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An environmental assessment of *Agaricus bisporus* ((J.E.Lange) Imbach) mushroom production systems across Europe

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A R T I C L E I N F O

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A B S T R A C T

Mushrooms have become a relevant part of our diet globally, as non-animal sources of proteins; but data on their value chain and environmental impact are still scarce. Therefore, a good understanding of the environmental impacts of mushroom production, the environmental hotspots throughout the value chain, comparisons between production systems and regions, and an assessment of the improvement potential of mushroom production is required. This paper carried out a life cycle assessment (LCA) to estimate the environmental impacts of three *Agaricus bisporus* mushroom production systems in three different European countries: Spain, Poland and Serbia. We found that there is a large variability in the composition of the substrates, which is in all cases a combination of compost (mainly straw and animal manure) topped by casing materials (mainly peat), and a large variability in energy use, substrate use and yields. Especially the Serbian organic dried mushroom case distinguishes from the other conventional fresh mushroom cases. This is also reflected in the life cycle impact assessment results. The composting processes resulted in the largest contribution to environmental impact (about 49.6% on average ranging between 16.4% and 84.4% across all impacts assessed), followed by the electricity production and the casing (respectively 20.3% and 10.3% on average across all systems and impact categories analysed). Thus, optimizing composting and casing production together with switching to renewable energy sources appeared to be the most effective to reduce the overall environmental impacts of mushroom production. This paper provided a comprehensive assessment across Europe which could be further expanded to have a broader and more representative overview of the impact of mushroom production at European level.

1. Introduction

Global production of mushroom has increased 30-fold between 1978 and 2016 (Royse et al., 2017), reaching more than 44 million metric tonnes in 2021 (FAOSTAT, 2022; Robinson et al., 2019). China, the largest mushroom producer, produced over 40 million t of mushrooms, while the European Union produced more than 1 million t in 2021 (FAOSTAT, 2022). The global average consumption is about 100 g per capita per week (Royse et al., 2017), making mushrooms a relevant part of consumers' diet and an important source of non-animal protein.

Agaricus bisporus (J.E.Lange) Imbach is a mushroom belonging to the Agaricaceae family, division Basidiomycota, in the Fungi kingdom, which is one of the most cultivated mushroom species worldwide (Leiva et al., 2016; Robinson et al., 2019). This species has several common names including white mushroom, button mushroom or champignon mushroom. Throughout this paper, the term mushroom will be used instead.

This mushroom is used, besides medicinal and cosmetic purposes, mostly for human consumption (Usman et al., 2021). It is highly nutritious, as it is a source of carbohydrates and proteins, while having a

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limited fat content (Krishnamoorthi et al., 2022; Leiva et al., 2016). The mushroom has a low energy level due to its high moisture content and contains certain important elements such as potassium and phosphorus (Leiva et al., 2016), and is an important source of selenium, a micronutrient lacking in many diets (Prange et al., 2019). Mushrooms also have high concentrations of flavonoids, saponins, tannins and vitamin C (Krishnamoorthi et al., 2022). Further, they have antimicrobial properties against several human pathogens, including *Staphylococcus aureus* and *Candida albicans* (Krishnamoorthi et al., 2022).

Mushrooms are cultivated around the world under controlled environmental conditions where energy, water, peat, compost and other materials are used (Leiva et al., 2016, 2015; Robinson et al., 2019). Differently from other horticultural crops, mushrooms grow in darkness, using a substrate consisting of two layers: a specially formulated compost layer topped with a casing, normally peat-based, layer (Leiva et al., 2016, 2015; Robinson et al., 2019). As secondary decomposer, the button mushroom requires the action of other organisms during a process of composting which matures the substrate and is later colonized by the crop mycelium (Grimm and Wösten, 2018). After substrate preparation, the partially decomposed selective substrate will present low soluble sugar content, to avoid the growth of competitive/parasitic bacteria and moulds. The compost is generally made up of chicken manure and straw and/or horse manure, depending on the availability of the latter (Gruda, 2019; Leiva et al., 2016; Robinson et al., 2019). The covering layer, commonly called casing layer, has several functions including keeping the mushroom crop continuously hydrated and facilitating the transport of dissolved nutrients to the carpophores (Pardo-Giménez et al., 2017). Although different materials have been employed as casing, the most commonly used casing layer is composed of peat amended with chalk or limestone to buffer the pH slightly alkaline (Gunady et al., 2012; Leiva et al., 2016; Robinson et al., 2019). Depending on the requirements for mushroom production the selection of the casing material for instance may influence the number of mushrooms harvested positively (regarding total density and porosity) or negatively (regarding the water holding capacity) (Zied et al., 2014). The mushroom life cycle starts with spore germination which leads to monokaryotic hyphae growth mating for the formation of an extensive dikaryon and fertile mycelium network (Choi et al., 2023). Commercially, the compost/selective substrate is inoculated with commercial spawn (cereal grains covered by mycelium) which colonizes the compost and the casing, eventually producing the fruiting bodies from mushroom primordia, which ripen, sporulate, and then go to senescence (Carrasco et al., 2021). The post-harvest period is paramount for the mushroom's commercial quality (Leiva et al., 2015). Mushrooms are consumed fresh, dried or canned. Fresh mushrooms have a short shelf-life, therefore mushrooms can only be stored for a short period, depending on the conditions and treatments (Zhang et al., 2018).

Mushrooms take in between 37–46% of their water from the casing which is regularly watered (Herman and Bleichrodt, 2022). Therefore depending on the destination for the mushroom production, either fresh product (where a higher water content can maximise profit) or dried product (where a lower content can reduce costs of processing), moisture content can be optimised. There has been concerns of the environmental impact of mushroom production both at European and at global level (Gunady et al., 2012; Leiva et al., 2015; Robinson et al., 2019), because of its energy intensive process, peat use, composting, water and pesticide consumption. Certain challenges are now affecting the negatively the industry including the reduced quality and availability of the raw materials together with increasing prices of raw materials and energy costs, narrow margin of benefits or even limited accessibility to work labour in rural areas.

Life cycle assessment (LCA) has been used to assess the environmental impact of mushroom production in California, U.S.A. (Robinson et al., 2019), La Rioja, Spain (Leiva et al., 2015), and Australia (Gunady et al., 2012). With exception of Robinson et al. (2019), carbon dioxide emissions related to the use of peatland were not accounted for in the other previous studies (Gunady et al., 2012; Leiva et al., 2015). Considering the limited number of studies published on mushroom production, deep knowledge of the environmental impacts, insights into environmental hotspots and tools with improvement potential for different mushroom production systems are scarce. In this study, the environmental impact of two conventional mushroom production systems in Spain and Poland and one organic mushroom production system in Serbia are assessed. The goal was i) to assess their environmental performance with a cradle to cultivation gate approach; ii) to identify the contribution of each process involved in the mushroom production to the environmental impacts; iii) to identify potential improvements of the environmental performance of mushroom production.

2. Material and methods

2.1. System description, functional unit, allocation, and impact assessment

The mushroom production systems analysed included an average production system for the La Rioja region in northern Spain, a small-scale organic farm in Serbia close to Belgrade and a conventional farm in South-Eastern Poland. In the Spanish and Polish cases, the mushrooms were sold fresh, while most of the mushrooms in the Serbian case were sold dried. These were selected as the Polish and the Spanish systems have quite distinct value chains and locations. Further, both countries are major producers, representing 26% and 12% of the overall European market in 2020 (GEPC, 2024). Instead, the Serbian system was analysed because of its different geographical location, final product and market (organic dry mushroom). The LCA included processes from cradle to cultivation gate. Thus, the system included the production and consumption of inputs and energy, including fossil fuel and crude oil production, electricity production, peat extraction, transport and manufacturing, compost processing and transport, spawn production, and mushroom cultivation, as it was carried out in previous research (Gunady et al., 2012; Robinson et al., 2019) (Fig. 1).

Two functional units were used to account for the productivity of the mushroom cultivation: 1 kg of freshly harvested mushrooms and 1 € of mushroom output to consider the economic function of mushroom cultivation, in agreement with LCA methodology focused on agricultural systems (Goglio et al., 2017; Nemecek et al., 2011; Ponsioen and van der Werf, 2017).

Economic allocation was applied to the data used for inputs, such as grains for spawn production and straw for compost production. Manure used to produce compost was treated as a waste, thus no upstream environmental impact was allocated at the animal farm gate to manure production (ISO, 2006a, 2006b).

2.2. Primary data collection

Primary data were collected for the foreground processes: mushroom production (including room cleaning), compost production, and casings production from mushroom producers in Spain, Poland and Serbia. Greenhouse gas (GHG) emission rates are based on the IPCC Guidelines (Gavrilova et al., 2019; Hergoualc'h et al., 2019) (2.3.5). The data survey consisted of questions on the on-site yield and use of inputs. The LCIs generated based on the data collected for this study were presented per kg of harvested mushroom product (one of the functional units used in this study) (Table 1). In each case, the mushroom growing period was 30 days. Each growing period consists of two or three flushes. After the final flush the room is cleaned (Leiva et al., 2015; Robinson et al., 2019).

Traditionally, the colonization of the selective composted material (phase II compost) takes place in the growing facilities before adding the casings layer onto the fully colonized compost (phase III), as it is done in the Serbian case. However, producing phase III compost at the mushroom facilities of the compost makers allows to scale up to 10 cycles per year of mushroom production in the growing chambers while reducing

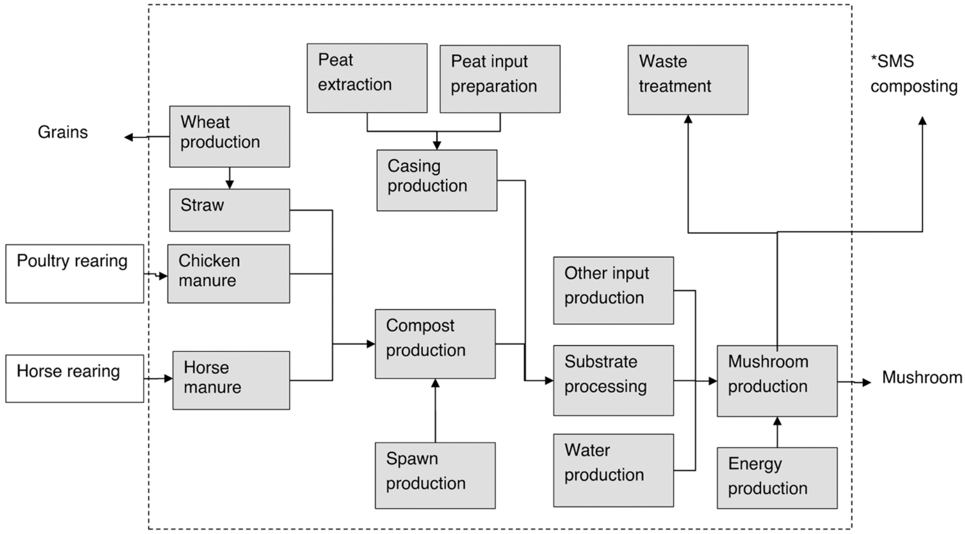

Fig. 1. Flow chart representing the system boundary (dashed line) and the processes included in this LCA. In the Spanish case, the compost consists of chicken manure and straw; in the Serbian case, the compost consists of horse manure; in the Polish case, the compost consists of horse and chicken manure and straw. *SMS: Spent Mushroom Substrate (agricultural by-product generated after mushroom cultivation).

Table 1
Life cycle inventory per kg of harvested mushrooms.

Input/output	Unit (kg of harvested mushroom^{-1})	Spain	Poland	Serbia	Source
Casing	Kg	0.62	0.29	1.8	primary data
Casings - black peat (moisture 65%)	Kg	0.11	0.29	1.8	primary data
Casings - blond peat (moisture 65%)	Kg	0.27	0	0	primary data
Casings - lime	Kg	0.09	0	0	primary data
Casings - water	Kg	0.14	0	0	primary data
Casings - High Density PolyEthylene (HDPE)	G	0.03	0	0	primary data
Compost	Kg	2.62	2.63	4.50	primary data
Compost - chicken manure (moisture 90%)	Kg	0.38	1.08	0	primary data
Compost - horse manure (moisture 30%)	Kg	0	0.15	4.43	primary data
Compost - wheat straw (tel quel weight)	Kg	0.48	1.22	0	primary data
Compost- water	Kg	1.7	0.15	0	primary data
Compost- gypsum	G	45.8	0	75.0	primary data
Compost - Champfood	G	0	34.6	0	primary data
Compost - mycelium spawn	G	26.2	26.3	36.0	primary data
Compost - spawn - electricity	MJ	0.024	0.024	0.033	Leiva et al. (2015)
Compost - spawn - diesel	MJ	0.090	0.090	0.124	Leiva et al. (2015)
Compost - spawn - rye grains	Kg	0.017	0.017	0.023	Leiva et al. (2015)
Compost- electricity	kWh	0.17	0.00	0.41	primary data
Compost - diesel	MJ	81.4	0	12.0	primary data
Compost - N_2O emissions	G	0.33	0.34	0.42	Calculated (Gavrilova et al., 2019)
Compost - Ammonia emissions	G	8.6	15.6	20.3	Calculated (Gavrilova et al., 2019)
Compost - Nitrate emissions	G	4.2	5.4	6.8	Calculated (Gavrilova et al., 2019)
Water	Kg	14.6	27.8	3.1	primary data
Electricity	kWh	0.20	0.36	0.38	primary data
Heat from wood	kWh	0.90	0	0	primary data
Heat from diesel	kWh	0.90	0	0	primary data
Heat from natural gas	MJ	0	0	0.58	primary data
Prochloraz-Mn (Sporgon ®)	g (a.i.)	15	23	0	primary data
Metrafenone (Vivando ®)	g (a.i.)	31	0	0	primary data
Room area	cm^2	3.62	3.65	3.65	primary data
CO_2 emissions (peat)	G	29	226	98	Calculated (Blain et al., 2007; Lovelock et al., 2019;Ogle et al., 2019b)
N_2O emissions	G	1.66	3.0	2.3	calculated (Hergoualc'h et al., 2019)
Ammonia emissions	G	27.0	43.8	45.0	calculated (Hergoualc'h et al., 2019)
Nitrate emissions	G	26.6	49.6	38.4	calculated (Hergoualc'h et al., 2019

the mushroom cycle by up to 14 days per cycle (Carrasco and Preston, 2020). Thus, this is more often applied in modern large conventional production facilities (Polish case). For calculating the environmental impact per Euro, 5-year average prices were used (2017–2021): in the Polish case 1.07 € per kg at 13.5% dry matter content; in the Spanish case, 1.50 € per kg at 13.5% dry matter content; in the Serbian case, 50.00 € per kg at 95.0% dry matter content. The dry matter content at harvest was estimated at 10.0% for the Polish and Spanish cases, where the mushrooms are sold fresh and 13.5% for the Serbian case, where the mushrooms were dried. For the Serbian case, the drying of the mushroom was excluded from the assessment. These data were collected from the mushroom growers in the three countries.

2.3. Data processing and assumptions

2.3.1. Reference life cycle inventories and allocation procedures

Data were processed using SimaPro 9.5 (Simapro 9.5, 2023). Background data were collected from SimaPro LCI databases Ecoinvent 3.9 (Wernet et al., 2016), Agri-footprint 6 (Blonk, 2021), and World Food Lifecycle Data Base (WFLDB) 3.5 (Quantis, 2020). Data for spawn production were taken from a previous mushroom assessment and adapted for the systems analysed here (Leiva et al., 2015). Agricultural inputs were sourced from Agri-footprint and other inputs from Ecoinvent 3.9 (Wernet et al., 2016). Where available, LCIs based on Spanish, Polish and Serbian conditions were selected.

Some inputs to mushroom production do not have existing LCI datasets: metrafenone (Vivando®) and prochloraz-Mn (Sporgon®) used as fungicides; quaternary ammonium used as disinfectant, and nutritional supplement for mushroom growth (Champfood®). For Champfood®, the soybean meal from the Netherlands dataset in the Agri-footprint database was used as proxy. For all the pesticides used during mushroom production, as data were not available for the specific active ingredient, data from similar chemical compounds or from the same pesticide class were used, in agreement with previous LCA research (Goglio et al., 2012, 2018).

Several datasets of LCI databases were adapted to make them more specific for our case study. For Champfood®, the soybean meal from the Netherlands dataset in the Agri-footprint database was used as proxy. For quaternary ammonium, the global market for non-ionic surfactant dataset from the ecoinvent database was used (as a proxy for cationic surfactants), as carried out in similar manner in previous research (Goglio et al., 2022; Helmes et al., 2022). The dataset in WFLDB for *Agaricus bisporus* spawn production (commercial mycelium in grain cereals) in the Netherlands was adapted by replacing the rye (*Secale cereale* L.) grains and electricity by country specific data.

The dataset in Agri-footprint 6 containing the co-products wheat grain and wheat straw was adapted by replacing the allocation factors with more specific price information for Spain in the period of 2018–2021 (€ 0.200 per kg of wheat grains, based on MAPA, 2022; and € 0.043 per kg of wheat straw, based on information from the mushroom producers in Spain). The dataset in Agri-footprint 6 containing the co-products wheat grain and wheat straw was adapted by replacing the allocation factors with more specific price information for Poland in the period of 2018–2021 (PLN 0.78 per kg of wheat grains, based on Statistics Poland, 2023 (Statistics Poland, 2023); and PLN 0.24 per kg of wheat straw, based on information from the mushroom producers in Poland). The dataset from ecoinvent for peat production in the NORDEL countries (Denmark, Finland, Iceland, Norway and Sweden) was adapted by adding peat oxidation emissions (see Section 2.3.3), and by replacing the electricity mixes by Serbian and Polish electricity for the case studies in these countries. The dataset from ecoinvent for tap water in Europe and underlying datasets for conventional treatment and direct filtration treatment were adapted for the three case study countries by replacing the electricity grid mix and water flows, as carried out in previous research (Goglio et al., 2022; Helmes et al., 2022).

2.3.2. Transport

The average transport distances for material inputs are based on the material manufacturer and location information reported by the mushroom growers, as previously carried out in agricultural LCA (Goglio et al., 2018; Leiva et al., 2015). For all materials, the transport distances from manufacturer to facility for each material type is specified (Table 2). Transport within facilities was accounted for using reported annual fuel use data from the mushroom growers. Combustion emissions are accounted for in the relevant selected datasets from the ecoinvent database for fossil fuel use (Wernet et al., 2016).

Table 2
Transport distances per input (km). Empty cells indicate that the product was not used in the analysed systems, therefore no transport distance was assumed.

Input/output	Spain	Poland	Serbia
Manure	40	0	0
Compost	0	30	30
Gypsum	80	80	
Urea	500		
Wheat straw	200		
Black peat road transport	281	300	300
Black peat sea transport	1010		
White peat road transport	281		0
White peat sea transport	1010		
Lime	80		
HDPE plastic	400		
Pesticides	10		
Casings	1450	30	
Prochloraz-Mn	20	20	
Metrafenone	20		
Deltametrin	20		

2.3.3. Casing production and peat emissions

For peat production, the peat production dataset of ecoinvent was adapted by adding peat oxidation emissions following IPCC methodology (Blain et al., 2007; Lovelock et al., 2019). Carbon dioxide emissions per kg of peat were determined using the default carbon dioxide and nitrous oxide emission factors for extraction from boreal and temperate nutrient-rich peatland and taking into account the land occupation and dry matter content of the ecoinvent peat data. For Poland and Serbia, the dataset was also modified for local production (replacing the electricity mix).

2.3.4. Compost related GHG emissions

Compost may be produced on site at the mushroom growing facility or may be produced at a compost-only facility and transported to mushroom growers. In the Spanish case base materials are composted near the mushroom production site. In the Polish and Serbian cases, the case materials were composted elsewhere.

Compost emissions are released during the composting process, and these emissions include methane (CH_4), nitrous oxide (N_2O), ammonia (NH_3), and nitrate (NO_3^-). Composting emissions can vary based on composting practices (e.g., aeration and turning) and conditions (e.g., temperature, moisture content). We applied the IPCC (2019) emission factors with the emission and conversion factors shown in Table 3, by multiplying the amount of dry matter or nitrogen in the used manure by the relevant factors per type of emission.

2.3.5. Mushroom production and related GHG emissions

Peat is a carbon-rich material used as casing in mushroom production. Peat forms in wetlands where plant matter accumulates, rather than degrading, and captures and stores carbon for thousands of years (Blain et al., 2007). The IPCC Guidelines assume all peat is oxidised in 1 year (Blain et al., 2007; Lovelock et al., 2019), thus the total peat oxidation emissions were therefore divided by the number of mushroom production cycles per year.

Nitrous oxide and ammonia emissions from the substrate are based on IPCC (2019), where we only take direct N_2O emissions into account (0.01 kg N_2O-N kg^{-1} N in the substrate). We applied an average of 0.11 kg NH_3-N kg^{-1} synthetic N fertilizer and 0.21 kg NH_3-N kg^{-1} organic N fertilizer. Leaching of nitrate is assumed the same as for composting. The calculated emissions for mushroom production following these guidelines are per year, thus they are divided by the number of cycles per year (Ogle et al., 2019a).

Table 3
Parameters for calculating composting and mushroom production emissions.

Parameter	Unit	Value	Reference
Maximum methane producing capacity B0 of chicken manure	m^3 CH_4 kg^{-1}	0.37	based on layers (Gavrilova et al., 2019)
Maximum methane producing capacity B0 of horse manure	m^3 CH_4 kg^{-1}	0.30	default for horse (Gavrilova et al., 2019)
Effective methane conversion factor MCF of compost	kg kg^{-1}	0.02	temperate climate, static pile or passive windrow (Gavrilova et al., 2019)
Fraction of compost N leached	kg NO_3-N kg^{-1} N	0.06	static pile (Gavrilova et al., 2019)
Fraction of compost N volatilized	kg NH_3-N kg^{-1} N	0.65	poultry, static pile (Gavrilova et al., 2019)
Fraction of compost N as N_2O	kg N_2O-N kg^{-1} N	0.01	static pile (Gavrilova et al., 2019)
Fraction of substrate N leached	kg NO_3-N kg^{-1} N	0.06	assumed same as composting (Gavrilova et al., 2019)
Fraction of substrate N volatilized	kg NH_3-N kg^{-1} N	0.15	average of synthetic and organic fertilizer) (De Klein et al., 2006)
Fraction of substrate N as N_2O	kg N_2O-N kg^{-1} N	0.01	(Hergoualc'h et al., 2019)
Carbon fraction in peat	kg C kg^{-1} dry weight	0.45	(Blain et al., 2007; Lovelock et al., 2019)

2.3.6. Waste treatment

Spent mushroom substrate (agricultural by-product generated after mushroom cultivation) was removed from the facilities to be processed and used for other purposes than mushroom production. The energy consumption for spent mushroom substrate removal until the gate of the mushroom production facility was included. All other environmental impact from transport and processing were deemed to be attributed to the further uses (i.e. horticultural crops). The impact of plastic packaging for the mushroom growing substrate and waste treatment was included using ecoinvent data for plastic waste treatment in the producing countries (Leiva et al., 2015; Robinson et al., 2019).

2.4. Life Cycle Impact assessment

Life cycle impact assessment (LCIA) converts the life cycle inventory (LCI) emissions, resource extractions and land use into environmental impact categories. Six relevant impact categories, which are often reported for mushrooms, were selected in this research: climate change with a 100 year horizon, freshwater and marine eutrophication, acidification, abiotic resource depletion-fossil fuels and water scarcity. For climate change, we applied the most recent Global Warming Potential (GWP) 100 characterization factors of the Intergovernmental Panel for Climate Change (IPCC) (Forster et al., 2021). For freshwater and marine eutrophication, and for acidification, we applied the ReCiPe 2016 method (Huijbregts et al., 2016). For fossil resource energy carrier use we applied the abiotic resource depletion – fossil fuels (ADP-fossil) method of CML 2002 (Guinée et al., 2002; Van Oers et al., 2002). For water scarcity, we applied the user deprivation potential method (deprivation weighted water consumption) Available WAter REmaining (AWARE) of Boulay et al. (2018). The fossil resource use and water scarcity, indicators are used as implemented in the Environmental Footprint 3.0 Method (Fazio et al., 2018). The selected LCIA indicators were used as implemented in SimaPro 9.5 (Simapro 9.5, 2023).

2.5. Contribution and sensitivity analysis

A contribution analysis was carried out to identify environmental hotspots which contribute to the environmental impacts assessed in this research, in agreement with the ISO standards (ISO, 2006b, 2006a). On the basis of the contribution analysis results and previous research (Leiva et al., 2016, 2015; Robinson et al., 2019), a series of sensitivity scenarios were conceived to assess how results are affected by changes in parameters with regards to the baseline systems (BS). In the YIELD scenario, an increase in yield of 10% was considered for all the scenarios. In the PHO scenarios, all the electricity consumed during mushroom production, including casing and compost production, was assumed to be produced out of photovoltaic panels within the farm premises.

Due to the large amount of peat consumed during mushroom cultivation and the importance of peat with regards to climate change (Blain et al., 2007; IPCC, 2022; Paustian et al., 2016; Robinson et al., 2019), in the PEAT scenarios a 10% reduction in peat consumption was considered. Previous research also pointed out how the composting process can have a large environmental impact contribution (Gunady et al., 2012; Leiva et al., 2015; Robinson et al., 2019), thus in the COMP scenarios a 10% reduction in compost used for all the three systems was assumed. Following the ISO standard (ISO, 2006b, 2006a) and considering the large amount of straw used in some systems, a STRAW scenario was conceived where a physical allocation was carried out using dry biomass weights of straw and grain instead of economic allocation.

3. Results

3.1. Environmental impacts per mass of harvested products

Fig. 2 shows the absolute environmental impact results and contribution of important sources of environmental impact for the three cases in Spain, Poland and Serbia. The organic production in Serbia resulted in the largest impact on climate change (1.55 kg of CO_2eq kg^{-1} of harvested mushrooms). However, it must be noted that the dry matter content of these mushrooms is higher (approximately 13.5%) than in the other cases (approximately 10%), where the mushrooms are sold fresh, while in Serbia the organic mushrooms are sold dried. Taking this into account, the Polish system has a similar impact on climate change (0.931 kg CO_2eq kg^{-1} of harvested mushrooms) and the Spanish system has a lower impact (0.521 kg CO_2eq kg^{-1} of harvested mushrooms). Freshwater eutrophication was the largest in the Serbian system (2.34 g of Peq kg^{-1} of harvest mushrooms, while the Polish and the Spanish systems had much lower potential (70.3% and 91.3% less than the Serbian system, respectively, based on the harvested mass) even when taking the differences in dry matter content at harvest into account (Fig. 2).

On the contrary, for marine eutrophication, the Polish system had highest potential (0.966 g of N eq kg^{-1} of harvested mushrooms), while the Serbian system had 20.0% lower impact and the Spanish system 59.7% less marine eutrophication, based on harvested mass. The Serbian system had the largest resource use (29.9 MJ kg^{-1} of harvested mushrooms), while the Polish system had 68.8% lower value and the Spanish system 69.6% lower potential than the Serbian system. The impact on terrestrial acidification was also the largest in Serbia (25.5 g SO_2 kg^{-1} of harvested mushrooms), followed by Polish system with 24.3% less impact and the Spanish systems with 78.7% lower impact value. The Spanish system resulted in the largest water use impact (1.71 m^3 deprivation kg^{-1} of harvested mushrooms) while the Serbian and Polish case had lower impacts (88.1% and 92.1% less respectively) (Fig. 2).

The contributions of the different sources per kg of harvested mushrooms to the environmental impacts analysed here are shown in Table 4. The largest contributor to the impact categories analysed are compost production (about 49.6% on average and ranging between 16.4% and 84.4% across all the systems and environmental impacts, though less for fossil resource and water use), electricity (about 20.3% on average and ranging between 0.5% and 61.8% across all the systems and environmental impacts) and casings (about 10.3% on average and ranging between 0.1% and 61.0% across all the systems and environmental impacts). The main source of impact to climate change for all the systems here analysed was compost production (46.1%–49.5%), mainly because of the use of wheat straw, which carries some environmental

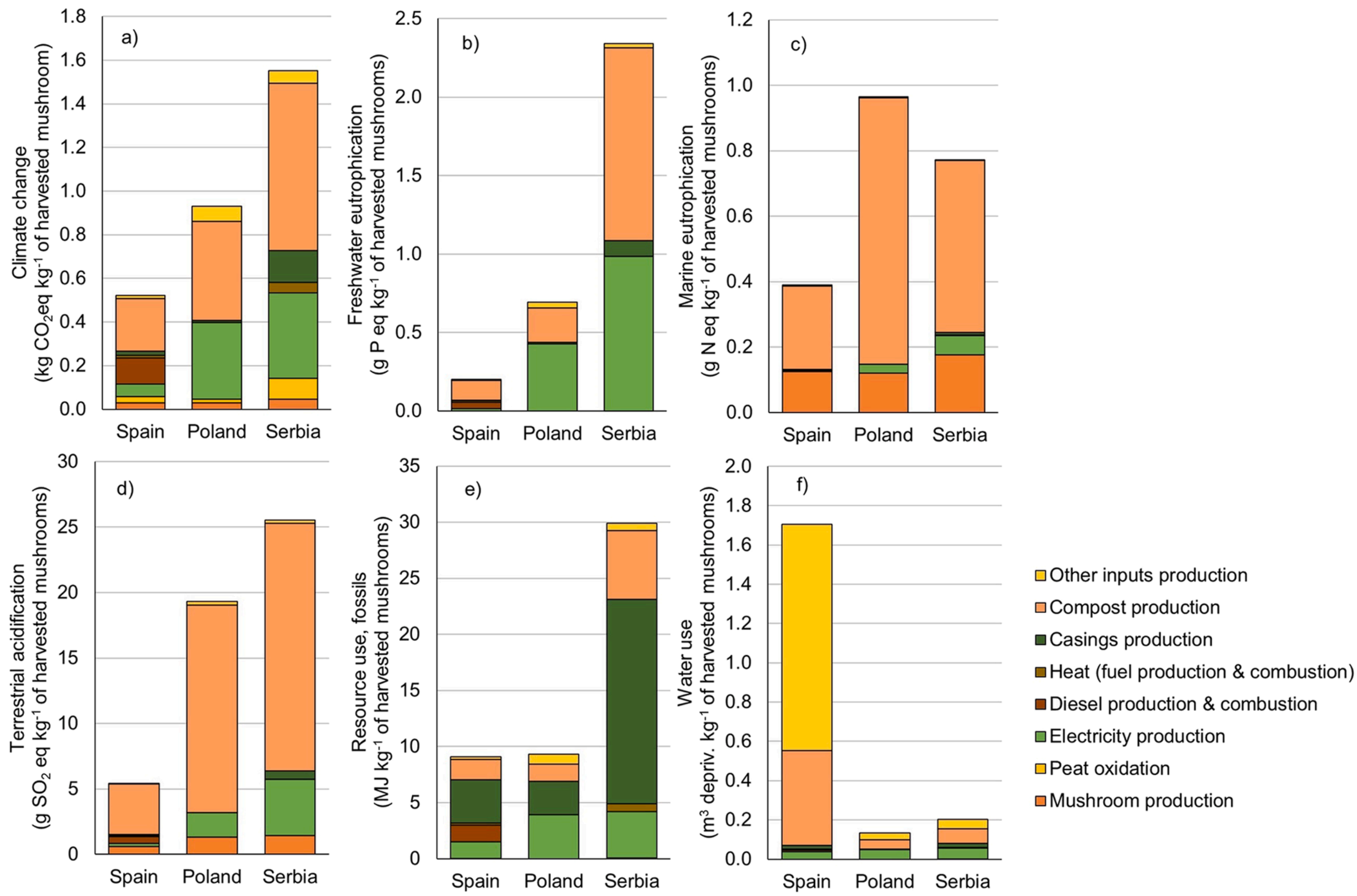

Fig. 2. Impact assessment results for climate change (kg carbon dioxide equivalents), freshwater eutrophication (kg phosphorus equivalents) and marine eutrophication (kg nitrogen equivalents), terrestrial acidification (kg sulphur dioxide equivalents), resource use, fossil (MJ) and water use (cubic metre water deprivation) per functional unit (kg harvested mushroom) (Peat oxidation includes only carbon dioxide emissions due to peat oxidation from mushroom production; Mushroom production includes only on-site emissions from the substrate other than peat oxidation).

burden from wheat grain cultivation (about 8%, based on economic allocation) and electricity consumption for composting (Fig. 2). Electricity use in mushroom cultivation had the largest share of impact on climate change in the Polish case (37.9%), while in the Serbian case, the absolute contribution of electricity use for mushroom cultivation had a slightly lower contribution (25.2%), followed by Spain (10.3%) (Table 4).

The eutrophication and acidification impacts were mainly caused by compost production (electricity) (52.5 to 84.4%, except for freshwater eutrophication in Poland with 31.9%), electricity production (varying between 0.5% and 61.8%), and direct emissions from mushroom production (12.5–32.1% to marine eutrophication; 5.7–11.3% to acidification) (Table 4). Fossil resource use is dominated by casings production (31.9–61.0%), which mainly consist of energy use and to a lesser extent by peat extraction. Water use is dominated by electricity production (2.2–36.5%) and other inputs (24.2–67.6%; the other inputs category included water input in the mushroom production process). There was a much higher impact on water use in the Spanish case, due to the much higher water scarcity (Boulay et al., 2018). However, in the Serbian and Polish systems, compost production resulted in the largest water use contribution (35.9% and 35.4%, respectively) (Table 4).

3.2. Environmental impacts per monetary unit

The impacts per monetary unit (€) per harvested product make a comparison between the three different cases possible in a different perspective than per mass unit (kg). With the monetary functional unit, the economic function of producing income for the mushroom grower was encompassed, as suggested for LCA of agricultural systems (Goglio et al., 2017; Nemecek et al., 2011). The Polish system resulted in the largest impact per € of harvested mushrooms on climate change (0.838 kg of CO_2eq €$^{-1}$ of harvested mushroom), freshwater eutrophication (0.625 g of Peq €$^{-1}$ of harvested mushrooms), marine eutrophication (0.869 g N eq €$^{-1}$ of harvested mushrooms), terrestrial acidification (17.4 g SO_2eq €$^{-1}$ of harvested mushrooms) (Fig. 3). For climate change, marine eutrophication, terrestrial acidification, the Spanish system resulted in the second largest impact with 21.3–60.4% lower impacts than the Polish system except for water use and fossil resource use. The Spanish system had also the lowest freshwater eutrophication impact per € of harvested product (0.257 kg of Peq €$^{-1}$ of harvested mushrooms). On the other hand, the Serbian system had the lowest climate change impact, marine eutrophication, terrestrial acidification, resource use, fossil fuel, and water use (with a 49.8–98.7% lower value than the Polish case) (Fig. 3).

For the resource use fossil fuels and water use impact category, the highest impact per Euro of harvested mushrooms belonged to the Spanish case (11.5 GJ €$^{-1}$ of harvested mushrooms, 2.16 m^3 of water deprivation €$^{-1}$ of harvested mushrooms), while the Polish and the Serbian case had a smaller impact (>94.4% lower impact for water use and >27.0% less resource use fossil fuels) (Fig. 3). This is because the organically cultivated mushrooms in Serbia have an about six times higher price (0.79, 1.11, and 7.11 € kg^{-1} harvested mushrooms for the Spanish case corrected at 10% dry mass, Polish case corrected at 10% dry mass and Serbian case corrected at 13.5% dry mass, respectively).

Table 4
Contribution of environmental impact results in percentages (%) to the total climate change, freshwater eutrophication, marine eutrophication, terrestrial acidification, resource use, and water use impacts for each of the three systems (Spanish, Polish, Serbian).

Source	Climate change			Freshwater eutrophication			Marine eutrophication			Terrestrial acidification			Resource use, fossils			Water use		
	Spain	Poland	Serbia	Spain	Poland	Serbia	Spain	Poland	Serbia	Spain	Poland	Serbia	Spain	Poland	Serbia	Spain	Poland	Serbia
Mushroom production emissions[a]	5.8	3.1	2.9	< 0.1	< 0.1	< 0.1	32.1	12.5	22.9	11.3	6.9	5.7	< 0.1	< 0.1	0.2	< 0.1	< 0.1	< 0.1
Peat oxidation[a]	5.5	1.8	6.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Electricity production[b]	10.7	37.9	25.2	7.9	61.8	42.0	0.5	2.7	7.7	4.2	9.5	16.7	16.4	42.4	13.8	2.2	36.5	27.9
Diesel production and combustion[b]	23.0	< 0.1	< 0.1	20.1	< 0.1	< 0.1	0.9	< 0.1	< 0.1	9.6	< 0.1	< 0.1	16.3	< 0.1	< 0.1	0.6	< 0.1	< 0.1
Heat (fuel production & combustion)[b]	2.6	< 0.1	3.2	4.1	< 0.1	0.1	0.3	< 0.1	< 0.1	1.6	< 0.1	0.1	2.4	< 0.1	2.4	0.2	< 0.1	0.7
Casings production	3.7	0.9	9.4	1.9	1.0	4.2	0.1	< 0.1	1.0	1.3	0.2	2.4	42.3	31.9	61.0	1.1	0.9	11.3
Compost production	46.1	48.7	49.5	62.0	31.9	52.5	65.4	84.4	68.1	71.0	82.0	74.1	20.1	16.4	20.4	28.3	35.4	35.9
Other inputs production	2.6	7.5	3.6	3.9	5.3	1.1	0.8	0.3	0.2	1.0	1.4	1.0	2.4	9.3	2.2	67.6	27.3	24.2

[a]during cultivation
[b]used during mushroom cultivation

3.3. Sensitivity analysis

The results of the sensitivity analyses are shown in Fig. 4. Among the sensitivity scenario assessed, PHO scenarios resulted in the largest reduction of the environmental impact analysed (24.2% on average ranging between 0.3% and 86.7% across the analysed impact categories), followed by the YIELD scenarios (9.1% for all the analysed impact categories). The COMPOST scenarios had lower changes (4.9% on average ranging between 1.7% and 8.4% across impact categories) and the PEAT scenario also had lower changes (2.9% on average ranging between 0.0% and 29.6% across impact categories). The STRAW scenarios had on average a 39.9% larger impact ranging between 12.2% and 125.2% in comparison to the baseline (Fig. 4).

The PHO scenarios showed larger impact decrease on climate change in Serbian case (49.5% reduction) and the Polish case (35.8% reduction), on freshwater eutrophication (60.3% and 86.7% reduction in the Polish and Serbian cases, respectively). In Spain, using photovoltaic electricity for mushroom production was responsible only to 8.9% of the overall difference on climate change impact.

4. Discussion

4.1. Overall mushroom impacts

The climate change impact obtained in this study (0.521–1.55 kg of CO_2eq kg^{-1} of harvested mushroom) were at least 27.1% lower than Robinson et al. (2019). These differences can be attributed to a series of factors, including a shift towards renewable electricity sources of the national electricity grid in some countries (IEA, 2021). In addition, this study accounted for emission from compost production and peat extraction, following the most recent IPCC guidelines (Gavrilova et al., 2019; Hergoualc'h et al., 2019; Lovelock et al., 2019; Ogle et al., 2019a, 2019b), in contrast to the other studies (Gunady et al., 2012; Leiva et al., 2016; Robinson et al., 2019). Finally, this difference in results can also be due to a different impact assessment method for climate change. Here we used 6th assessment report characterization factors while in previous studies the 5th assessment report (Forster et al., 2007, 2021; Gunady et al., 2012; Leiva et al., 2016; Robinson et al., 2019).

Resource use results for fossils fuels were comparable to the total primary energy accounted by Robinson et al. (2019) (29.2 MJ kg^{-1} of harvested mushrooms vs 29.9 MJ kg^{1} of harvested mushrooms of the Serbian system). Variations with the other systems could be attributed to the transport and sourcing of the casing material and the different impact assessment methods. In contrast, due to different methodology, the present results on water use impact cannot be compared with Robinson et al. (2019). The latter assessed freshwater use, just accounting for water use, surface water (lakes and rivers), and ground water use, while this study assessed water use following the AWARE method (Boulay et al., 2018), which takes in consideration water scarcity.

Terrestrial acidification obtained here (5.44–25.5 g of SO_2eq kg^{-1} of harvested mushroom) was within range with previous results obtained for mushroom cultivation in California, US (12.4 g of SO_2eq kg^{-1} of mushroom) (Robinson et al., 2019) and in a previous assessment in Spanish conditions (7.95 g of SO_2eq kg^{-1} of mushrooms (Leiva et al., 2015)). In contrast with terrestrial acidification results, the freshwater eutrophication potentials obtained for the three systems (0.203–2.34 g of Peq kg^{-1} of harvested mushroom) had a larger range than the corresponding range estimated in previous research (corresponding to 0.777–0.783 g of P kg^{-1} of mushroom) (Leiva et al., 2015; Robinson et al., 2019).

Compared to the agricultural production of other food products, mushroom production has a distinct environmental impact profile, both in terms of process contribution and environmental impact which is not comparable to animal production or crop cultivation, except, to some extent, for substrate-based horticulture (Leiva et al., 2015). Among the various processes accounted for, most of the environmental impact was

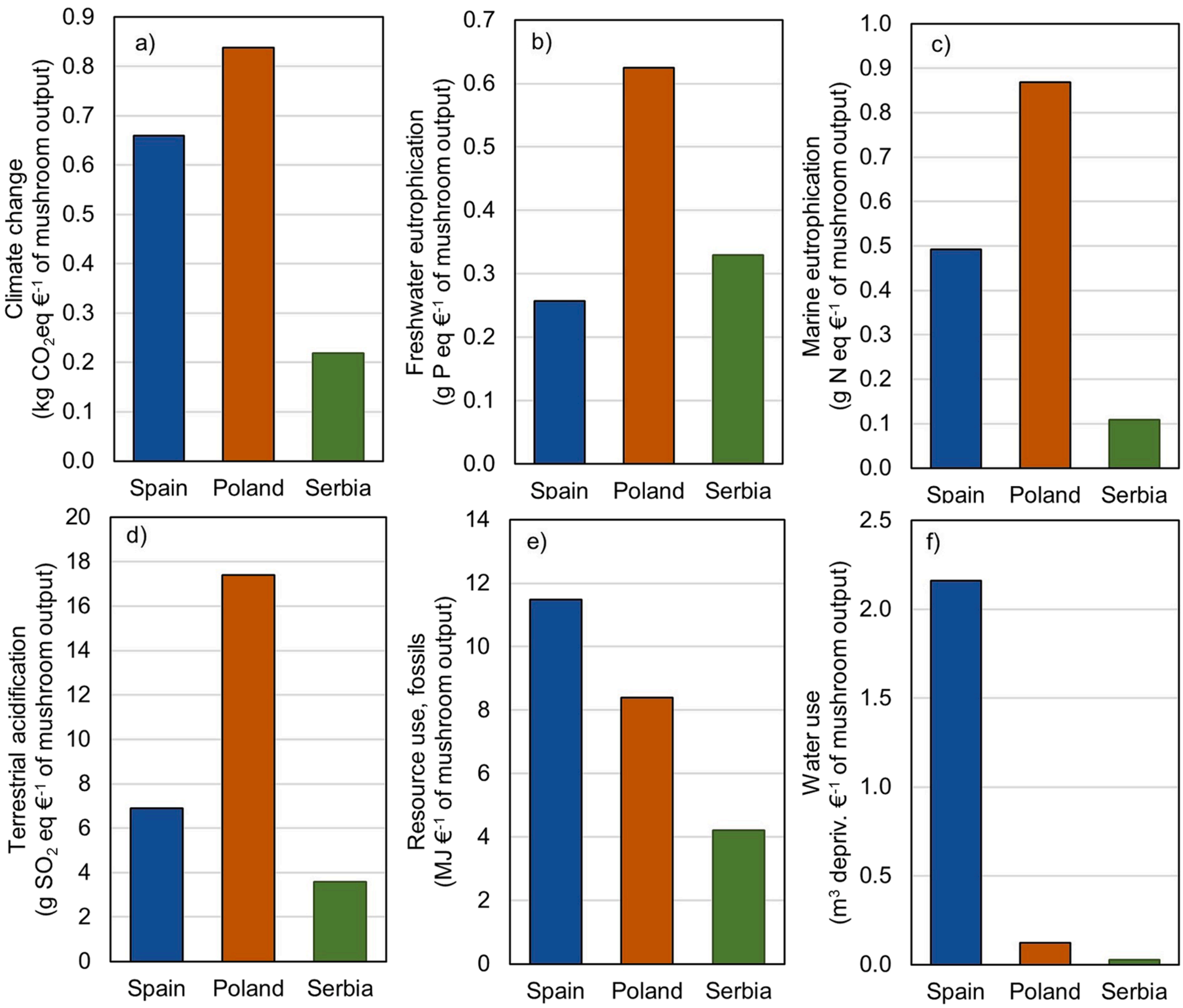

Fig. 3. Impact assessment results for climate change (kg carbon dioxide equivalents), freshwater eutrophication (kg phosphorus equivalents) and marine eutrophication (kg nitrogen equivalents), terrestrial acidification (kg sulphur dioxide equivalents), resource use, fossil (MJ) and water use (cubic metre water deprivation) per functional unit (€ of harvested mushroom).

due to compost and casing production which was in agreement with Gunady et al. (2012). However, our results were in contrast with Robinson et al. (2019) who reported electricity as the main source of impact for mushroom production. In this study, electricity use was still a large contributor towards the climate change and freshwater eutrophication impact (>7.9% contribution) across systems. Thus, the mushroom cultivation is an energy intensive process (for instance mushroom cultivation requires an environmentally controlled process for requirements of T up to 25℃, relative humidity up to 95% and ventilation to induce fructification (Carrasco et al., 2021), depending on the climatic conditions of the production and location with regards to electricity grid (Robinson et al., 2019). In addition, for the Spanish case, the contribution towards climate change of diesel consumption used during the mushroom farm operation (23.0%) was larger than electricity use (10.7%).

We observed also large differences in the type and amount of compost used, which also explains some of the differences in impact. This may be dependent on the availability of sources of compostable materials in the vicinity of the mushroom production facility, and the quality of those sources. Thus logistics and production of both peat and compost is particularly important as both process and related transport contributed 49.6% (compost) and 10.3% (casings) on average across systems and impact categories analyzed here. This was also reported for waste management logistics and bioenergy systems previously (Goglio and Owende, 2009; Kouloumpis et al., 2020). This confirmed the large importance that these upstream processes have with regards to mushroom production, as previously reported (Robinson et al., 2019). Besides the number of harvested mushrooms and the yield per unit area may increase with rising the density of compost load per unit area according to Pardo-Giménez et al. (2017), however the biological efficiency was not significantly modified which suggest that the compost load could be also amended to reduce the amount of compost used per unit area and eventually the environmental impact related to the compost.

The main alternative use of straw beside compost production for mushroom would be soil burial to maintain soil fertility and enhance soil C sequestration (Brady and Weil, 2002; Lal and Stewart, 2018; Paustian et al., 2016). This would also lead to large amount of CO_2 to the atmosphere due to organic matter humification (Brady and Weil, 2002; Lal and Stewart, 2018). Potential other uses include also the biochar production or bioenergy production (Giuntoli et al., 2016; Lychuk et al., 2021). However, a proper comparison of the environmental impacts of various straw uses both in Spain and in Poland was beyond the scope of the present paper.

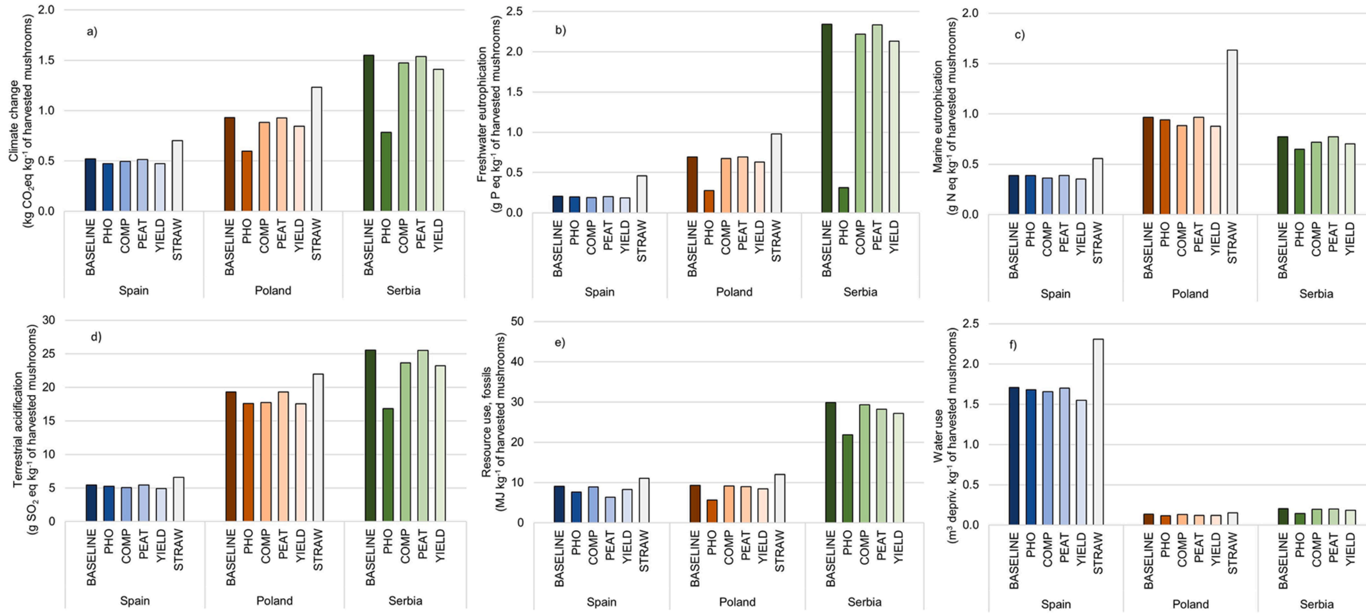

Fig. 4. Climate change, freshwater eutrophication and marine eutrophication, terrestrial acidification, resource use, fossil and water use results per functional unit (kg harvested mushroom) for the different scenarios (PHO = photovoltaic electricity replacing national electricity mix; COMP = 10% reduction in compost use; PEAT = 10% reduction in peat use; YIELD = 10% increase in yield; STRAW = dry mass allocation).

4.2. Mushroom growers management effect and systems consideration

This study showed that the environmental impacts of mushroom productions were affected by a series of factors which could be influenced by the mushroom growers choices. These are partially dictated by the productivity objectives but also by the background systems, such as the compost and casing production and the electricity grid. For instance, increasing the overall productivity of the mushroom production will reduce the environmental impacts by 9.1% for all the three systems under analysis (i.e. Spain, Poland, Serbia). These results suggest that nutritional supplementation in the compost could be a recommended agronomical measurement to boost productivity and reduce the environmental impact (Carrasco et al., 2018). However, this should be adapted following local conditions dependent on logistics, target market (ie. Dry mushroom, fresh mushroom).

The business strategy of the mushroom grower must also be taken into consideration: medium to large scale conventional fresh mushroom production involves very different technology and scale than organic farmers. Organically grown foods are often associated with environmentally friendly production, which is not often captured in LCA of agricultural systems(van der Werf et al., 2020). However, as in this case, the impact is generally lower when calculated from an economic perspective (i.e. per Euro of produce).

For all types of producers, the overall composition of the electricity mix can have a large effect on the environmental impact, as previously discussed for fish farming (Song et al., 2019). As the electricity grid mix composition shifts towards more renewable sources (IEA, 2021), the environmental impact and contribution of electricity use can potentially be reduced. The present paper showed that using photovoltaic electricity could reduce the impact of mushroom production with 24.2% across mushroom production systems and impact categories. Another choice the mushroom growers could undertake would be recycling the spent casing (Zied et al., 2020). Indeed, it has been suggested that recycling spent casing increases productivity by 11% and reduces the environmental impact by 28% (Banasik et al., 2017). However, each potential solution should be evaluated by the mushroom growers considering the specific conditions and the logistics (Zied et al., 2020). For instance, this choice could be affected by the spent casing availability, agronomic performance, pathogenic presence and technological issues to remove spent casing from spent compost (Taparia et al., 2021). Overall, an increased use of alternative products to peat and peat recycling have the potential to decrease climate and biodiversity impact related to peatland exploitation and resource use (Renou-Wilson et al., 2019; Zied et al., 2020).

Considering the high nutritional value of cultivated mushroom in particular as protein source (Krishnamoorthi et al., 2022; Leiva et al., 2015), the climate change impact of the studied mushrooms was 20.8, 45.7, and 39.4 kg CO_2eq per kg of protein (for the Spanish, Serbian and Polish case, assuming 10% dry matter at harvest in Spain and Poland and 13.5% in Serbia and 0.251 kg of protein per kg of dry matter; protein content from Dimopoulou et al., 2022). Comparing these results with other protein rich food, the climate change impacts are similar to eggs and broiler meat, much smaller than most other animal products, but larger than pulses and nuts (Poore and Nemecek, 2018). Therefore, in potential environmentally driven dietary changes, mushrooms are a reasonable alternative for animal products as protein source on dry matter basis. However, it should be considered that mushrooms also have other nutritional benefits beside high protein content, including bioactive compounds such as polyphenols, polysaccharides and micronutrients (Krishnamoorthi et al., 2022; Leiva et al., 2015), which constitute the main nutritional benefits of mushrooms.

4.3. Methodological issues

The assessment carried out here was highly reliant on several assumptions with regards to the logistics of mushroom production inputs, as previously discussed (Robinson et al., 2019). Further, in this paper the IPCC methodology has been used to assess emissions, which has been proven to be less accurate than other methods in assessing emissions from agricultural systems (FAO, 2018; Goglio et al., 2018). Other research used direct emission measurements (Leiva et al., 2016; Renou-Wilson et al., 2019), which can be considered more accurate than IPCC emission factors, but they can be time consuming and costly (Goglio et al., 2015, 2018), also considering the large variety of processing involved (e.g. industrial processes, land based processes). Furthermore, this assessment, in contrast to previous assessments (Gunady et al., 2012; Leiva et al., 2015), did not consider storage at the farm as the system boundary was set at the mushroom harvest to avoid comparability issues (ISO, 2006a, 2006b) for the final product (i.e., dry mushrooms and fresh mushrooms).

There is a continuous discussion among LCA experts on how to allocate upstream environmental impact among co-products (Anex and Lifset, 2014; Plevin, 2017). In the STRAW scenario, physical allocation was carried out, which resulted in significantly higher impacts of the mushroom production in the Spanish and Polish cases, where straw was employed. However, there is no consensus on whether economic allocation should be preferred over physical allocation (Moretti et al., 2020), even though the general LCA ISO standards stated that physical causalities need to be explored first (ISO, 2006a, 2006b). The main argument for applying economic allocation instead of a physical causality is that if allocation is based on a physical parameter this "results in the attribution of a large proportion of burdens to low-value co--products" (Pelletier et al., 2015). In this study, the allocation can affect the overall performance of the Spanish case, thus it was decided in agreement with the LCA objectives to calculate the results using both economic and physical allocation to provide a broader overview of the results.

Several LCA's highlight that agricultural activities have often a multifunctional role which include production, land occupation and income generation (Goglio et al., 2017; Nemecek et al., 2011). The analysis of using monetary units as functional unit instead of mass showed that the choice of the functional unit highly affects comparisons of environmental impact between different cases. This was partially due to the fact that organic mushrooms had higher prices. Using two functional units contributes in having a larger perspective on the environmental impact of mushroom production, as previously discussed for agricultural systems (Goglio et al., 2017; Nemecek et al., 2011; Ponsioen and van der Werf, 2017).

5. Conclusions

This research aimed at assessing different mushroom production systems across Europe to identify the environmental impacts of mushroom management practices. The results show a large variability in the environmental impacts of *Agaricus bisporus* mushroom production systems across Europe, where there may be different solutions to reduce the impact. A more efficient substrate use may reduce the overall environmental impact, for instance by adding nutritional supplements in the compost or selecting more productive commercial strains to increase the biological efficiency of the crop. Together with switching to renewable sources of electricity both for compost production (compost yards) and mushroom growth (mushroom growing facilities).

LCA results of mushrooms were affected by the business strategy of the mushroom growers (conventional fresh or organic dried mushrooms), location of the mushroom plant, logistics and mushroom input availability (availability of raw materials for compost production, peat extraction and transportation). In this sense, the selection of locally available ingredients for mushroom compost and casing blends can largely contribute to decrease the environmental impact of mushroom production.

This paper provided a comprehensive assessment across Europe which could be further expanded to have a more representative overview of the impact of mushroom production at European level. This study contributed in providing life cycle inventory and impact assessment data for potential technical innovations in different contexts.

CRediT authorship contribution statement

Kiwala Lukasz: Data curation, Funding acquisition, Investigation, Project administration, Writing – review & editing. **Milenkovi Ivanka:** Data curation, Funding acquisition, Investigation, Project administration, Writing – review & editing. **Pérez Margarita:** Funding acquisition, Project administration. **Carrasco Jaime:** Data curation, Funding acquisition, Investigation, Project administration, Writing – review & editing. **Ponsioen Thomas:** Data curation, Investigation, Methodology, Writing – original draft, Writing – review & editing. **Goglio Pietro:** Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Writing – original draft, Writing – review & editing. **Oosterkamp Elsje:** Project administration, Writing – review & editing. **Tei Francesco:** Writing – review & editing. **Helmes Roel:** Data curation, Investigation, Writing – original draft, Writing – review & editing. **Van Mierlo Klara:** Data curation, Investigation, Writing – original draft, Writing – review & editing.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: All the authors report financial support was provided by H2020 Food Security Sustainable Agriculture and Forestry Marine Maritime and Inland Water Research and the Bioeconomy. Ivanka Milenkovi reports a relationship with Ekofungi that includes: employment. Lukasz Kiwala reports a relationship with UGLK Lukasz Kiwala that includes: board membership.

Data Availability

Data will be made available on request.

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