Science of the Total Environment 964 (2025) 178624

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Environmental impact of peat alternatives in growing media for European mushroom production

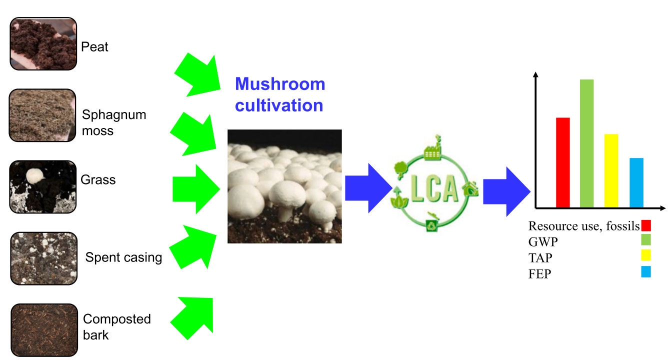
Pietro Goglio [a,*], Thomas Ponsioen [a,b], Jaime Carrasco [c,d], Francesco Tei [a], Elsje Oosterkamp [b], Margarita Pérez [e], Jan van der Wolf [f], Nancy Pyck [g]

[a] *Department of Agricultural, Food and Environmental Sciences, University of Perugia, Borgo XX Giugno 74, 06121 Perugia (PG), Italy*
[b] *Wageningen Economic Research, Wageningen University & Research, Droevendaalsesteeg 4, 6708 PB Wageningen, Netherlands*
[c] *IRIAF – Agri-food and Forestry Regional Research and Development Centre, CIAF - Agroforestry Research Centre, Carretera Toledo-Cuenca km 174, Cuenca 16194, Spain*
[d] *Department of Biology, University of Oxford, S Parks Rd, Oxford OX1 3RB, UK.*
[e] *Centro Technologico de Investigaction de Champinon de la Rioja, Carretera Calahorra km 4, 26560 Autol, Spain*
[f] *Wageningen Plant Research, Wageningen University & Research, Droevendaalsesteeg 4, 6708 PB Wageningen, Netherlands*
[g] *INAGRO, Department Edible Mushrooms, Ieperseweg 87, 8800 Rumbeke-Beitem, Belgium*

HIGHLIGHTS

- LCA of mushroom production was carried out with peat alternatives as casing material.
- Peat alternatives caused environmental trade-offs as casing for mushroom production.
- Peat alternatives caused <13.5 % impact change except for fossil resource use.
- Fossil resource use was largely reduced by peat alternatives.
- Trade-offs of reducing peat use need to be identified for more alternatives.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Jacopo Bacenetti

Keywords:
LCA
Mushroom
Button mushroom
Trade-offs
Peat
Casing

ABSTRACT

Button mushrooms are an important protein source with a production of >48 million metric tonnes in 2021. Several life cycle assessments (LCAs) have been employed in assessing mushroom cultivation. This paper assessed potential impacts of relevant alternatives (sphagnum moss, grass fibres, spent casing and bark) to peat as casing materials for mushroom production across Europe by using LCA using a cradle to farm gate approach. Here, we: i) compared the environmental impacts of mushroom produced with different growing media across Europe ii) identified environmental hotspots across the value chains of mushroom growing media and iii) provide insights on the sustainability of mushroom growing media production. Two functional units have been used the kg and € of harvested mushrooms. Data were gathered from mushroom producers and casing processors across Europe.

Changes in casing material for mushroom production caused environmental trade-offs, by reducing the resource use for fossil and by moderately changing (<7.7 %) all other impacts assessed here except the bark used

* Corresponding author.
E-mail address: pietro.goglio@gmail.com (P. Goglio).

https://doi.org/10.1016/j.scitotenv.2025.178624
Received 26 October 2024; Received in revised form 11 December 2024; Accepted 21 January 2025
Available online 26 January 2025

as casing material which caused a reduction for all impact categories. Thus, each of the casing alternative material could substitute peat in mushroom production with limited environmental impacts if productivity does not decrease. LCA demonstrated advantages or disadvantages of replacing part of the peat casings by other alternatives (e.g., sphagnum moss, spent casing, grass fibre, bark). As switching to alternatives becomes more urgent in the near future, potential trade-offs, advantages and disadvantages of using less peat need to be identified using a broader range of alternatives.

1. Introduction

The environmental impact of mushroom cultivation has become of increasing importance in the public society (Goglio et al., 2024; Robinson et al., 2019), as mushroom consumption is about 100 g per capita per week globally (Royse et al., 2017), and mushrooms are an important source of non-animal proteins (Goglio et al., 2024). According to FAOSTAT (2024), mushroom production reached over 48 million metric tonnes in 2021, with China accounting for 45 Mt. and Europe producing more than a million tonnes.

Mushroom cultivation requires a lot of external inputs including electricity, casing substrate, compost and water (Leiva et al., 2015; Robinson et al., 2019). Button mushroom (*Agaricus bisporus* ((J.E.Lange) Imbach) (hereafter referred to as mushroom) is commonly cultivated on compost topped with a casing layer with peat (Goglio et al., 2024; Grimm and Wösten, 2018; Robinson et al., 2019). As a secondary decomposer, the button mushroom grows on a pre-composted material (Grimm and Wösten, 2018), with low soluble sugar content, to avoid the growth of competitive/parasitic bacteria and moulds.

At commercial scale, phase III compost, occupied by the mushroom mycelium, is covered by the casing, then fruiting bodies from mushroom primordia are produced during cultivation (Carrasco et al., 2021). The compost is generally made up from different types of manure, straw combinations and gypsum (Gruda, 2019; Leiva et al., 2016; Robinson et al., 2019). The covering layer, commonly called casing layer, contributes in maintaining mushroom hydration, facilitates nutrient transport to the carpophores, and hosts a diverse microbial community important in the fructification (Pardo-Giménez et al., 2017; Taparia et al., 2021).

The casing material affects the mushrooms harvested positively (size and quality) or negatively (regarding the water holding capacity) (Zied et al., 2014). Among the casing materials, peat is still the first choice material for use in the casing growing media for production of button mushrooms because of its economic and technical features. It has favourable physical and microbiological characteristics at a favourable price-quality ratio (Taparia et al., 2021; Wever et al., 2005). However, peat extraction from natural peatland for horticultural use and mushroom production comes at an environmental cost. The main environmental concerns are removal of peat leading to CO_2 emission due to peat degradation (Rumpel et al., 2018), loss of ecohydrological functions and biodiversity (Räsänen et al., 2023; Šimanauskienė et al., 2019). Thus, several countries across the world restrict the exploitation of peatlands and stimulate exploration of alternatives (Airaksinen and Albrecht, 2019; Chen et al., 2023; Nordbeck and Hogl, 2023; Strack et al., 2022). Peatland excavation for horticultural purposes is more limited than other peatland uses (i.e. cultivation) (Clarke and Rieley, 2019). On the other hand, alternatives to peat as a casing material may also cause environmental burdens (Legua et al., 2021; Paoli et al., 2022; Roy et al., 2020).

Peatlands are considered high carbon reservoirs and high biodiversity areas and the extraction of peat is responsible for its reduction (Paustian et al., 2016; Renou-Wilson et al., 2019). It is estimated that 32 to 46 % of the global soil carbon is contained in peatland areas (Rumpel et al., 2018). After it has been exploited, the restoring of peatland is a very long process which can take up to several decades (Stichnothe, 2022). Further on, restoring peatlands can be difficult and it is largely affected by environmental and management variables (Renou-Wilson et al., 2019). Nielsen et al. (2023) reported that the acidity (pH) and water holding capacity affect the overall greenhouse gas emissions of the rewetted peatland.

Life cycle assessments (LCA) have been broadly employed in the environmental assessment of agricultural systems (Goglio et al., 2017), food products (Poore and Nemecek, 2018) and waste material (Olofsson and Börjesson, 2018; Vinci et al., 2023). LCA is a very powerful tool to identify hotspots and environmental trade-offs in long value chains or waste management systems (Kouloumpis et al., 2020; Robinson et al., 2019). Several LCAs have been employed in assessing button mushroom conditions in different countries and cultivation conditions (conventional vs organic) (Goglio et al., 2024; Gunady et al., 2012; Leiva et al., 2015; Robinson et al., 2019; Vinci et al., 2023), compost and casing (Leiva et al., 2016; Stichnothe, 2022).

Several peat casing substrate alternatives have been tested for their suitability for mushroom cultivation. For instance, spent mushroom substrate has been proposed and assessed to be reused in the next cycle (Grimm and Wösten, 2018; Vinci et al., 2023). Other substrates such as autoclaved sawdust, pangola grass (*Digitaria eriantha* Steud.), primavera tree (*Roseodendron donnell-smithii* (Rose) Miranda) and maize-cobs (Grimm and Wösten, 2018) were tested for button mushroom cultivation. Other substrates assessed for oyster mushrooms include reed (*Phragmites australis* (Cav.) Trin. ex Steud.) straw (Ye et al., 2023). Several other casing materials have been tested as an alternative to peat in button mushroom cultivation such as coco-peat, fly-ash, tea waste, pine bark, green waste compost, perlite and recycled rock wool (Taparia et al., 2021; Young et al., 2024). Due to the mushroom growth requirements, the range of casing material is more limited than horticultural growing media (Hashemi et al., 2024; Young et al., 2024). Indeed, many of these growing materials resulted in a low performance in comparison to peat due to presence of pests and pathogen, or in an intrinsic poor quality due to low physical characteristics (e.g. low water holding capacity) or chemistry (e.g. high salinity, accumulation of toxic residues). The presence of pests and pathogens often required the use of steaming the material. In a recent productivity and disease assessment, it was found that grass fibres from agricultural waste, sphagnum moss (from *Sphagnum sp*), and recycled spent casing soil can be used to successfully substitute peat in mushroom cultivation (Taparia et al., 2021).

Vinci et al. (2023) carried out an LCA of mushroom production using several compost materials (i.e., poultry manure, horse manure and straw), however to our knowledge no paper has assessed different casing materials for mushroom cultivation using LCA. Thus, the present research has the following objectives: i) compare the environmental impact of mushroom produced with different growing media across Europe; ii) identify environmental hotspots across the value chains of mushroom growing media and iii) provide insights on the sustainability of mushroom growing media production.

2. Materials & methods

2.1. Life-cycle assessment

2.1.1. System description

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 Poland. In the Spanish and Polish cases, the conventional mushrooms were sold fresh, while most of the organic mushrooms in the Serbian case were sold dried. In all three cases, the base casing material was peat. A detailed description of the systems, the scope, and allocation methods can be found in Goglio et al. (2024). The functional unit used included kg of harvested mushrooms and € of mushroom outputs, in agreement with previous research (Goglio et al., 2024, 2017). Fig. 1 provides an overview of the scenarios assessed in the present paper and the related system boundary.

2.1.2. Scenarios

In the baseline scenario peat is used as the casing material in mushroom cultivation for three mushroom systems, previously assessed in Goglio et al. (2024). Peat alternatives analysed in this paper include grass fibres, sphagnum moss, and spent casing, also called spent mushroom substrate (SMS) as relevant alternatives, as they resulted as valid alternatives to peat (Taparia et al., 2021). In the Serbian case, decomposed bark was also assessed as this resulted as a valid alternative in mushroom cultivation at the case study mushroom farm. The technical performance of these materials in mushroom production (except for bark) has been discussed by previous research (Taparia et al., 2021; Young et al., 2024). The use of grass fibres as partial replacement of peat in the casings layer for mushroom production has been developed in the Netherlands (Taparia et al., 2021). The alternatives described in Taparia et al. (2021) are currently not applicable to a small-scale organic mushroom farm in Serbia, due to the geographical availability of this material, the logistics and the organic production regulation (EC, 2018). This Serbian farm is producing mushrooms with the complete replacement of peat by decomposed bark from paper industry in the region. The steaming of spent casing was accounted for all spent mushroom substrate scenarios, considering that on-farm or collective steaming of spent mushroom substrate may be implemented more widely in the future as a sanitary measure (Cunha Zied et al., 2020), despite the economic constraints related to the energy prices. In all alternative scenarios, it was assumed that all inputs and outputs not related to the casing remain the same as the baseline.

Changes in substrate characteristics due to changes in composition affect soil respiration. Steaming and inoculation were therefore applied to the materials to make the alternatives to peat equally productive and control diseases (Taparia et al., 2021). As there is increasing indications that peat alternatives can be equally performing without steaming (Young et al., 2024), the contribution of steaming was reported separately for the scenarios where it was assumed to be carried out. In Table 1, the different scenarios are described.

2.1.3. System boundaries and functional units

The study uses system boundaries from cradle-to-gate, including the environmental impacts of the processes upstream of the production chain until the cultivation gate (Fig. 1). The processes included are production of energy required for cultivation, compost production, casings (including inoculum) production, spawn production, and other inputs, transport of inputs, mushroom cultivation, and waste treatment. Capital inputs including the facility and machinery manufacturing and their associated environmental impacts are also included (Fig. 1). Two functional units have been employed for this LCA scenario assessment 1 kg of harvested mushrooms and 1 € of harvested mushrooms according to previous research (Goglio et al., 2024, 2017; Nemecek et al., 2011).

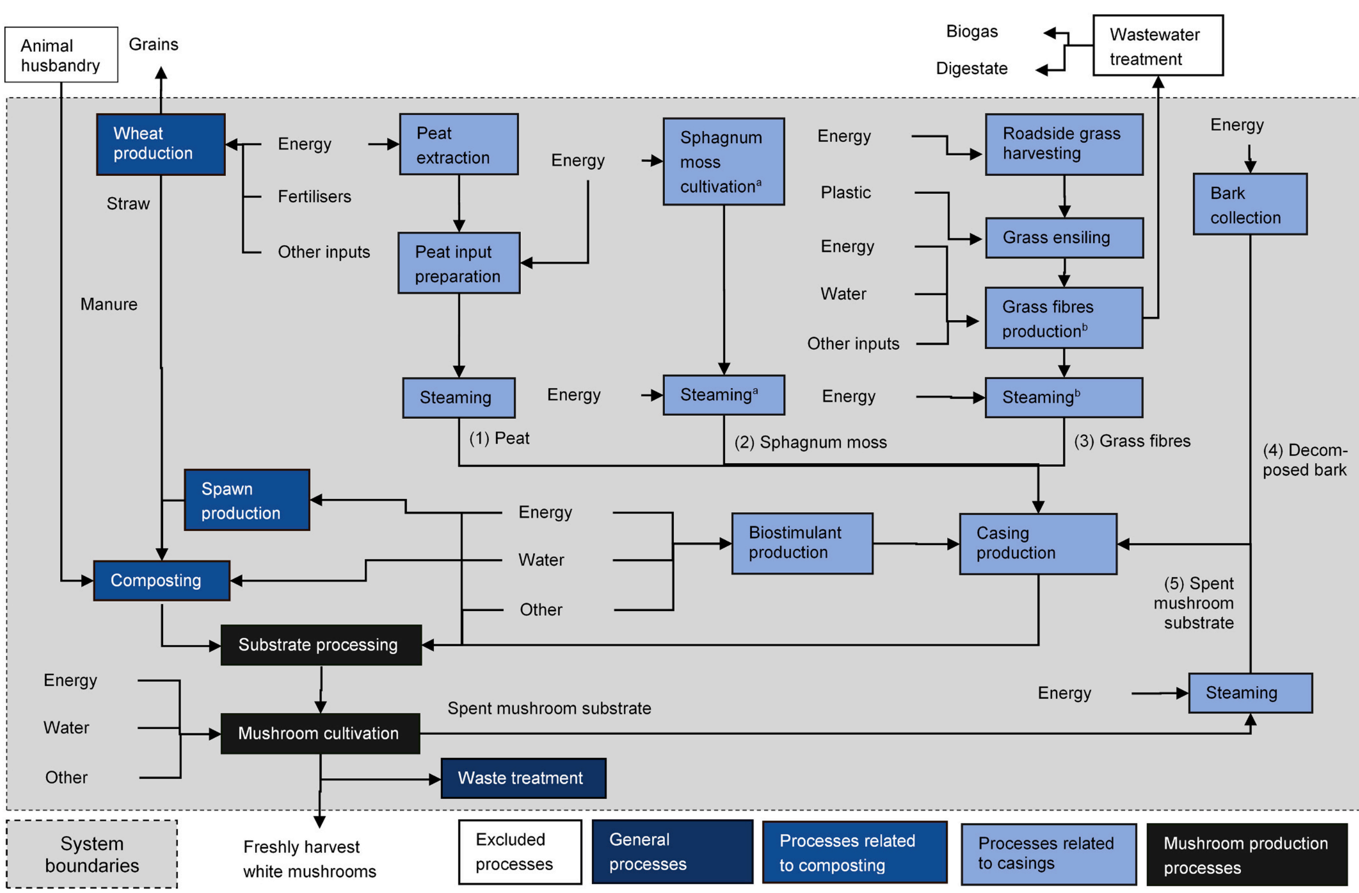


Fig. 1. Flow chart of processes included in the life cycle assessment (production of capital goods, energy, water and material inputs and transport not shown); (1) Peat (baseline for all 3 cases); (2) Sphagnum moss (alternative for Spanish & Polish cases); (3) Grass fibres is alternative (Spanish & Polish cases); (4) Decomposed bark (alternative for Serbian case); (5) Spent mushroom substrate (alternative for all three cases (Spain, Poland, Serbia)).

Table 1
Scenario descriptions.

Scenario	Spanish case	Polish case	Serbian case
BASELINE	30 % black and 70 % blond/white peat	100 % black peat	100 % white peat
GRASS_NL	50 % of the peat volume is replaced by grass fibres produced in the Netherlands from grass residues, which are transported to Spain by road	50 % of the peat volume is replaced by grass fibres produced in the Netherlands from grass residues, which are transported by boat to Poland	Not applicable
GRASS_ROAD	Not applicable	50 % of the peat volume is replaced by grass fibres produced in the Netherlands from grass residues, which are transported by road to Poland	Not applicable
GRASS_PL	Not Applicable	50 % of the peat volume is replaced by grass fibres produced in Poland from grass residues	Not applicable
GRASS_ES	50 % of the peat volume is replaced by grass fibres produced in Spain from available residual plant material	Not applicable	Not applicable
MOSS_BC	50 % of the black and white peat volume substrate in the casings is replaced by sphagnum moss; due to the large variation in soil carbon sequestration and release from Sphagnum moss cultivation as different levels of drainage or water saturation can be found, best case (MOSS_BC) scenarios are applied.[a]		Not applicable
MOSS_WC	50 % of the black and white peat volume substrate in the casings is replaced by sphagnum moss; due to the large variation in soil carbon sequestration and release from Sphagnum moss cultivation as different levels of drainage or water saturation can be found, worst case (MOSS_WC) scenarios are applied.[a]		Not applicable
SPENT casing	50 % of the peat volume is replaced by spent substrate. The spent substrate is steamed by the producer. The impact of 25 km transport to and from the steaming facility is considered.		
BARK		Not applicable	100 % of the peat in the Serbian case is replaced by several years old decomposed bark from cellulose paper industry in Serbia.

[a] Details of the scenarios were presented in Section 2.2.3.

2.1.4. Allocation

Economic allocation was applied to the data used for inputs, such as grains for spawn production and straw for compost production, in agreement with Goglio et al. (2024). 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). Roadside grass/verge harvesting is carried out for road safety reasons. Thus, the environmental impact related to this activity was attributed to road maintenance. The harvested grass/verge can be considered as a waste and sent to waste treatment. Alternatively, it can be transported to processing facilities where it is converted into products, such as grass fibres, as considered in the present research. In that case, the additional transport was attributed to the grass fibre product. Fibre grass production results in wastewater which is then anaerobically digested in anaerobic digestor to produce biogas. The same way the environmental impact of transport and processing of the roadside grass/verge is completely attributed to the grass fibres, the environmental impact of wastewater processing (anaerobic digestion) is completely attributed to the biogas. Thus, no environmental impact of any activities related to the wastewater after leaving the grass fibre substrate material production process is attributed to the grass fibre substrate material.

2.1.5. Impact assessment

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, as previously carried out (Goglio et al., 2024). The results were calculated using the Environmental Footprint 3.1 Method (Andreasi Bassi et al., 2023) for climate change, fossil resource energy carrier use, and water scarcity; ReCiPe 2016 Midpoint (H) for freshwater and marine eutrophication and terrestrial acidification, Van Oers et al. (2002) for abiotic resource depletion and Boulay et al. (2018) for water scarcity as implemented in SimaPro 9.5 (2023).

2.2. Inventory data and data processing

2.2.1. Mushroom production

The data for the three case studies are extensively described in Goglio et al. (2024) which are the baseline (BASE) scenarios. Table 2 shows the most relevant inputs and yield data. The following subsections describe the data and assumptions for the materials alternative to peat as a casing material.

2.2.2. Grass fibre production

The process of producing grass fibre substrate is described in detail by Taparia et al. (2021). In summary, the grass fibres are produced by a patented biorefining process which converts non-woody biomass into lignocellulosic fibres (Vos and Rustenburg, 2016). The grass fibre substrate is produced from roadside grass silage. The heat energy demands for the process are met by producing and combusting biogas from the leftover grass-juice. All the water used in this process is cleaned and recycled within the biorefinery. Grass fibres can proportionally replace 50 % of the peat volume in the casing soil without affecting the productivity.

To control pathogens, it was assumed that the grass fibres requires steaming which was assumed to be carried out by a substrate producer with natural gas as the energy source (Taparia et al., 2021). It was also assumed that the grass fibres were inoculated with the biostimulants containing *Bacillus velezensis* CM5, CM19 and CM35 (Carrasco and Preston, 2023), *Pseudomonas fluorescens* SBW25 (Rainey and Bailey, 1996), and *Pseudomonas putida* PMS118R and PMS118S (Rainey et al., 1991) for making the material suitable for mushroom production. Data for the production of the bacterial inoculum were based from the upscaling of the lab experiment data (Carrasco and Preston, 2023; Moni et al., 2020). The inputs and outputs data of the steamed grass fibres are shown in Table 3.

In the GRASS PL/ES and GRASS NL scenarios, 50 % of the peat volume in the Polish and Spanish cases baseline system casings described by Goglio et al. (2024) was replaced by grass fibres. The grass fibres are currently produced in the Netherlands, which means that the material needs to be transported to the production locations in Poland and Spain. These were the assumptions for the GRASS NL scenarios. Instead, in the GRASS PL/ES scenario, it was considered that grass fibres are produced locally with local sourcing of residual materials, such as grass silage or plant residues.

Table 2
Most relevant inputs and yield data. The table presents data for the scenarios for each specific case (Spanish, Polish and Serbian case).

Input/output	Unit	Spanish case (conventional)	Polish case (conventional)	Serbian case (organic)	Source
Yield	kg m^{-2} $year^{-1}$	276	274	274	Primary data
Number of cycles per year	# $year^{-1}$	8	9.36	8	Primary data
Compost use per year	kg m^{-2} $year^{-1}$	723	721	1233	Primary data
Compost use per cycle	kg m^{-2} $cycle^{-1}$	90	77	154	Primary data
Compost use per kg mushroom	kg kg^{-1} mushroom	2.62	2.63	4.50	Primary data
Casings use per year	kg m^{-2} $year^{-1}$	170	168	482	Primary data
Casings use per cycle	kg m^{-2} $cycle^{-1}$	21	18	60	Primary data
Casings use per kg mushroom	kg kg^{-1} mushroom	0.62	0.62	1.8	Primary data
Peat use (in casings)	kg kg^{-1} mushroom	0.38	0.62	1.8	Primary data
Electricity use	kWh kg^{-1} mushroom	0.20	0.36	0.38	Primary data
Heat use from wood	kWh kg^{-1} mushroom	0.90	0	0	Primary data
Heat use from diesel	kWh kg^{-1} mushroom	0.90	0	0	Primary data
Heat use from natural gas	MJ kg^{-1} mushroom	0	0	0.58	Primary data
Peat bulk density	kg d.m. m^{-3}	300	300	300	Primary data
Grass bulk fibre density	kg d.m. m^{-3}	255	255	–	Primary data
Moss bulk density	kg d.m. m^{-3}	100	100	–	Martineau (2013)
Bark bulk density	kg d.m. m^{-3}	–	–	300	Assumption
Spent substrate density	kg d.m. m^{-3}	300	300	300	Assumption

Table 3
Inputs and outputs of the grass fibre substrate production and Sphagnum moss production. This data have been used in the life cycle inventory for the grass fibres scenarios.

Scenario	Input/output	Amount	Unit	Source
Grass fibre	Moisture of the grass fibre casings	30	%	Confidential, primary data
	Density of the grass fibre casings	0.255	kg d.m. dm^{-3}	Confidential, primary data
	Roadside grass (16.5 % dry matter)	4.55	kg kg^{-1} casing	Confidential, primary data
	Distance from grass supplier	40	Km	Confidential, primary data
	Packaging film for silage	5.68	g kg^{-1} casing	Remmelink et al. (2011)
	Electricity (from the grid)	0.156	kWh kg^{-1} casing	Confidential, primary data
	Heat from biogas	0.0675	MJ kg^{-1} casing	Confidential, primary data
	Natural gas for steaming	0.60	MJ kg^{-1} casing	Baar et al. (2005)
	Triptone (Sigma)	2.2	g m^{-2}	Primary data
	Yeast extract (Sigma)	1.1	g m^{-2}	Primary data
	Salt (naCl)	2.2	g m^{-2}	Primary data
	distilled water	4	L m^{-2}	Primary data
	Electricity	0.54	kWh m^{-2}	Primary data
Sphagnum moss	Yield	2.5	t dry matter ha^{-1}	Wernet et al. (2016)
	Density	100	kg dry matter m-3	Wernet et al. (2016)
	CO_2 exchange at high water table	−0.6	t CO_2-C ha^{-1} y^{-1}	Oestmann et al. (2022)
	CO_2 exchange at low water table	2.2	t CO_2-C ha^{-1} y^{-1}	Oestmann et al. (2022)

2.2.3. Sphagnum moss production

For the sphagnum moss peat scenarios (MOSS), it was assumed that 50 % of the peat volume in the casings can be replaced by sphagnum moss, following indications from previous research (Taparia et al., 2021; Young et al., 2024). This substrate material is harvested from peatlands which have been managed to grow *Sphagnum sp* moss species. For this scenario Ecoinvent 3.9 on sphagnum moss production for horticultural use (Von Post scale of H1-H4; (Van Post, 1924)) in Quebec, Canada was modified for Finnish and Sweden conditions (Wernet et al., 2016). In particular, the electricity market and geography-specific water flows were adapted for production in Finland and Sweden (assuming 50 % from each country). The carbon dioxide exchange between the peat soil and the air in the process was also adapted for European conditions. As only the net emissions are reported in the dataset and is specific for the situation in Quebec, Canada, we replaced the net CO_2 emissions by another source for European conditions. Oestmann et al. (2022) found a range of CO_2 exchanges in sphagnum moss cultivation between −0.6 and 2.2 t CO_2-C ha^{-1} y^{-1} in Northwestern Germany. These values were used as best MOSS_BC and worst case MOSS_WC estimates for the carbon dioxide exchange between the peat soil and the air in the Swedish and Finnish sphagnum moss cultivation in the MOSS scenarios (Table 3).

2.2.4. Spent casing and decomposed bark

In the SPENT scenarios, it was assumed that 50 % of the peat in the casings can be replaced by steamed spent mushroom substrate, in agreement with previous findings (Young et al., 2024). The spent casing was assumed to be transported 25 km to and from the steaming facility. Data for steaming were taken from Baar et al. (2005).

Considering the organic production regulation for the Serbian case and the small scale of the operations there, the grass fibre, moss and spent substrate scenarios cannot be applied (EC, 2018). However, SPENT casing scenarios were still assessed as potential scenarios, following discussions with the mushroom growers. As several years old decomposed bark from cellulose paper industry in Serbia is available and the performance of the mushroom production appears to be successful in this case, we considered this scenario to the Serbian case in the BARK scenario. As the bark is a waste product of the paper industry, no impact from production was considered. So, only packaging and transport were included.

2.2.5. Carbon balance

The compost and casings in the substrate contain substantial quantities of carbon, which is mainly released as carbon dioxide due to oxidation and partly as methane due to anaerobic conditions of the compost during composting (Fryda et al., 2018; Hashemi et al., 2024; Saer et al., 2013; Stichnothe, 2022). The carbon stored in the peat substrate is considered as long-term storage, in agreement with the IPCC definitions (IPCC, 2019). The carbon of all other substrate materials were considered as short-term storage, which means that carbon dioxide emissions from these materials did not have any impact to climate change (Saer et al., 2013). The complete balances of the different substrate materials were calculated, including carbon dioxide uptake upstream and release downstream of the mushroom production.

The carbon content of compost at start was assumed to be 14 %. Spent casing and decomposed bark substrate are assumed to have the same carbon content as peat substrate. The dry matter carbon content of grass fibre substrate and sphagnum moss substrate was assumed the same as of peat substrate (0.45 kg carbon per kg dry matter) (Adamovics et al., 2018; Garnier and Vancaeyzeele, 1994; Shetler et al., 2008; Vingiani et al., 2004). All loss of stored carbon in the trees (including the bark) was attributed to the main product (paper), as decomposed bark is a waste of the paper industry, in agreement with the ISO standards (ISO, 2006a, 2006b).

2.3. Contribution and uncertainty analyses

A contribution and uncertainty analysis was carried out in agreement with the ISO standards (ISO, 2006a, 2006b). The changes in the contribution of different environmental impact sources as reported in Goglio et al. (2024) were analysed to get a better understanding of the main environmental impact sources in the different scenarios.

An uncertainty analysis was carried out in line with previous LCA research (Bisinella et al., 2021; Manzano et al., 2023; Mendoza Beltran et al., 2018) and the LCA related standards (ISO, 2006a, 2006b). For the uncertainty analysis, triangular distributions with a minimum and maximum of 10 % below and 10 % above the measured material and energy inputs were assigned. For water, we assigned minimum and maximum of 20 % below and above the measured amount, because this input is known to vary more than the other inputs. The mushroom yield was assigned a triangular with a minimum of 25 % below the measured and a maximum of 20 % above, because the yield depends on many factors that cannot be controlled completely, in some cases with higher yields, but likely more often with lower yields.

Monte Carlo analysis was applied comparing each scenario with the baseline for the three case studies, using 10,000 runs in SimaPro 9.5 (2023). The 95 % confidence intervals of the difference in impact are shown in the results. The uncertainty analysis was carried out for the parameters which could affect the differences among the scenarios on the basis of the contribution analysis and the overall results. The statistical significance between the baseline and each scenario was tested with the Welch test using R software (R Development Core Team, 2005; Rosner, 2011).

3. Results

3.1. Absolute results

The absolute results per kg of harvested mushrooms and per € of harvested mushrooms (Tables 4, 5) showed only small differences compared to the baseline <7.7 %, with some exceptions. These were due to the limited contribution of the casing production to the overall mushroom impacts for most of the categories excluding resource use (<16.9 %). These included the BARK scenario resulted in a 13.5 % lower impact on climate change than the BASE scenario for the Serbian mushroom systems and the impacts on fossil resource use (2.6 %–59.5 % change for the scenario analysed vs the corresponding BASE scenario) (Table 4). Further, the water deprivation of the BARK scenario vs the BASE scenario was reduced for the Serbian systems (10.3 % less, Tables 4, 5). As the bark required limited processing for casing production in comparison to the other material; resource use impact was largely affected by the peat consumption which was considered in the impact assessment method as a non-renewable material (Andreasi Bassi et al., 2023); while the other impacts were less influenced (Andreasi Bassi et al., 2023; Boulay et al., 2018; Van Oers et al., 2002).

In particular for resource use, energy carriers, the Serbian case has the highest impact of fossil resource use (29.9 MJ kg^{-1} of harvested mushrooms, Table 4) and thus the largest potential for reduction. Considering the mushroom price, however, the Spanish case has the highest impact (11.5 MJ $€^{-1}$ of harvested mushrooms, Table 5) and thus the highest potential for reduction. In the following sections, the differences compared to the baseline are analysed in more detail per case.

3.2. Contribution analysis results

The relative contribution of casing production and peat oxidation in the Spanish case BASE scenario to fossil resource use on the overall mushroom production was 42.3 %, for climate change 9.2 %, while to all the other environmental impacts analysed here was <2 % on a per kg of mushroom basis. In the case of the Polish system in the BASE scenario, the casing production resulted in a large relative contribution only for the fossil resource use (31.9 %) but far less (<2.8 %) to all the other environmental impacts. In a similar manner to the Spanish case, for the Serbian system, the casing production contributed in the BASE scenario largely only to fossil resource use (62.0 %), climate change (16.9 %), water use (11.9 %) and to a more limited extent to the other analysed environmental impact.

This was due to the larger use of casing in comparison to the other two systems (1.8 kg of casing kg^{-1} of mushrooms for the Serbian systems vs 0.62 kg of casing kg^{-1} of mushrooms). The large contribution to fossil resource use is because peat was accounted as a fossil resource, therefore

Table 4
Environmental impacts per functional unit (FU) (FU: 1 kg of harvested mushrooms at harvest).

	GWP100	Freshwater eutrophication	Marine eutrophication	Terrestrial acidification	Resource use, fossils	Water use
	kg CO_2-eq. FU^{-1}	g P eq. FU^{-1}	mg N eq. FU^{-1}	g SO_2 eq. FU^{-1}	MJ FU^{-1}	m^3 depriv. FU^{-1}
Spanish case						
BASE	0.521	0.203	264	5.44	9.08	1.71
GRASS_NL	0.535	0.211	265	5.47	7.76	1.71
GRASS_ES	0.523	0.206	264	5.46	7.69	1.71
MOSS_BC	0.499	0.203	264	5.44	7.29	1.70
MOSS_WC	0.512	0.203	264	5.44	7.29	1.70
SPENT	0.511	0.203	264	5.42	7.37	1.70
Polish case						
BASE	0.931	0.695	966	19.3	9.32	0.135
GRASS_ROAD	1.000	0.708	968	19.5	9.07	0.138
GRASS_NL	0.983	0.706	967	19.5	8.83	0.137
GRASS_PL	0.989	0.748	969	19.6	8.79	0.138
MOSS_BC	0.893	0.695	966	19.4	8.10	0.131
MOSS_WC	0.978	0.695	966	19.4	8.10	0.131
SPENT	0.951	0.692	966	19.3	8.07	0.128
Serbian case						
BASE	1.55	2.34	772	25.5	29.9	0.203
BARK	1.34	2.25	765	25.0	12.1	0.182
SPENT	1.48	2.30	769	25.3	21.5	0.193

Table 5
Environmental impacts per functional unit (FU) (FU: 1 € of harvested mushrooms at harvest).

	GWP100	Freshwater eutrophication	Marine eutrophication	Terrestrial acidification	Resource use, fossils	Water use
	kg CO_2-eq. FU^{-1}	g P eq. FU^{-1}	mg N eq. FU^{-1}	g SO_2 eq. FU^{-1}	MJ FU^{-1}	m^3 depriv. FU^{-1}
Spanish case						
BASE	0.659	0.257	334	6.88	11.5	2.16
GRASS_NL	0.677	0.267	335	6.93	9.82	2.16
GRASS_ES	0.662	0.261	335	6.91	9.73	2.16
MOSS_BC	0.632	0.257	334	6.88	9.23	2.16
MOSS_WC	0.648	0.257	334	6.88	9.23	2.16
SPENT	0.647	0.257	334	6.86	9.32	2.15
Polish case						
BASE	0.838	0.625	869	17.4	8.39	0.121
GRASS_ROAD	0.900	0.637	871	17.5	8.17	0.124
GRASS_NL	0.885	0.635	870	17.5	7.95	0.123
GRASS_PL	0.890	0.673	872	17.6	7.91	0.124
MOSS_BC	0.804	0.625	869	17.4	7.29	0.118
MOSS_WC	0.880	0.625	869	17.4	7.29	0.118
SPENT	0.856	0.623	869	17.4	7.26	0.115
Serbian case						
BASE	0.262	0.395	130	4.31	5.05	0.0343
BARK	0.227	0.379	129	4.22	2.05	0.0308
SPENT	0.250	0.388	130	4.26	3.63	0.0326

non-renewable (IPCC, 2019).

In the different alternative scenarios to the BASE scenarios for all the three cases analysed here (Spain, Poland and Serbia), the relative contributions of casing production to fossil resource use of mushroom production ranged from 6.2 to 47.1 % (in all cases a considerable reduction compared to the baseline), climate change −1.3 to 12.9 % (in the best case, carbon is sequestered during sphagnum moss cultivation (Oestmann et al., 2022)), freshwater eutrophication 0.2 to 8.1 %, water use 0.8 to 8.0 %, marine eutrophication range from 0.0 to 0.6 %, terrestrial acidification from 0.1 to 1.6 %. Thus, different peat alternatives affected mostly resource use and climate change, while the other impact categories were minimally changed. Indeed, the change in casing material influenced the greenhouse gas emissions, the fuel and the peat use during the casing material production.

Limited changes were obtained with regards to electricity and compost contribution to climate change in the Spanish scenarios which showed a 10.4–11.2 % range for the first process and 44.9 %–48.1 % range for the second process. Instead, the contribution of electricity production in the Polish case to climate change was 37.9 % in the baseline and decreased down to 35.3–35.9 % in the grass fibre scenarios, to 36.1 % in the MOSS_WC scenarios, and to 37.1 % in the spent substrate scenario. Further, the contribution of compost production in the Polish case to climate change was about 49.4 % in the baseline and was reduced down to 46.0–46.8 % in the grass fibre scenarios, to 47.1 % in the MOSS_WC scenarios, and to 48.4 % in the spent casing scenario. Instead for both MOSS_BC for the Spanish and Polish Systems, the contribution to the overall climate change impact of compost production increased up 39.5 % in the Spanish system and 51.5 % in the Polish system. Conversely in the Serbian case, the contribution of electricity production in the Serbian case on climate change, which was 25.2 % in the baseline, increased to 29.1 % in the BARK scenario, and 26.4 % in the SPENT scenario. A similar pattern was found in the Serbian case for compost production which contributed to 48.4 % in the baseline but higher contribution were observed in the bark scenario (55.9 %) and the spent substrate scenario (50.7 %).

In all three cases, the contribution of electricity production to fossil fuel use was subject to an increase in all the alternative scenarios to the baseline. The largest change was obtained for the Serbian system with a contribution increase from 13.8 % of the baseline to >19.2 % for all the other alternative scenario for the Serbian case, while the other cases (i.e. the Spanish and the Polish case) have a smaller change up to 5.4 % for all the alternative scenarios to the baseline.

In a similar manner, also for the compost production the contribution to fossil resource use increased in the alternative scenarios for all the three cases. The lower change in contribution was observed between the contribution in the Polish BASE scenario and the other Polish scenarios (0.5 %), while the largest change in contribution was between the BASE scenario in Serbia and the alternative scenarios in Serbia (28.8 %). The Spanish case resulted in a change in contribution of the compost production between the previous two cases (3.4 %–4.7 %).

3.3. *Grass fibre scenarios*

In the grass fibre scenarios of the Spanish and Polish cases, we found small increases compared to the baselines scenario in the climate change (between 0.5 and 7.8 %) and freshwater eutrophication (between 1.4 and 8.0 %), based on the uncertainty analysis results (Fig. 2a, ES: Spanish system; PL: Polish system; RS: Serbian system). This is partly due to a higher energy use for production of grass fibres and steaming (representing 1.9 % of the total climate change for the Spanish systems and 1.6–1.7 % in the Polish systems). In the Spanish case, the effect was smaller when the grass fibre was produced in Spain (0.5 % for climate change and 1.4 % for freshwater eutrophication) than in the Netherlands (2.8 % for climate change and 4.0 % for freshwater eutrophication), due to transport distances. For the Polish case, the increase is also due to a larger transport distance when produced in the Netherlands for the GRASS scenarios (between 5.9 and 7.8 % for climate change and 1.7 and 2.0 % for freshwater eutrophication), or when the grass fibres are produced in Poland (6.5 % for climate change and 8.0 % for freshwater eutrophication) (Fig. 2a). The fossil resource use on the other hand, decreased between 2.6 and 15.9 %. This was because peat was considered as a fossil resource (IPCC, 2019). In the Spanish case, the reduction was much larger (between 15.3 and 15.9 %) than in the Polish case (between 2.6 and 5.8 %) (Fig. 2a), because the contribution to fossil resource use of the peat use was smaller in the Polish case, and the reduction was partly compensated in the Polish case by larger transport distance or higher fossil energy use for the electricity. For fossil resource use, steaming represented 1.9 % of the overall impact while for the Polish systems 2.6–2.7 %.

The confidence intervals between each scenario and impact categories overlapped except for fossil resource use for the scenarios evaluated. Further the Welch test showed significant differences across

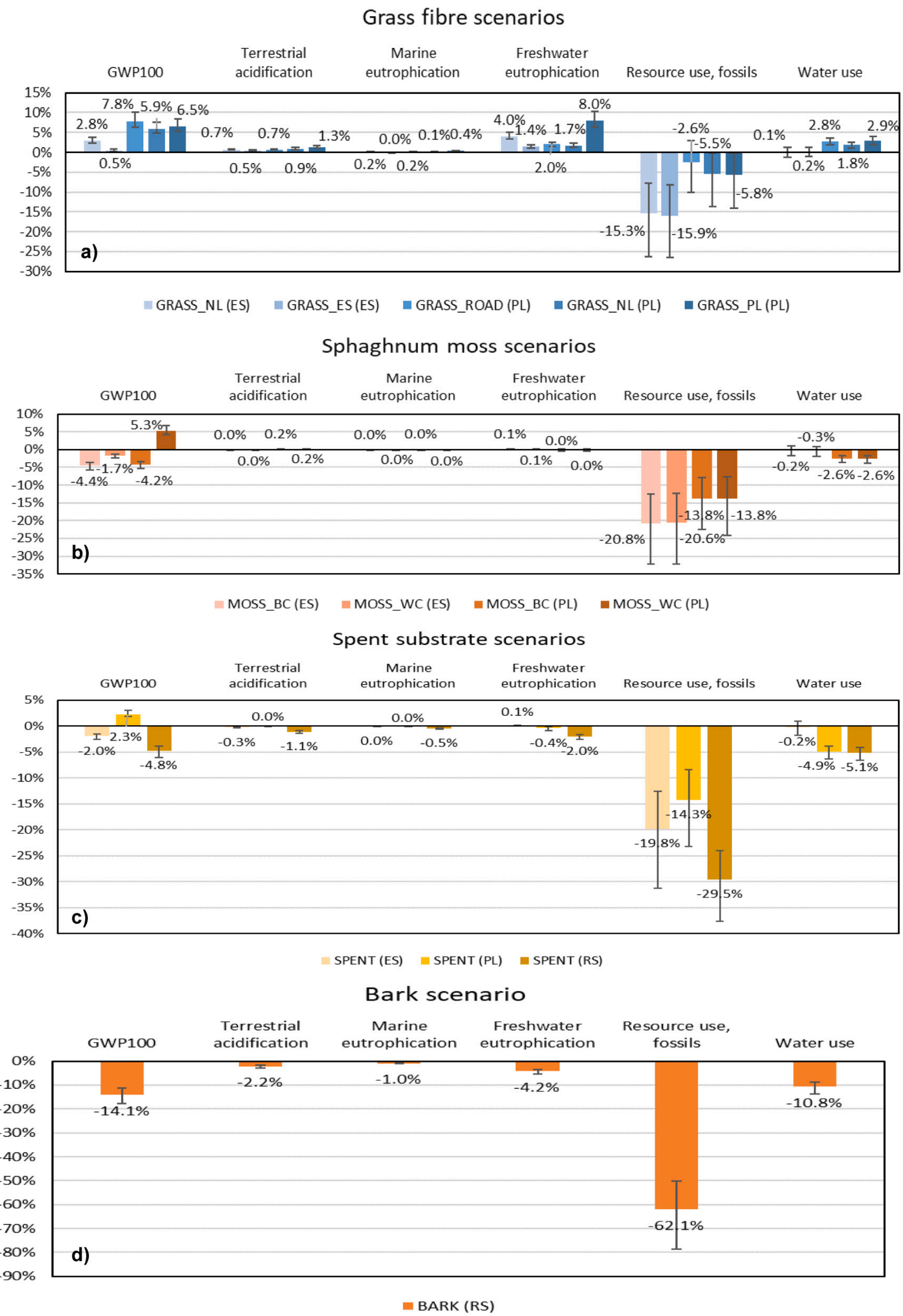


Fig. 2. Percentage change in environmental impact of the peat alternative vs the baseline (SP: Spanish case; PL: Polish case; RS: Serbian case); a) grass fibre scenarios compared to the baseline (GRASS_NL: grass fibre produced in the Netherlands; GRASS_ROAD: grass fibre produced in the Netherlands transported to Poland by road; GRASS_ES: grass fibre produced in Spain; GRASS_PL: grass fibre produced in Poland); b) Sphagnum moss scenarios compared to the baseline (MOSS_BC: Sphagnum moss best case scenario with net carbon sequestration in the soil; MOSS_WC: Sphagnum moss worst case scenario with net carbon release from the soil); c) spent substrate scenarios (SPENT) compared to the baseline; d) bark scenario (BARK) compared to the baseline; error bars represent the 95 % confidence interval on the absolute differences between the alternative scenarios and the baseline.

scenarios and impact categories, as previously discussed (Greenland et al., 2016).

3.4. Sphagnum moss scenarios

The sphagnum moss scenarios had a smaller impact on climate change (between 1.7 and 4.4 % reduction) than the baseline scenarios, except for the worst case scenario in Poland, which had a slightly larger impact (5.3 %) (Fig. 2b). The impact depended upon water saturation level in the sphagnum moss field, affecting CO_2 emissions, which can vary highly between sphagnum moss cultivation areas. Changes of sphagnum moss scenarios on freshwater eutrophication, marine eutrophication, terrestrial acidification, and water use were smaller than 0.3 %, except for water use in Poland, which decreases by 2.6 % (for both best and worst case scenarios). Fossil resource use decreased by between 20.6 and 20.8 % in the Spanish case and 13.8 % in the Polish case (Fig. 2b). The 95 % confidence interval did not overlap in all sphagnum moss scenario cases where the differences compared to the baseline are larger than 1 % and in all of these cases the confidence interval overlaps and the Welch test showed significant differences at $p < 0.05$.

3.5. Spent casing scenarios

The spent casings scenarios had a slightly lower impact than the baseline in most cases and impact categories, mainly because less long distance transport of peat is required. Only in the Polish case, the climate change impact was slightly higher (2.3 %) (Fig. 2c). The fossil resource use impact was considerably smaller, similar to the sphagnum moss scenarios (between 14.3 and 29.5 % reduction), where the largest decrease is found in the Serbian case due to the large contribution of peat to this impact category in the BASE scenario (62.0 %). Further reduction (up to 3.1 % for all the impact categories, while for the impact on climate change 1.7 %–3.1 %) could be achieved without steaming.

The 95 % confidence interval did not overlap in all spent substrate scenario cases where the differences compared to the baseline are larger than 1 %. In all of these cases, the Welch test showed significant differences at $p < 0.05$.

3.6. Bark scenario

The BARK scenario in the Serbian case resulted in a 14.1 % reduction in climate change impact, 62.1 % reduction in fossil resource use, 10.8 % reduction in water use, and smaller reductions in the other impact categories (Fig. 2d). This was due to the limited processing necessary for casing production from bark. The 95 % confidence intervals did not overlap in all impact categories, except for marine eutrophication. However, the Welch test showed significant differences between the BARK scenario and the baseline at $p < 0.05$.

4. Discussion

4.1. Environmental benefits and trade-offs

The potential environmental benefits of the casing alternatives to peat are substantial for the fossil resource use impact category. However, this only represented a decrease in demand of peat as fossil resource. Potential loss of biodiversity and ecosystems services by extracting peat from natural wetlands are not represented by the selected environmental impact indicators, but there is no reliable indicator available at the moment to apply in LCA (Curran, 2013; van der Werf et al., 2020). Nonetheless it is quite well known that peatlands are responsible for a large amount of greenhouse gas emissions when they are dried (Oestmann et al., 2022; Paustian et al., 2016; Rumpel et al., 2018). However, horticultural uses have limited contribution to peatland exploitation in comparison to other uses (Clarke and Rieley, 2019).

The main reason for evaluating alternative casings materials is the decreasing peat availability due to increasing governmental restrictions and peatland conservation policies essential to preserve biodiversity (Airaksinen and Albrecht, 2019; Chen et al., 2023; Renou-Wilson et al., 2019; Strack et al., 2022). Besides, environmental cost of peat used in mushroom cultivation is also significant while covering long distances from peatlands to growing facilities (Navarro et al., 2021; Robinson et al., 2019). Thus, differently than common LCA carried out, this research did not focus only on the performance of the alternatives compared to peat, but also identifying substantial trade-offs among impact categories. The differences among the various scenarios resulted in <7.7 % change among the peat alternatives tested here and across all impact categories, excluding resource use with 59.5 % change and the BARK scenarios which had up to 13.5 % impact reduction. In some Polish case scenarios there may be an increase of >5 % in climate change and freshwater eutrophication due to electricity use for grass fibre production, transport of grass fibre, or release of soil carbon in worst case Sphagnum moss cultivation. However, these trade-offs may be addressed by switching to renewable energy sources when these become economically viable (Cunha Zied et al., 2020), avoiding steaming (Young et al., 2024) and ensuring water saturation in sphagnum moss cultivation. In the Serbian BARK scenario, there is likely even a substantial additional benefit to climate change and water use (10.3–13.5 %). The availability of decomposed bark, however, can be limited. The present results showed that composted bark where available can be a valid alternative to peat from the environmental standpoint.

The potential changes and trade-off are often dependent on the logistics and the location of the mushroom farm to the source of casing material. As shown in Fig. 2a, the same material (grass fibre) can have a different change in several impacts including climate change, freshwater eutrophication and water use depending on whether this peat alternative is produced locally or imported from other countries, as discussed in previous research (Leiva et al., 2016; Robinson et al., 2019). Further investigation could be carried out addressing other alternatives material and addressing the specific logistics related to the casing transport and processing using site-specific and site-dependent assessment. The overall performance is largely dependent on local conditions, as previously discussed (Goglio et al., 2024). Peatland management can also affect the overall GHG emissions, as previously reported (Stichnothe, 2022).

4.2. Comparison with previous research

Several previous reports assessed the impact of mushroom production and casing production using a LCA approach (Gunady et al., 2012; Hashemi et al., 2024; Leiva et al., 2015; Robinson et al., 2019). The overall impact for climate change for mushroom production was at least 37.3 % lower than described by Robinson et al. (2019). This was due to different assumptions including the location of the farms in Europe while in Robinson et al. (2019) the mushroom farms were located in California. The geographical location also affects the electricity grid impacts related to electricity consumption during mushroom cultivation as previously highlighted (Goglio et al., 2024). Climate change impact ranges (0.499–1.55 kg of CO_2eq kg^{-1} of mushrooms) were comparable with Vinci et al. (2023) who reported an impact of climate of 1.11 kg of CO_2eq kg^{-1} of mushroom. In particular, the GRASS_ROAD scenario was only 11 % lower than Vinci et al. (2023) who assessed a mushroom system located in Italy. Robinson et al. (2019) only used peat casing while here different alternatives to casing were tested. Large differences were found with the impact on climate change reported by several previous researchers (Gunady et al., 2012; Leiva et al., 2015), with at least 2.9 fold differences with the present results. In this case too, the present study was assessing different alternatives to peat while Leiva et al. (2015) only used peat in Spanish conditions and Gunady et al. (2012) carried out an assessment in Australian conditions. In addition, different methodologies were used to account for substrate oxidation. In our study, CO_2 estimation with IPCC methodology, the IPCC 2019 emission factors and the most recent IPCC characterisation factors were used (Forster et al., 2021; IPCC, 2019), while Leiva et al. (2015) used direct measurements and the CML Leiden 2000 methodology (Leiva et al., 2015).

Regarding the acidification potential, the range obtained here for the Spanish systems was at maximum 46.7 % lower than the value reported by Leiva et al. (2015) (7.95 g of SO_2eq kg^{-1} of mushrooms). The other systems had larger values (19.3–25.5 g of SO_2eq kg^{-1} of mushrooms). Acidification values obtained here were − 49.5 % to 68.1 % different from those reported by Vinci et al. (2023) and in Robinson et al. (2019). The freshwater eutrophication impacts for the Polish systems (0.692–0.748 kg of Peq kg^{-1} of mushrooms) were at maximum 12.0 % below that reported by Leiva et al. (2015). The Spanish systems had at least 73.2 % lower values; while the Serbian systems at least 2.9 fold larger freshwater eutrophication values than those obtained by Leiva et al. (2015). The corresponding eutrophication potential accounted for by Robinson et al. (2019) was very close (<10.9 %) to the freshwater EP reported for the Polish systems. The Spanish and the Serbian systems resulted in larger differences (>72.8 %). These differences can be related to the different composting process and compost type (e.g., either from poultry manure and straw or horse manure). Freshwater eutrophication was at least 11.1 fold higher than Vinci et al. (2023); while a 8.7 folds difference was observed between results presented here and Vinci et al. (2023) for marine eutrophication. In Vinci et al. (2023), mushroom cultivation was carried out using a growing substrate from agricultural waste and recycled water while for the SPENT and BARK scenarios spent casing and decomposed bark was used and no water recycling occur.

Several papers assessed the impact of the production of casing material for the horticultural sector (Hashemi et al., 2024; Stichnothe, 2022). Hashemi et al. (2024) reported a 1.09 fold difference in climate change using alternatives to peat, while our results had a larger effect (2.69 folds difference). However the type of casing materials in our research were quite different than those in the studies of Hashemi et al. (2024) who assessed hydrochar, wood fibre, compost and degassed agricultural waste fibre, as mushrooms have specific physical and chemical requirements (Young et al., 2024). Peat-based growing substrate resulted in a climate change impact with a broader range than the corresponding climate change impact reported in Stichnothe (2022) (−0.0187–0.153 kg of CO_2eq kg^{-1} here vs 0.158–0.309 kg of CO_2eq kg^{-1} in the studies of Stichnothe (2022)). Nevertheless, the GRASS_ROAD scenario was 2.9 % smaller than the white peat production in Stichnothe (2022). Freshwater (2.66–97.6 mg of Peq kg^{-1} of harvested mushrooms) and marine eutrophication potentials (0.483–6.18 mg of Neq kg^{-1} of harvested mushrooms) exhibited at least 1.69 fold difference with the peat growing media assessed by Stichnothe (2022). Resource use values (0.416–10.3 MJ kg^{-1} of harvested mushrooms) were at maximum 15.5 fold higher than the peat value reported in the same research (Stichnothe, 2022) except for the BARK scenario which was at least 21 % less than Stichnothe (2022).

4.3. Confidence of the results and general overview

In the Spanish and Polish cases, the scenarios are feasible for large scale implementation in the near future. Similar systems have been partially tested previously such as wood fibres and degassed fibres from food waste in Hashemi et al. (2024). In the Serbian case, however, the question is whether the bark will support satisfactory yields and will be feasible for medium to large scale production. The organic mushroom farmer consulted reported satisfactory performance, however, this was not tested in an experimental setting. Nonetheless, where available it is a potential alternative to peat which needs further investigation (Young et al., 2024).

The effects of replacing 50 % of the peat in the casings by grass fibres, sphagnum moss or spent substrate on mushroom growth parameters were assumed to be negligible in this study, following recent productivity testing and recent literature evidence (Young et al., 2024). Substantial differences in soil respiration and adhesiveness were found in the grass fibres compared with peat-based casing. Another factor likely affecting the productivity is the microbiological composition, which varies highly between the alternatives (Taparia et al., 2021). It is questionable if steaming is required or even desirable. On the one hand, it will eliminate populations of competitors, pathogens and pest organisms, but it will also reduce beneficial microbial buffering potential of the substrate (Carrasco and Preston, 2020), and make it more vulnerable for pathogens introduced from the mushroom ecosystem.

Recent studies in commercial mushroom cultivation facilities, showed that replacement of peat by grass fibres or sphagnum moss can also result in a good productivity without pasteurization of the constituents. This aligns with recent literature evidence which found no substantial performance in peat alternatives tested (Young et al., 2024). The resilience of casings may increase via microbial buffering with specific beneficial organisms. In this assessment, data from a lab production were used to estimate the impact of inoculations with beneficial bacteria. However, pilot scale investigation did not fully succeed in producing the inoculum at industrial scale.

Uncertainties are quite common when assessing future emerging technologies in which LCA can become an essential tool for the environmental assessment (Bisinella et al., 2021; Goglio et al., 2019; Moni et al., 2020). In this work, an effort was made to minimise the overall uncertainty by refining the life cycle inventory in several iterations, by identifying the substantial contributions of preliminary results and collecting higher quality data which affect these contributions, from primary and secondary sources, in agreement with the ISO standards (ISO, 2006a, 2006b).

5. Conclusions

The results of this paper showed that a mix of peat and some relevant alternatives instead of using complete peat casings, reduced fossil resource use as peat is considered a fossil material. Replacement of peat by alternatives did not substantially increase the impact on climate change, freshwater and marine eutrophication, terrestrial acidification, and water use in three distinct case studies across Europe. Relatively small trade-offs (<7.7 % increased impact) to potential benefits of reducing the amount of peat in the casing are related to the energy required to produce the alternative material with steaming corresponding to <3.1 % of the overall impact, the transport distance and mode, and the (local) availability of residual material suitable for making mushroom casing. With mushroom productivity similar to peat casing, the substrates are considered as a valid alternative to peat, even though the production process and the logistic could be further improved. The choice of casing material is highly dependent on the physical, chemical and biological characteristics of the material, casing material availability, processing and logistics of the supply chain.

The quantitative assessment demonstrated the potential advantages or disadvantages of replacing part of the peat casings by peat alternatives on a selection of environmental impacts. As switching to alternatives becomes more urgent in the near future, potential trade-offs of the advantages of using less peat need to be conducted for a broader range of alternatives. The results of this paper provided insights and examples on how to assess this.

CRediT authorship contribution statement

Pietro Goglio: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Thomas Ponsioen:** Visualization, Software, Methodology, Investigation, Formal analysis, Data curation. **Jaime Carrasco:** Writing – review & editing, Supervision, Resources, Project administration, Conceptualization. **Francesco Tei:** Writing – review & editing. **Elsje Oosterkamp:** Writing – review & editing, Project administration. **Margarita Pérez:** Project administration, Funding acquisition, Data curation. **Jan van der Wolf:** Writing – review & editing, Supervision. **Nancy Pyck:** Writing – review & editing.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Pietro Goglio reports financial support was provided by H2020 Food Security Sustainable Agriculture and Forestry Marine Maritime and Inland Water Research and the Bioeconomy. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

A special thanks goes to the funders of the BIOSCHAMP Horizon 2020 project (Grant Agreement n. 101000651).

Data availability

Background data from this assessment can be made available upon request to the corresponding author. *B. velezensis* CM5, CM19 and CM35 are protected as biocontrol agents against mushroom parasites in a patent published as EP4200400A1; WO2021255181A1PCT, 2023/06/28.

References

Adamovics, A., Platace, R., Gulbe, I., Ivanovs, S., 2018. The content of carbon and hydrogen in grass biomass and its influence on heating value. Eng. Rural Dev. 17, 1277–1281.

Airaksinen, J., Albrecht, E., 2019. Arguments and their effects – a case study on drafting the legislation on the environmental impacts of peat extraction in Finland. J. Clean. Prod. 226, 1004–1012. https://doi.org/10.1016/j.jclepro.2019.04.161.

Andreasi Bassi, S., Biganzoli, F., Ferrara, N., Amadei, A., Valente, A., Sala, S., Ardente, F., 2023. Updated Characterisation and Normalisation Factors for the Environmental Footprint 3.1 Method. Publications Office of the European Union, Luxembourg. https://doi.org/10.2760/798894.

Baar, J., Amsing, J.G.M., Rutjens, A.J., 2005. Reductie energiegebruik in de champignonteelt, P.P.O. 2005-5. Parktijk onderzoek Plant & Omgeving B.V, Horst, The Netherlands.

Bisinella, V., Christensen, T.H., Astrup, T.F., 2021. Future scenarios and life cycle assessment: systematic review and recommendations. Int. J. Life Cycle Assess. 26, 2143–2170. https://doi.org/10.1007/s11367-021-01954-6.

Boulay, A.-M., Bare, J., Benini, L., Berger, M., Lathuillière, M.J., Manzardo, A., Margni, M., Motoshita, M., Núñez, M., Pastor, A.V., Ridoutt, B., Oki, T., Worbe, S., Pfister, S., 2018. The WULCA consensus characterization model for water scarcity footprints: assessing impacts of water consumption based on available water remaining (AWARE). Int. J. Life Cycle Assess. 23, 368–378. https://doi.org/10.1007/s11367-017-1333-8.

Carrasco, J., Preston, G.M., 2020. Growing edible mushrooms: a conversation between bacteria and fungi. Environ. Microbiol. 22, 858–872. https://doi.org/10.1111/1462-2920.14765.

Carrasco, J., Preston, G.M., 2023. Bacteria. Patent Number EP4200400A1. European Patent Office, Munich. https://worldwide.espacenet.com/patent/search/family/071835520/publication/EP4200400A1?q=pn%3DEP4200400A1.

Carrasco, J., Zied, D.C., Navarro, M.J., Gea, F.J., Pardo-Giménez, A., 2021. Commercial cultivation techniques of mushrooms. In: Advances in Macrofungi. CRC Press, Boca Raton, pp. 11–40. https://doi.org/10.1201/9781003096818-3.

Chen, C., Loft, L., Matzdorf, B., 2023. Lost in action: climate friendly use of European peatlands needs coherence and incentive-based policies. Environ. Sci. Policy 145, 104–115. https://doi.org/10.1016/j.envsci.2023.04.010.

Clarke, D., Rieley, J. (Eds.), 2019. Strategy for Responsible Peatland Management, 6th edited. International Peat Society, Jyväskylä, Finland.

Cunha Zied, D., Sánchez, J.E., Noble, R., Pardo-Giménez, A., 2020. Use of spent mushroom substrate in new mushroom crops to promote the transition towards a circular economy. Agron 10, 1239. https://doi.org/10.3390/agronomy10091239.

Curran, M.A., 2013. Life Cycle Assessment: a review of the methodology and its application to sustainability. Curr. Opin. Chem. Eng., Energy and Environmental Engineering/Reaction Engineering and Catalysis 2, 273–277. https://doi.org/10.1016/j.coche.2013.02.002.

EC, 2018. Regulation (EU) 2018/848 of the European Parliament and of the council of 30 May 2018 on organic production and labelling of organic products and repealing Council Regulation (EC) No 834/2007. Off. J. Eur. Union L 150, 1–92.

FAOSTAT, 2024. FAOSTAT, Statistics division, Food and Agriculture Organization of the United Nations, Rome. http://www.fao.org/faostat/en/#home (accessed 23 May 2024).

Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D.J., Mauritsen, T., Palmer, M.D., Watanabe, M., Wild, M., Zhang, H., 2021. The Earth's energy budget, climate feedbacks, and climate sensitivity. In: Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S.L., Péan, C., Berger, S., Caud, N., Chen, Y., Goldfarb, L., Gomis, M.I., Huang, M., Leitzell, K., Lonnoy, E., Matthews, J.B.R., Maycock, T.K., Waterfield, T., Yelekçi, O., Yu, R., Zhou, B. (Eds.), Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 923–1054. https://doi.org/10.1017/9781009157896.009.

Fryda, L., Visser, R., Schmidt, J., 2018. Biochar replaces peat in horticulture: environmental impact assessment of combined biochar & bioenergy production. Detritus 05, 132–149. https://doi.org/10.31025/2611-4135/2019.13778.

Garnier, E., Vancaeyzeele, S., 1994. Carbon and nitrogen content of congeneric annual and perennial grass species: relationships with growth. Plant Cell Env. 17, 399–407.

Goglio, P., Brankatschk, G., Knudsen, M.T., Williams, A.G., Nemecek, T., 2017. Addressing crop interactions within cropping systems in LCA. Int. J. Life Cycle Assess. 1–9. https://doi.org/10.1007/s11367-017-1393-9.

Goglio, P., Williams, A., Balta-Ozkan, N., Harris, N.R.P., Williamson, P., Huisingh, D., Zhang, Z., Tavoni, M., 2019. Advances and challenges of life cycle assessment (LCA) of greenhouse gas removal technologies to fight climate changes. J. Clean. Prod. 118896. https://doi.org/10.1016/j.jclepro.2019.118896.

Goglio, P., Ponsioen, T., Carrasco, J., Milenkovi, I., Kiwala, L., Van Mierlo, K., Helmes, R., Tei, F., Oosterkamp, E., Pérez, M., 2024. An environmental assessment of Agaricus bisporus ((J.E.Lange) Imbach) mushroom production systems across Europe. Eur. J. Agron. 155, 127108. https://doi.org/10.1016/j.eja.2024.127108.

Greenland, S., Senn, S.J., Rothman, K.J., Carlin, J.B., Poole, C., Goodman, S.N., Altman, D.G., 2016. Statistical tests, P values, confidence intervals, and power: a guide to misinterpretations. Eur. J. Epidemiol. 31, 337–350. https://doi.org/10.1007/s10654-016-0149-3.

Grimm, D., Wösten, H.A.B., 2018. Mushroom cultivation in the circular economy. Appl. Microbiol. Biotechnol. 102, 7795–7803. https://doi.org/10.1007/s00253-018-9226-8.

Gruda, N., 2019. Increasing sustainability of growing media constituents and stand-alone substrates in soilless culture systems. Agronomy 9, 298. https://doi.org/10.3390/agronomy9060298.

Gunady, M.G.A., Biswas, W., Solah, V.A., James, A.P., 2012. Evaluating the global warming potential of the fresh produce supply chain for strawberries, romaine/cos lettuces (Lactuca sativa), and button mushrooms (Agaricus bisporus) in Western Australia using life cycle assessment (LCA). In: J. Clean. Prod., Working Towards a More Sustainable Agri-food Industry: Main Findings From the Food LCA 2010 Conference in Bari, Italy, 28, pp. 81–87. https://doi.org/10.1016/j.jclepro.2011.12.031.

Hashemi, F., Mogensen, L., Smith, A.M., Larsen, S.U., Knudsen, M.T., 2024. Greenhouse gas emissions from bio-based growing media: a life-cycle assessment. Sci. Total Environ. 907, 167977. https://doi.org/10.1016/j.scitotenv.2023.167977.

IPCC, 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Glossary. Intergovernmental Panel for Climate Change (IPCC), Geneva, Switzerland.

ISO, 2006a. SS-EN ISO 14040 Environmental Management-Life Cycle Assessment, Principles and Framework. International Organization for Standardization, Geneva.

ISO, 2006b. ISO 14044 Environmental Management — Life Cycle Assessment — Requirements and Guidelines.

Kouloumpis, V., Pell, R.S., Correa-Cano, M.E., Yan, X., 2020. Potential trade-offs between eliminating plastics and mitigating climate change: an LCA perspective on Polyethylene Terephthalate (PET) bottles in Cornwall. Sci. Tot. Environ. 727, 138681. https://doi.org/10.1016/j.scitotenv.2020.138681.

Legua, P., Hernández, F., Tozzi, F., Martínez-Font, R., Jorquera, D., Jiménez, C.R., Giordani, E., Martínez-Nicolás, J.J., Melgarejo, P., 2021. Application of LCA methodology to the production of strawberry on substrates with peat and sediments from ports. Sustainability 13, 6323. https://doi.org/10.3390/su13116323.

Leiva, F.J., Saenz-Díez, J.C., Martínez, E., Jiménez, E., Blanco, J., 2015. Environmental impact of Agaricus bisporus cultivation process. Eur. J. Agron. 71, 141–148. https://doi.org/10.1016/j.eja.2015.09.013.

Leiva, F., Saenz-Díez, J.-C., Martínez, E., Jiménez, E., Blanco, J., 2016. Environmental impact of mushroom compost production. J. Sci. Food Agric. 96, 3983–3990. https://doi.org/10.1002/jsfa.7587.

Manzano, P., Rowntree, J., Thompson, L., del Prado, A., Ederer, P., Windisch, W., Lee, M.R.F., 2023. Challenges for the balanced attribution of livestock's environmental impacts: the art of conveying simple messages around complex realities. Anim. Front. 13, 35–44. https://doi.org/10.1093/af/vfac096.

Mendoza Beltran, A., Chiantore, M., Pecorino, D., Corner, R.A., Ferreira, J.G., Cò, R., Fanciulli, L., Guinée, J.B., 2018. Accounting for inventory data and methodological choice uncertainty in a comparative life cycle assessment: the case of integrated multi-trophic aquaculture in an offshore Mediterranean enterprise. Int. J. Life Cycle Assess. 23, 1063–1077. https://doi.org/10.1007/s11367-017-1363-2.

Moni, S.M., Mahmud, R., High, K., Carbajales-Dale, M., 2020. Life cycle assessment of emerging technologies: a review. J. Ind. Ecol. 24, 52–63. https://doi.org/10.1111/jiec.12965.

Navarro, M.J., Carrasco, J., Gea, F.J., 2021. The role of water content in the casing layer for mushroom crop production and the occurrence of fungal diseases. Agronomy 11, 2063. https://doi.org/10.3390/agronomy11102063.

Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G., 2011. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agric. Syst. 104, 217–232. https://doi.org/10.1016/j.agsy.2010.10.002.

Nielsen, C.K., Elsgaard, L., Jørgensen, U., Lærke, P.E., 2023. Soil greenhouse gas emissions from drained and rewetted agricultural bare peat mesocosms are linked to geochemistry. Sci. Total Environ. 896, 165083. https://doi.org/10.1016/j.scitotenv.2023.165083.

Nordbeck, R., Hogl, K., 2023. National peatland strategies in Europe: current status, key themes, and challenges. Reg. Environ. Chang. 24, 5. https://doi.org/10.1007/s10113-023-02166-4.

Oestmann, J., Tiemeyer, B., Düvel, D., Grobe, A., Dettmann, U., 2022. Greenhouse gas balance of sphagnum farming on highly decomposed peat at former peat extraction sites. Ecosyst 25, 350–371. https://doi.org/10.1007/s10021-021-00659-z.

Olofsson, J., Börjesson, P., 2018. Residual biomass as resource – life-cycle environmental impact of wastes in circular resource systems. J. Clean. Prod. 196, 997–1006. https://doi.org/10.1016/j.jclepro.2018.06.115.

Paoli, R., Feofilovs, M., Kamenders, A., Romagnoli, F., 2022. Peat production for horticultural use in the Latvian context: sustainability assessment through LCA modeling. J. Clean. Prod. 378, 134559. https://doi.org/10.1016/j.jclepro.2022.134559.

Pardo-Giménez, A., Pardo González, J.E., Zied, D.C., 2017. Casing materials and techniques in *Agaricus bisporus* cultivation. In: Diego, C.Z., Pardo-Giménez, A. (Eds.), Edible and Medicinal Mushrooms. John Wiley & Sons, Ltd, Chichester, UK, pp. 149–174. https://doi.org/10.1002/9781119149446.ch7.

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G.P., Smith, P., 2016. Climate-smart soils. Nature 532, 49–57. https://doi.org/10.1038/nature17174.

Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Sci 360, 987–992. https://doi.org/10.1126/science.aaq0216.

R Development Core Team, 2005. A Language and Environment for Statistical Computing. R Foundation for Statistical Computing.

Rainey, P.B., Bailey, M.J., 1996. Physical and genetic map of the *Pseudomonas fluorescens* SBW25 chromosome. Mol. Microbiol. 19, 521–533. https://doi.org/10.1046/j.1365-2958.1996.391926.x.

Rainey, P.B., Brodey, C.L., Johnstone, K., 1991. Biological properties and spectrum of activity of tolaasin, a lipodepsipeptide toxin produced by the mushroom pathogen Pseudomonas tolaasii. Physiol. Mol. Plant Pathol. 39, 57–70. https://doi.org/10.1016/0885-5765(91)90031-C.

Räsänen, A., Albrecht, E., Annala, M., Aro, L., Laine, A.M., Maanavilja, L., Mustajoki, J., Ronkanen, A.-K., Silvan, N., Tarvainen, O., Tolvanen, A., 2023. After-use of peat extraction sites – a systematic review of biodiversity, climate, hydrological and social impacts. Sci. Total Environ. 882, 163583. https://doi.org/10.1016/j.scitotenv.2023.163583.

Remmelink, G., Blanken, K., Van Middelkoop, J., Ouweltjes, W., Wemmenhove, H., 2011. Handbook Melkveehouderij 2011 (No. Handboek 22). Wageningen UR Livestock Research, Lelystad, The Netherlands.

Renou-Wilson, F., Moser, G., Fallon, D., Farrell, C.A., Müller, C., Wilson, D., 2019. Rewetting degraded peatlands for climate and biodiversity benefits: results from two raised bogs. Ecol. Eng. 127, 547–560. https://doi.org/10.1016/j.ecoleng.2018.02.014.

Robinson, B., Winans, K., Kendall, A., Dlott, J., Dlott, F., 2019. A life cycle assessment of Agaricus bisporus mushroom production in the USA. Int. J. Life Cycle Assess. 24, 456–467. https://doi.org/10.1007/s11367-018-1456-6.

Rosner, B., 2011. Fundamentals of Biostatistics, 7th edition. Brooks/Cole Cengage Learning, Australia, Brazil, Japan, Mexico Korea, Singapore, UK, USA.

Royse, D.J., Baars, J., Tan, Q., 2017. Current Overview of Mushroom Production in the World. In: Cunha Zied, D., Pardo-Giménez, A. (Eds.), Edible and Medicinal Mushrooms. J Wiley & Sons, Hoboken, NJ, USA, pp. 5–13. https://doi.org/10.1002/9781119149446.ch2.

Roy, P., Dutta, A., Gallant, J., 2020. Evaluation of the life cycle of hydrothermally carbonized biomass for energy and horticulture application. Renew. Sustain. Energy Rev. 132, 110046. https://doi.org/10.1016/j.rser.2020.110046.

Rumpel, C., Amiraslani, F., Koutika, L.-S., Smith, P., Whitehead, D., Wollenberg, E., 2018. Put more carbon in soils to meet Paris climate pledges. Nat 564, 32–34. https://doi.org/10.1038/d41586-018-07587-4.

Saer, A., Lansing, S., Davitt, N.H., Graves, R.E., 2013. Life cycle assessment of a food waste composting system: environmental impact hotspots. J. Clean. Prod. 52, 234–244. https://doi.org/10.1016/j.jclepro.2013.03.022.

Shetler, G., Turetsky, M.R., Kane, E., Kasischke, E., 2008. Sphagnum mosses limit total carbon consumption during fire in Alaskan black spruce forests. Can. J. For. Res. 38, 2328–2336.

Šimanauskienė, R., Linkevičienė, R., Bartold, M., Dąbrowska-Zielińska, K., Slavinskienė, G., Veteikis, D., Taminskas, J., 2019. Peatland degradation: the relationship between raised bog hydrology and normalized difference vegetation index. Ecohydrology 12, e2159. https://doi.org/10.1002/eco.2159.

Simapro 9.5, 2023. PRé Consultants: Life Cycle Consultancy and Software Solutions. Amesfoort, The Netherlands.

Stichnothe, H., 2022. Life cycle assessment of peat for growing media and evaluation of the suitability of using the Product Environmental Footprint methodology for peat. Int. J. Life Cycle Assess. 27, 1270–1282. https://doi.org/10.1007/s11367-022-02106-0.

Strack, M., Davidson, S.J., Hirano, T., Dunn, C., 2022. The potential of peatlands as nature-based climate solutions. Curr. Clim. Chang. Rep. 8, 71–82. https://doi.org/10.1007/s40641-022-00183-9.

Taparia, T., Hendrix, E., Nijhuis, E., de Boer, W., van der Wolf, J., 2021. Circular alternatives to peat in growing media: a microbiome perspective. J. Clean. Prod. 327, 129375. https://doi.org/10.1016/j.jclepro.2021.129375.

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

Van Oers, L., de Koning, A., Guinee, J.B., Huppes, G., 2002. Abiotic Resource Depletion in LCA. Road and Hydraulic Engineering Institute, Ministry of Transport and Water, Amsterdam.

Van Post, L., 1924. Das genetische system der organogenen Bildungen Schwedens. Quatrième commission, commission pour la nomenclature et la classification des sols, Rome. Com. Int. Pédologie 287–304.

Vinci, G., Prencipe, S.A., Pucinischi, L., Perrotta, F., Ruggeri, M., 2023. Sustainability assessment of waste and wastewater recovery for edible mushroom production through an integrated nexus. A case study in Lazio. Sci. Total Environ. 903, 166044. https://doi.org/10.1016/j.scitotenv.2023.166044.

Vingiani, S., Adamo, P., Giordano, S., 2004. Sulphur, nitrogen and carbon content of Sphagnum capillifolium and Pseudevernia furfuracea exposed in bags in the Naples urban area. Environ. Pollut. 129, 145–158. https://doi.org/10.1016/j.envpol.2003.09.016.

Vos, D.J., Rustenburg, S., 2016. C12P1/00 - preparation of compounds or compositions, not provided for in groups C12P3/00 - C12P39/00, by using microorganisms or enzymes. RS54366B1.

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8.

Wever, G., Van Der Burg, A.M.M., Straatsma, G., 2005. Potential of adapted mushroom compost as a growing medium in horticulture. Acta Hortic. 171–177. https://doi.org/10.17660/ActaHortic.2005.697.21.

Ye, D., Hu, Q., Bai, X., Zhang, W., Guo, H., 2023. Increasing the value of Phragmites australis straw in a sustainable integrated agriculture model (SIAM) comprising edible mushroom cultivation and spent mushroom substrate compost. Sci. Total Environ. 869, 161807. https://doi.org/10.1016/j.scitotenv.2023.161807.

Young, G., Grogan, H., Walsh, L., Noble, R., Tracy, S., Schmidt, O., 2024. Peat alternative casing materials for the cultivation of Agaricus bisporus mushrooms – a systematic review. Clean. Circ. Bioeconomy 9, 100100. https://doi.org/10.1016/j.clcb.2024.100100.

Zied, D.C., Pardo Giménez, A., Pardo González, J.E., Souza Dias, E., Carvalho, M.A., Minhoni, M.T. de A., 2014. Effect of cultivation practices on the β-glucan content of Agaricus subrufescens Basidiocarps. J. Agric. Food Chem. 62, 41–49. https://doi.org/10.1021/jf403584g.