{-1}	3043.2	2420.3	-20.5	
Freshwater ecotoxicity (kg 1.4-DCB)	1 t^{-1}	4.8	2.6	-46.4	NO
	1 ha^{-1}	20.2	7.8	-61.6	
	1000 CAD^{-1}	20.9	6.4	-69.6	
Marine ecotoxicity (kg 1.4-DCB)	1 t^{-1}	5.4	3.9	-27.3	NO
	1 ha^{-1}	22.6	11.8	-47.9	
	1000 CAD^{-1}	23.4	9.7	-58.8	
Human carcinogenic toxicity (kg 1.4-DCB)	1 t^{-1}	3.5	3.6	2.7	YES
	1 ha^{-1}	14.8	10.9	-26.3	
	1000 CAD^{-1}	15.3	8.9	-41.7	

Human non-carcinogenic toxicity (kg 1.4-DCB)	1 t^{-1}	124.7	89.3	-28.4	NO
	1 ha^{-1}	523.9	268.8	-48.7	
	1000 CAD^{-1}	542.4	220.1	-59.4	
Land use (m^2a^{-1} crop eq)	1 t^{-1}	1239.7	1740.1	40.4	YES
	1 ha^{-1}	5206.6	5220.4	0.3	
	1000 CAD^{-1}	5389.9	4275.5	-20.7	
Mineral resource scarcity (kg Cu eq)	1 t^{-1}	0.7	0.6	-10.7	NO
	1 ha^{-1}	2.7	1.8	-35.9	
	1000 CAD^{-1}	2.8	1.4	-49.3	
Fossil resource scarcity (kg oil eq)	1 t^{-1}	54.9	54.2	-1.3	YES
	1 ha^{-1}	230.6	163.0	-29.3	
	1000 CAD^{-1}	238.8	133.5	-44.1	
Water consumption (m^3)	1 t^{-1}	4.3	0.8	-82.7	NO
	1 ha^{-1}	18.3	2.3	-87.6	
	1000 CAD^{-1}	18.9	1.8	-90.2	

The impact variation (%) reflects the difference of organic relative to conventional (e.g, a positive value reflects a higher impact of organic compared to conventional). The "FU sensitivity" column refers to the ability of at least one FU to reverse the results of an impact category in favor of an agricultural production system.

Table 5
Results using the mass-based, land-based and income-based functional units for endpoint impact categories.

Impact category	Functional unit (FU)	Conventional	Organic	Impact variation (%)	Sensitive to FU?
Human health (DALY)	1 t^{-1}	1.05E-03	1.66E-03	57.9	YES
	1 ha^{-1}	4.43E-03	5.00E-03	12.9	
	1000 CAD^{-1}	4.58E-03	4.09E-03	-10.7	
Ecosystems (species.yr)	1 t^{-1}	1.36E-05	1.98E-05	45.1	YES
	1 ha^{-1}	5.73E-05	5.94E-05	3.7	
	1000 CAD^{-1}	5.93E-05	4.87E-05	-18.0	
Resources (USD)	1 t^{-1}	2.15E+01	2.27E+01	5.6	YES
	1 ha^{-1}	9.03E+01	6.82E+01	-24.4	
	1000 CAD^{-1}	9.35E+01	5.59E+01	-40.2	

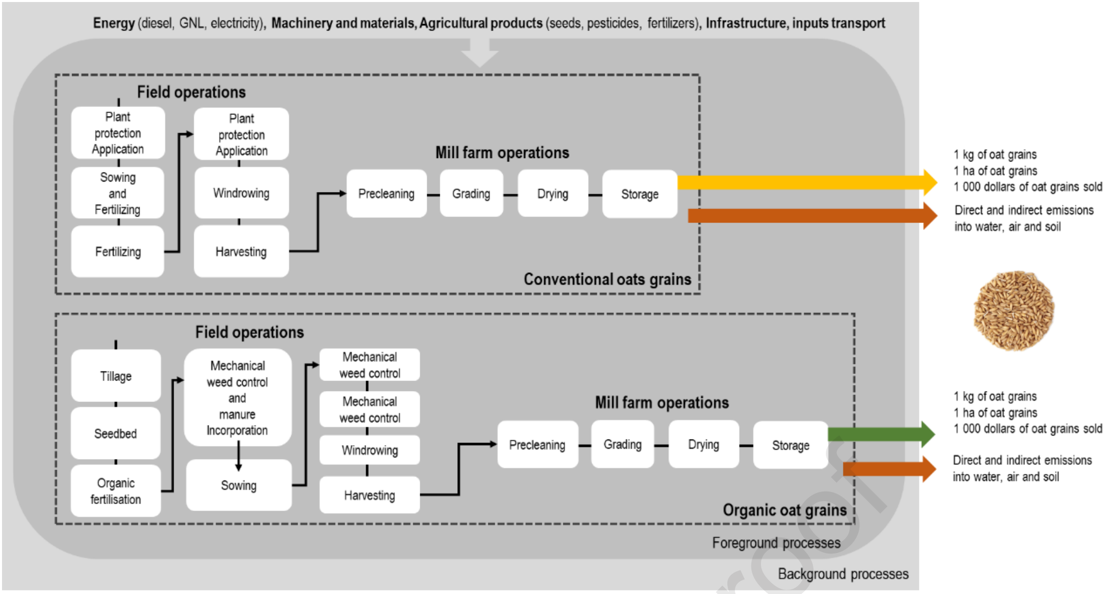

Figure 1. System boundaries for the organic and conventional grain production systems

A) Conventional grains

Impact category	Agricultural inputs						Field				Mill farm
	Seeds	Herbicides	Urea	Monoammonium phosphate (MAP)	Potassium chloride	All input transportation	Agricultural operations	Herbicides emissions	Fertilizers emissions	Land use	Grain processing
Global warming	7,2%	1,3%	14,3%	4,3%	1,3%	2,4%	9,9%	0,0%	51,2%	0,0%	8,2%
Stratospheric ozone depletion	6,9%	0,1%	0,3%	0,1%	0,0%	0,1%	0,3%	0,0%	92,2%	0,0%	0,1%
Ionizing radiation	11,2%	8,8%	28,4%	22,0%	3,8%	4,1%	13,0%	0,0%	0,0%	0,0%	8,8%
Ozone formation, human health	9,5%	1,3%	9,0%	6,1%	1,7%	4,9%	52,4%	0,0%	0,0%	0,0%	15,2%
Fine particulate matter formation	3,9%	0,9%	8,0%	5,9%	0,7%	1,0%	9,3%	0,0%	66,9%	0,0%	3,4%
Ozone formation, terrestrial ecosystems	9,5%	1,3%	9,2%	6,1%	1,7%	4,9%	52,2%	0,0%	0,0%	0,0%	15,2%
Terrestrial acidification	2,6%	0,3%	3,7%	2,6%	0,3%	0,4%	3,0%	0,0%	85,8%	0,0%	1,2%
Freshwater eutrophication	4,0%	2,8%	5,3%	5,2%	1,5%	0,4%	1,9%	0,0%	78,4%	0,0%	0,4%
Marine eutrophication	15,6%	0,1%	0,2%	0,1%	0,0%	0,0%	0,0%	0,0%	83,9%	0,0%	0,0%
Terrestrial ecotoxicity	8,8%	2,1%	36,8%	18,3%	5,0%	15,5%	9,3%	1,3%	0,0%	0,0%	2,7%
Freshwater ecotoxicity	11,1%	3,3%	20,9%	22,2%	6,1%	2,6%	7,0%	25,4%	0,0%	0,0%	1,4%
Marine ecotoxicity	13,1%	3,4%	28,9%	29,6%	7,8%	4,2%	9,3%	1,7%	0,0%	0,0%	2,0%
Human carcinogenic toxicity	13,2%	5,0%	29,7%	19,7%	7,7%	4,9%	16,3%	0,0%	0,0%	0,0%	3,4%
Human non-carcinogenic toxicity	15,3%	3,1%	27,5%	32,0%	7,9%	3,8%	9,0%	0,1%	0,0%	0,0%	1,3%
Land use	3,5%	0,0%	0,0%	0,3%	0,0%	0,0%	0,0%	0,0%	0,0%	96%	0,0%
Mineral resource scarcity	15,2%	10,2%	22,8%	26,4%	6,1%	2,5%	15,5%	0,0%	0,0%	0,0%	1,3%
Fossil resource scarcity	6,5%	2,5%	35,0%	12,0%	2,8%	5,3%	20,2%	0,0%	0,0%	0,0%	15,8%
Water consumption	2,8%	1,7%	61,3%	24,2%	2,1%	0,5%	1,7%	0,0%	0,0%	0,0%	5,7%

B) Organic grains

Impact category	Agricultural inputs		Field			Mill farm
	Seeds + transport	Manure transportation	Agricultural operations	Manure emissions	Land use	Grain processing
Global warming	9,4%	11,9%	15,6%	56,5%	0,0%	6,5%
Stratospheric ozone depletion	7,9%	0,2%	0,4%	91,4%	0,0%	0,1%
Ionizing radiation	19,3%	31,7%	38,2%	0,0%	0,0%	10,8%
Ozone formation, human health	10,4%	19,4%	60,5%	0,0%	0,0%	9,7%
Fine particulate matter formation	3,4%	3,4%	9,9%	81,4%	0,0%	1,9%
Ozone formation, terrestrial ecosystems	10,4%	19,4%	60,5%	0,0%	0,0%	9,7%
Terrestrial acidification	2,1%	1,2%	2,9%	93,3%	0,0%	0,6%
Freshwater eutrophication	3,6%	1,6%	3,8%	90,7%	0,0%	0,3%
Marine eutrophication	16,0%	0,0%	0,1%	83,9%	0,0%	0,0%
Terrestrial ecotoxicity	12,6%	68,9%	16,5%	0,0%	0,0%	1,9%
Freshwater ecotoxicity	26,2%	30,3%	40,8%	0,0%	0,0%	2,7%
Marine ecotoxicity	22,9%	35,6%	38,7%	0,0%	0,0%	2,8%
Human carcinogenic toxicity	18,6%	29,8%	48,3%	0,0%	0,0%	3,3%
Human non-carcinogenic toxicity	28,5%	33,3%	36,4%	0,0%	0,0%	1,8%
Land use	4,0%	0,1%	0,1%	0,0%	95,8%	0,0%
Mineral resource scarcity	24,7%	17,2%	56,6%	0,0%	0,0%	1,5%
Fossil resource scarcity	11,5%	33,2%	39,3%	0,0%	0,0%	16,0%
Water consumption	25,6%	19,0%	22,4%	0,0%	0,0%	33,0%

Figure 2: Detailed contribution analysis of each process for all midpoint impact categories for conventional (A) and organic (B) grains production. The total of each line is 100%.

Endpoint categories	Midpoint categories	Unit	Conventional	Organic
Human health	Global warming, Human health	DALY	31	24
	Stratospheric ozone depletion	DALY	0	0
	Ionizing radiation	DALY	0	0
	Ozone formation, Human health	DALY	0	0
	Fine particulate matter formation	DALY	64	73
	Human carcinogenic toxicity	DALY	1	1
	Human non-carcinogenic toxicity	DALY	3	1
	Water consumption, Human health	DALY	1	0
Ecosystems	Global warming, terrestrial ecosystems	species.yr	7	6
	Global warming, freshwater ecosystems	species.yr	0	0
	Ozone formation, terrestrial ecosystems	species.yr	1	1
	Terrestrial acidification	species.yr	10	14
	Freshwater eutrophication	species.yr	1	1
	Marine eutrophication	species.yr	0	0
	Terrestrial ecotoxicity	species.yr	0	0
	Freshwater ecotoxicity	species.yr	0	0
	Marine ecotoxicity	species.yr	0	0
	Land use	species.yr	81	78
	Water consumption, terrestrial ecosystem	species.yr	0	0
	Water consumption, aquatic ecosystems	species.yr	0	0
Resources	Mineral resource scarcity	USD	1	1
	Fossil resource scarcity	USD	99	99

Figure 3: Contribution of midpoint categories to damage categories. The sum of the columns of each endpoint category is 100%.

Highlights

- Organic and conventional oat production systems were compared through a full LCA
- Organic oat has low environmental impacts per area cultivated and sales income
- Conventional oat has low environmental impacts per product unit
- N_2O and P emission models strongly affect the comparison of both production systems
- Green manures strongly improve the environmental performance of both systems

Declaration of interests

☐ The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

☐The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

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