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Abstract

French organic vegetable farms are diverse, ranging from complex biodiversity-based systems with many vegetables to simple input-based systems with few vegetables, which suggests that their impacts on the environment may differ. We used life cycle assessment (LCA) to assess impacts of three contrasting farms: a microfarm (MF, high crop diversity and a low input level), a medium-sized farm specialised in sheltered production (SP, low crop diversity and a high input level), and a large farm specialised in open-field production (OP, intermediate input level and crop diversity). To manage the complexity of organic vegetable farms, we opted for a system LCA, based on farm inputs and output (i.e. « vegetables ») for a one-year period. Using functional units based on mass of output, area, and economic value, we analysed five impacts: climate change, cumulative energy demand, marine eutrophication, on-farm biodiversity, and the use of plastic. Farming-system LCA assessed environmental impacts of farms with different levels of agroecology, including complex systems with a large diversity of crops grown on small areas. The three functional units strongly influenced the ranking of the systems. Per ha, the systems differed greatly in climate change and energy demand: SP had the highest impacts, whereas OP had the lowest impacts. Per kg and per €, the systems differed much less in climate change and energy demand, and even ranked differently. OP used much less plastic but performed worse on biodiversity and yield. Despite its higher yield, SP performed no better than the other two farms for climate change, energy demand, and plastic use per kg and €. The impact on biodiversity contrasted with the other impacts, which highlighted the importance of semi-natural habitats. Quantification of plastic use echoed growing concerns about (micro-)plastic pollution in agricultural soils and landscapes, and the newly identified planetary boundary on novel entities. Estimating nitrate leaching was difficult, and the Intergovernmental Panel on Climate Change (IPCC)

Tier 1 model used to do so seemed unsatisfactory; thus, estimated marine eutrophication impacts had high uncertainty. Crop residues contributed greatly to marine eutrophication. In this perspective, models for estimating crop residues and nitrate leaching in a farming-system LCA approach need to be improved. In agroecological systems, semi-natural habitats are part of the farming system. The farming-system LCA approach requires clear rules for setting farm boundaries, which strongly influence impacts per ha and biodiversity impacts.

Key words: life cycle assessment, agroecology, horticulture, organic farming, biodiversity, farming-system LCA

1. Introduction

French organic vegetable farms are diverse, ranging from complex biodiversity-based systems, with many vegetables, to simple input-based systems, with few vegetables. Beyond this conceptual dichotomy, Pépin et al. (2021) described four types: microfarmers (high crop diversity and a low level of inputs), medium-sized market gardeners (high crop diversity and variable level of inputs), producers specialised in cultivation under shelter (low crop diversity and a high level of inputs), and large market gardeners specialised in open-field cultivation (low crop diversity and moderate input use). The heterogeneity of input use and farming practices, which can be interpreted as sign of bifurcation between "agroecological" and "conventionalised" organic farming (Pépin et al., 2021), may influence environmental impacts, but these have yet to be quantified. Organic farming claims to be environmentally sound, and conventionalisation of organic practices is seen as a threat to the identity of organic farming (Darnhofer et al., 2010). Quantifying environmental impacts of different types of organic farms will inform the debate about the conventionalisation of organic farming.

Life cycle assessment (LCA) has been used to assess environmental impacts of agricultural products and systems for several years. LCA studies of vegetable production have been conducted at the crop scale for open-field (e.g. Abeliotis et al., 2013) and sheltered production (e.g. Cellura *et al.*, 2012) and at the farm scale (Adewale et al., 2019; Markussen et al., 2014), for both conventional and organic farming, including studies that compared them (Foteinis and Chatzisymeon, 2016) LCA is a method that quantifies a variety of potential environmental and health impacts and resource depletion issues that are associated with goods or services (i.e. multi-criteria assessment). The environmental impacts relevant to agricultural production include climate change, eutrophication, biodiversity decline, and energy demand, among others. The first three impacts correspond to planetary boundaries that have already been exceeded (Rockström et al., 2009; Steffen et al., 2015). Agriculture contributes greatly to climate change (IPCC, 2019a). The runoff and leaching of nitrogen (N) from agricultural soils are the main cause of marine eutrophication (Le Moal et al., 2019). Biodiversity is rarely considered in LCA studies (Knudsen et al., 2019), although it is a key agri-environmental indicator (Haas et al., 2000) and is high on the political agenda (United Nations Environment Programme, 2021a). Organic farming claims to enhance biodiversity (European Commission, 2007) and rely more on natural regulation, which makes biodiversity an important issue. Current methods for assessing impacts on biodiversity, reviewed by Curran et al. (2016), include global approaches (Chaudhary et al., 2015; Chaudhary and Brooks, 2018; Knudsen et al., 2017), more detailed approaches (Jeanneret et al., 2014), and attempts to combine both in a case study (Bystricky et al., 2020).

Plastic pollution is an emerging concern worldwide, with for example a ban on certain single-use objects in the European Union (European Commission, 2019). The use of plastic in agriculture and the accumulation of microplastics in agricultural soil has been highlighted (United Nations Environment Programme, 2021b). Vegetable crop production, including in organic farming, is a major user of plastic, particularly as mulch and tunnels for several purposes (e.g. earlier production, higher yield, weed control, cleaner vegetables) (Lamont, 2017, 2005), which is a threat to long-term soil quality (Steinmetz et al., 2016). Massive use of plastic, especially in organic farming, has caused controversy (Held, 2019). Plastic pollution contributes greatly to exceeding the planetary boundary for novel entities (i.e. "novel in a geological sense and that could have large-scale impacts that threaten the integrity of Earth system processes") (Persson et al., 2022). However, there is no ready-made indicator in current LCA methods to assess plastic or microplastic pollution.

Organic vegetable farms can be complex systems that combine a wide range of crops, intercrops, and non-crop biodiversity to help maintain the farming system's health and resistance to disturbance (Morel and Leger, 2016). Within a field, farmers may grow several vegetables at the same time and that have different growing durations, which makes the spatial and temporal organization of crops complex (Aubry et al., 2011), with no clearly defined crop rotation (Morel and Leger, 2016). A crop that often has high value, such as tomato, may be preceded and/or followed by several other crops on the same part of the field.

Generally, fertilisers are applied to feed the soil (Fortier, 2012) rather than directly to the crops. Organic fertiliser such as compost may not be applied every year but instead every 2-4 years, which complicates the ability to allocate organic fertilisers to a given crop. Moreover, due to complex soil dynamics, allocating carbon dioxide (CO_2) and nitrous oxide (N_2O) emissions as well as nitrate (NO_3) leaching to a given crop is challenging (Goglio et al., 2018). Another challenge is that farmers often do not know the yield of each crop, as for many vegetables, small quantities can be harvested regularly over time. These challenges increase uncertainty in the allocation of inputs and estimates of outputs.

Given these challenges related to the complexity of organic vegetable farms, it seems relevant to assess their environmental impacts using a farming-system LCA rather than a product LCA, which focuses on individual crops. Farming-system LCA approaches the farm as a whole that produces different products, which helps assess and compare farms and understand mechanisms that influence environmental impacts (Goglio et al., 2018). In this perspective, all inputs and operations are estimated for the entire farm, and the output is the total production of vegetables.

The objectives of this study were to 1) assess environmental impacts of contrasting organic vegetable farms, 2) describe strengths and weaknesses of farming-system LCA, and 3) identify needs for research to better assess impacts of complex organic vegetable farms. To reach these objectives, we performed an LCA of three French organic vegetable farms, each being a specific case corresponding to a more general farm type, using farming-system LCA. Using functional units (FUs) based on mass of vegetables, area, and economic value, we analysed the climate change impact, cumulative energy demand, marine eutrophication impact, impact on biodiversity, and the use of plastic.

2. Materials and methods

2.1. Description of the farms

We chose three farms that participated in the survey of Pépin et al. (2021) used to characterize the diversity of French organic vegetable farms. We chose farms that were among the most typical of their types. Because these farms were specific cases with individual characteristics, however, they should not be considered as completely representative. The farms were 1) a microfarm (MF, high crop diversity and a low input level), 2) a medium-sized farm specialised in sheltered production (SP, low crop diversity and a high input level), and 3) a large farm specialised in open-field production (OP, intermediate input level and crop diversity) (**Table 1**).

Table 1. Characteristics of three French organic vegetable farms studied

Characteristic	Farm type		
	Microfarm	Sheltered production	Open-field production
Farm area	0.34 - 1.1 ha	3.2 ha	21.3 ha
Cultivated vegetable area	0.28 ha	2.0 ha	17.5 ha
Open-field vegetable area	0.16 ha	0 ha	17.5 ha
Sheltered vegetable area	0.12 ha	2.0 ha	0 ha
Labour (full-time equivalent)	1.3	5.0	4.3
Number of tractors	1	3	3
Number of vegetable crops	35	6	20
Main vegetable crops	Many	Tomato, cucumber, lettuce, strawberry	Potato, cabbage, carrot, squash, onion
Fertilisation	Mainly compost of green waste + manure, green manure	Commercial fertiliser + green manure	Cattle + poultry manure + green manure
Tillage	Shallow tillage or no-tillage	Deep non-inversion tillage	Deep non-inversion tillage
Weed control	Organic mulch or reusable plastic mulch	Single-use plastic mulch	Mechanical weeding + thermal weeding on carrot
Pest and disease control	Mainly natural biocontrol	Purchased biocontrol	None + insect-proof netting on turnip and radish
Seeds and seedlings	Some seeds and seedlings self-produced	Seeds and seedlings purchased	Some seedlings self-produced
Food supply chain	Direct selling, locally	Long supply chain, to France and Germany	Direct selling + short supply chain, locally
Turnover	33 000 €	475 000 €	380 000 €
Year of creation	2017	1987	1992
Organic since the beginning	Yes	No, since 2005	No, since 1997
Region of France	North-west	South-east	North-west
Biotechnical Index[1]	0.73	0.07	0.47
Socio-economic Index[2]	1	0	0.88

[1] The biotechnical index estimates the extent to which organic practices were agroecological (towards 1) or conventionalised (towards 0). Values are from Pépin et al. (2021).

[2] The socio-economic index estimates the extent to which the socio-economic context was territorially embedded (towards 1) or connected to the global market (towards 0). Values are from Pépin et al. (2021).

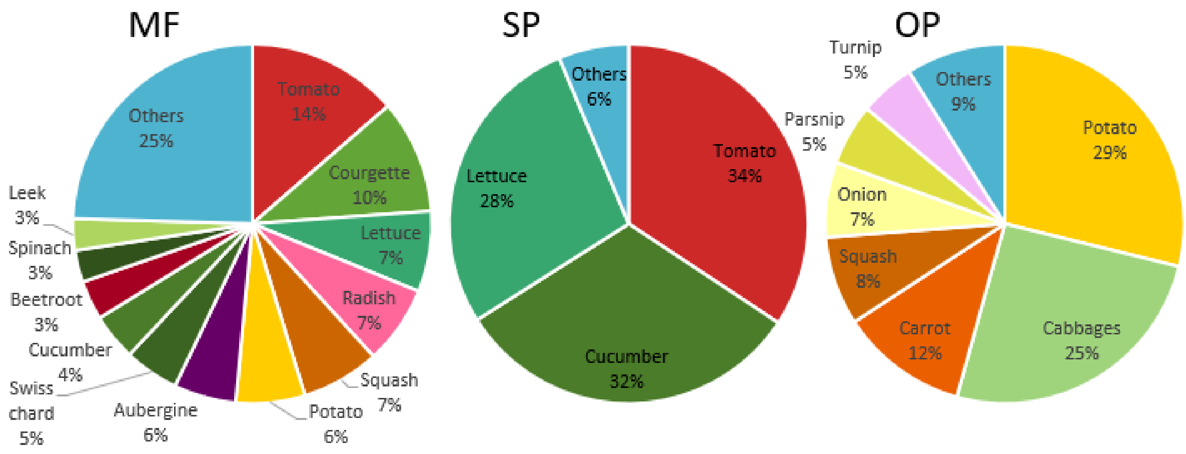

Fig. 1. Distribution of crops produced (fresh mass) by the three types of farms: microfarm (MF), sheltered production (SP), open-field production (OP)

MF was a recently established microfarm in the Brittany region that produced 35 types of vegetables (**Fig. 1**) in a 1200 m² tunnel and a 1600 m² open field (**Fig. 2**). The farmer was inspired by a French farming trend called “market gardening on living soil” (*maraîchage sur sol vivant*) that aims to protect and feed the soil - and its living organisms such as earthworms, bacteria and mycelia – by combining no-tillage and permanent cover of organic mulch and plants. Fertilisation consisted of compost of green waste, manure, and manure pellets to achieve long-term fertility and avoid short-term NO_3 deficiency due to microbial activity. Reusable insect-proof netting was used to decrease insect problems, and copper sulphate was used against blight on tomatoes (the only crop that received a treatment). The selling outlets were vegetable boxes, a local market, shops, and restaurants in the nearby village.

SP, in the Provence-Alpes-Côte-d'Azur region, produced mainly tomato and cucumber in summer and lettuce in winter (**Fig. 1**), in 33 tunnels, for a total cultivated area of 19 840 m² (

Fig.*3*). Sorghum cover crops were grown for 1-2 months each year in ca. 25% of the tunnels to add fresh biomass to the soil. Fertilisation consisted of industrial manure pellets and beet vinasse applied before each crop. Single-use plastic mulch was used against weeds. Purchased insects were released in the tunnels to control pests (*Macrolophus* spp., *Chrysopa* spp.) and for pollination (bumblebees). The farmer sold the vegetables to wholesalers in France and Germany under the biodynamic label.

OP produced vegetables on 24 ha of open fields in Brittany (**Fig. 4**) following a four-year rotation: potato / rye followed by turnip / cabbages (cauliflower, green cabbage, savoy cabbage, Brussels sprouts, kale) / various vegetables (e.g. carrots, onions, squash) (**Fig. 1**). Fertilisation consisted of cow and poultry manure applied three out of every four years. Weeding was mechanical and, in carrot crops, thermal (natural gas). Reusable insect-proof netting was used to decrease insect problems on some vegetables, but overall pest and disease control was limited. The farmer sold the vegetables locally to organic stores and wholesalers, and at local markets.

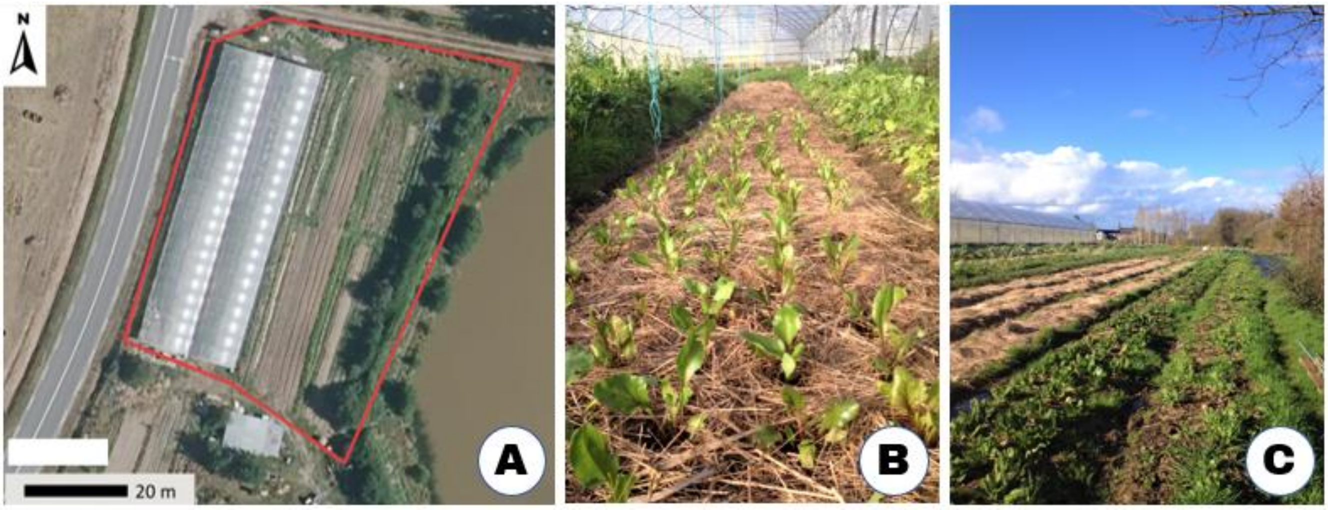

Fig. 2. (A) Satellite image (source: www.geoportail.gouv.fr) and (B, C) photographs (source: the authors) of the microfarm. The red line indicates the farm's boundary.

Fig.3. (A) Satellite image (source: www.geoportail.gouv.fr) and (B, C, D) photographs (source: the authors) of the sheltered production farm. The red line indicates the farm's boundary.

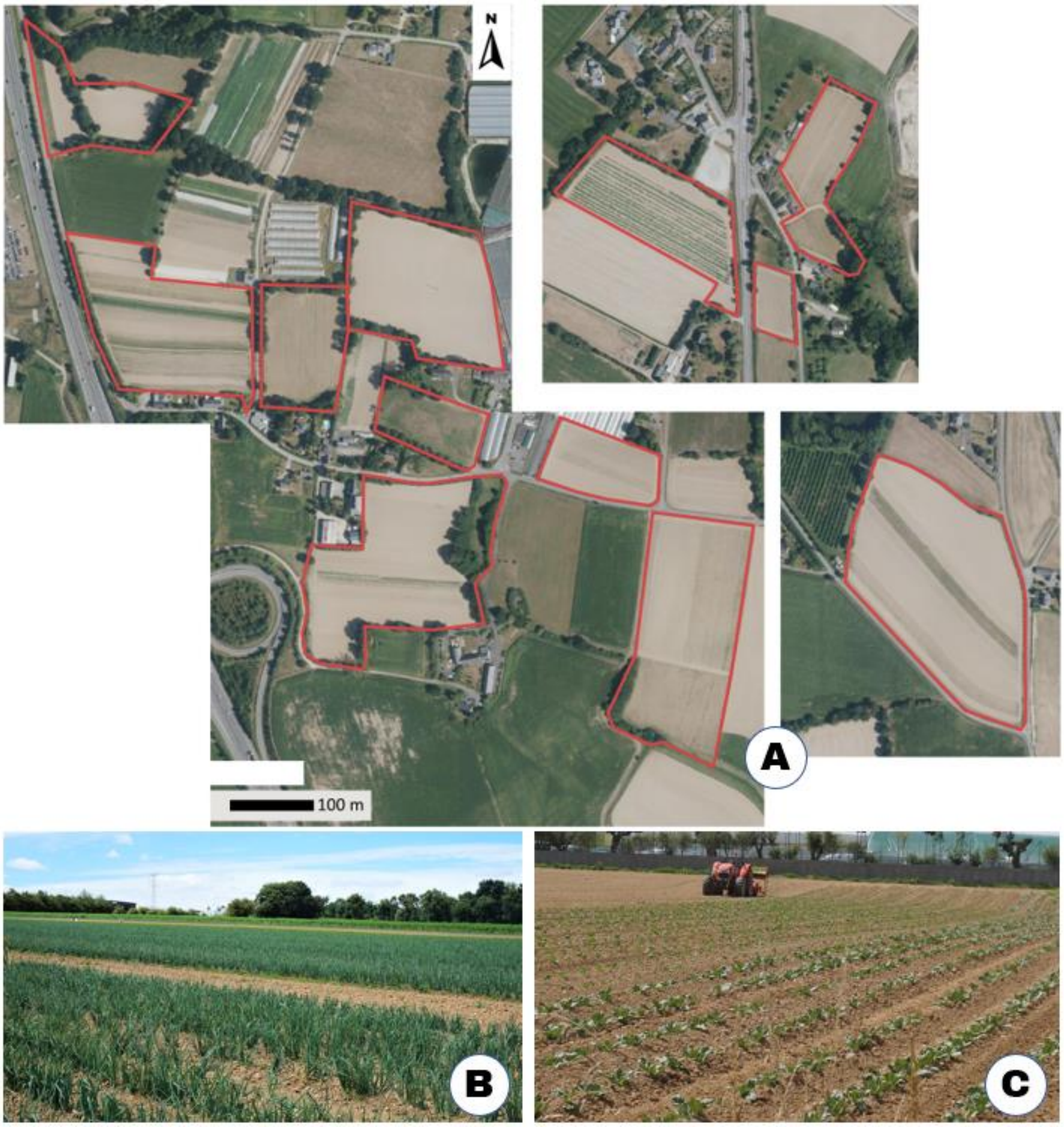

Fig. 4. (A) Satellite image (source: www.geoportail.gouv.fr) and (B, C) photographs (source: the authors) of the open-field production farm. The red lines indicate the farm's boundary.

2.2. Goal and scope definition

The aim of this study was to compare environmental impacts of three organic vegetable farms, each of which was a specific case that represented a more general farm type (Pépin et al., 2021). We analysed the farms as a whole in a farming-system LCA: we considered the total annual production of vegetables and the total inputs, without specifying which input was used for which crop.

The impact categories considered were climate change (CC) with a 100-yr temporal horizon, marine eutrophication (ME), cumulative energy demand (CED), and on-farm biodiversity; use of plastic was also assessed. The impact assessment method used for CC and ME was ReCiPe 2016 (Huijbregts et al., 2016). We calculated CED following Frischknecht et al. (2015). Biodiversity was assessed by adapting SALCA-BD (Jeanneret et al., 2014) to vegetable production. SALCA-BD assesses potential impacts on terrestrial biodiversity of 11 indicator-species groups of land-use types (including semi-natural habitats) and management practices. Field-scale impact scores were aggregated at the farm scale.

For biodiversity scores, the contribution of each land-use type equalled the land-use type's intrinsic score weighted by the proportion of the farm area it occupied. Thus, a large or small contribution could be due to a high or low intrinsic score, respectively, or to the occupation of a large or small proportion of the farm, respectively, or both. Because of the weighting, the farm score was not a simple sum of the values for each land-use type, as for LCA indicators. For this reason, we calculated scores of the entire farm separately from the scores of the cultivated land alone. A higher score indicated higher biodiversity.

Plastic use was calculated by summing the mass of plastic used per year on the farm (**Table Erreur** *! Il n'y a pas de texte répondant à ce style dans ce document.**2***). The mass of materials that lasted several years was divided by their respective life span, which yielded a mean annual value. Plastic materials included tunnel covers and plastic components, (fert)irrigation pipes and drips, mulching sheets, insect-proof netting for pest protection, pots and trays for purchased and farm-grown seedlings, and plant support clips and strings.

Table Erreur ! Il n'y a pas de texte répondant à ce style dans ce document.**2. Quantities of plastic used per year (kg) by the three farm types (microfarm (MF), sheltered production (SP), and open-field production (OP))**

Type of plastic	MF	SP	OP
Polypropylene	22.6	0.0	2.1
Polyethylene	7.1	1618.5	0.0
Polystyrene	11.8	38.9	0.0
Polyester resin	2.2	72.0	0.0
Phenolic resin	0.0	0.7	0.0
Polyvinylchloride	5.3	57.5	21.0
Ethylene vinyl acetate	52.3	1728.5	0.0
Polylactide	0.0	135.8	0.0
Nylon	0.0	0.0	11.9
Total	**101.3**	**3651.9**	**35.0**

2.2.1. Farming-system approach

As mentioned, given the complexity of organic vegetable farms, we opted for a system LCA based on farm inputs and output for a one-year period (**Fig. 5**).

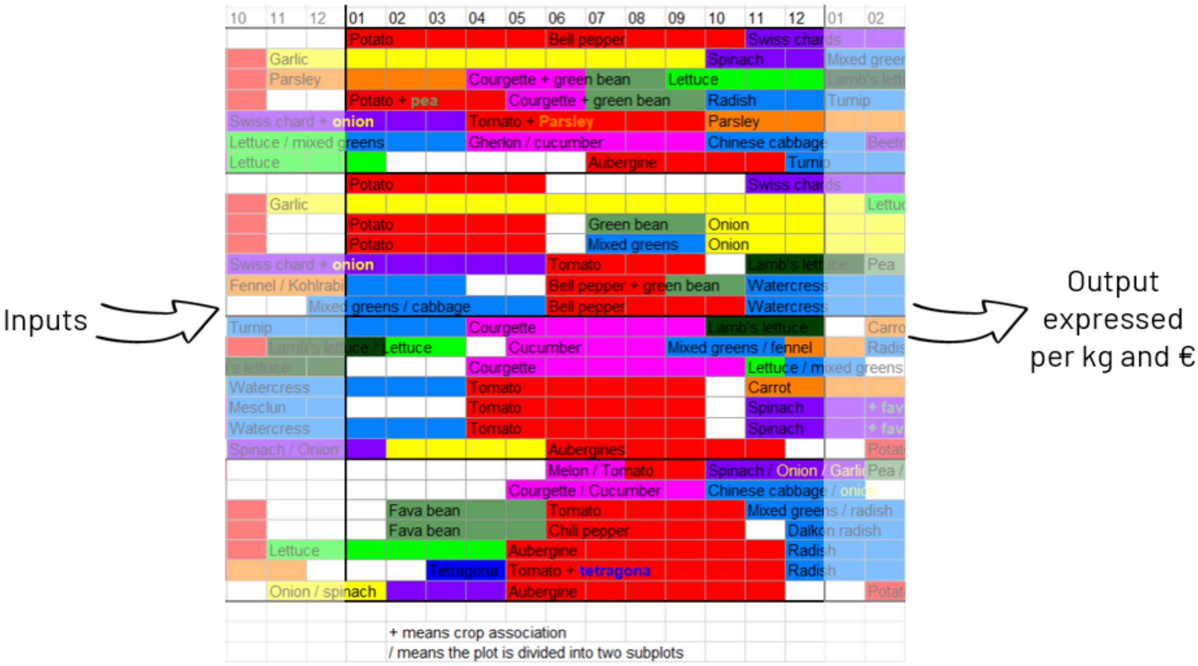

Fig. 5. System Life Cycle Assessment applied to a microfarm. The colours represent botanical families, each column is a month, and each line is a vegetable bed of 43 m² (1200 m²/28 beds).

2.2.2. Functional units

Agriculture has several functions, the first of which is to produce food. Product mass is often used as a FU to represent this function (Schau and Fet, 2008). Another function is to occupy land sustainably, for which an area-based FU was used. Last, agriculture has an economic function for farmers, and the economic value of products reflects their quality (van der Werf and Salou, 2015). In the present study, prices reflected value at the farm gate. An FU based on economic value is also a way to capture the heterogeneity of product mixtures among farms.

Except for impacts on biodiversity, which were expressed by a single score for the entire farm, impacts were expressed according to the following FUs:

- per ha of farmland occupied for one year, which included cultivated land and on-farm semi-natural habitats (e.g. hedges, pastures, ruderal areas, spaces between tunnels), as they may provide regulating ecosystem services, but excluded off-farm land associated with production of inputs
- per kg of vegetables produced during one year
- per € of vegetables produced during one year, based on sales to the first buyer, who may or may not have been the final consumer

2.2.3. System boundary

We assessed impacts from the cradle to farm gate. The foreground system included field preparation, fertilisation, sowing and planting, weeding, pest and disease control, irrigation, harvesting, and on-farm storage (**Fig. 6**). The background system included the production of fertilisers, main materials, and energy used for production and on-farm storage. The construction phase of tunnels was included but not the production phase of tractors or pumps. Processes beyond the farm gate, such as transportation, packaging, retail, use, and end of life, were not included.

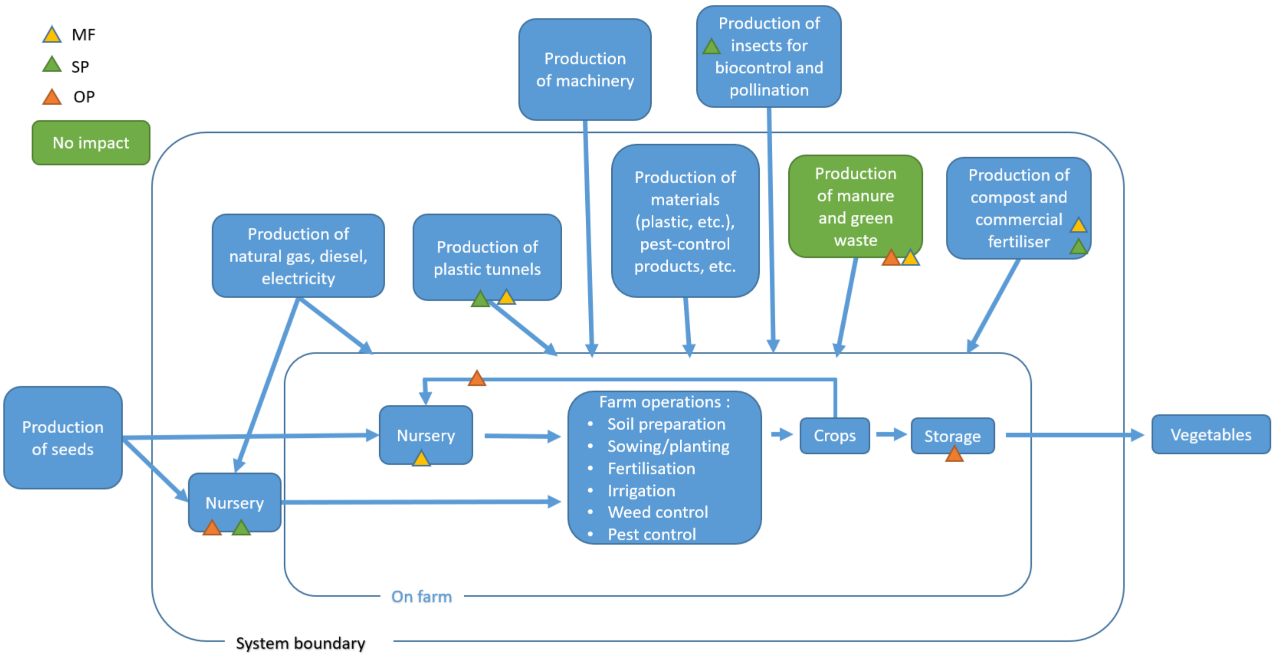

Fig. 6. Diagram of material flows of the three farm types (microfarm (MF), sheltered production (SP), and open-field production (OP)) and their system boundaries. Materials were common to all three types, except for those noted (triangles).

2.2.4. Estimation of emissions and biodiversity scores

For organic fertilisers, field emissions, including direct and indirect N_2O, ammonia (NH_3), and nitric oxide (NO) emissions, as well as NO_3 leaching, were calculated according to EMEP (2019) and IPCC (2019b). N and ammonium contents of fertilisers were obtained from their commercial documents when available or from Avadí and Paillat (2020) and Koch and Salou (2020). N supplied by crop residues and cover crops was calculated according to IPCC (2019b) using generic values. Emissions factors for NH_3 emissions were obtained from the French data base Agribalyse® (Koch and Salou, 2020), which follows EMEP methodology (Supplementary Material). Impacts of producing manure and raw green waste, which were considered as waste from upstream systems, were attributed to the producers of livestock and green waste, according to Agribalyse® methodology. Impacts of producing commercial fertiliser and composts were estimated using Avadí (2020). We adapted the green-waste composting process to divide environmental burdens between the producer of green waste (92.8%) and the production of compost (7.2%) using economic allocation based on the cost of composting and the price of compost. This type of allocation was suggested by Ekvall and Tillman (1997) and used by Christensen et al. (2018) for similar composts. A variety of organic plant-protection substances, including biodynamic preparations, sexual confusion substances, plant-stimulation products and natural pesticides (**Table 1**), were used in small quantities. Life cycle inventories (LCIs) for these inputs and for purchased insects for biocontrol and pollination were not available; we thus excluded them, except for copper sulphate, which existed in the ecoinvent® data base. Seedling production on- and off-farm was included, but seed production was not, due to the lack of data for it and its minor impact. When seedling LCIs did not exist in the Agribalyse® data base, we used the LCI of the seedling of a similar crop after modifying the amount of natural gas consumed in a heated greenhouse nursery. Natural gas consumption was estimated using the Hortinergy tool (www.hortinergy.com), with data on period of the year, duration, and seedling density based on expert opinion. We included plastic use on the farm but not off of the farm (e.g. plastic trays for purchased seedlings). We excluded boxes for harvest and storage and nets for selling potatoes. Biogenic carbon (i.e. in the biomass produced) was excluded, as the biomass produced was consumed (vegetables) or decomposed (crop residues) in the short term. Transport of inputs to the farms was not included except when it was included in ecoinvent® or Agribalyse® LCIs.

Using SALCA-BD adapted to vegetable production, farm-scale impact scores on biodiversity were calculated at two scopes: cultivated land alone, and both cultivated land and semi-natural habitats. For SP and OP, the crops were assessed individually and considered as different habitats. For MF, since many vegetables were intercropped in the tunnel and open field, we used the SALCA-BD category "intercropped vegetables".

2.3. Life cycle inventory and data collection

The foreground system was based on the use of inputs and infrastructure as reported by the farmers. LCI data for the inputs and infrastructure in the background system came from ecoinvent® 3.5 (Wernet et al., 2016) in SimaPro® 9.0. Farming practice data were collected through interviews with the farmers. In the first part of the interview, we asked general questions to obtain an overview of the farm (e.g. farm map; crops grown; farmer's approach to fertilisation; weed, pest and disease control; irrigation). In the second part, we asked about practices at the farm or crop scale, depending on the farm's management. For example, most data for SP was at the crop scale (e.g. fertilisation, pest management), while the rest was at the farm scale (e.g. diesel, disposable drip pipes). Quantitative records of practices were used when available. Follow-up phone calls and e-mails allowed us to obtain missing or inaccurate data.

Production quantities, farm area, and turnover were key data as they were used for the FUs. Turnover, calculated from farmers' accounts, was considered reliable. Farm areas were calculated from the French Géoportail web mapping service (www.geoportail.gouv.fr). Because OP grew rye in the rotation, its area (4.2 ha for 1 year, i.e. 20% of its cultivated land) was subtracted from the farm area, and consequently, 20% of fertiliser input was subtracted. Other inputs (mainly diesel) used to produce rye were not included. SP and OP had recorded the quantities of vegetables sold at the farm and crop scales, respectively. MF practiced direct selling (i.e. boxes and markets), with nearly daily harvest of small quantities, which were not weighed. We estimated the total production by dividing the farm turnover by the mean price per kg of vegetables (estimated from the boxes). This estimate was double-checked by multiplying the mean yields of organic vegetables by the area occupied by each, which yielded a difference of only 4%. Mean yields were provided by the farmer of MF, who estimated expected yields based on a variety of technical references.

2.4. Sensitivity analysis of farm area

We performed a sensitivity analysis of the farm area of MF by including semi-natural areas to differing degrees. The MF farmer owned 1.10 ha of land, but one field was not cultivated at the time of the survey (**Fig. 7A**). Excluding this field resulted in an area of 0.71 ha (**Fig. 7Fig. 7B**). With these boundaries, a large area was covered by pond banks that lay far from the farm's core. Excluding the furthest banks resulted in 0.46 ha (**Fig. 7C**). Considering only the closest semi-natural habitats resulted in an area of 0.34 ha (**Fig. 7D**), which was used as the reference area of MF when comparing it to the other farms. The sensitivity analysis was performed for CC and biodiversity.

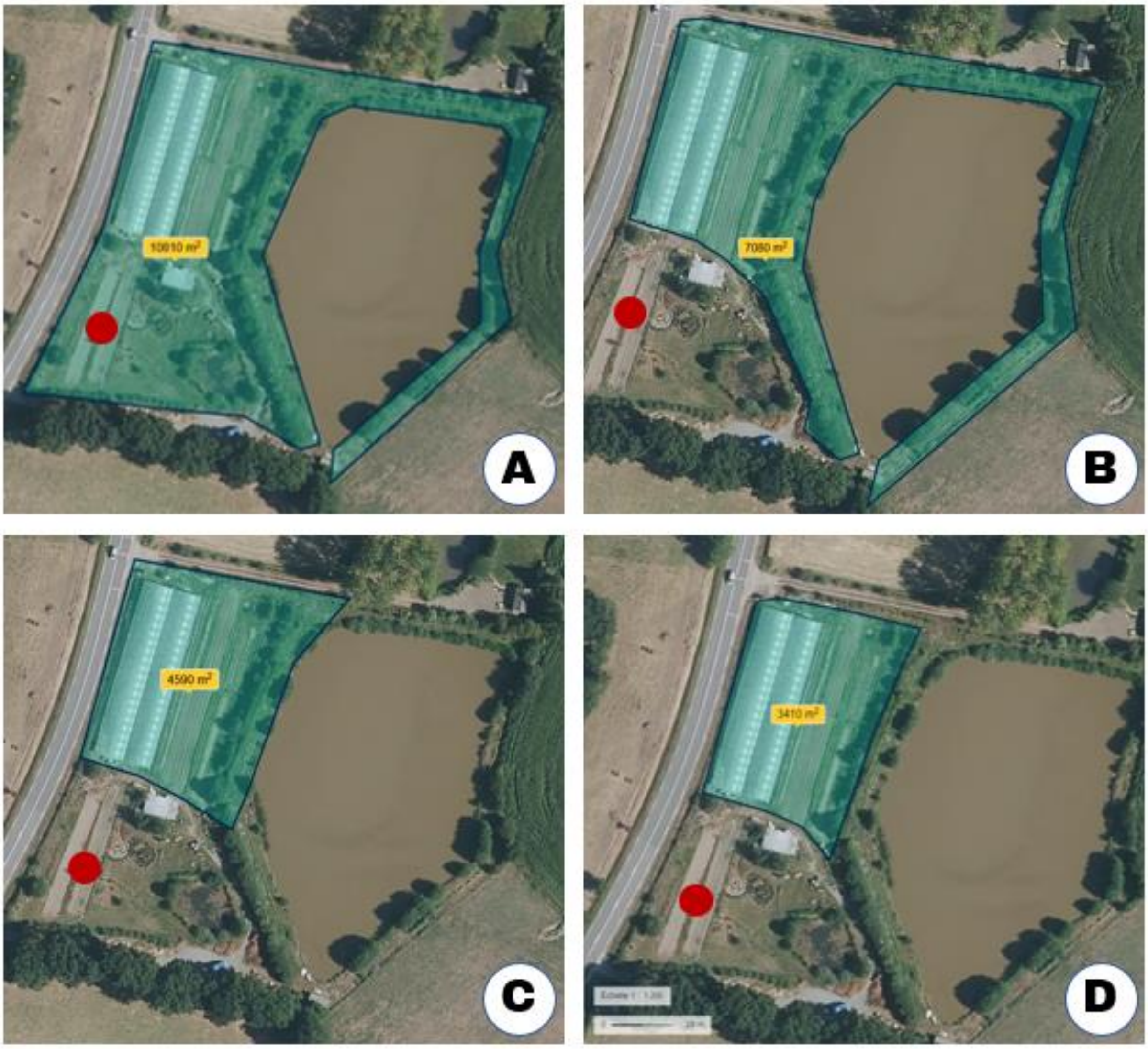

Fig. 7. Estimates of the area of the microfarm using different boundaries: *(A) 1.10 ha, (B) 0.71 ha, (C) 0.46 ha, and (D) 0.34 ha. The field identified with the red spot was not cultivated at the time of the survey.*

3. Results

3.1. Main input and output flows

The main input and output flows were expressed per ha of land occupied per year (**Table 3Table *3***). SP had the highest N fertiliser input (362 kg N/ha/yr), mainly from solid (90%) and liquid (10%) commercial fertilisers made with raw materials such as livestock manure, castor bean meal, bone meal, phosphate, or plant-based compost and waste. MF applied 141 kg N/ha/yr, which was provided mainly by slow-release N fertilisers: composted cow manure (40% of N input), shredded green waste (30%), and compost of green waste (25%). OP had the lowest N input, with 96 kg N/ha/yr from livestock manure. The direct energy used by SP (74 GJ/ha/yr) was composed mainly of electricity (72%), mainly for irrigation. The direct energy used by MF (48 GJ/ha/yr) was composed entirely of diesel, also mainly for irrigation. The direct energy used by OP (16 GJ/ha/yr) was composed of diesel for tractors (64%) and electricity for a storage refrigerator (33%). SP had the highest yield (109 t/ha/yr), followed by MF (43 t/ha/yr) and OP (11 t/ha/yr). MF had the highest mean vegetable price (2.83 €/kg), followed by SP (2.20 €/kg) and OP (1.90 €/kg).

Direct emissions of NO, NO_3, and N_2O were nearly proportional to fertiliser N input. NH_3 was also emitted after application of N fertiliser, but because fertiliser types varied, emissions were not proportional the amounts applied. CO_2 emissions were caused by lime used to whitewash tunnels.

Table 3. Main annual inputs and output flows of the three farms expressed per ha of total cultivated land

Type	Item	Unit	Microfarm	Sheltered production	Open-field production
Inputs	Fertiliser	kg N/ha/yr	141	362	96
	Electricity	GJ/ha/yr	0	53.4	5.1
	Diesel	GJ/ha/yr	48.4	20.3	10.0
	Natural gas	GJ/ha/yr	0	0	0.4
	Irrigation water	m^3/ha/yr	4622	4111	0
	Plastic	kg/ha/yr	363	1821	1
	Seedlings	no./ha/yr	0 (self-production of seedlings and direct sowing) Potting soil: 5357 kg/ha/yr	Tomato: 3355 Cucumber: 3097 Lettuce: 119 323 Strawberry: 4788 Celery: 3327 Fennel: 6397	Cabbage: 3903 Onion: 2857 Leek: 1029 Swiss chard: 263 Squash:143 Potato: 286 kg/ha/yr + self-production of seedlings and direct sowing
	Chemicals	-	Copper sulphate: 0.47 kg/ha/yr Iron phosphate: 36 kg/ha/yr	Wettable sulphur: 0.6 kg/ha/yr Neem oil: 1.8 L/ha/yr Plant-based biostimulant: 0.8 L/ha/yr Sex pheromone: 180 doses/ha/yr Horn silica (501): 20 g/ha/yr Prepared horn manure (500P): 140 g/ha/yr	-
	Purchased insects	no./ha/yr	-	*Macrolophus* spp. (box of 1000 insects): 6 *Chrysoperla carnea* (box of 10 000 insects): 28 Bumblebee hives: 17	-
	Shading paint	kg/ha/yr	0	590	0
	Plastic tunnel	ha/ha/yr	0.43	1	0
Output	Vegetables	t/ha/yr	43	109	11
	Price	€/kg	2.83	2.20	1.90
Emissions	N_2O	kg/ha/yr	2.75	7.30	1.73
	NH_3	kg/ha/yr	7.22	16.90	9.39
	NO	kg/ha/yr	5.69	13.92	3.54
	NO_3	kg/ha/yr	214	531	131
	CO_2	kg/ha/yr	0	259	0

3.2. Assessment of farm impacts

Per ha, SP had the highest CC impact, followed by MF and OP (13.3, 7.5, and 1.3 t CO_2 eq./ha, respectively) (**Fig. Erreur** *! Il n'y a pas de texte répondant à ce style dans ce document.**8***). The ranking of SP, MF, and OP was the same for CED (387, 157, and 29 GJ/ha, respectively) and ME (23.2, 12.2, and 7.2 kg N eq./ha, respectively).

For CC, the contribution of system components varied among farms. Field emissions contributed more for SP than for MF or OP (1.51, 0.68, and 0.52 t CO_2 eq./ha, respectively). Fertiliser production also contributed more for SP than for MF or OP (2.17, 0.75, and 0 t CO_2 eq./ha, respectively). Diesel contributed much more for MF than for SP or OP (3.70, 1.04, and 0.68 t CO_2 eq./ha, respectively). Conversely, plastic tunnels contributed more for SP than for MF or OP (4.49, 2.01, and 0 t CO_2 eq./ha, respectively). Plastic production (other than for tunnels) also contributed more for SP than for MF or OP (1.31, 0.23, and 0.01 t CO_2 eq./ha, respectively). Seedling production contributed much more for SP than for MF or OP (2.02, 0.15, and 0.06 t CO_2 eq./ha, respectively).

For MF, diesel was the main contributor (49%) to CC, followed by tunnels (27%), fertiliser (10%), and field emissions (9%). For SP, tunnels were the main contributor (34%), followed by seedling production (15%, mainly for greenhouse heating), fertiliser (16%), field emissions (11%), and plastic (10%). For OP, diesel and field emissions were the two main contributors (54% and 34%, respectively).

For CED, the contributions were generally similar to those to CC, except that field emissions did not contribute; instead, seedlings (because of the use of peat) contributed more, particularly for MF (2% of CC vs. 23% of CED), as did electricity, particularly for SP, where it was used for irrigation (4% of CC vs. 33% of CED). Field emissions dominated ME for MF, SP, and OP (98%, 96%, and 96%, respectively), followed by a modest contribution of fertiliser production (2% for SP) and seedlings (4% for OP).

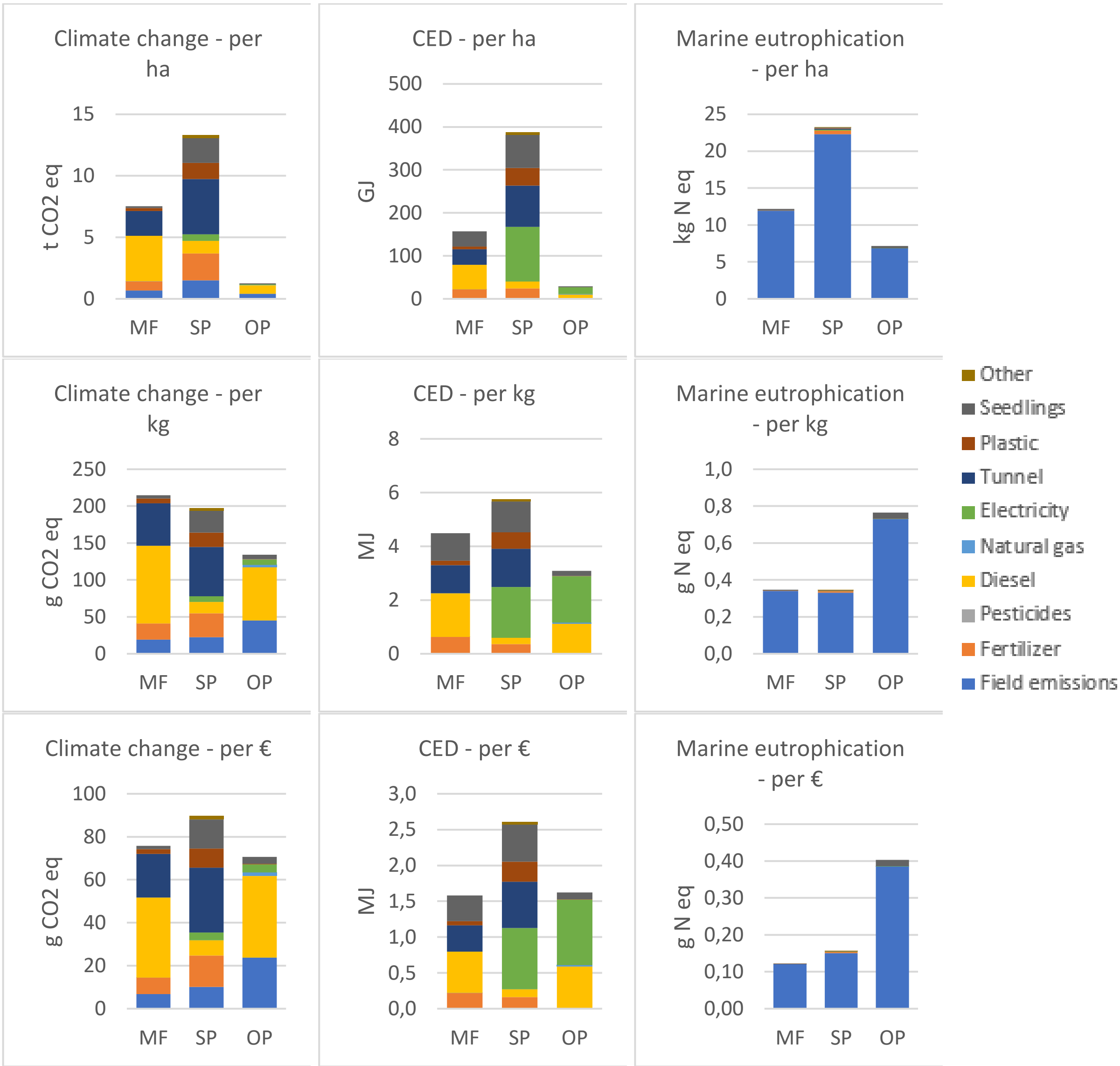

Fig. Erreur ! Il n'y a pas de texte répondant à ce style dans ce document.**8. Impacts per ha of farmland during one year, per kg of vegetables, and per € of vegetables; and contributions of inputs and field emissions for the microfarm (MF), sheltered production farm (SP), and open-field production farm (OP)**

Per kg of vegetables, MF had the highest CC impact, followed by SP and OP (215, 198, and 134 g CO_2 eq./kg, respectively) (**Fig. Erreur** *! Il n'y a pas de texte répondant à ce style dans ce document.**8***). The ranking of MF, SP, and OP was the same for CED (4.5, 5.7, and 3.1 MJ/kg, respectively). Conversely, OP had the highest ME impact, followed by OP and SP (0.77, 0.35, and 0.35 g N eq./kg, respectively).

Per € of vegetables, SP had the highest CC impact, followed by MF and OP (90, 76, and 71 g CO_2 eq./€, respectively) (**Fig. Erreur** *! Il n'y a pas de texte répondant à ce style dans ce document.**8***). SP also had the highest CED, followed by MF and OP (2.6, 1.6, and 1.6 MJ/€, respectively). OP had the highest ME impact, followed by SP and MF (0.40, 0.16, and 0.12 g N eq./€, respectively).

3.3. Biodiversity

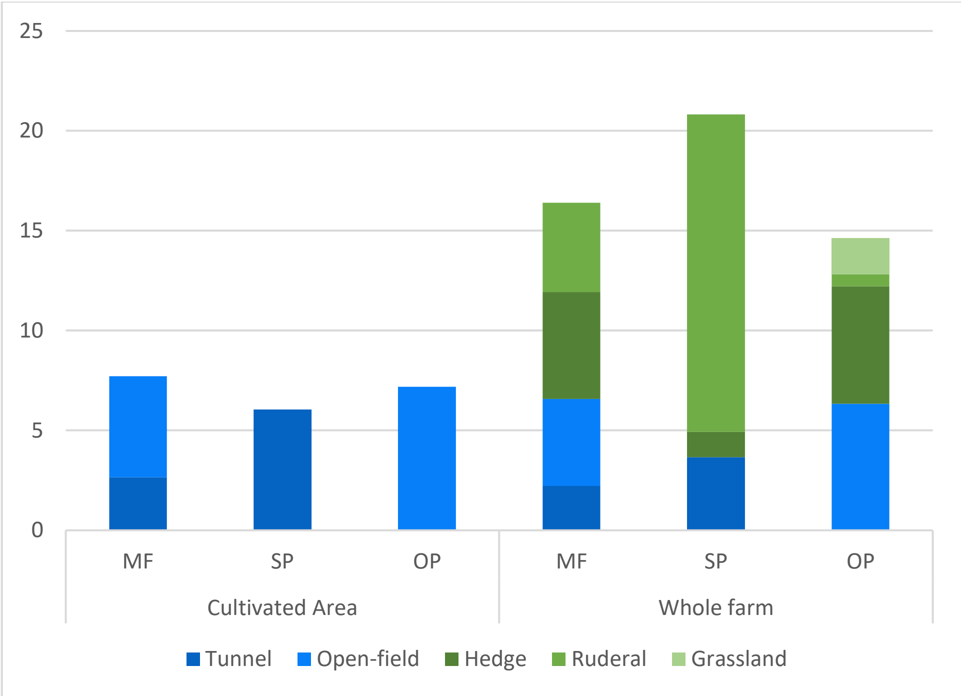

Fig. 9. On-farm biodiversity scores (higher = better for biodiversity) and contributions of land-use types for the three types of farms (microfarm (MF), sheltered production (SP), and open-field production (OP)) for the entire farm (cultivated land (blue bars) and semi-natural habitats (green bars)) and for the cultivated land alone

Differences in the biodiversity score among the farms were smaller when considering cultivated land alone (7.7, 6.0, and 7.2 for MF, SP, and OP, respectively) than when considering the entire farm, for which SP had the highest score (20.8), with a contribution of 82% from semi-natural habitats (especially ruderal areas (76% of the total)) and 18% from tunnels (**Fig. *9***). MF had a score of 16.4, with relatively equal contributions from cultivated land (40%) and semi-natural habitats (60%). OP had a score of 14.6, with cultivated land contributing 43% and semi-natural habitats contributing 57%, of which 40% of the total was due to hedges.

3.4. Plastic use

Depending on the FU, SP used 2-4 times as much plastic as MF (1129 and 299 kg/ha, 16.8 and 8.5 kg/t of vegetables, and 7.6 and 3.0 kg/k€ of vegetables, respectively), whereas OP used little plastic (2 kg/ha, 0.2 kg/t of vegetables, and 0.1 kg/k€ of vegetables) (**Fig. 10**). Plastic was used mainly in tunnels (60% of plastic use for MF and SP). For MF, insect-proof netting and pots and trays represented 17% and 12% of plastic use, respectively. For SP, plastic mulch in tunnels and disposable water pipes represented 28% and 9% of plastic use, respectively.

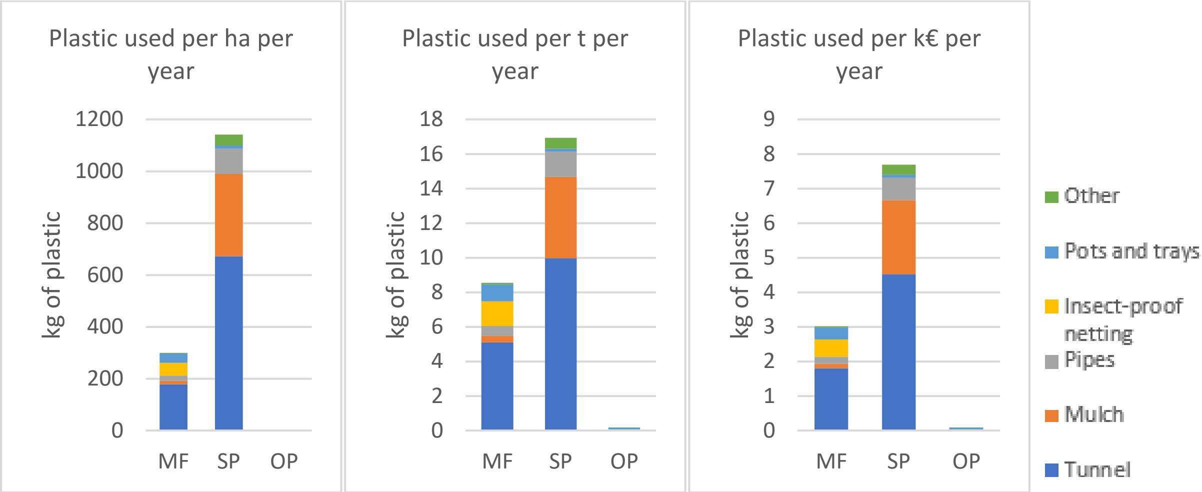

Fig. 10. On-farm plastic use per ha of farmland, per t of vegetables, and per k€, and contributions of plastic uses for the three types of farms: microfarm (MF), sheltered production (SP), and open-field production (OP).

3.5. Effects of farm area on impacts per ha

In the sensitivity analysis of farm area, the cultivated area of 0.28 ha represented 25%, 39%, 61%, and 82% of the farm area when farm area equalled 1.10, 0.71, 0.46 and 0.34 ha, respectively. These four areas resulted in CC impacts of 2.3, 3.6, 5.6, and 7.5 t CO_2 eq./ha, respectively (**Fig. 11A**), while biodiversity scores were 28.8, 24.3, 20.0, and 16.4, respectively (**Fig. 11B**).

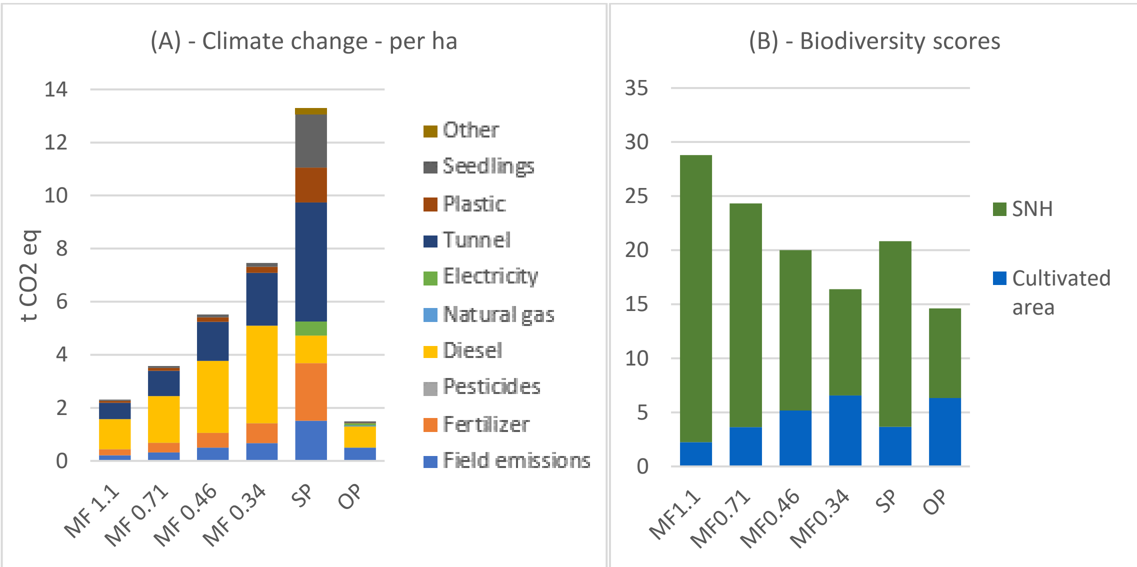

Fig. 11. (A) Climate change impact expressed per ha of farmland and (B) biodiversity scores for four scenarios for the microfarm (MF) (for areas of 0.34, 0.46, 0.71, and 1.10 ha), sheltered production farm (SP), and open-field production farm (OP). SNH: semi-natural habitats.

4. Discussion

4.1. Comparison of the farms

4.1.1. Climate change and cumulative energy demand

Environmental impacts of the three farms differed among impact categories and FUs. Per ha of land occupied, OP had the lowest CC impact and CED, due to its low input use. Conversely, SP had the highest CC impact and CED per ha because it produced 2-3 crops per year, which led to higher input use. MF had an intermediate CC impact and CED. Part of this farm had one crop per year (open field), and the other part had two crops per year (tunnel). The CC impact of SP per ha was 10.6 times as high as that of OP. Major contributors to CC impact and CED included the use of diesel (MF) and electric (SP) pumps for irrigation, the tunnel structure (MF and SP), the use of plastic water pipes and mulch (SP), and seedling production in heated greenhouses (SP); these inputs were not used by OP. Impacts of tunnels were due mainly to their galvanized steel structures, which was assumed to last 20 years, and plastic covers, which were assumed to last 4-8 years, depending on the farm. Using the same tunnel longer would reduce impacts.

Per kg of vegetables, MF and SP had a similar CC impact, while that of MF was 1.6 times as high as that of OP. This difference was much smaller than for the CC impact per ha because OP had a lower total yield than MF. The higher productivity of SP gave it a similar or slightly lower CC impact per kg despite using more inputs per ha; however, for CED, SP had higher impact than MF per kg. SP relied more on direct (diesel and electricity) and indirect (plastic and seedlings) energy than MF and OP. Per €, the highest CC impact (SP) was 1.3 times as high as the lowest CC impact (OP). MF practiced direct selling of several vegetables, including those with high value (e.g. tomatoes, mixed greens), for a mean price of 2.83 €/kg. SP sold in a long supply chain, which dilutes the value among more stakeholders. Nevertheless, SP's high-value vegetables (e.g. tomatoes, strawberries, lettuces) under a biodynamic label sold for a mean price of 2.20 €/kg. OP produced mainly less valuable vegetables (e.g. potatoes, cabbages, turnips, carrots) that were sold in a short supply chain for a mean price of 1.90 €/kg.

4.1.2.Marine eutrophication

For ME, the ranking of the farms depended on the FU, with OP having the lowest impact per ha but the highest impact per kg and €. NO_3 leaching, which is the main contributor to ME, was estimated using proportions of fertiliser and crop residue N (IPCC, 2019b) and ignoring N output. The farms' yields differed greatly, which suggests that their levels of crop N output did as well. The type of fertilisers also differed among the farms, because some mineralise faster (e.g. commercial fertiliser used by SP, poultry manure used by OP) than others (e.g. shredded green waste used by MF), which results in differing rates of N release in the soil. According to Qasim et al. (2021), incorporating straw into a greenhouse soil tended to reduce NO_3 leaching by stimulating denitrification. The Agence de l'eau Seine Normandie (2018) found low NO_3 leaching under vegetable crops grown with "market gardening on living soil" principles, as on MF. The high C:N ratio of the fertilisers used by MF enhances the activity of soil microbes and immobilises NO_3 (Kirchmann et al., 2002), and these fertilisers increase water retention (Zemánek, 2014), which decreases leaching. Furthermore, soil sequestration of N decreases NO_3 leaching (Knudsen et al., 2019) and depends on fertiliser properties. NO_3 leaching may also depend on whether fertiliser is applied in a greenhouse (i.e. a controlled water supply) or an open field (i.e. rainfall) (Koch and Salou, 2020). The IPCC Tier 1 emission factor we used to estimate NO_3 leaching is rudimentary and easy to apply in a farming-system LCA, but it seems too coarse given the variety of the farms' fertilisation strategies. Consequently, estimated ME impacts had high uncertainty, which calls for considering fertiliser properties and soil sequestration to improve estimates of NO_3 leaching.

4.1.3. Plastic use

SP used the most plastic, particularly to cover its 33 tunnels. MF also covered its tunnel in plastic, but used less, for two reasons: 1) only some of the cultivated land was under shelter, whereas all was under shelter on SP, and 2) the plastic lifetime was 8 years for MF and 4 for SP. The smaller tunnel area of MF allowed the farmer to repair plastic when damaged. In south-eastern France, where SP was located, plastic on small farms similar to MF had a lifetime of 6-7 years (Oriane Mertz, Agribio 84, pers. comm.); thus, the climate may influence this practice, along with the effect of the farming system. SP also used more plastic for mulching than MF and OP. On SP, all crops were mulched with single-use plastic, whereas on MF, straw mulch, manual weed control, and reusable plastic mulch were combined.

Plastic use is not an LCA indicator, and to our knowledge it has not been included before in an environmental assessment of vegetable production. In our study, it revealed major differences among systems. Plastic use in agriculture is a growing concern (United Nations Environment Programme, 2021b). Plastic mulch is a major source of microplastics (Bläsing and Amelung, 2018; Campanale et al., 2022) as it is thin and hard to remove from the soil (Qi et al., 2020). Microplastics may have detrimental effects on plant growth (Liu et al., 2021), soil properties (Zhang et al., 2020), and the fitness of soil bacteria and earthworms (Jiang et al., 2020), and can be found in fruit and vegetables at worrying concentrations (Oliveri Conti et al., 2020). An alternative to single-use plastic mulch that SP used the year after the survey is biodegradable plastic mulch, which is a common substitution approach (Hill and MacRae, 1995). Its benefits remain uncertain, as some studies conclude that it has no noxious effects on soil organisms (Sforzini et al., 2016), while others state that single-use and biodegradable plastic mulch have the same effects on earthworms (Ding et al., 2021; Kumar et al., 2020).

Plastic use included all types of items, from thin single-use items (e.g. mulch, drip tape) to long-lasting items (e.g. hard pipes). All types of plastic, regardless of their life span, can generate microplastics because the breakdown process starts on the surface. However, plastic used on the soil is more likely to be a source of soil microplastics (United Nations Environment Programme, 2021b). We included on-farm plastic but excluded up-stream plastic and products unintentionally contaminated with plastic (e.g. compost). Considering these sources of plastic would improve the indicator.

4.1.4. Biodiversity

Assessing biodiversity on the cultivated land alone or on the entire farm gave contrasting results, which highlighted the importance of a farm's semi-natural habitats for biodiversity (Chiron et al., 2010; Jeanneret et al., 2021; Rischen et al., 2021). On SP, the cultivated land yielded a low biodiversity score, which was offset by the high proportion of ruderal area (i.e. spaces between tunnels that are left to ruderal organisms). On OP, fields were generally surrounded by a ruderal strip or hedge. As its fields were large, its proportion of semi-natural habitat was lower, which yielded a lower biodiversity score at the whole-farm scale. On MF, the cultivated land yielded a biodiversity score similar to those of the other systems. Out of a maximum score of 45 in SALCA-BD for semi-natural habitats such as hedges, biodiversity-friendly managed grasslands and pastures can reach a score of 25 (Lüscher et al., 2017), which was the case for the SP grassland. Such scores are far higher than those of the vegetable fields studied here (3-8).

Consequently, for all farms, semi-natural habitats obviously contributed more to the biodiversity score than cultivated land. This result is in line with ecological studies that concluded that semi-natural habitats were important for spiders (e.g. Šálek et al., 2018), carabid beetles (e.g. Knapp and Řezáč, 2015), butterflies (e.g. Dover et al., 2000), birds (e.g. Billeter et al., 2007), and vascular plants (e.g. Billeter et al., 2007). The benefits of small farms for biodiversity are also acknowledged by Ricciardi et al. (2021), since the fields of smaller farms have a higher perimeter:area ratio than those of larger farms. Smaller farms are also more likely to create heterogeneous landscapes.

SALCA-BD analysed impacts of land-use type, farmer practices, and elements of spatial organisation of the farms. Other biodiversity assessment methods (Chaudhary and Brooks, 2018; Knudsen et al., 2017; Koellner and Scholz, 2008; Mueller et al., 2014) quantify impacts on biodiversity based on land-use classes and the distinction between organic and conventional farming. These methods are not adapted for assessing organic farms that have the same land use (arable land) but different farming practices.

4.1.5. Ranking and farm-specific effects

Considering the different impacts and FUs, a clear ranking of the farms did not emerge. OP had the lowest impacts, except for CED per € and for ME per kg and per €, and it was not best for biodiversity.

However, OP had a much lower yield than MF and SP (75% and 90% lower, respectively), which required more land to produce the same quantity of vegetables. Although the three farms are typical of the variety of such farming systems, farm-specific effects cannot be ignored. For example, MF used a diesel pump for irrigation, which contributed strongly to its CC impact. MF tried to limit input use, whereas some microfarms inspired by "bio-intensive" practices may use commercial fertilisers or plastic mulch intensively. On MF, the tunnel had large impacts, but some microfarms, particularly in southern France, do not use tunnels. On SP, ruderal areas between tunnels occupied a large proportion of the farm, but farms similar to SP use glasshouses or multispan greenhouses rather than tunnels, without inter-tunnel areas. Microfarms, often inspired by permaculture design methods, include semi-natural areas (e.g. hedges, ponds, woodland) (Morel et al., 2019) that would increase their biodiversity score. OP reduced its use of plastic close to zero, but some farms that grow vegetables on large open fields such as OP use plastic mulch or small plastic "caterpillar" tunnels.

According to the biotechnical index (Pépin et al., 2021), the level of agroecology of the farm was highest for MF (0.73), intermediate for OP (0.47), and lowest for SP (0.07). Per ha, SP had the highest CC and ME impacts, and the highest CED and plastic use, which is in line with its low biotechnical index, which corresponds to intensive use of inputs. However, MF (high biotechnical index) did not have lower impacts per ha than OP (intermediate biotechnical index). For MF, the tunnel and diesel, used mainly used for irrigating, contributed 75% of CC and 60% of CED, but tunnels and irrigation were not considered in the biotechnical index, which appears to be a methodological oversight. Because their total yields differed, the farms ranked differently when expressing impacts per kg and per € than per ha. The ranking per kg and per ha did not correlate with the biotechnical index, which ignores the yield.

Microfarming is often promoted as a solution to produce food with lower environmental impacts, but the LCA results in this case study suggest that this benefit is not obvious. However, microfarms may be a good compromise by having higher yields than large open-field farms and lower impacts per ha, and promoting biodiversity by having a high ratio of semi-natural habitats to cultivated land and diversified crops.

4.2. Advantages and disadvantages of the farming-system approach

The farming-system LCA approach was able to estimate several environmental impacts of complex farms by considering the system as a whole, without modelling every single crop. It also compared farms and identified hotspots (i.e. the main contributors to impacts). For example, in unheated greenhouse production, we found fertilisers (including compost), the greenhouse structure, and heated seedling production to be major contributors to CC, which confirms other results in the literature (Boulard et al., 2011; Cellura et al., 2012; Martinez-Blanco et al., 2011). The FUs identified differences in environmental impacts and eco-efficiency. Expressing impacts per kg and per unit of economic value are two ways to relate impacts to products. The mass-based FU considered production but introduced a bias when comparing farms that produced vegetables with different characteristics and value. In contrast, the value-based FU can compare any vegetables.

The farming-system approach followed the rationale of agroecology, in which inputs are farm-oriented rather than crop-oriented (e.g. fertilising the soil rather than the crop (Gliessman, 2021)). On MF, the goal of fertilisation was to have a fertile soil that was rich in organic matter and soil organisms. In a product LCA approach, MF's tunnel would be allocated to the vegetables grown in tunnels and not to those grown on open fields. However, it is difficult for a microfarm in this region to earn a sufficient turnover with only open fields, which means that the vegetables require a tunnel. On OP, rye production in the crop rotation had the main functions of producing rye and reducing pest and disease pressure. Because the latter function influences the (non-)use of inputs for vegetable production, it would make sense to include rye impacts in vegetable LCAs, with the challenge of allocating impacts between the two functions.

Some vegetables may have higher impacts than others due to specific needs (e.g. seedlings, fertiliser, water, pest control), lower yields, and/or longer cropping periods, but the farming-system approach cannot identify such "hotspot" vegetables. Identifying specific operations that have high impact requires detailed information about farmer practices. For example, knowing total diesel consumption does not provide information about how it was used for individual operations.

4.3. Comparison to similar studies

CC impacts of the farms studied are consistent with the few studies of similar systems (**Table 4Table *4***). CC impact of a small-scale organic farm in Washington, USA, (Adewale et al., 2016) was 1.7-2.7 t CO_2 eq./ha/yr and 45-623 g CO_2 eq./kg, depending on the vegetable. Irrigation contributed strongly to CED, as for MF and SP. The greenhouse contributed 7-10% to the CC impact of vegetables produced under shelter. When assessing a small and a large organic farm, Adewale et al. (2019) estimated a CC impact of 7.1 and 3.4 t CO_2 eq./ha/yr, respectively. For onion and winter squash, they estimated 188 and 276 g CO_2 eq./kg, respectively, for the small farm and 50 and 68 g CO_2 eq./kg, respectively, for the large farm. Christensen et al. (2018) studied community-supported vegetable farms in California, USA, and estimated CC impacts of 1.72-6.69 kg CO_2 eq./kg and 1.4-6.3 t CO_2 eq./ha/yr. These farms had very low yields (534-949 kg/ha/yr), which explains their very high CC impact per kg. Cellura et al. (2012) estimated a CC impact of 740 g CO_2 eq./kg for conventional tomatoes produced in an unheated tunnel in Italy. These impacts are higher than those for SP, partly because of a wider scope that included packaging and transport, and a shorter tunnel life span. He et al. (2016) estimated a CC impact for organic tomatoes in China of 208 g CO_2 eq./kg, and Martinez-Blanco et al. (2011) estimated 182 and 289 g CO_2 eq./kg for conventional tomatoes produced in tunnels and on open fields, respectively, both with compost and mineral fertilisers. Tomatoes produced in a heated greenhouse had CC impacts 10-50 times as high as those of vegetables produced in unheated tunnels, and heating and lighting contributed 97% of the impact (Williams et al., 2006). In open-field production in Oregon, USA, which is likely similar to OP, Venkat (2012) estimated a CC impact of 409 and 268 g CO_2 eq./kg for organic broccoli and lettuce, respectively. As a comparison to other organic crops, Nitschelm et al. (2021) estimated a mean CC impact for 106 cereals and legumes (e.g. spring and winter barley, spring and winter wheat, winter pea, fava bean) of 0.8 ±0.2 and 258 ±112 g CO_2 eq./kg, respectively.

Table 4. Literature results for climate change impact (100-year horizon) per ha during 1 year and per kg of vegetables

Type of farm	Vegetable	Country	t CO_2 eq./ha/yr	g CO_2 eq./kg	Greenhouse (GH)/open-field (OF)	Organic (Yes/No)	Source
Microfarm (MF)	Various	France	7.5	215	GH+ OF	Yes	Present study
Sheltered production (SP)	Various	France	13.3	198	GH	Yes	
Open-field production (OP)	Various	France	1.3	134	OF	Yes	
Small vegetable farm	Winter squash	USA	1.9	101	OF	Yes	Adewale et al. (2016)
	Potato		2.7	45	OF	Yes	
	Dry bush beans		1.7	623	OF	Yes	
	Chard		1.7	101	OF	Yes	
	Summer squash		2.1	62	OF	Yes	
	Peppers		2.6	65	OF	Yes	
	Onion		2.1	79	OF	Yes	
	Cauliflower		2.7	155	OF	Yes	
Small vegetable farm	Various	USA	7.1	-	OF	Yes	Adewale et al. (2019)
	Onion		-	188	OF	Yes	
	Winter squash		-	276	OF	Yes	
Large organic vegetable farm	Various	USA	3.4	-	OF	Yes	
	Onion		-	50	OF	Yes	
	Winter squash		-	68	OF	Yes	
Community-supported agriculture	Various	USA	2.9	3290	GH+ OF	Yes	Christensen et al. (2018)
	Various		1.3	1720	OF	Not certified	
	Various		6.4	6690	GH+ OF	Yes	
	Various		2.0	3720	GH+ OF	Not certified	
	Various		3.7	3980	GH+ OF	Not certified	
Mediterranean greenhouse	Tomato	Italy	-	740	GH	No	Cellura et al. (2012)
Mediterranean greenhouse	Tomato	Spain	-	182	GH	No	Martinez-Blanco et al. (2011)
Open field	Tomato	Spain	-	289	OF	No	
Heated greenhouse	Tomato	UK	-	9400	Heated GH	No	Williams et al. (2006)
Mediterranean greenhouse	Tomato	Morocco	-	220	GH	No	Payen et al. (2015)
Heated greenhouse	Tomato	France	-	1750	Heated GH	No	
Urban greenhouses	Tomato	China	-	208	GH	Yes	He et al. (2016)
Urban greenhouses	Tomato	China	-	261	GH	No	
National data base (practices considered typical)	Broccoli	USA	-	409	OF	Yes	Venkat (2012)
	Broccoli		-	353	OF	No	
	Lettuce		-	268	OF	Yes	
	Lettuce		-	192	OF	No	

4.4. The farm area and production area

In a product LCA, the area of the field or greenhouse is usually used for an area-based FU. In a farming-system LCA, the farm area is used, but farms often have uncultivated semi-natural areas. On microfarms, farmers may leave land uncultivated due to a lack of time or labour, or to enhance biodiversity and/or regulating ecosystem services (Morel et al., 2019). On farms with tunnels, areas between tunnels are rarely considered in yields or for an area-based FU. Microfarms and small farms that specialise in sheltered production have small areas; thus, uncultivated land may represent a much higher proportion of the area than that on a larger farm. The sensitivity analysis of farm area showed large differences in CC per ha and yielded a different ranking of the farms for biodiversity, depending on the area considered for MF. It is therefore important to establish clear rules for defining farm area, in particular when comparing farms of different types and sizes. Consequently, results with area-based FUs must be interpreted cautiously. It is reasonable to consider semi-natural areas as part of the farming system, as they provide regulating ecosystem services.

4.5. Methodological concerns when assessing organic vegetable farms

Estimates of NO_3 leaching influence the ME impact greatly. On agroecological farms that use organic fertiliser with high stability and slow mineralisation, the IPCC Tier 1 equation for NO_3 leaching (IPCC, 2019b) we used seems inappropriate, although it was easy to use. This calls for improving modelling of NO_3 leaching in a farming-system approach for systems that use organic fertilisers.

N added to the soil by crop residues was estimated using a generic coefficient for all vegetables, although the N content of residues differs among vegetables. N_2O emissions from crop residues and cover crops represented 5-12% of farm N_2O emissions and 1-2% of CC impacts. Crop residues and cover crops caused 18-28% of NO_3 leaching and 17-27% of ME impacts. These results suggest that improving estimates of crop residues is a priority for estimating ME impacts but not CC impacts, which agrees with Akkal-Corfini et al. (2021), who observed a large contribution of crop residues to NO_3 leaching from artichoke and cauliflower.

Biocontrol, particularly using macro-organisms against pests or for pollination, is commonly used in organic vegetable production. Producing them requires infrastructure, feed, heat, and rapid transportation, but to our knowledge, their impacts are not known or no data are available. Studies that mention the use of insects for pest control or pollination excluded their impacts (Cellura *et al.*, 2012). Estimating impacts of these insects would improve estimates of impacts of organic vegetable production, as the biocontrol market is growing rapidly.

5. Conclusion

Farming-system LCA assesses environmental impacts of farms that have different levels of agroecology, including complex systems with a wide variety of crops grown on small areas. The three FUs strongly influenced the ranking of the systems. Depending on the FU and the impact, each farm ranked first, second, or third. Per ha, differences in the CC impact and CED among the systems were large. SP had the highest impacts, whereas OP had the lowest impacts, which correlated with the intensity of input use. Per kg and per €, differences in the CC impact and CED among the systems were much smaller. OP had a lower CC impact and CED per kg, but not per €. OP used much less plastic, but had a lower biodiversity score and total yield. Despite its high total yield, SP did not perform well for CC impact, CED, or plastic use per kg or per €.

Our results show that the biotechnical index developed by Pépin et al. (2021) did not correlate with LCA-based estimates of environmental impacts. Including the use of greenhouses and irrigation infrastructure in the index would likely improve estimates of the CC impact and CED. Microfarms may be a good compromise, however, by having higher yields than large open-field farms and lower impacts per ha, and promoting biodiversity with a high ratio of semi-natural habitats to cultivated land and diversified crops. Although we selected farms that were typical of three farming systems, their potential farm-specific effects cannot be ignored. Farming-system LCA was able to assess farms using a relatively moderate amount data, and it can compare contrasting farming systems and identify hotspots within them.

The impact on on-farm biodiversity, which highlighted the importance of semi-natural habitats, contrasted with the other impacts. The quantification of plastic use echoes growing concerns about (micro-)plastic pollution in agricultural soils and landscapes, and the newly identified planetary boundary for novel entities. Estimating NO_3 leaching accurately was difficult, and IPCC Tier 1 modelling seemed inappropriate; consequently, estimates of the ME impact had high uncertainty. Because crop residues contributed greatly to this impact, models that estimate NO_3 leaching in a farming-system LCA need to be improved. In agroecological systems, semi-natural habitats are important for natural regulation, as they are part of the farming system. The farming-system LCA approach requires clear rules for setting farm boundaries, which strongly influence impacts per ha and biodiversity impacts.

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