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BUILDINGS COMPONENTS AND BUILDINGS

Environmental potential of fungal insulation: a prospective life cycle assessment of mycelium-based composites

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Abstract

Purpose Bio-based insulation materials are one of the most promising solutions for reducing the environmental impacts of building envelopes. Among these materials, the environmental benefits of mycelium-based materials have merely been investigated, despite their promising technical and thermal properties. In this paper, we perform a first prospective cradle-to-grave life cycle assessment (LCA) of mycelium-based composite blocks.

Methods An attributional cradle-to-gate LCA of the laboratory production of mycelium-based composites was first performed, including 11 environmental impact indicators. Then, scenarios were defined to scale up the technology to the level of industrial production, including the remaining life cycle modules to perform a cradle-to-grave analysis. Biogenic and metabolic carbon were considered by applying the static −1/+1 approach and following the current LCA standards. Future-oriented energy and transport mixes were also included as an additional scenario, systematically modifying both the foreground and background data. Finally, the industrially scaled-up technology and alternative insulation materials were compared with these future conditions (as applied to both materials).

Results and discussion Considering climate change, the results are encouraging in comparison to those for traditional plastic insulation, but do not necessarily surpass those for other existing materials such as rock wool. However, trade-offs are observed in other indicators, for which mycelium-based composites tend to perform worse than traditional insulation materials. The industrial scale-up reduced impacts for most indicators, but a considerable trade-off was observed with regard to terrestrial ecotoxicity. The main driver for the remaining greenhouse gas (GHG) emissions was found to be the electricity use during the manufacturing phase. We consider the inclusion of the other life cycle stages as relevant, as this increased the GHG emissions by 10%. Limitations of the current LCA standards, however, are noted and discussed, especially regarding the cascading use of biogenic materials, and highlight the relevance of this case study.

Conclusions Mycelium-based composites show a potential for future development, but careful attention should be paid to reducing electricity needs in their manufacturing process. Further improvements could also be made by using fast-growing biogenic materials as a substrate. In particular, we encourage researchers to include all of the life cycle stages in future studies, especially if biogenic emissions are considered.

Keywords Attributional life cycle assessment (A-LCA) · Prospective LCA (pLCA) · Ex ante LCA · Biogenic carbon · Mycelium-based composites · Circular economy · Whole life cycle · Cradle-to-grave

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Highlights

- A first prospective cradle-to-grave life cycle assessment (LCA) of mycelium-based composites is provided.
- Future-oriented energy and transport mixes are included, modifying both the foreground and background data.
- The main driver for greenhouse gas (GHG) emissions is the electricity used during the manufacturing phase.
- Limitations of current LCA standards regarding the cascading use of biogenic materials are discussed.

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1 Introduction

The impact of buildings on climate change is undeniable. These are responsible for about 40% of the global energy consumption (European Commission 2020) and 37% of the related greenhouse gas (GHG) emissions (UNEP 2022). Ambitious emission reduction targets have been set in the European Union (EU), aiming for a 60% decrease in GHG emissions for buildings by 2030 (European Commission 2020). As about 80% of the buildings which will be in use in 2050 have already been built (Vilches et al. 2017), strategies to reduce these emissions will mostly rely on a necessary "renovation wave", which is seen as the backbone of the EU Green Deal (European Commission 2020). Indeed, the decarbonisation potential of the building stock through renovation measures is considerable (OIB 2020). One of the measures which was identified as the most effective regarding renovation is the improvement of the building envelope (Pomponi et al. 2015; Amini Toosi et al. 2020). This improvement is usually performed by adding thermal insulation to the existing building (Obrecht et al. 2023). However, emerging technologies or innovative materials which could be used for thermal renovation are rarely considered in such political agendas. The focus is usually directed toward the operational emissions, which mainly come from the energy used in the building. Thus, this tends to overlook the embodied emissions, which can be caused, e.g. by building material manufacturing, even though these have been shown to significantly contribute to the GHG emissions of buildings (Röck et al. 2020; IPCC 2022).

Strategies to reduce the embodied emissions of building materials have recently appeared more frequently in the literature (Alaux et al. 2022, 2023; Scherz et al. 2022; Greene et al. 2023). Especially concerning insulation, the use of bio-based materials is perceived as a promising strategy for reducing the environmental impacts of buildings due to their ability to store potentially large amounts of carbon (Asdrubali et al. 2015; Pittau et al. 2018; Kuittinen et al. 2021; Carcassi et al. 2022a; Galimshina et al. 2022). Innovative bio-based insulation materials are thus expected to be further developed in the next few years. One of these materials is mycelium-based composites (MBCs). The mycelium represents the vegetative part of fungal organisms, which penetrate the soils and can be grown into different shapes by using casting methods (Vašatko et al. 2022). When mixing the mycelia with other bio-based lignocellulosic materials (called substrates), such as wood sawdust, the mycelial network will begin spreading, consuming the substrate itself and acting as a "natural glue", subsequently binding the material together (Jones et al. 2017). Once this has dried, the material hardens considerably (Vašatko et al. 2022).

Mycelium-based insulation has multiple advantages and is mostly seen as an alternative to plastic-based insulation, such as expanded polystyrene (EPS), which has similar thermal characteristics (Jones et al. 2020). The advantages offered by these composites include their high thermal and acoustic properties, fire safety effectiveness and their ability to "upcycle waste" by digesting almost any substrate used. These substrates can range from waste products from the wood and paper industry to plant-based agricultural waste materials; their use, therefore, contributes to the transition to a circular economy (Robertson et al. 2020; Rafiee et al. 2021; Vašatko et al. 2022). In addition to the fact that they are entirely degradable, MBCs can also be reused at their end of life in a similar manner as a spent mushroom substrate, e.g. for energy production, as a soil amendment to improve soil quality and regulate pesticide behaviours in soil, for enzyme recovery, as fertilisers in agriculture and horticulture, or when directly reincorporated into the MBC production process (Phan and Sabaratnam 2012; Marín-Benito et al. 2016; Owaid et al. 2017; Grimm and Wösten 2018). Nevertheless, some obstacles are still being faced when attempting to industrially scale-up the use of such material, such as its rapid water absorption (Robertson et al. 2020) or the need to design appropriate industrial equipment for efficiently cultivating and drying the composite material (Rafiee et al. 2021). Although technical studies on these materials exist in the literature, most of these contain environmental claims, and proper scientific sustainability assessments of fungal-based composites are lacking (Elsacker et al. 2020).

Scientific environmental assessments can be performed by using life cycle assessment (LCA), a robust method based on the international standards ISO 14040/14044 (ISO 2006a, b), which were adapted to create the European standards EN 15978 for buildings and EN 15804 for building products (CEN 2012, 2019). Most LCAs are retrospective in that they are used to assess a product which has already been well defined in the market and for which (past) data have been collected; it is praxis based (di Bari et al. 2023). A prospective LCA, on the contrary, is carried out to assess a technology that is at a low technological readiness level (TRL) and to investigate how it might develop in the future. Arvidsson et al. (2018) defined a prospective LCA as a method that is applied "when the (emerging) technology studied is in an early stage of development (e.g. small-scale production), but the technology is modelled at a future, more-developed stage (e.g. large-scale production)". Within the realm of prospective LCA studies, a multitude of different approaches have been taken, and designations have been made (di Bari et al. 2023). Some authors might call this approach explorative LCA, ex ante LCA or anticipatory LCA (Cucurachi et al. 2018; Guinée et al. 2018; Thonemann et al. 2020).

The LCA literature on MBCs is both scarce and recent. Ng et al. (2021) performed the first cradle-to-gate LCA of fungal-like adhesive materials in Singapore, examining different environmental impact indicators. Although they identified influential hotspots in the transportation of these materials, they did not attempt to scale up the laboratory production to an industrial scale and did not include the whole life cycle of the product. Stelzer et al. (2021) then performed a cradle-to-gate LCA of the lab-scale production and scaled up the industrial production of fungal-based composite bricks in Germany, considering six impact indicators. They determined that most of the GHG emissions came from the electricity used and provided initial insights that revealed how the scaled-up technology could perform. The authors also suggested that other studies perform a cradle-to-cradle analysis in order to consider the recycling potential of the material and that they expand the scope of the industrial scale-up (as they had mostly considered improvements in energy efficiency). More recently, Carcassi et al. (2022b) performed a laboratory-scale cradle-to-gate LCA of a bamboo-mycelium-based composite material, including dynamic LCA calculations to additionally account for the carbon removal potential and the product valorisation at its end of life. These authors determined that the energy source used for drying the material was critical for the LCA results. They did not scale up the material to meet industrial-scale production requirements, but mention that this could be explored in future research, as could the use of more advanced low-energy technologies in the drying process. Finally, Livne et al. (2022) performed a lab-scale cradle-to-gate LCA of MBCs, considering the metabolic CO_2 emissions which result from the mycelium growth process. These authors found that the metabolic emissions accounted for 21% of all emissions, highlighting their relevance. They also discussed possibilities for scaling up some of the processes to a factory level.

In a review of the literature, no prospective cradle-to-grave LCA of mycelium-based composites was found, meaning that the technology has not been scaled up to an industrial production level as projected in the future, including a calculation of the life cycle biogenic and metabolic emissions. This poses a problem, because the environmental impacts of such materials might be highly underestimated, on the one hand, due to the lack of information about industrial upscaling and, on the other hand, due to the omitted life cycle stages. The goal of carrying out the current study was to perform a cradle-to-grave attributional and prospective LCA of mycelium-based composites, including 11 environmental impact indicators, and to provide future projections for the technology. To our knowledge, this is a novel goal. The assessment process was divided in four steps: (1) a traditional cradle-to-gate life cycle assessment of the material was performed at the laboratory scale, (2) scenarios were defined to scale up the technology to an industrial production level and to assess the remaining life cycle impacts of the product (for the cradle-to-grave analysis), (3) the planned increase of renewables in the energy and transport mix were included in the analysis as a future projection, and (4) the industrially scaled-up technology and alternative insulation materials were compared with these future conditions (as applied to both materials). The outcomes of this assessment are reported in this paper.

2 Methods

2.1 Goal and scope definition

This case study was performed to assess the potential impacts of MBCs (which can be used for insulation) at the lab and industrial scales, taking into account changes that may occur in the future. In this study, environmental hotspots and opportunities for reducing emissions were also identified. MBCs were also compared with alternative insulation materials, namely, plastic insulation and rock wool materials. To achieve these purposes, two functional units (FU) were considered:

- A $10 \times 10 \times 10$-cm block of insulation material. This reflects the lab-scale production most accurately and is equivalent to 0.212 kg of mycelium-based material;
- A mass of insulation material for an area of 1 m^2 which has a thermal resistance value of 1 m^2 K/W and a lifetime of 30 years. Choosing this mass enabled us to make a more relevant comparison with alternative insulation materials (considering only their thermal characteristics) and is equivalent to 11 kg of MBC material. The quantities of insulation needed to make up this FU, as well as the densities, thermal properties and references used for the calculation, are provided in the Supplementary Information (SI) for all of the compared insulation materials.

First, a cradle-to-gate LCA of the lab-scale production was performed. (The TRL was estimated to be 4–5 for this technology.) Then, scenarios were defined to project the technology on an industrial scale and to extend the prospective LCA to a cradle-to-grave analysis. Options that will potentially be available in the future, such as an increase of renewable sources for the energy and transport mix, were then included in the LCA, modifying both the foreground and background data. The cradle-to-grave LCA was performed in accordance with European standards regarding environmental assessments of construction products (CEN 2019). The assessment results were compared with those for other insulation materials. Figure 1 summarises the main methodological steps. In terms of the cutoff, the cloth bags used during the substrate preparation step and which can be reused without limit were not included in the LCA. The production of capital goods, such as infrastructure and equipment (e.g. laboratory machines) was also not considered, as previous researchers found that their influence on the LCA of construction materials was not significant (Silva et al. 2018). Apart from these two aspects, all available data were taken into consideration.

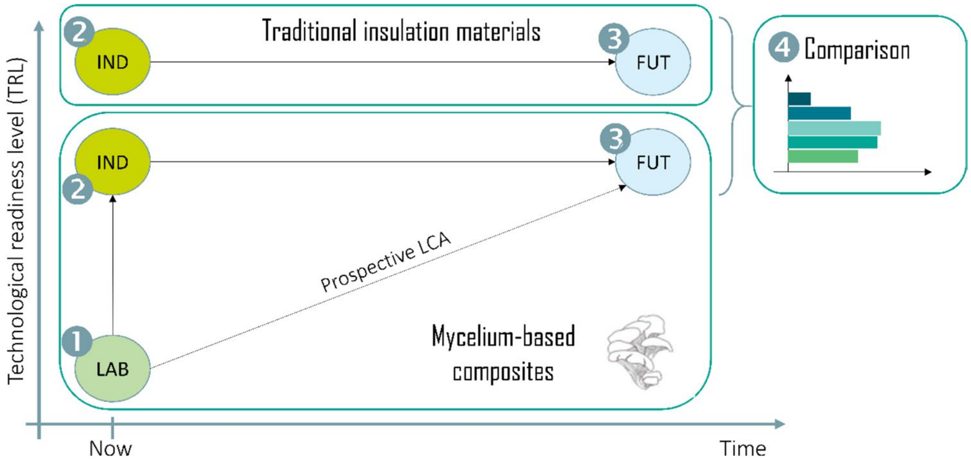


Fig. 1 Graphical representation of the methodology used in this study, inspired by Fig. 7 in Thonemann et al. (2020). Step 1 is the lab-scale LCA of the MBCs. Step 2 is the scale-up to the industrial scale for these composites and the cradle-to-grave LCA calculation. Step 3 is the future projection using future-oriented energy and transport mixes (for both the MBCs and traditional insulation materials). Step 4 shows the comparison of the cradle-to-grave LCAs of the MBCs with traditional insulation materials. The technological readiness level of both MBCs and traditional insulation materials is supposed to be the same in steps 2 and 3; they appear as one under the other only for graphical purposes

2.2 Description of the technology

The production of MBCs follows several steps, which can be initially divided into three merging courses: (1) the substrate preparation, (2) mycelium extraction and (3) the mould fabrication. These three stages are combined under sterile working conditions, i.e. inoculation, which is followed by the growth process and post-processing steps (Fig. 2). The following description corresponds to the images displayed in the provided figure.

Choosing the substrate depends on the mycelia strain involved and on the scale of the substrate pieces. For this research, wood of the European beech (*Fagus sylvatica* (S1)) was taken in the form of sawdust (S2) with pieces ranging from 1 to 2 mm in diameter. The substrate is prepared by soaking it for 24 h (S3), then manually draining the excess water by placing the wet material into a cloth bag, which can be reused as many times as needed (S4). The drained material is then placed into a polypropylene microfilter bag (S5), which is placed in an autoclave prior to the sterilisation process. The sterilisation process lasts 20 min once the autoclave has reached a temperature of 121 °C. The substrate is then cooled to room temperature and is ready for inoculation.

Mycelium extraction is a process that is carried out by mushroom producers by cloning, i.e. extracting a piece of tissue from a mushroom fruiting body (MY1) and allowing it to further grow (MY2) before inoculated grain spawn are produced (MY3). In this research, MY3 was taken as one of the most widespread forms of mycelia commercially available. The inoculated grain spawn was purchased from the Austrian *Pilzzucht*[1], and the chosen strain was an oyster mushroom mycelium strain (Pleurotus ostreatus).

The mould fabrication involves a process of laser cutting a PVC/PE foil in the dimensions of the desired object (M1), considering the shrinkage rate of the chosen substrate for the MBC. In this case, preliminary research showed that the shrinkage rate of the beech sawdust amounts to 10% (Vašatko et al. 2022). Perforating the mould surface is necessary, because it allows sufficient oxygen and carbon dioxide exchange during the growth process (M2).

Inoculation is a process that involves mixing the sterilised substrate with grain spawn. This process takes place in a sterile environment, usually in a laminar flow hood. The ratio of spawn to substrate is calculated by weighing the damp substrate and taking 5–10% of this weight for the grain spawn. The mould surface is then cleaned with a 70% ethanol solution. Once the ethanol evaporates from the mould's surface, the inoculated substrate is manually put into the mould and hand-pressed layer by layer. The filled moulds are then placed in a dark environment with temperatures ranging from 20 to 24 °C (growth process). After 14 days, the composite blocks are unmoulded and left to dry. In order to make sure that the fungal growth is terminated, the sample should be exposed to a minimal temperature of 40 °C for 30 min, which can be carried out in a conventional kitchen oven. Afterwards, the remaining moisture should be allowed to evaporate the composite block. Currently, passive solar heating is being used for this step.

[1] https://www.pilzzucht.at/

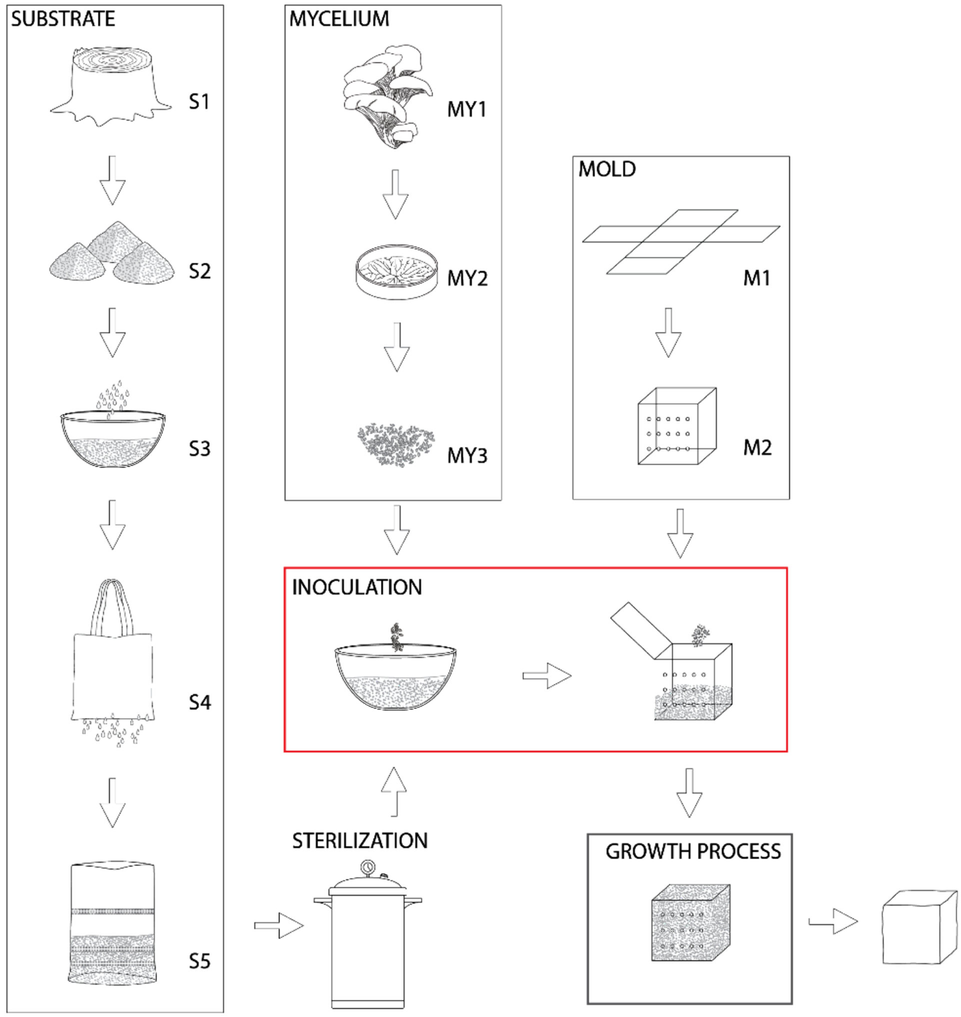


Fig. 2 Mycelium-based composite production process diagram for a 10 × 10 × 10-cm block

2.3 Life cycle inventory (LCI)

The data collection was primarily performed by carrying out laboratory experiments and by relying on the literature if no experimental data were available. The LCI, including the quantities and data sources, is present in the SI. The data quality was additionally assessed by using the adapted pedigree matrix (Ciroth et al. 2016) and is also provided in the SI. The ecoinvent database v3.8 (Wernet et al. 2016) was used as a background database. As LCA software, the open-source framework for LCA brightway2 was adopted (Mutel 2017) in combination with its activity browser (Steubing et al. 2020). In the following subsections, more detailed information is provided concerning the developed scenarios. Figure 3 represents the main processes and inputs of both the laboratory and industrial production of the mycelium-based composites, highlighting the additional steps included in the technological upscaling.

2.3.1 Lab-scale production

The material quantities used for the lab-scale production were measured on-site (in the laboratory) when producing one 10 × 10 × 10-cm block of MBC material. The energetical inputs in kilowatt-hour (kWh) (e.g. for sterilising, laser cutting) were calculated as follows: The process duration was measured in the laboratory (in hours) and multiplied by the power requirements (in kW) as provided in the manufactures' technical datasheets for the used equipment. Regarding the transportation distances, the mycelium-inoculated rye grain was directly bought from a local provider situated 70-km away from Graz (Austria). For the wood-based products constituting the substrate (sawdust), an average 100-km transportation distance was considered based on the latest published environmental product declarations (EPDs) for wood products in Austria (Kielsteg GmbH 2019). The modelling of

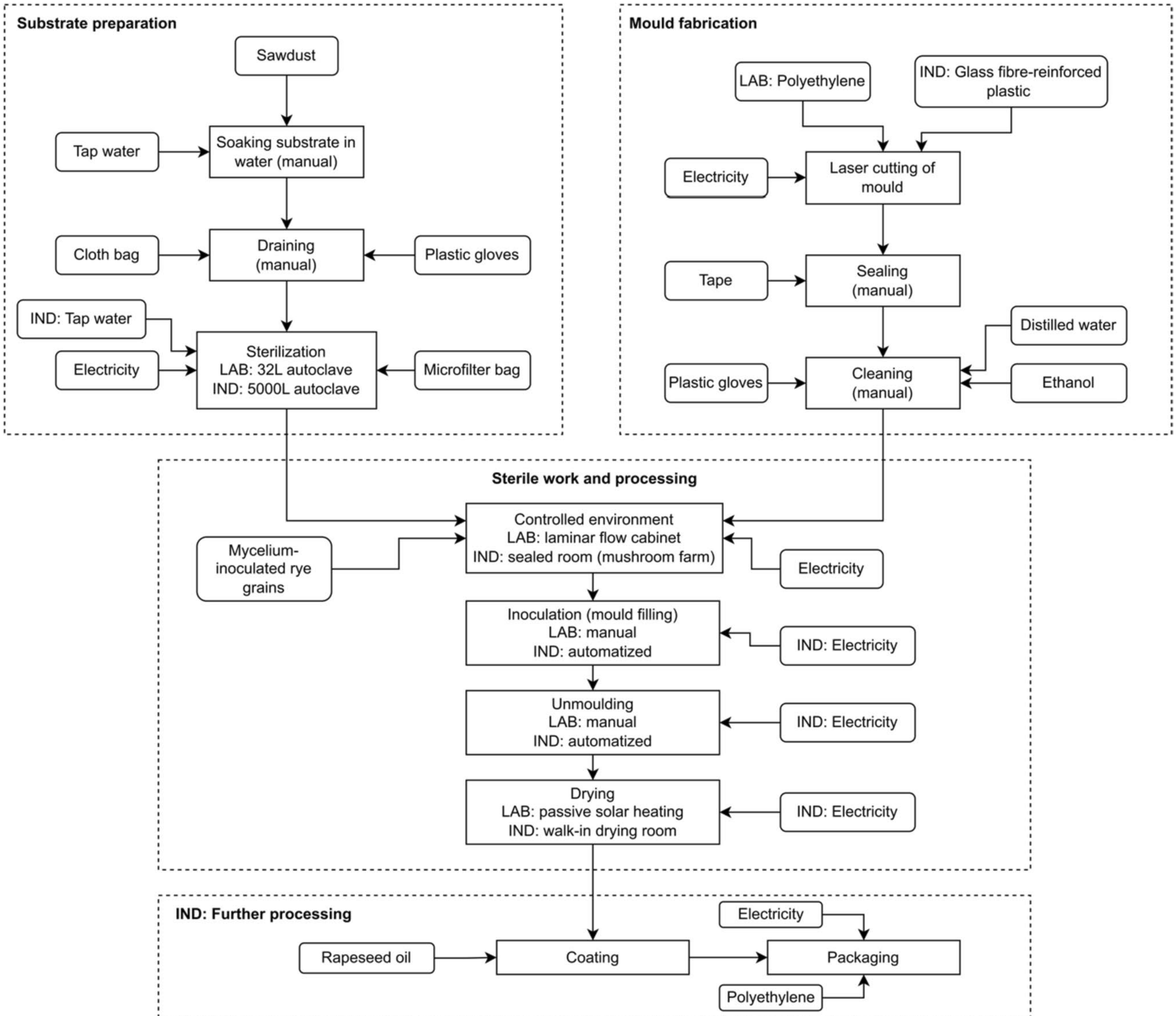


Fig. 3 Flowchart representing the main processes (rectangles with sharp edges) and inputs (rectangles with soft edges) of both the laboratory and industrial production of the mycelium-based composites (A1–A3). "LAB" (respectively, "IND") means that this process or input is specific to the laboratory (respectively, industrial) production

mycelium-inoculated rye grain was carried out based on previous studies (Leiva et al. 2015; Stelzer et al. 2021) but was adapted to the Austrian context. Finally, laboratory material such as plastic bags and gloves were modelled based on the recommendations of Stelzer et al. (2021) and Dorr et al. (2021).

2.3.2 Industrial scale-up: scenarios

In this section, the scenarios used to scale up the lab-scale production to an industrial-scale production are described. Our approach was to base the scenarios on those in existing industrial mushroom farms which have similar needs, e.g. in terms of sterilisation or control of air temperature and humidity.

- Substrate preparation: The main step in substrate preparation is the sterilisation process. Currently, a 32-L laboratory autoclave is being used for lab-scale production. At a large scale, we assumed the use of an industrial autoclave with a 5000-L capacity, which is specially manufactured for substrate sterilisation in mushroom farms (ROOETECH 2022). We adapted the electricity inputs by using the given power from the manufacturer, assuming that the autoclave would be filled to 80% capacity and that the duration of the sterilisation process remained the same. We also included additional water inputs of 600 kg/cycle, as specified in the technical data. We assume that the substrate would still be placed in propylene microfilter bags before being sterilised and that a pair of gloves would still be used during the whole substrate preparation process.

- Mould fabrication: Instead of low-quality plastic moulds which could only be reused four times, we assumed that, at an industrial scale, high-quality moulds such as those made out of acrylic glass or metal would be used and theoretically reused endlessly. Instead of following a cut-off like Stelzer et al. (2021), we assumed the use of glass fibre–reinforced plastic moulds which could be reused 100 times.
- Sterile working and processing: The processes of mixing the substrate with the mycelium and of moulding, growing and unmoulding the material usually take place in a sterile environment. In the lab, we used a laminar flow hood. On an industrial mushroom farm, it is likely that a sealed room with a controlled environment would be dedicated to these processes. To model this situation, we took the electricity inputs of cultivation in mushroom farms from Dorr et al. (2021) and adapted these based on the functional unit and the duration of the process, assuming a linear relationship. (Their cultivation process lasted 2 months, and, in our case, we only needed 2 weeks.) These electricity inputs included air purification, ventilation, temperature and humidity regulation, as well as the mixing and cooling of the substrate. We did not take the lighting into account, because the growing phase was almost entirely performed in the dark in our study. We also considered the additional electricity needed to fill in the moulds (i.e. this is currently being done by hand). As this part is usually not considered on the mushroom farms, we took the electricity input from the "injection moulding" process by ecoinvent as a proxy, which is normally used for moulding plastics. We assumed the same input for the unmoulding process. Finally, like Stelzer et al. (2021), we assumed that the drying would take place in a walk-in drying room (Totech 2020), and we used the technical data from the manufacturer to estimate the additional electricity inputs.

2.3.3 Further processing, transport and installation

As reported by Elsacker et al. (2020), post-processing methods for MBCs have been investigated to ensure, for instance, that they remain weather proof and to increase their durability. An example is the use of natural oils or polymers as a coating for the material. Shellac or linseed oil was used as examples. For this work, a coating from rapeseed oil was used as a proxy for linseed oil. Concerning the transport to the construction site and installation, the transport distances were assumed to be similar to those for other insulation materials produced in Austria. Based on published local EPDs (Fachverband Strohballenbau Deutschland e.V. 2019; Saint-Gobain ISOVER Austria GmbH 2019), an average transport distance of 150 km was considered. Wooden pallets were assumed to be used to transport the blocks. A reuse of 22 times was considered an average observed in LCA studies of pallets (Deviatkin et al. 2019). For this application, we presumed that the loaded pellets would be wrapped in plastic to prevent their possible degradation due to unwanted water exposure. The rapid water absorption capability of such composites was discussed in the Section 1 (Robertson et al. 2020). Finally, a material loss of 5% was assumed on site from sawing the blocks, as is commonly expected in the EPDs of insulation materials (Fachverband Strohballenbau Deutschland e.V. 2019; Saint-Gobain ISOVER Austria GmbH 2019). This material waste was assumed to be incinerated (see 2.3.5).

2.3.4 Use phase and reference service life

Provided that they are properly planned and correctly built, insulating materials manufactured from renewable raw materials should not alter in composition over time and should not contribute to environmental impacts during their use phase. However, a high degree of uncertainty regarding the reference service life (RSL) exists for MBCs, since they have merely been used in practice and mostly for short-term demonstration projects. As a first assumption, the RSL of the MBCs was set at 30 years, like that of other bio-based materials used as insulation (Lasvaux et al. 2020).

2.3.5 End of life and module D

Mycelium-based composites are created to be 100% natural and biodegradable. Multiple end-of-life treatments are possible, such as composting, incinerating with energy recovery or producing biogas by anaerobic digestion. However, one of our interests in performing this work was to explore the potential to recycle the composites as a new substrate in MBC production. Therefore, two end-of-life scenarios were investigated:

- The material is incinerated and the energy recovered, following the current practice for insulation produced with renewable materials (Bau EPD GmbH 2017). In module D, we assumed that the MBCs would have an upper calorific value of 17.3 MJ/kg (Bau EPD GmbH 2017) and that average emissions from district heating would be avoided through their incineration (FGW 2022).

- The composites are recycled to form a new substrate. However, it is unknown if the substrate could only be composed of a waste mycelium block, as some nutrients are consumed during the growth process. For this analysis, we assumed that 70% of the MBC could be recycled as substrate for a new one. In module D, we accounted for grinding the waste mycelium block and the benefit of not using virgin sawdust.

The demolition from the building was expected to be manual, assuming that the blocks are not glued or mechanically fixed. Similarly to the installation phase, a material loss of 10% was considered, the material was expected to be incinerated and a transport distance to the processing facility of 50 km was considered (Fachverband Strohballenbau Deutschland e.V. 2019; Saint-Gobain ISOVER Austria GmbH 2019).

2.3.6 Future-oriented technological scenarios and systematic inventory modification

In a prospective LCA, it is common to study emerging technologies which are not yet on the market. Such technologies might take decades to fulfil all the technical and legal requirements which would allow them to be sold and produced at an industrial scale. By that time, different technologies and markets than those currently available today will have been established. In this study, we included the future evolution of the electricity, energy and transport market mixes. We assumed that mycelium-based composites would need at least 10–15 more years before they could be produced on a full industrial scale. For this reason, we chose 2035 as a time horizon. To consistently modify the electricity generation, freight transportation and the supply of fuels to consider all foreground and background processes listed in the ecoinvent database, we relied on *premise*, a plug-in for brightway2 which was developed to align life cycle inventories with the results of integrated assessment models (IAMs) (Sacchi et al. 2022). IAMs can be used to explore possible future socioeconomic and technological pathways and have a great potential for supporting prospective LCA studies (Mendoza Beltran et al. 2020). These models rely on the Shared Socioeconomic Pathways (SSPs), a family of climate scenarios which feed into the work of the Intergovernmental Panel on Climate Change (IPCC). More detailed information about the SSPs can be found in Hausfather (2018). In this study, we used the Integrated Model to Assess the Global Environment (IMAGE) (Stehfest 2014) as an IAM and the "SSP2-RCP26" as a scenario, meaning that the SSP number 2 "Middle of the Road" was combined with the upper climate target of the Paris Agreement as a boundary condition (just below + 2 °C of atmospheric temperature increase by 2100) (Hausfather 2018). This scenario is not meant to be used as a prediction of the future, but rather to highlight the possible influence of future electricity, energy and transport mixes on the prospective LCA of MBCs.

2.4 Biogenic carbon

In this study, biogenic carbon was considered by applying the static −1/+1 approach, following the current version of the EN 15804 standard (CEN 2019). The quantification of the biogenic carbon contained in a wooden product is based on the EN 16449 (CEN 2014).

In general, to calculate the amount of CO_2 sequestered in a wooden product, it is necessary to know the carbon content, the moisture content, the raw density of the wooden biomass at the respective moisture content and the volume of the wooden product. For this study, a biogenic carbon factor (C-factor$_{wood}$) given by Diestel and Weimar (2014) is applied, which already incorporates an average moisture content for the wooden material, allowing the direct use of the C-factor$_{wood}$ to quantify the biogenic carbon content in the material. Applying this factor, the sequestered CO_2 can then be calculated by taking the molar masses of C and CO_2 into account. CO_2 has a molar mass of 44 g per mole, while C has a molar mass of 12 g per mole, resulting in a mass ratio of CO_2 to C of 44/12 = 3.67. To quantify the sequestered amount of CO_2, the given C-factor$_{wood}$ from Diestel and Weimar (2014) is multiplied by this factor 3.67.

- Module A1–A3: The CO_2 sequestered by the substrate material beech sawdust (*Fagus sylvatica*) is calculated via the C-factor$_{wood}$ of one ton of beech sawnwood as given by Diestel and Weimar (2014). According to this source, the C-factor$_{wood}$ of beech sawnwood is 0.463. Due to the solely mechanical treatment process of sawing, the C-factor$_{wood}$ of the beech sawnwood is assumed to be equal to the beech sawdust used in this study. The use of this C-factor$_{wood}$ also allows the influence of moisture on the substrate to be neglected in the substrate preparation step (step S3 in the production process). This C-factor$_{wood}$ is multiplied by the used substrate mass and the factor 3.67, which results in the biogenic CO_2 sequestered in the block. This biogenic CO_2 is booked into the system via the factor "−1", under the assumption that the wood has been extracted from a non-native forest.

  The CO_2 emissions occurring due to the biological respiration process of the mycelium in A1–A3 are based on the study by Pavlík et al. (2020). The published average respiration emissions for the growth of *Pleurotus ostreatus* on beech sawdust (*Fagus sylvatica*) for a growing duration of 14 days (i.e. week 1 and week 2 in the source) are taken into account. Those metabolic biogenic CO_2 emissions are booked out of the system via the factor "+1".

- Module A4–A5: For module A4, it is assumed that the pallet used for transportation of the blocks entered and left the system boundary as a whole. Therefore, carbon sequestered via the wood used for the pallet was not considered in the biogenic carbon assessment of this study. In module A5, material losses of 5% are assumed to occur due to sawing of the blocks on site. This installation waste is assumed to be incinerated, thus emitting biogenic CO_2 to the atmosphere. The biogenic CO_2 is booked out via the factor of "+1".
- Module C1–C4: It is assumed that 10% material losses occur in module C1 due to the deconstruction of the blocks. This 10% loss is assumed to be incinerated; thus, 10% of the biogenic carbon content in the block is emitted to the atmosphere as biogenic CO_2 in C1 and booked out of the system via the factor "+1".

  As mentioned in Section 2.3.5, two distinct scenarios for the end of life are assumed. Scenario one represents the incineration of the block to thermally recover the heat energy (incineration scenario), and scenario two represents the recycling of 70% of the mycelium block as a substrate for a new mycelium block (recycling scenario).

  In the incineration scenario, the block is transported to an incineration facility and incinerated for energy recovery, leading to the full emission of the remaining biogenic carbon as biogenic CO_2 to the atmosphere. Following the logic, the biogenic CO_2 is booked out of the system via the factor "+1".

  In the recycling scenario, a transition of 70% of the mycelium material to a downstream product system (i.e. the second life cycle of the mycelium substrate) is assumed, while only 30% of the material are assumed to be incinerated for energy recovery. Nevertheless, according to the standards, all biogenic CO_2 must be characterised with the factor "+1" when leaving the system boundary (CEN 2019). Hence, the remaining biogenic CO_2 is fully booked out of the system at the end of the life cycle, although the material is not incinerated.
- Module D: In the incineration scenario, an upper calorific value of 17.3 MJ/kg (Bau EPD GmbH 2017) is taken into account, resulting in the avoidance of average emissions from district heating.

  In the recycling scenario, the substitution of 70% of the raw sawdust material by the recycled mycelium block substrate is assumed. As described by Ouellet-Plamondon et al. (2022) and also indicated by Hoxha et al. (2020), if the material is reused or recycled, the biogenic carbon is passed to the next life cycle. The remaining biogenic carbon for recycling is thus once again booked into module D as biogenic CO_2 with the factor of "−1".

Figure 4 provides an overview of the biogenic CO_2 fluxes as accounted for by applying the static "−1/+1" method in this study.

2.5 Life cycle impact assessment (LCIA) and interpretation

The impact assessment was conducted using the CML 2001 baseline method (version 4.8) (Guinée 2002) and included the following impact indicators: abiotic depletion (material and energy resources), acidification, eutrophication, global warming potential (GWP 100a), ozone layer depletion, human toxicity, freshwater aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity and photochemical oxidation. For readability, the results in this paper are mostly shown with regard to the GWP, but the same information concerning the other impact indicators is available in the SI. The biogenic carbon calculation provided here is solely incorporated for the GWP 100a impact indicator.

Due to the high levels of uncertainty associated with the RSL and the fact that the RSL was identified as one of the most influential uncertainties in LCAs of buildings and construction materials (Goulouti et al. 2020), both 15 and 30 years were considered a RSL when comparing the industrially scaled-up technology to other insulation materials in the sensitivity analysis. Finally, a Monte Carlo analysis based on the data quality assessment (pedigree matrix) was also performed for the industrially scaled-up technology in the SI.

3 Results

3.1 Main results for one mycelium-based composite block

3.1.1 Lab scale and industrial scale: cradle-to-gate

In this section, the cradle-to-gate results for the production of one MBC block at both the lab and industrial scales are presented. Biogenic and metabolic emissions are not included here. Figure 5 presents the evolution of the environmental indicators when progressing from the lab scale to the industrial scale. For 9 out of the 11 indicators, a reduction in impacts is observed, with a maximum of 55% for abiotic depletion. Conversely, upscaling the process results in an increase in the impacts for terrestrial ecotoxicity and eutrophication, respectively, of 120% and 6%.

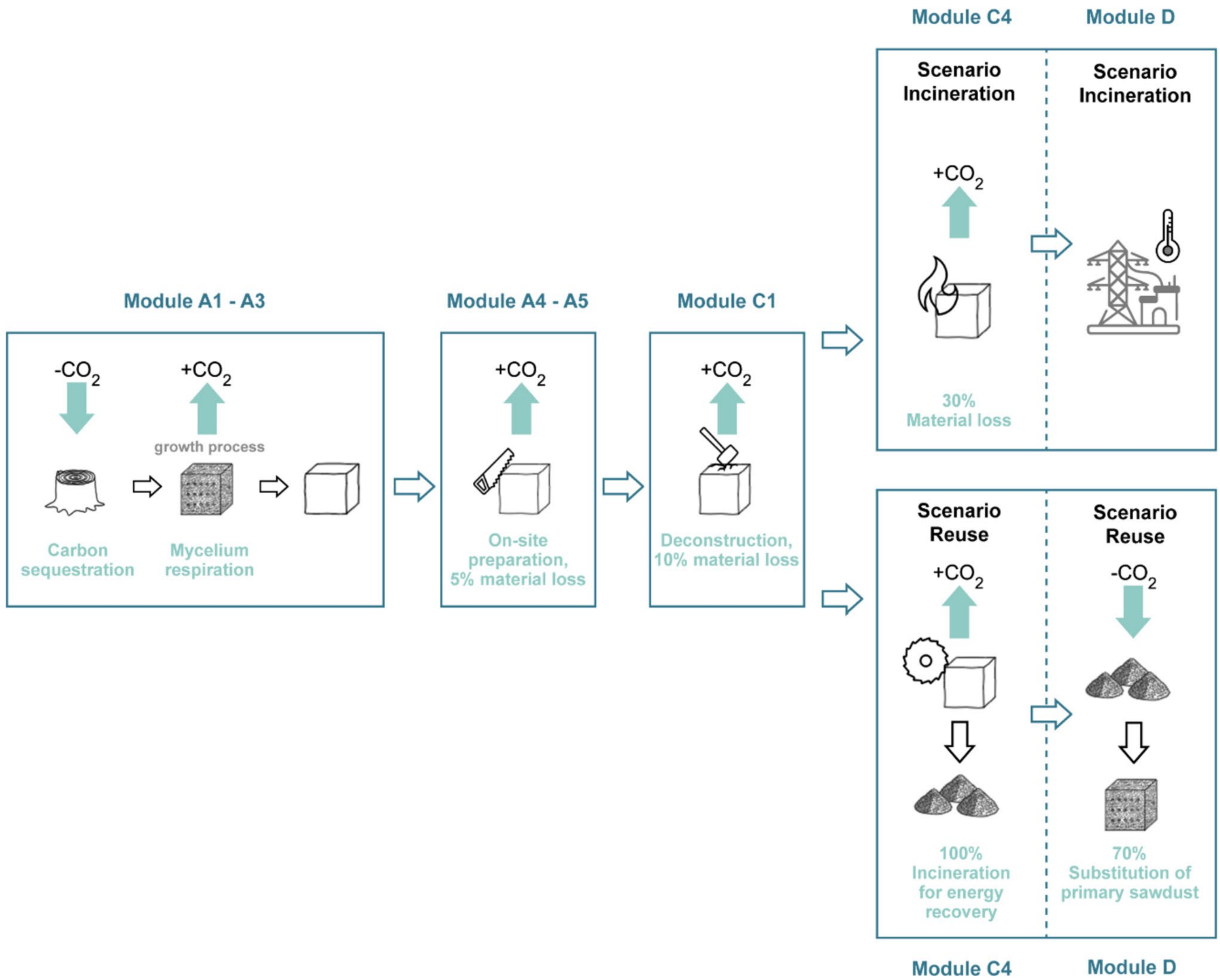


Fig. 4 Overview of biogenic CO_2 uptake and emissions throughout the life cycle of the mycelium-based composite block, applying the static "−1/+1" accounting approach

Figure 6 displays the results of a contribution analysis of the cradle-to-gate GHG emissions from a MBC block on the lab and industrial scales. For readability, the results are only shown for climate change, but the results for the remaining indicators can be found in the SI. In all, 39% of the GHG emissions from the lab-scale production originated from the electricity used. The remaining emissions mostly come from the ethylene (27%) and its disposal (19%), a material which is used to make the bags used for sterilising the substrate, but also to create the moulds (which can only be reused four times). In the case of the industrial production, an overall reduction of 45% in the GHG emissions can be achieved. The emissions are dominated by the electricity needs, which account for 64%. Additional electricity inputs have been added for the scaling up, for example, for mixing the substrate or drying the blocks. Ethylene and waste polyethylene are only responsible for 7% and 5% of the emissions, respectively; we assumed in the scaling up that the moulds were of higher quality and could be reused 100 times. The "rest" part of the emissions is processes which do not account for more than 5% of the total.

3.1.2 Whole life cycle emissions at the industrial scale

In this section, the cradle-to-grave GHG emissions of the MBCs are presented. The prospective background database has not yet been used, and the emissions are given per block with 10 × 10 × 10-cm dimensions. The results concerning other impact indicators can be found in the SI. In Fig. 7, the GHG emissions are allocated to the life cycle stage in which they occur, based on the usual decomposition process defined in the EN 15804 (CEN 2019). Modules A1 to A3 include the raw material supply, transport of the raw

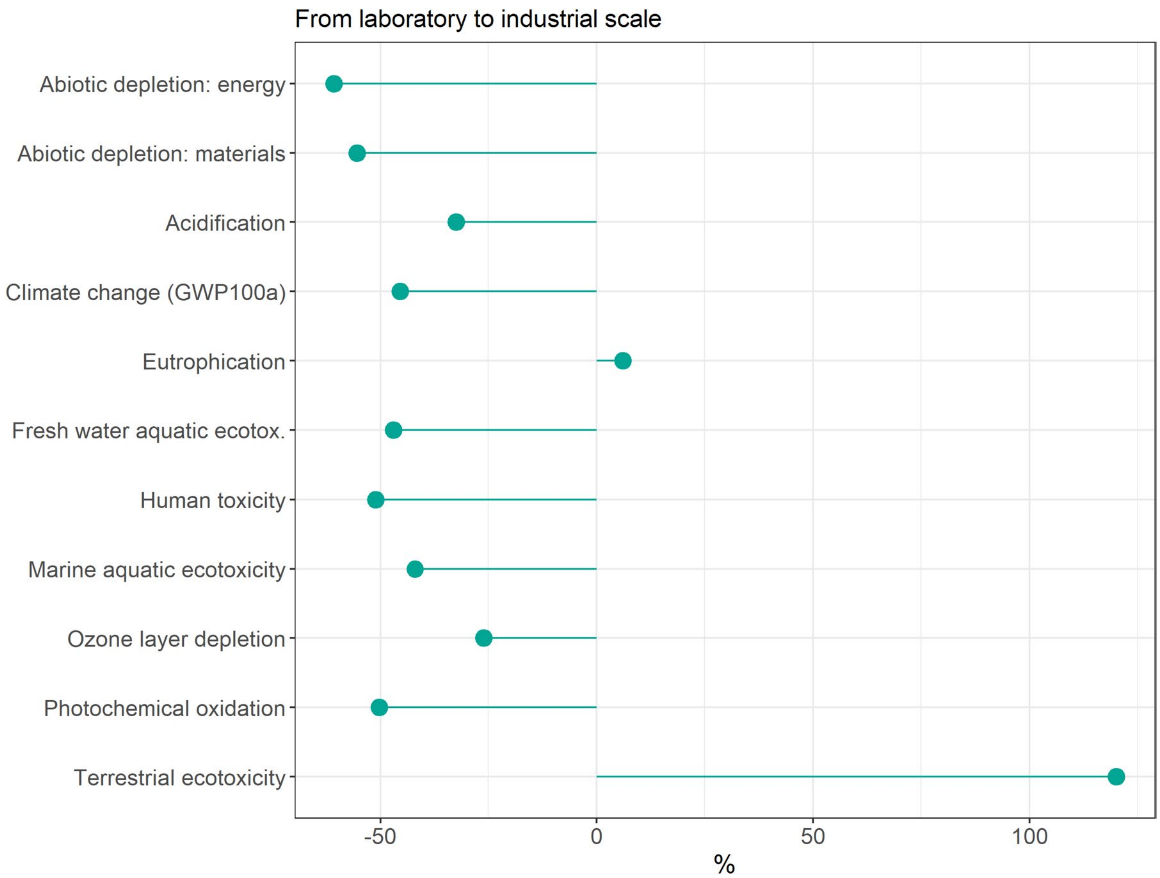
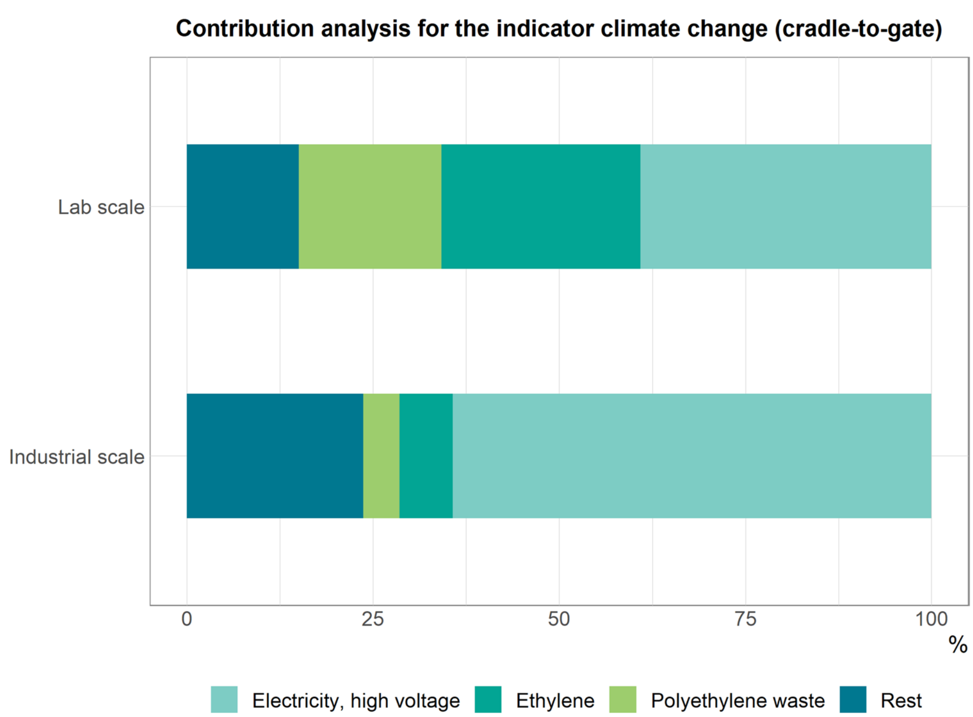


Fig. 5 Evolution of the cradle-to-gate environmental impacts when scaling up the production of a mycelium-based composite block

Fig. 6 Comparison of the cradle-to-gate GHG emissions from a mycelium-based composite block at the lab and industrial scales



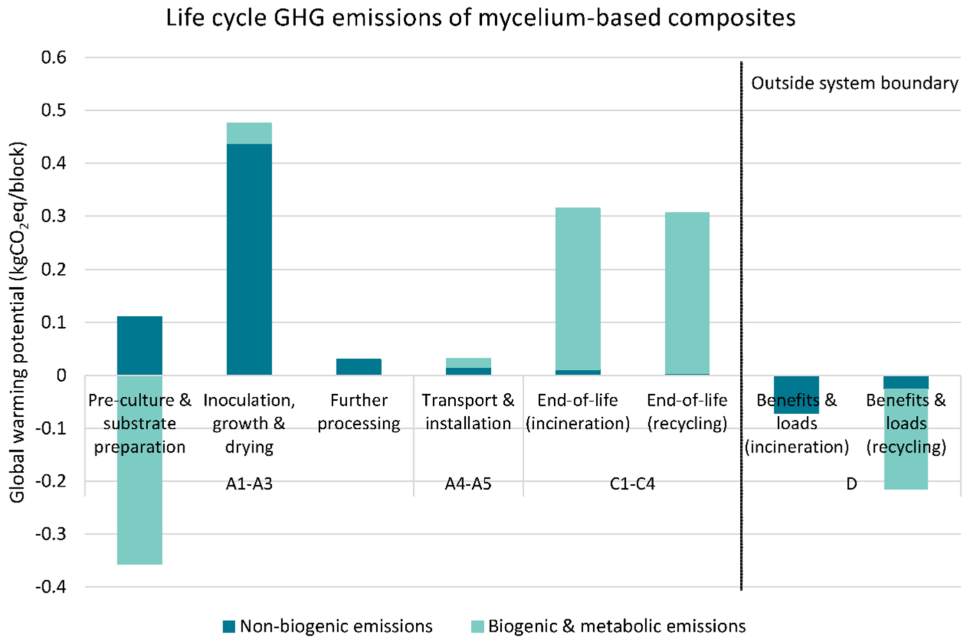


Fig. 7 Cradle-to-grave greenhouse gas emissions (with highlighted biogenic and metabolic) of the industrially scaled-up mycelium-based composites, using the current background database (not projected into the future) and including two end-of-life scenarios. The emissions are given per block with 10 × 10 × 10-cm dimensions

materials and product manufacturing (cradle-to-gate). Modules A4 and A5 represent the transport to the construction site and the installation in the building. Finally, modules C1 to C4 describe the end of life of the product, and module D accounts for benefits and loads that occur outside of the system boundary, such as typical benefits from recycling the material. No emissions occur during the use of the material, which is why modules B1–7 are not presented here. As mentioned in the Section 2, two end-of-life scenarios are considered (therefore affecting modules C1–4 and D), either incineration of the material with energy recovery or recycling part of the material (70%) as a new substrate in the mycelium production process, while the rest is being incinerated as well. Additionally, biogenic and metabolic emissions are highlighted.

Most of the non-biogenic GHG emissions (95%) can be attributed to the manufacturing stage and especially to the inoculation, growth and drying procedures. As previously shown, most of the emissions come from the electricity needed to ensure a sterile working environment with controlled light, humidity and temperature. The remaining emissions originate from the transport and installation (3%), as well as the end-of-life processing (2%). In the currently published studies, only the manufacturing phase is considered without including further processing. Doing so would, in our case, lead to an underestimation of the life cycle GHG emissions by about 10%. Regarding biogenic emissions, the amount of carbon stored in the substrate is considerable, representing more than 60% of the manufacturing fossil GHG emissions. Part of this carbon (about 10%) is already emitted during the inoculation, growth and drying phases in the form of metabolic emissions from the growing phase of the mycelium, which emits CO_2. The rest is emitted when incinerating the waste material from the installation or at the end of life. Based on the EN 15804 (CEN 2019), the total sum of these biogenic/metabolic emissions is zero.

Regarding the end of life, in the first scenario (incineration), module D takes into account the replacement of district heat by the thermal energy produced when incinerated the blocks at their end of life, which explains the negative value. In the second scenario (recycling), 30% of the block is also incinerated, and 70% of the block is recycled. For this recycled part, the additional energy needed for grinding was included as a load, and the production of wood avoided was seen as a benefit. According to the standard, the biogenic carbon has to be accounted for as a "+1" in module C because the material crosses the system boundary. Furthermore, as indicated in previously cited sources (Hoxha et al. 2020; Ouellet-Plamondon et al. 2022) when using the "−1/+1" approach, the biogenic carbon is transferred to a subsequent product system as "−1" if it is recycled, hence the negative bar of GWP for the recycled scenario.

3.2 Comparison with traditional technologies

Figure 8 presents a comparison of the environmental impacts of the mycelium-based composites, PUR, XPS and rock wool. To ensure fairness in the comparison, the second functional unit was used, i.e. the mass of insulation material which involves a thermal resistance value of 1 m^2 K/W for an area of 1 m^2 and for a lifetime of 30 years. (The calculation of the reference flow is provided in the SI.) The modified ecoinvent database reflecting the prospective "SSP2-RCP26" scenario in 2035 was applied to

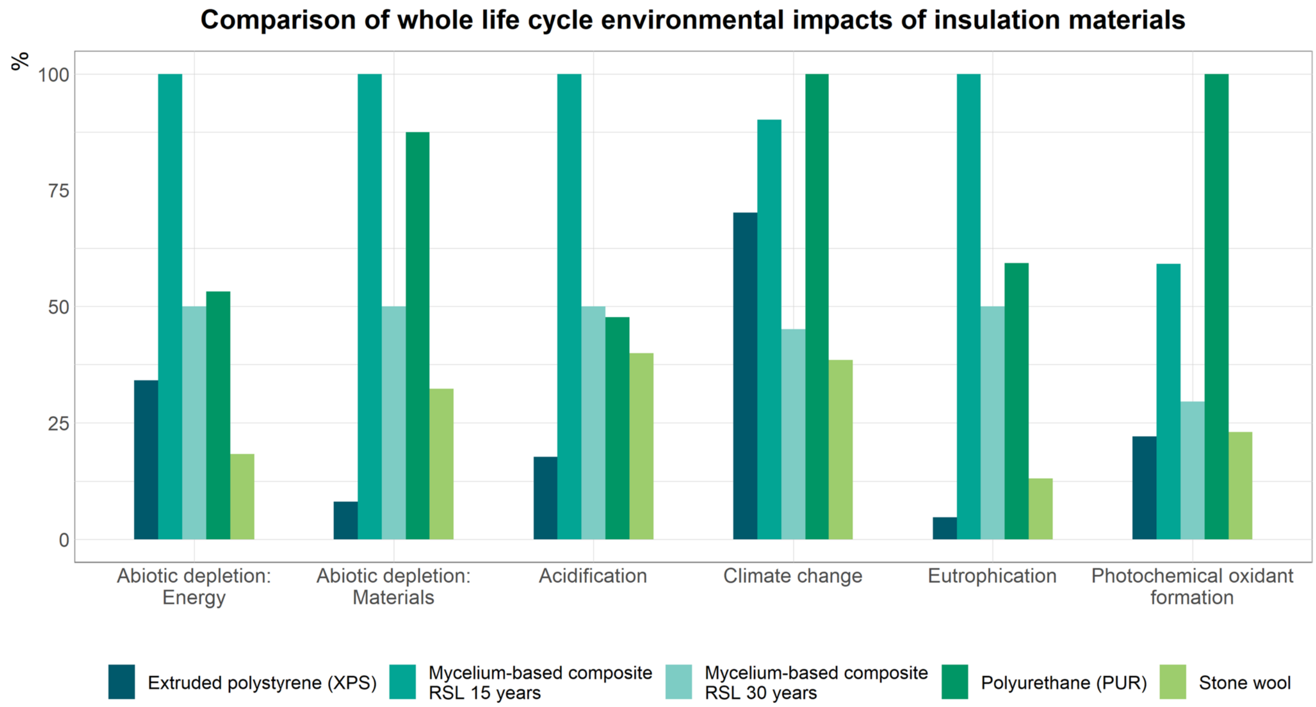


Fig. 8 Comparison of the cradle-to-grave LCAs with prospective background database of mycelium-based composites with traditional insulation materials

all insulation materials. The results for two possible RSLs for the MBCs are shown as 15 or 30 years. For each impact indicator, the results were normalised by the material which had the highest impact. Neither the ecotoxicity nor the ozone depletion indicators are displayed; the SSPs are based on climate models. The results for indicators other than climate change are highly uncertain, and especially for the previously mentioned ones; hence, we chose not to provide results for these indicators. For example, planned bans on certain chemicals were not included in the model. The future potential recyclability of metals was also not included, which is why careful interpretation of the results for the displayed indicators is also recommended.

The RSL of the MBCs was confirmed to have a high influence on their environmental impacts; if the RSL is only 15 years, then, the MBCs have the highest impacts for most indicators except for climate change and photochemical oxidant formation, for which polyurethane shows higher impacts. If the material is proven to be long lasting, it might represent a more interesting alternative to the currently used plastic insulation. Regarding climate change, the GHG emissions from MBCs are about half those from PUR and XPS. However, they are still slightly lower than the emissions from rock wool. Higher impacts for MBCs are also observed with respect to other indicators, such as acidification or eutrophication. MBCs indeed lead to higher impacts than stone wool and extruded polystyrene for all impact indicators, except for climate change, which signals a possible negative trade-off. Concerning acidification and eutrophication, these seem to be driven by agricultural products, such as the rapeseed oil used as a coating and the rye grains used to grow the mycelium in the pre-culture stage. For the other impact indicators, most of the impacts are triggered by the polyethylene and electricity use. Such trade-offs should be examined when developing the technology further. However, as mentioned, the results for these other impact indicators might not entirely be accurate.

4 Discussion

4.1 Obtained results in light of the published literature

Our cradle-to-gate lab-scale results show considerable differences from those of previous studies (Stelzer et al. 2021; Carcassi et al. 2022b; Livne et al. 2022). While the GWP results reported in these studies range from 250 to 370 $kgCO_2$ eq/m^3, our GHG emissions are about three times higher. The exact reasons for these variations are not yet known; the level of detail and transparency regarding the inventories provided in the respective studies did not allow for a precise comparison. Small differences could, nonetheless, be spotted: In some cases, for example, the disposal of the gloves or plastic bags used in the lab was not considered. Alternative LCA databases were also used for the modelling. In any case, the cubic metre is not an adequate functional unit for making comparisons, as the mycelium blocks have different mechanical and thermal properties and the authors of these studies did not always use the material for the exact same purpose. The substrate used was also not always the same: While we used sawdust, other studies examined straw, bamboo, wood chips and other materials.

When scaling up the technology, we increased the level of detail as compared to that used in previous studies by including more processes, such as additional energy inputs for, e.g. filling the blocks and for controlling the room temperature. The GHG emissions could be decreased by 46%. Stelzer et al. (2021) performed a simplified scaling up of their mycelium block and observed a decrease of 68% in the GHG emissions. This higher reduction could be due to the fact that they only scaled up the sterilisation and drying processes, but did not include additional energy inputs, for example, for filling the moulds. These authors also assumed that no further substrate bags would be needed, an assumption which we could not verify and did not include in our study. Compared to our study, these authors did not observe an increase in the impacts for eutrophication, although the reduction in emissions is quite small (13%). They also did not include terrestrial ecotoxicity, in which we observed a high trade-off of impacts. Livne et al. (2022) also briefly discussed potentially scaling up their mycelium block, mainly in terms of energy savings. These authors estimated energy savings of 7.6% for the sterilisation process and of 8.2% for drying. However, they acknowledged that the method cited by Stelzer et al. (2021) is more accurate, and we also based our scaling-up procedure on this approach for this reason. One limitation of our scaling-up procedure is that we did not consider advanced industrial processes, such as reusing the heat from sterilisation for the drying process or using solar heating system for drying, which could be envisioned, as the necessary heating temperature is relatively low (minimum of 40 °C).

In this paper, we proposed an initial approach that can be used to perform whole life cycle modelling for mycelium-based composites. Only including emissions from the production would underestimate the life cycle emissions by at least 10%. We encourage researchers to include all of the life cycle stages in future studies and to refine the scenarios regarding the installation and the end of life of the material. In particular, we strongly disfavour including biogenic carbon emissions if the whole life cycle is not considered or at least the end-of-life biogenic emissions, following the approach described by Carcassi et al. (2022b). In addition to not entirely adhering to the existing LCA standards, studies such as those by Livne et al. (2022) may mislead future technological advances by creating the false impression that MBCs are negative emission technologies. A more detailed discussion regarding biogenic carbon accounting can be found in the following section. If the technology is to be developed further, attention should be paid to the energy inputs during the inoculation, growth and drying phases of the block, as well as the material used for the moulds, which were hotspots in our analysis.

4.2 Biogenic carbon modelling and recycling: limits of the current standard

In the course of assessing the biogenic carbon of the mycelium block, certain methodological questions arose in relation to the end of life accounting when using the static −1/+1 method. When observing the results in module C, it is rather surprising that the results regarding the emissions in both scenarios seem to be nearly the same. Although the methods were applied correctly, these results do not represent the actual reality, as emissions would only occur when the biogenic material is incinerated. "Booking out" the biogenic carbon of the product system as +1 in module C certainly does not represent the reality in the recycling scenario, where the carbon remains stored in the material during the second product life cycle. This "artificially" +1 of emissions accounted in module C can lead to misinterpretations of the results and indicate certain weaknesses of the static −1/+1 approach when performing biogenic carbon accounting.

For the second end-of-life scenario "recycling", we applied the approaches described in the literature to deal with the biogenic material in module D. When "booking out" the biogenic carbon in module C as a +1 emission, the amount of biogenic carbon recycled (in our case, 70%) has to be "booked back in" again in the subsequent product cycle as −1 in module D in order to proper reflect the cascading use of the material. This also seems to be the path that standardisation will be taking with regard to the environmental declarations of wood products (CEN 2023). What is not reflected by taking this approach is the fact that the primary material (in our case, sawdust from wood) is substituted. In a secondary life cycle, this approach would not enable the user to distinguish between the use of primary raw sawdust material and the use of a recycled substrate from the previous life cycle. Both would result in the same −1 accounting of biogenic carbon. Since the use of secondary material is the more favourable option, allowing a cascading use of biogenic material and a storage of biogenic carbon in the built environment for a longer time, using this secondary material should actually lead to a better result when accounting the emission. Here, we identified a potential for improving the end-of-life accounting of biogenic materials to push the cascading use for biogenic materials. A next step could be to apply a more detailed approach, such as that conducted by Allacker et al. (2017), but for biogenic materials.

As discussed, some methodological drawbacks can be observed when applying static assessment methods such as the −1/+1 approach for biogenic carbon accounting. In this respect, the latest literature uniformly points toward dynamic assessment methods, e.g. the development by Levasseur et al. (2010), being the favourable option when dealing with biogenic carbon (Hoxha et al. 2020; Arehart et al. 2021). Thereby, more transparent results can be obtained by incorporating time-dependent aspects in the assessment, such as the growth rate of the used biogenic material. In relation to this study, we noted that research gaps are opening up that will allow the incorporation of a dynamic LCA assessment of the mycelium block by using fast-growing bio-based materials. Regarding limitations of the biogenic carbon assessment in this study, we only considered CO_2 emissions for the incineration, but in reality, some minor amounts of methane and nitrous oxide are also emitted. These emissions are assumed to be negligible with respect to the focus of our study.

4.3 Prospective background database: influence and limitations

To consider the time that might be required before MBCs can be produced on an industrial scale, we included the potential future evolution of the electricity, energy and transport market mixes in 2035, a time by which we assumed the technology might be available on the market. We automatically adapted all the foreground and background processes of the ecoinvent database based on the “SSP2-RCP26” scenario, a pathway derived from the upper climate target defined in the Paris Agreement as a boundary condition. By doing so, we observed an 84% reduction in the GHG emissions from the MBCs, which reveals how heavily MBC production depends on energy. This result is similar to that of Carcassi et al. (2022b) who observed a roughly 70% reduction in the GHG emissions for their MBC when changing the foreground energy inputs from the current energy mix to a 100% renewable one. Previous work on construction materials has already provided information about high GHG emission reductions for insulation materials when changing the electricity sources in the foreground and background to renewable sources, i.e. up to 83% (Potrč Obrecht et al. 2021; Zhang 2022). One limitation of this study and the application of the SSPs to the ecoinvent database is that these do not yet take into account technologies which might be available in the future. Future studies could include new production routes, for example, including carbon capture and storage or new heating technologies, based on the work of Alaux et al. (2023).

By only changing the foreground energy inputs, we could achieve a 64% reduction in GHG emissions, which gives an indication of the influence of the background data. Adapting the background data seems to be responsible for roughly 20% of the observed GHG reduction. This is fairly influential, but not as much as in other contexts; the influence of the background was thus far mostly investigated in relation to transportation (and more specifically, electric vehicles), where it was found that changing the electricity source in the background could influence the climate change impacts by up to 80% (Cox et al. 2018, 2020; Mendoza Beltran et al. 2020). In the case of some metals, the influence of the background electricity source was found to be up to 63% regarding climate change and up to 43% for human toxicity (Harpprecht et al. 2021). The number of investigations on the background’s influence on building materials is, however, rather limited. Apart from the previously cited studies (Potrč Obrecht et al. 2021; Zhang 2022), Zhong et al. (2021) conducted a study translating the SSPs into life cycle inventories for building materials on the global scale and including background data modification. These authors also investigated ways to meet the targets defined in the Paris Agreement. Therefore, we recommend further investigating the influence of systematic background data modification on impact results for building materials and buildings.

5 Conclusion

We performed the first prospective cradle-to-grave LCA of mycelium-based composites, including an industrial scale-up and future-oriented energy and transport mixes. We found that the industrial scale-up led to a reduction in impacts for most indicators, including one of 45% for the GWP, but a considerable trade-off was observed for terrestrial ecotoxicity. The main driver for the remaining GHG emissions was found to be the electricity used during the manufacturing phase. Possibilities to reduce the electricity needs could be further investigated, such as cold sterilisation processes, or alternative heating technologies, such as infrared lighting. Further improvements could come from using fast-growing biogenic materials (such as straw) or biochar as a substrate instead of sawnwood. We also encourage researchers to include all of the life cycle stages, not to dismiss at least 10% of all GHG emissions, and to refine the scenarios regarding the installation and the end of life of the material in future studies. Finally, because the performed work includes many scenarios and assumptions, it is highly specific to the investigated technology and involves multiple inherent uncertainties. The projected future presented in this study relies on a set of assumptions and was only constructed to identify environmental hotspots and opportunities for improvement. More research is needed to refine the prospective scenarios, the LCA method and to improve the manufacturing process of mycelium-based composites. The economic feasibility of large-scale production (including market opportunities) should also be investigated to ensure the transfer of this new material to industry.

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Declarations

Competing interests The authors declare no competing interests.

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