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Comparative life cycle assessment of bio-based insulation materials: Environmental and economic performances

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Abstract

Insulation materials decrease the final energy consumption of buildings. In Germany, fossil and mineral insulations dominate the market despite numerous life cycle assessments (LCAs) showing that bio-based insulations can offer environmental benefits. Evaluating the results of such LCAs is, however, complex due to a lack of comparability or costs considered. The objective of this study is comparing bio-based insulations under equal conditions to identify the most environmentally friendly and cost-efficient material. For this purpose, a comparative LCA and life cycle costing (LCC) were conducted from "cradle to grave" for four bio-based and two nonrenewable insulations. The bio-based insulation materials evaluated were wood fiber, hemp fiber, flax, and miscanthus. The nonrenewable insulations were expanded polystyrene (EPS) and stone wool. Key data for the LCA of the bio-based insulations were obtained from preceding thermal conductivity measurements under ceteris paribus conditions. Eighteen environmental impact categories were assessed, and direct costs were cumulated along the life cycle. Results show that the most environmentally friendly bio-based insulation materials were wood fiber and miscanthus. A hotspot analysis found that, for agriculturally sourced insulations, cultivation had the largest environmental impact, and for wood fiber insulation, it was manufacturing. The use phase (including installation) constituted a cost hotspot. The environmental impacts of end-of-life incineration were strongly influenced by the fossil components of the materials. Overall, bio-based insulations were more environmentally friendly than EPS and stone wool in 11 impact categories. The LCC found EPS and miscanthus insulation to be most cost-efficient, yet market integration of the latter is still limited. It can be concluded that miscanthus biomass is an environmentally and economically promising bio-based insulation material. Comparability of the environmental performance of the bio-based insulations was increased by applying the same system boundary and functional unit, the same impact assessment methodology, and the preceding ceteris paribus thermal conductivity measurements.

KEYWORDS
bio-based, comparative LCA, hotspot analysis, insulation materials, life cycle costing, thermal conductivity measurement

1 | INTRODUCTION

In the residential sector of the European Union (EU), it is expected that heating will still account for 64% of final energy consumption in the year 2020, and by 2050 will still share 54% (EC, 2013). These figures underline the need for insulation materials to reduce the environmental and economic impact of the building stock (Al-Homoud, 2005) in the EU, of which 75% is currently energy inefficient (EC, 2019a).

There are various insulation materials available that can increase the energy efficiency of buildings. On the German market, fossil (i.e., petrol-based) materials currently have a share of 48% and mineral-based materials 45% (FNR, 2014). Renewable (i.e., bio-based) insulation materials account for only 7%. Of these, wood fiber constitutes the largest proportion (51%), followed by cellulose (41%). Hemp (5%) and other biomass sources (2%) make only minor contributions. Nonrenewable insulation materials, such as polystyrene and mineral wool, have a considerable cost advantage over bio-based materials through their sheer market dominance. However, bio-based insulations can reduce environmental impacts in comparison to nonrenewable materials (Torres-Rivas et al., 2018).

Bio-based insulation materials represent a form of CO_2 storage and, as such, can help decrease the climatic burden while installed (EC, 2018; FAO, 2016; Lawrence, 2015). By contrast, nonrenewable materials such as expanded polystyrene (EPS) contribute to an immediate increase in global warming potential (GWP), for example through their energy-intensive production process (Pittau et al., 2019). Bio-based insulation materials can help mitigate depletion of nonrenewable resources and also contribute to the passive control of indoor air conditions (temperature and humidity) via hygroscopicity (Romano et al., 2019; Torres-Rivas et al., 2018). They are non-irritant, rendering them more user-friendly, and being biodegradable lead to less pollution when disposed of (Lawrence, 2015). Finally, bio-based insulation materials have higher specific heat capacities than, for example, stone wool, resulting in a slower response to temperature changes (Lawrence, 2015).

Of the increasingly wide range of bio-based materials available, wood fiber and hemp fiber have become established feedstocks for insulation applications due to their beneficial physical properties (Kosiński et al., 2017; Kymäläinen & Sjöberg, 2008) and high market shares (FNR, 2014). However, less established feedstocks, such as flax (Kymäläinen, 2004; Kymäläinen & Sjöberg, 2008; Zach et al., 2013) and miscanthus (Hesch, 2000; Murphy et al., 1995; Nichtitz et al., 2016; Pude, 2005), also display favorable properties. The utilization of miscanthus as an insulation material has shown a very promising environmental performance (Wagner et al., 2017), whereas that of flax insulation has been more varied (Schmidt et al., 2004; Struhala et al., 2016).

Wood fiber, hemp fiber, flax, and miscanthus are thus four bio-based feedstocks that represent a broad spectrum of the bio-based insulation materials available, but with widely varying market shares. However, each material differs in its thermal conductivity, environmental performance, and production costs.

High-performance bio-based insulation products are vital if the environmental impacts of the production processes are to be reduced and higher energy savings in buildings to be achieved. Several studies have assessed the environmental performance of bio-based insulation materials, but also reveal a number of limitations to such assessments. These include a lack of comparability between materials due to dissimilar assessment conditions, the assessment of only a small number of environmental impact categories (i.e., non-holistic analysis; Beus & Piotrowski, 2017; Carcassi et al., 2020; Gellert, 2010; Ingrao et al., 2015; Klingler et al., 2018; Lazzarin et al., 2008; Pennacchio et al., 2017; Silvestre et al., 2011), no inclusion of nonrenewable materials in the environmental assessment (Sierra-Pérez et al., 2018), or no cost assessment given (Batouli & Zhu, 2013; Gellert, 2010; Kono et al., 2016; Pennacchio et al., 2017; Schmidt et al., 2004; Silvestre et al., 2011; Uihlein et al., 2008; Usubharatana & Phungrassami, 2019; Zampori et al., 2013).

The objective of this study is to fill this research gap by comparing bio-based insulations under equal conditions to identify the most environmentally benign material with the most cost-efficient performance. For this purpose, a comparative life cycle assessment (LCA) of the bio-based insulations was conducted following the ISO 14040 and 14044 (ISO, 2006a, 2006b) standards, and an environmental hotspot analysis performed. In addition, a relative environmental impact comparison with two nonrenewable insulations and a life cycle costing (LCC) were conducted. Key data for the LCA of the bio-based insulation materials under study were obtained from preliminary thermal conductivity measurements under ceteris paribus conditions. This determined the amount of each material required for a functionally equivalent comparison. In addition, the study employed the same system boundary, the same functional unit (FU), and the same life cycle impact assessment (LCIA) method including 18 different impact categories for all insulations under investigation. It thus provides a comprehensive environmental and economic assessment of bio-based insulation materials with best possible comparability of their performance.

2 | MATERIALS AND METHODS

2.1 | Scope

Wood fiber, hemp fiber, flax, and miscanthus were selected as bio-based insulations, and EPS and stone wool as

nonrenewable insulations for the assessment of the environmental and economic performance, based on their differing market shares.

The study examines the life cycle from "cradle to grave," and thus, the system boundary of the LCA and LCC encompasses four systems (Figure 1). The first is the raw material system, which includes cultivation or extraction of the resources. The second is the manufacturing system, which ends with the fabricated insulation material. The third is the use phase system, which takes place at the construction site and which is divided into installation and demolition. The fourth is the end-of-life system, which covers the disposal of the insulation materials either by incineration or landfill. Transportation between systems is included, assuming use of a EURO 5 truck and 100 km each, based on average national transport distances for Germany (Eurostat, 2020). For the LCA modeling of the bio-based insulation materials, the description of each life cycle system is based on a literature review and expert interviews, while key data were ascertained from thermal conductivity measurements under ceteris paribus conditions. The OpenLCA 1.8 software and the ecoinvent database 3.5 (Wernet et al., 2016) were used.

The LCC was conducted following the approach described by Swarr et al. (2011). A product life cycle cost model was
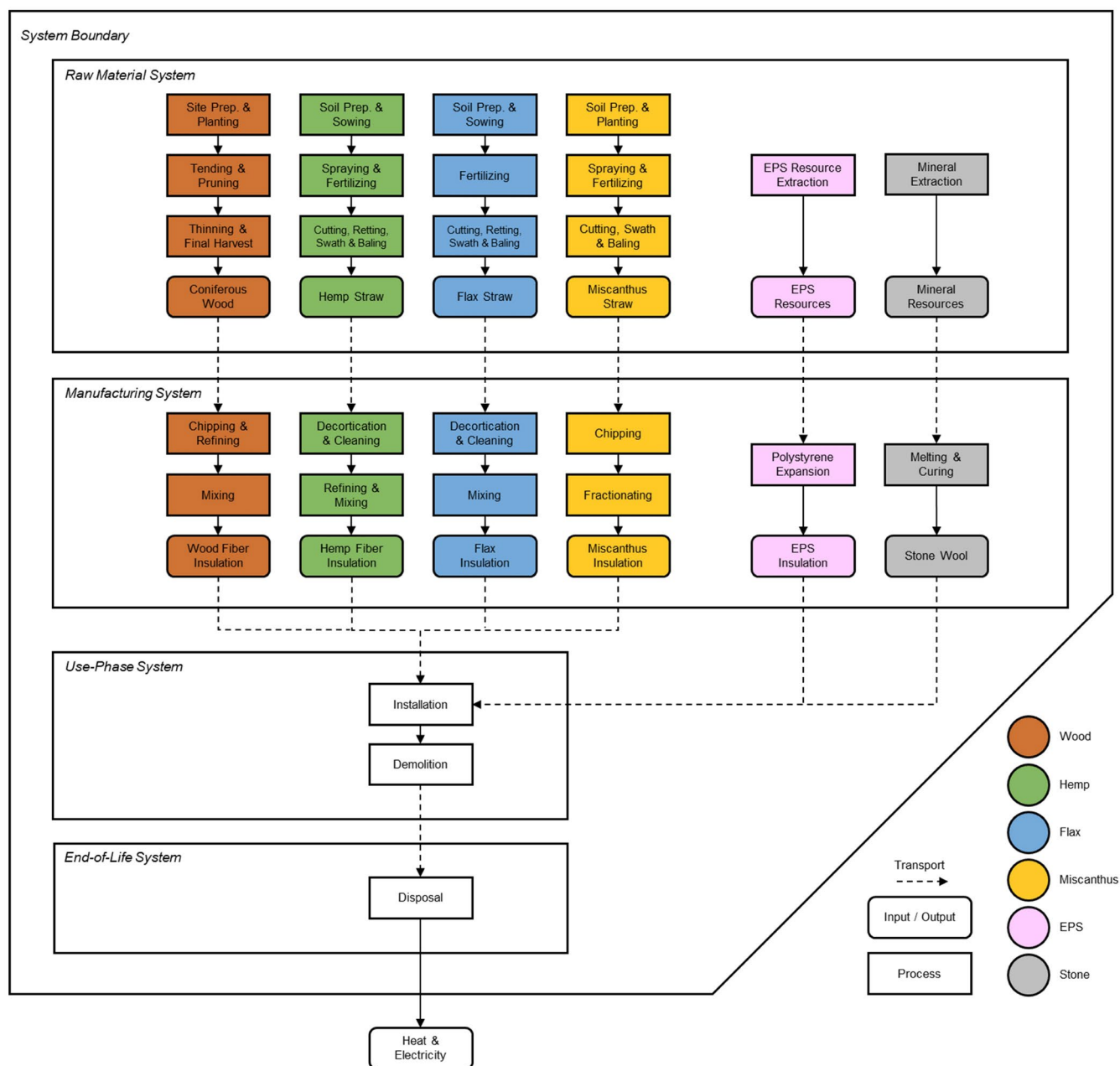

FIGURE 1 Life cycle system boundary of the insulation materials encompassing the raw material system, the manufacturing system, the use phase system, and the end-of-life system

applied including various stakeholders (farmer, manufacturer, consumer, waste disposal facility, etc.), and the costs were added up along each step of the life cycle. Only *direct* costs entering or leaving one of the four systems within the system boundary were accounted for.

The temporal reference of the LCA and LCC was assumed to be under steady-state conditions, and thus, the analysis is based on current conditions. No discount rate was applied for costs resulting from future processes (use phase system, end-of-life system) as this would not have influenced the objective of the study. For reasons of conformity, biogenic CO_2 sequestration (e.g., via soil organic carbon (SOC)), the differing CO_2 storage in the insulation materials, and biogenic CO_2 emissions from combustion were not accounted for. Geographically, the LCA and LCC were set for central European conditions (i.e., Germany), since the cultivation of the feedstocks and manufacture of the insulation materials were mainly set within this geographical boundary. Here, yield levels can differ significantly due to a variety of factors (temperature, precipitation, SOC, etc.). For this reason, a sensitivity analysis was performed with a range of yield levels to represent the effects of varying site conditions on the environmental impacts.

In the EU, legal statutes set mandatory requirements for the energy efficiency of buildings and thus of insulation materials (EC, 2019a). In Germany, these standards are defined by the Energy Saving Ordinance (EnEV; EnEV, 2015). For external walls of residential buildings, which in Germany have a designated economic use phase of 70 years (BMJV, 2015), the EnEV defines a maximum heat transmittance of $0.24 \text{ W m}^{-2} \text{ K}^{-1}$ (EnEV, 2015). In addition, fire resistance is set as a legal requirement for thermal insulations. To obtain legal approval, insulation materials must be categorized into European class E or higher, as defined by DIN EN 13501-1. Furthermore, DIN EN 16516-01 stipulates that insulation materials must also "fit and fill out the construction without air gaps, and ideally should remain unchanged in all three dimensions during the building lifetime. […] The material[s] must be stable to moisture and resistant to biological attack […] [and] shall not emit or radiate substances in hazardous concentrations to the indoor climate" (Schmidt et al., 2004). These legal and technical requirements were taken into account in the functional unit (FU) of this study. The FU was defined as *insulating 1 m² of external wall of a residential building with 0.24 W m⁻² K⁻¹ for 70 years, fulfilling legal fire resistance and health and safety standards.*

Since the materials differ in thermal conductivity, the different insulation types require varying insulation layer thicknesses and mass amounts, and thus have varying reference flows of the materials (Lakatos & Kalmár, 2013). This is because the thickness [m] of the insulation material required is given by the thermal conductivity [W m⁻¹ K⁻¹] divided by the thermal transmittance [W m⁻² K⁻¹]. The mass [kg]

required for insulation of 1 m² is given by the insulation layer thickness [m] times density of the material [kg m⁻³]. Thus, it follows that the FU determines the mass of an insulation material needed to fulfill the legally determined thermal transmittance requirements. These different masses result in varying environmental impacts and costs.

2.2 | Thermal conductivity measurement

To determine the required mass of insulation material per FU, a ceteris paribus thermal conductivity measurement was conducted for each bio-based material. The results were used to assess the environmental and economic performance of the bio-based insulation materials based on their individual physical properties. Each material was first dried in a drying chamber at 105°C for a minimum of 24 h, and then, the thermal conductivity was measured using a λ-meter EP500e (Lambda-Messtechnik GmbH Dresden) in accordance with DIN EN 12667 (Lambda-Meßtechnik GmbH Dresden, 2019). The testing temperature gradient was 15 K, ranging from 8°C to 23°C, at constant room temperature. Each specimen filled a volume of 0.20 m*0.20 m area and 0.08 m height. Thus, as the materials have different densities, the specimens being measured varied in density. Two measurements were taken for each of the bio-based insulation materials.

For wood fiber, the blow-in insulation STEICOzell was used (STEICO SE, 2019). It is derived from coniferous wood (DIBt, 2017) and has received European technical approval (ETA 12/0011). It consists of 83% wood and 17% additives (IBU, 2016). The product has a declared nominal thermal conductivity of $0.038 \text{ W m}^{-1} \text{ K}^{-1}$ and is categorized as fire resistance class E (STEICO SE, 2019). The hemp fiber product investigated was THERMO Stopfwolle Hanf from THERMO NATUR GmbH & Co.KG, which is listed by DIBt (2015) in class C of the technical regulations for buildings. In addition to hemp fiber, it contains up to 5% soda as a flame retardant, classifying it as fire resistance class E (THERMO NATUR GmbH & Co. KG, 2014). The flax insulation investigated was the Austrian blow-in insulation Flachsfloc, which has received technical approval from ETA 12/0037 with fire resistance class E and a nominal thermal conductivity of $0.041 \text{ W m}^{-1} \text{ K}^{-1}$ (EC, 2019b). It is composed entirely of flax, with a minimum share of two-thirds fiber and maximum one-third shives. For miscanthus, eight insulation specimens from Technical Service Kuehn GmbH were subjected to the thermal conductivity measurements, and the best performing specimen was selected for assessment in the LCA and LCC. A certified miscanthus insulation product is currently not available, resulting in a lack of market integration. Assumed properties of EPS are a thermal conductivity of $0.033 \text{ W m}^{-1} \text{ K}^{-1}$ at a given density of 33 kg m⁻³ (Klingler, 2011; Styrochem, 2015). For stone wool, a thermal

conductivity of 0.038 W m^{-1} K^{-1} is assumed at a density of 80 kg m^{-3}, in accordance with Abdou and Budaiwi (2005) and Schiavoni et al. (2016).

Table 1 provides key data from the thermal conductivity measurement for the LCA of the bio-based insulation materials under study. The thermal conductivity and density values were measured, and the thickness and mass values calculated to meet a thermal transmittance of 0.24 W m^{-2} K^{-1} (EnEV, 2015). For each bio-based insulation material, two thermal conductivity and density measurements were taken and the corresponding calculations performed (giving two columns for each material in Table 1). For the LCA and LCC, the average mass per FU was taken. This was the smallest for wood fiber insulation at 12.46 kg m^{-2}, followed by hemp fiber insulation at 12.87 kg m^{-2}. For flax insulation, an average mass of 13.50 kg m^{-2} was required to fulfill the FU, and only slightly more for the miscanthus insulation at 13.66 kg m^{-2}.

For EPS, 4.54 kg m^{-2} was needed to fulfill the FU and for stone wool 12.67 kg m^{-2} (calculated from properties given above).

2.3 | Life cycle inventory

2.3.1 | Raw material system

In the case of the bio-based insulations, the raw material system is the cultivation system, as shown in Figure 1.

The wood cultivation system is a sustainably managed forestry site for coniferous wood (i.e., spruce) production located in Germany. It was modeled by the ecoinvent process for spruce pulpwood production based on Albrecht et al. (2008) and in accordance with Klein et al. (2016). The lifespan of the system covers one rotation period and includes the following processes: site establishment, breeding and planting of tree seedlings (3,000 ha^{-1}), pruning and tending with a power saw, thinning and forwarding with a forest harvester and forwarder, and a manual final cutting with a power

saw and subsequent skidding to forest road. No fertilizer or pesticide application was given. This results in 1 m^3 coniferous wood with a density of 379 kg m^{-3} (dry wood matter/fresh wood volume; Kollmann, 1982), which was assumed to be transported in the form of roundwood departing from the forest road. Labor costs of 36.50 € h^{-1} were taken based on BMEL (2019), while other costs were derived from the ecoinvent database 3.5 (Werner, 2013), in accordance with Klein et al. (2016).

The hemp cultivation system for fiber production represents a typical central European setting (Beus & Piotrowski, 2017). Cultivation processes and data for agricultural inputs, yields, and biomass properties were derived from Beus and Piotrowski (2017) and Beus et al. (2019). The following processes were considered: soil preparation via harrowing (incl. sowing), fertilizing, pesticide spraying, cutting, retting (incl. two times turning), swathing, and baling. An average sowing rate of 48 kg ha^{-1} was assumed. Fertilizing consisted of 80 kg N ha^{-1}, 57.50 kg P ha^{-1}, and 115 kg K ha^{-1} and herbicide application corresponded to 2.57 kg glyphosate ha^{-1}. This resulted in a dry matter straw yield of 7500 kg ha^{-1} consisting of 30% fibers, 65% shives, and 5% other, with a moisture content of 15% (Gusovius & Pecenka, 2008).

The flax cultivation system and its corresponding costs were derived from Schmidt et al. (2004), Beus and Piotrowski (2017), Beus et al. (2019), and interviews with flax insulation manufacturer M. Mahringer. The required cultivation processes consisted of plowing, harrowing, fertilizing, sowing, mechanical weeding, cutting, retting (two times turning), swathing, and baling. The sowing rate was 110 kg seeds ha^{-1} and the fertilizing regime included 40 kg N ha^{-1}, 40 kg P ha^{-1}, and 80 kg K ha^{-1}. The corresponding dry matter yield amounted to 5550 kg ha^{-1} flax straw, consisting of 30% fibers, 65% shives, and 5% other, with a moisture content of 15% (Gusovius & Pecenka, 2008).

The miscanthus cultivation processes including agricultural inputs and yield were derived from Wagner et al. (2017) and Wagner et al. (2018). Since miscanthus is a perennial crop,

TABLE 1 Properties of the bio-based insulations (from thermal conductivity measurements) used to determine the required amount per FU. The two values shown for each bio-based insulation material derive from the two repetitions of the thermal conductivity measurement and the corresponding calculated values

Property	Unit	Wood fiber		Hemp fiber		Flax		Miscanthus	
Thermal conductivity	W m^{-1} K^{-1}	0.040	0.041	0.051	0.053	0.046	0.047	0.047	0.046
Density	kg m^{-3}	68.82	78.76	56.25	62.40	56.25	83.00	70.00	70.79
Thickness for FU	m	0.17	0.17	0.21	0.22	0.19	0.20	0.19	0.19
Mass per FU	kg m^{-2}	11.61	13.31	12.06	13.67	10.69	16.32	13.60	13.71
Average mass per FU	kg m^{-2}	12.46		12.87		13.50		13.66	

all agricultural processes were divided by 20 years (the length of the entire cultivation period) to give the values for a 1 year cultivation period. For example, the planting process occurs only once in 20 years and thus accounts for 0.05 ha year^{-1}. Accordingly, costs were calculated for 1 year using a discount rate of 6% (Wagner et al., 2018). The cultivation processes consisted of plowing, harrowing, planting, herbicide spraying, fertilizing, mulching (after the first and last year), mowing, swathing, baling, and chiseling. The amount of herbicide used was 1.26 kg glyphosate ha^{-1} year^{-1}, 0.68 kg pendimethalin ha^{-1} year^{-1}, and a mix of 0.27 kg ha^{-1} year^{-1} mesotrione, tritosulfuron, and dicamba. Fertilization consisted of 40 kg N ha^{-1} year^{-1}, 30 kg P ha^{-1} year^{-1}, and 120 kg K ha^{-1} year^{-1}. This led to a dry matter yield of 15,316 kg miscanthus stalks ha^{-1} year^{-1} with a moisture content of 15% (Lewandowski et al., 2016). Costs incurred by the cultivation system were based on Beus and Piotrowski (2017) and the KTBL database (KTBL, 2019).

For all agricultural cultivation systems, fertilizer and pesticide induced emissions were calculated. Direct N_2O and NO emissions from nitrogen fertilizer were calculated based on Bouwman et al. (2002) to account for the specific type of mineral fertilizer (i.e., calcium ammonium nitrate). Indirect N_2O and NO emissions from nitrate leaching and volatized ammonia emissions were calculated using factors taken from De Klein (2006). Ammonia emissions were calculated according to EMEP/CORINAIR (2001), and phosphate and phosphorous emissions using factors from the SALCA P model by Prasuhn (2006), given in Nemecek and Schnetzer (2011). An annual land rent of 328 € ha^{-1} was assumed for all cultivation systems, based on the German average land rent from 2016 (Destatis, 2018). Transport losses of 2% were assumed for the transportation following the cultivation system, adapted from Caixeta-Filho and Thiago Guilherme (2018). Transportation processes occurring between each subsystem within the system boundary and corresponding costs were based on Valsasina (2018) and ZDB (2018).

The data for the EPS and stone wool raw material systems were taken from the ecoinvent database (Althaus, 2010a; Klingler, 2011). The impacts associated with the raw material system of the nonrenewable materials are attributed to the manufacturing system.

2.3.2 | Manufacturing system

Following the raw material system and initial transport, the materials are processed into the insulation products in factory facilities (Figure 1), for which the terms "processing" and "manufacturing" are used interchangeably. Energy values are taken from the German electricity mix except for flax processing, which occurs in Austria, and thus values for the Austrian electricity mix were taken here. Costs for the electricity within the manufacturing system amount to 0.12 € kWh^{-1} for Germany and 0.09 € kWh^{-1} for Austria (Destatis, 2019; IEA, 2020).

The manufacturing system of the wood fiber insulation includes the following processes (IBU, 2016): First, chipping of the roundwood that arrives at the factory gate. Second, heating of the wood chips, followed by a defibration process. Third, a drying process via a cyclone dryer, and finally, mixing with additives and subsequent packaging. For modeling, the ecoinvent process of wood wool production in Europe was adapted according to IBU (2016) and Althaus (2010b). The resulting wood fiber blow-in insulation consisted of 83% wood fiber, 6.3% recycled paper, 6% water, 2.4% aluminum sulfate acting as flame retardant, 1.3% polyethylene (PE) and polypropylene (PP) fiber, and 1.2% sodium silicate serving as an adhesive. Costs for the manufacturing system of the wood fiber insulation were adapted from Beus and Piotrowski (2017), in compliance with Baunativ GmbH & Co. KG (2019) and bausep GmbH (2019).

For hemp and flax, the manufacturing system was based on the pilot-scale machinery line of Gusovius and Pecenka (2008). An upscaled mass flow of 4 t h^{-1} was assumed, which makes the production economically sustainable (Pauls & Carus, 2008). The same production line was assumed for both crops; however, not all process steps are identical. In both cases, the straw was first broken up and the resulting straw mix split into its components, namely fibers, shives, and dust. The shives were redirected via an air cyclone. The hemp fiber was further purified by a cleaning machine and a refiner, whereas the flax fiber was only cleaned (M. Mahringer, personal communication). The hemp fibers were subsequently mixed with soda in a mechanical process. For the flax insulation, the shives were mixed with the fibers. In both cases, dust was assumed to be disposed of as biowaste by the factory as it has no economic value (Pauls & Carus, 2008). The final hemp fiber insulation consisted of 95% fiber and 5% soda and was packed in cartons (THERMO NATUR GmbH & Co. KG, 2014). The finished flax blow-in insulation consisted of 66% fiber and 33% shives and was packed in PE film. Economic allocation was applied after the step of separating the straw into fibers (0.6 € kg^{-1}), shives (0.2 € kg^{-1}), and dust (0 € kg^{-1}; Pauls & Carus, 2008). Costs resulting from the hemp and flax manufacturing systems were adapted from Pauls and Carus (2008) and Beus and Piotrowski (2017).

Data on the energy demand for the manufacturing system of miscanthus stalks were provided by Technical Service Kuehn GmbH (2019). From the eight specimens subjected to the thermal conductivity measurements, the best performing specimen was taken for the assessment. It was produced via fractionation of the chipped miscanthus stalks, of which 1.6% ended as dust. The specimen consisted entirely of miscanthus with no additives being used in the production. After processing, the insulation material was packed in PE film and transported.

For the EPS and stone wool manufacturing systems, processing data including costs for both extraction and manufacturing are taken from the ecoinvent database (Althaus, 2010a; Klingler, 2011). For EPS, this includes the expansion of the polystyrene, and for stone wool, the melting, fiberizing, and curing of the minerals (Eurima, 2012).

2.3.3 | Use phase system

A time period of 70 years is assumed between installation and demolition (Figure 1). However, this study assumes current conditions for all activities occurring during this stage. Time requirements for installation are derived from ZDB (2002) and for demolition from Motzko et al. (2016). Costs for both operations were taken from ZDB (2018). The wall into which the insulations are fitted was assumed to be in the classification WH according to DIN 4108-10 for residential buildings.

The hemp fiber, miscanthus, EPS, and stone wool insulations were assumed to be installed manually, with a vapor barrier sheet fixed with screws. The amount of sheet used was calculated based on Luyt et al. (2006), SPAX (2018), and Isover (2019). The electricity demand for attaching the sheet to the wall with screws using a drill was calculated based on Giseke GmbH and Co. KG (2015) and BOSCH (2019). The energy requirements for the mechanical installation of the blow-in flax and wood fiber insulations were taken from Beus and Piotrowski (2017) and amounted to $0.021\ kWh\ kg^{-1}$. Labor costs and the time requirements were based on X-Floc GmbH (2017) and ZDB (2018), with the assumption that two persons were required for installation. After manual demolition, all insulations were transported to the end-of-life stage of the life cycle.

2.3.4 | End-of-life system

This study assumes that, after the last transport from the construction site, the insulation materials were disposed of in a German waste incineration plant, with the exception of stone wool, which was disposed of to landfill. The incineration processes were modeled based on the corresponding waste treatment processes of the different components of the insulation materials (Hischier, 2010a, 2010b, 2010c). For hemp fiber and wood fiber insulation, the product composition was considered excluding the flame retardants (i.e., incineration of only 95% hemp for hemp fiber insulation, and 83% wood, 6% wastepaper, and 1.3% PE for wood fiber insulation). All incineration processes were considered to have the same net energy efficiency of 44.6% in accordance with Flamme et al. (2018). Average costs for waste incineration in Germany were assumed (Statista, 2018). Different energy contents

were considered for the different insulations. The lower heating values (LHV) taken are $17.4\ MJ\ kg^{-1}$ for hemp (Prade et al., 2011), $17.7\ MJ\ kg^{-1}$ for flax (Boukaous et al., 2018), and $17.9\ MJ\ kg^{-1}$ for miscanthus (ECN, 2019a). The LHV taken for coniferous wood (i.e., spruce) is $18.08\ MJ\ kg^{-1}$ assuming a density of $379\ kg\ m^{-3}$ (Hahn et al., 2011; Kollmann, 1982). For EPS, the LHV of $37.44\ MJ\ kg^{-1}$ based on ECN (2019b) was taken.

2.4 | Environmental impact indicators

The life cycle impact assessment (LCIA) was conducted using the indicator approach ReCiPe, from which all 18 midpoint-impact categories were included, and a hierarchical perspective was applied (Huijbregts et al., 2016): Fine particulate matter formation (FPM), that is, air pollution from fine particles, is expressed in $kg\ PM_{2.5}$ eq.; fossil resource scarcity (FRS) and mineral resource scarcity (MRS) are given in kg oil eq. and kg Cu eq., respectively; freshwater ecotoxicity (FET) and freshwater eutrophication (FE) account for the emission of kg 1.4-dichlorobenzene equivalents (kg 1.4-DCB eq.) and kg P eq. to freshwater aquatic systems; Global warming (GW) represents GWP and is expressed in $kg\ CO_2$ eq.; Human carcinogenic toxicity (HCT) and human noncarcinogenic toxicity (HNT) are each given in kg 1.4-DCB eq.; Ionizing radiation (IR), expresses radioactive radiation in kBq Co-60 eq.; Land use (LU) refers to land occupation expressed in m^2*year of a crop eq.; Marine ecotoxicity (MET) and marine eutrophication (ME) account for the environmental burden on marine aquatic systems and are expressed in kg 1.4-DCB eq. and kg N eq., respectively; ozone formation, human health (OFH), and ozone formation, terrestrial ecosystems (OFT) are both given in $kg\ NO_x$ eq.; stratospheric ozone depletion (SOD), expressed in kg chlorofluorocarbon equivalents (kg CFC-11 eq.), cause greater organismal vulnerability to ultraviolet radiation from the sun; terrestrial acidification (TA) accounts for the SO_2 equivalents emitted into the air that cause soil pH deviation; terrestrial ecotoxicity (TET) refers to the accumulation and persistence of chemicals in the environment and is expressed in kg 1.4-DB eq.; Water consumption (WC) is given in m^3.

3 | RESULTS

A comparison of the total environmental impacts of the bio-based insulation materials is first presented, followed by the results of the environmental hotspot analysis. The bio-based materials are then compared to the nonrenewable insulations, EPS and stone wool. Finally, the results of the LCC are presented.

3.1 | Comparative life cycle assessment

Table 2 summarizes the environmental performances of the bio-based insulation materials under study per FU.

The comparison of biomaterials shows flax insulation to be the least well-performing material, as it has the highest values in 14 of the 18 impact categories. However, the difference between flax and hemp fiber insulation is only marginal in seven categories while hemp fiber has the largest impact in four categories. Thus, overall, hemp fiber and flax insulation display a comparable impact pattern. This stands in contrast to the other two materials evaluated, wood fiber and miscanthus insulation, which also have similar impact levels. In fact, wood fiber and miscanthus insulation both have considerably lower values than hemp fiber and flax insulation in 16 of the 18 impact categories, except for HCT and WC. Thus, wood fiber and miscanthus insulations were found to have the most environmentally friendly performance.

3.2 | Environmental hotspot analysis

Figures 2 and 3 give a breakdown of all environmental impacts of the bio-based insulation materials per FU according to the life cycle stage of the system boundary.

The results demonstrate that the cultivation system is a substantial environmental hotspot. For those materials sourced from agriculture, it is the major environmental hotspot in 11 categories. In all of these, the burden appears to be mainly induced by fertilizer production (GW, MRS, TET) and its application, including subsequent emissions into the environment in the form of N_2O (GW, SOD, OFH, OFT), ammonia (FPM, TA), phosphate (FE), and nitrate (ME). In addition, pesticide application via glyphosate (FET) and agricultural operations (e.g., harvesting, baling) (FPM) account for a large proportion of the burden in this life cycle stage. Evidently, the cultivation system is also the major hotspot for LU. The other impact categories account for substantial proportions as well, particularly for the hemp fiber and flax insulation. The reasons for this are, again, fertilizer production (FRS, HCT, HNT, MET, SOD, WC), agricultural and silvicultural operations (e.g., harvesting and baling, skidding and using power saws; FPM, FRS, IR, OFH, OFT), as well as seed production for hemp and flax, and rhizome production for miscanthus (HNT, SOD, TA).

The manufacturing system also partially constitutes an environmental hotspot. This displays a moderate impact in 12 of the categories with either hemp fiber or flax insulation, or both, having a higher value than wood fiber and miscanthus insulation in 10 of those (FPM, FRS, FET, GW, HNT, IR, MET, TA, TET, WC). However, wood fiber has the greatest environmental impact in the categories HCT and MRS. Miscanthus insulation has the lowest impact level in 14 categories, and a moderate impact in the other four. In general, the processes

TABLE 2 Total environmental impacts of the bio-based insulation materials per FU

Impact category	Unit	Wood fiber	Hemp fiber	Flax	Miscanthus
Fine particulate matter formation (FPM)	kg $PM_{2.5}$ eq.	0.004	0.018	0.022	0.005
Fossil resource scarcity (FRS)	kg oil eq.	0.80	2.16	2.73	0.61
Freshwater ecotoxicity (FET)	kg 1.4-DCB	0.09	0.28	0.30	0.09
Freshwater eutrophication (FE)	kg P eq.	0.001	0.070	0.070	0.009
Global warming (GW)	kg CO_2 eq.	2.72	11.70	13.62	2.95
Human carcinogenic toxicity (HCT)	kg 1.4-DCB	0.27	0.34	0.38	0.14
Human noncarcinogenic toxicity (HNT)	kg 1.4-DCB	1.84	10.69	16.12	1.75
Ionizing radiation (IR)	kBq Co-60 eq.	0.21	0.53	0.49	0.26
Land use (LU)	m^2*a crop eq.	10.45	44.59	59.17	6.44
Marine ecotoxicity (MET)	kg 1.4-DCB	0.12	0.39	0.43	0.14
Marine eutrophication (ME)	kg N eq.	0.0003	0.0335	0.0253	0.0054
Mineral resource scarcity (MRS)	kg Cu eq.	0.02	0.06	0.07	0.01
Ozone form., Human health (OFH)	kg NOx eq.	0.01	0.05	0.06	0.01
Ozone form., Terrestrial ecosystems (OFT)	kg NOx eq.	0.01	0.05	0.06	0.01
Stratospheric ozone depletion (SOD)	kg CFC11 eq.	2E-06	2E-04	2E-04	3E-05
Terrestrial acidification (TA)	kg SO_2 eq.	0.01	0.07	0.08	0.02
Terrestrial ecotoxicity (TET)	kg 1.4-DCB	4.34	24.92	26.95	4.83
Water consumption (WC)	m^3	0.031	0.041	0.028	0.028

FIGURE 2 Environmental hotspot analysis of the bio-based insulation materials per FU. C, cultivation system; EoL, end-of-life system; M, manufacturing system; Σ, total; T, transports; U, use phase system. Categories are fine particulate matter formation (FPM), fossil resource scarcity (FRS), freshwater ecotoxicity (FET), freshwater eutrophication (FE), global warming (GW), human carcinogenic toxicity (HCT), human noncarcinogenic toxicity (HNT), ionizing radiation (IR), and land use (LU)
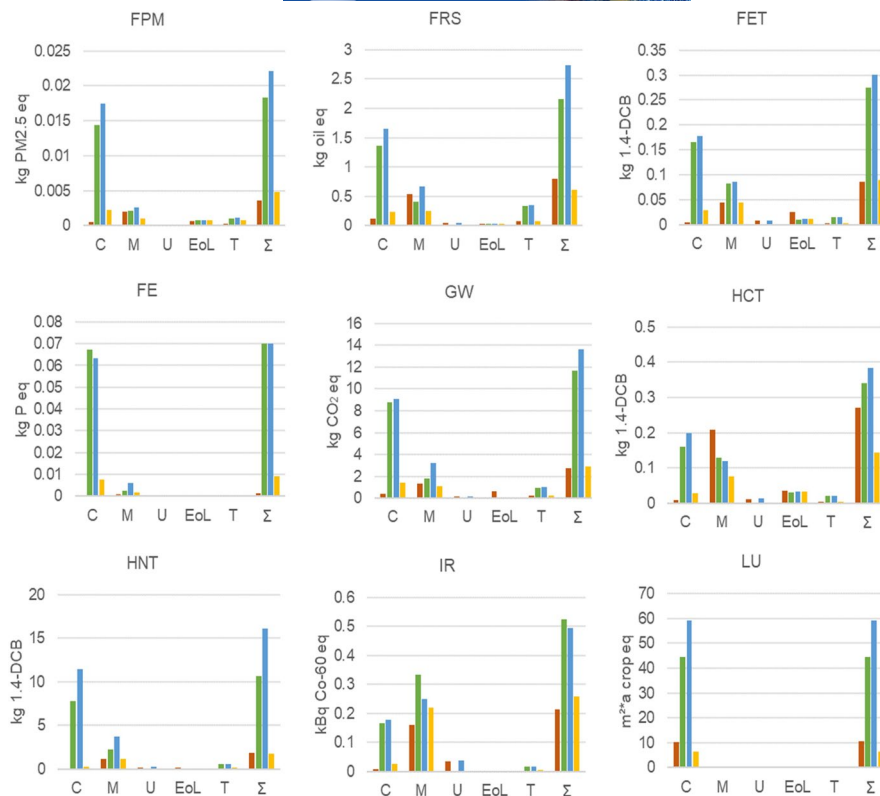

FIGURE 3 Environmental hotspot analysis of the bio-based insulation materials per FU. C, cultivation system; EoL, end-of-life system; M, manufacturing system; Σ, total; T, transports; U, use phase system. Categories are marine ecotoxicity (MET), marine eutrophication (ME), mineral resource scarcity (MRS), ozone formation, human health (OFH), ozone formation, terrestrial ecosystems (OFT), stratospheric ozone depletion (SOD), terrestrial acidification (FA), terrestrial ecotoxicity (TET), and water consumption (WC)
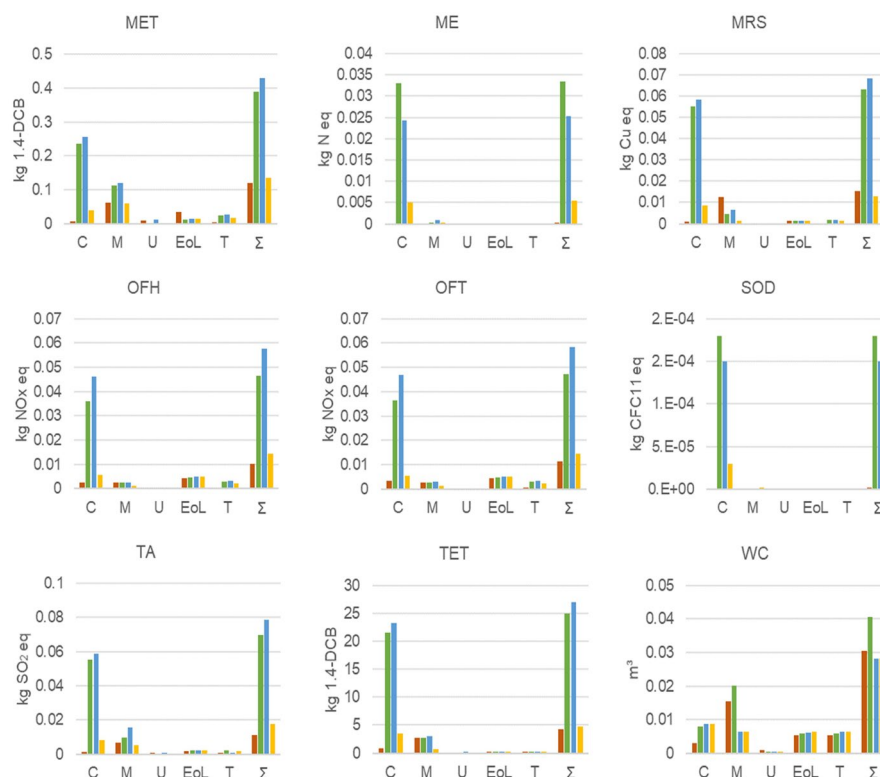
responsible for the environmental impacts in this life cycle stage are mostly related to background processes of electricity generation, for example, lignite combustion for FRS.

The use phase system does not show a hotspot for any impact category. Instead it provokes the least environmental burden of all of the life cycle stages.

In the end-of-life system, the environmental burdens caused by the incineration process are most substantial in FET, GW, HCT, MET, OFT, and WC. The wood fiber insulation has the highest contribution to FET, GW, and MET due to its fossil components.

Lastly, transportation between the life cycle stages accounts for moderate burdens in five categories, caused by both the production (WC) and combustion of fossil fuel (FPM, FRS), as well as road construction (OFH, OFT). In the remaining categories, transportation has only a marginal impact.

3.3 | Comparison to nonrenewable insulations

Figures 4 and 5 summarize the relative environmental performance of the bio-based and nonrenewable insulation materials under study.

The nonrenewable insulation materials show maximum absolute values in 11 of the 18 impact categories. Of these 11 categories, stone wool has the highest burden in seven, making it the least environmentally benign insulation material under study. EPS performs the least well in four categories, while flax insulation represents the highest environmental burden in four impact categories and hemp fiber insulation in

three. Wood fiber and miscanthus insulation do not have the highest burden in any of the impact categories. When compared to the nonrenewable materials, wood fiber and miscanthus insulation were found to be the most environmentally benign materials.

3.4 | Life cycle costing

Table 3 shows the costs per FU over the life cycle of all insulation materials under study. Total costs are highest for the life cycle of hemp fiber insulation, followed by wood fiber, stone wool insulation, and then flax insulation. The lowest costs are incurred along the life cycle of miscanthus insulation and EPS.

With respect to the cultivation system, flax and hemp are the renewable resources with the highest costs per FU. If the average of these two is taken, the miscanthus cultivation system incurs only 36% of these costs. The costs of wood cultivation have a value in between. In the manufacturing system, wood and hemp fiber have the highest costs for the processing of the corresponding insulation material. During the use phase, costs of the materials that were attached mechanically by a blowing machine account for about 22% of the costs of the manually attached materials. In the end-of-life stage, costs are comparable among all bio-based insulations, while
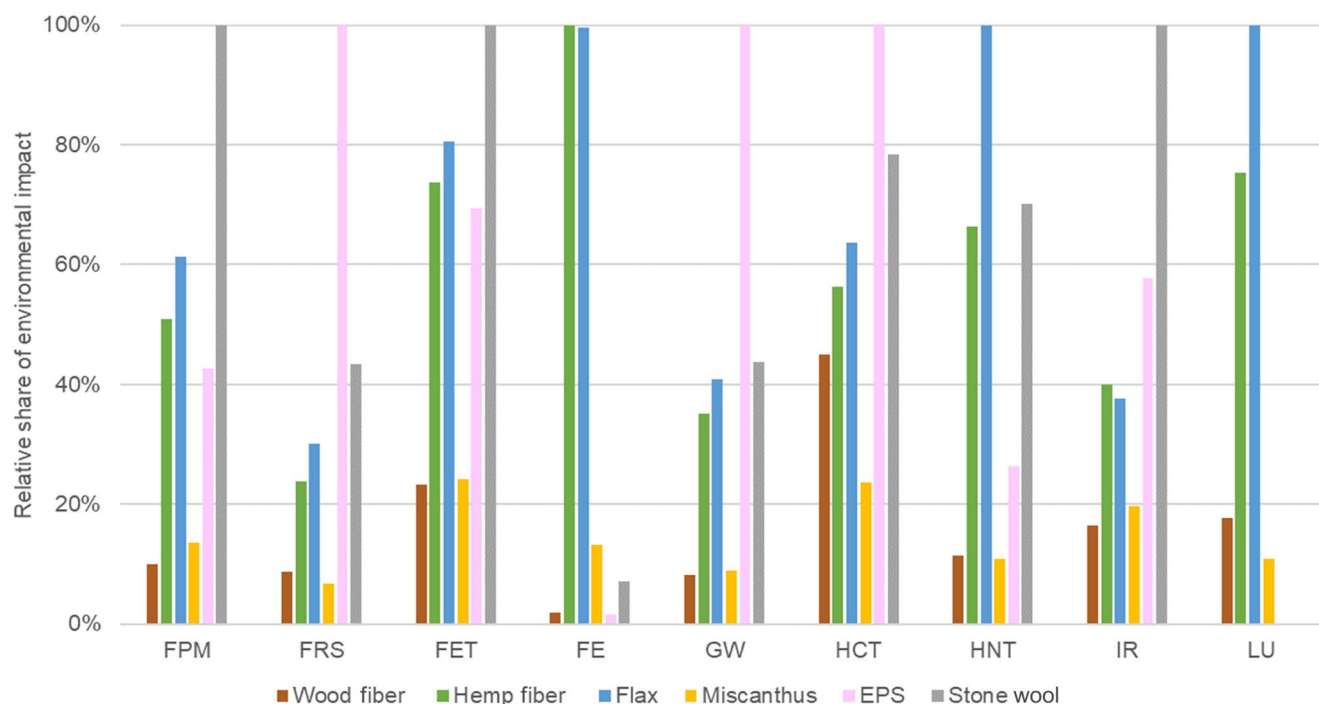

FIGURE 4 Comparison of the relative environmental burdens of the bio-based and nonrenewable insulations. The material with the highest impact in a category represents 100%. Categories are fine particulate matter formation (FPM), fossil resource scarcity (FRS), freshwater ecotoxicity (FET), freshwater eutrophication (FE), global warming (GW), human carcinogenic toxicity (HCT), human noncarcinogenic toxicity (HNT), ionizing radiation (IR), and land use (LU)
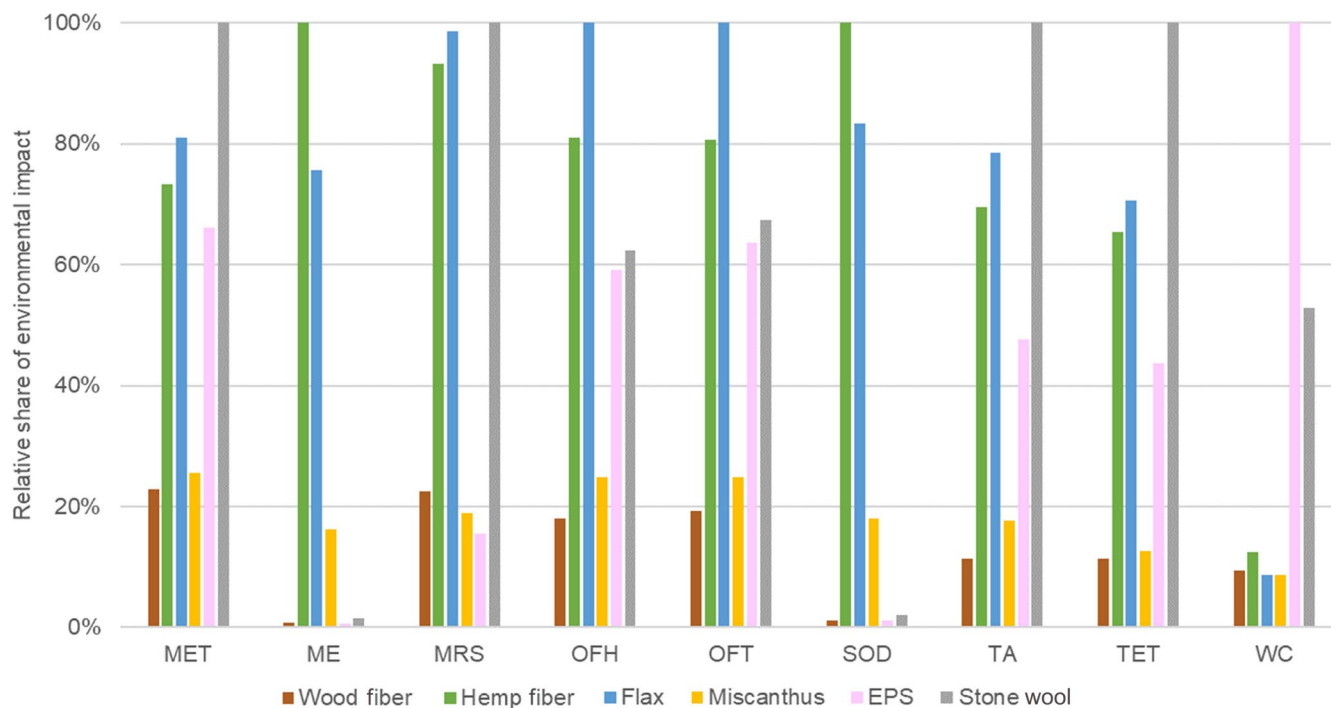
FIGURE 5 Comparison of the relative environmental burdens of the bio-based and nonrenewable insulations. The material with the highest impact in a category represents 100%. Categories are marine ecotoxicity (MET), marine eutrophication (ME), mineral resource scarcity (MRS), ozone formation, human health (OFH), ozone formation, terrestrial ecosystems (OFT), stratospheric ozone depletion (SOD), terrestrial acidification (TA), terrestrial ecotoxicity (TET), and water consumption (WC)

TABLE 3 Life cycle costs of all insulation materials per FU

Life cycle stage	Costs in € per FU					
	Wood fiber	Hemp fiber	Flax	Miscanthus	EPS	Stone wool
Cultivation system	1.51	2.21	2.79	0.90	0	0
Manufacturing system	10.33	9.19	6.35	2.76	4.13	8.35
Use phase system	0.81	3.70	0.81	3.74	3.70	3.64
End-of-life system	1.22	1.31	1.37	1.39	0.46	0.81
Transports	0.37	0.39	0.40	0.41	0.09	0.25
Total	14.24	16.79	11.72	9.18	8.39	13.05

the value for EPS is substantially lower. Transportation costs are similar among the bio-based materials and lower for EPS and stone wool.

4 | DISCUSSION

In the following sections, the assumptions made to increase the comparability and conformity of the LCA are validated and limitations are identified. Subsequently, key drivers determining the varying environmental and economic performance of the bio-based insulation materials are examined over the life cycle stages. In parallel, comparisons with other LCAs and LCCs of insulation materials are made to evaluate

the environmental and economic performances revealed in this study.

4.1 | Validation of comparability and limitations

In contrast to most other LCAs and LCCs of bio-based insulation materials, this study was based on a preliminary thermal conductivity measurement which provided key data for the determination of the FU. This allowed the physical property needed for a comparison on the basis of the FU, that is, the thermal conductivity, to be determined while all other parameters that could influence the

results were set aside (ceteris paribus). It was found that the thermal conductivity testing considerably increased the conformity and comparability of the environmental and economic performance of the investigated bio-based insulations. Thus, in the determination of the FU, the inclusion of the identification of key physical properties, such as thermal conductivity under ceteris paribus conditions, is to be recommended for future comparative LCAs of bio-based insulation materials.

In the subsequent LCA and LCC, various assumptions were made. One was an economic allocation of the multifunctional processes (i.e., processes with multiple outputs) during the manufacturing stage of hemp fiber and flax. This allocation method was used, because the study focused on economic costs following the LCC approach based on Swarr et al. (2011). Nevertheless, in line with the findings of Zampori et al. (2013), it was found that a physical allocation gave only slightly different results than the economic allocation.

Another assumption was that biogenic CO_2 sequestrations, emissions, and storages in the materials were neglected for reasons of conformity among the bio-based insulations. As no general consensus on the assessment of biogenic CO_2 is given in life cycle assessment, no universal procedure for this debated issue exists (Breton et al., 2018; Pawelzik et al., 2013). However, reporting the effect of biogenic CO_2 on the total GWP is advised, for example, for environmental product declarations (EPDs), as highlighted by Tellnes et al. (2017) for wood products. Methods for accounting for biogenic CO_2 that include time considerations are, for example, the dynamic LCA framework, and GWP_{bio}, which are emerging LCIA developments (Breton et al., 2018). These methods are useful for analyzing the effect of biogenic CO_2 on the total GWP of a bio-based product; however, they increase the complexity of the LCA (Anand & Amor, 2017; Breton et al., 2018) and can, thus, represent an additional barrier to comparability between LCAs.

Accounting for biogenic CO_2 could demonstrate two additional benefits of bio-based materials. First, the cultivation of bio-based feedstocks promotes SOC sequestration. In addition to its beneficial climatic effect, SOC sequestration also benefits overall soil properties for which cultivation of perennial crops is considered one suitable method (among others) for this purpose (Blanco-Canqui et al., 2013). Thus, if biogenic CO_2 had been accounted for, the environmental performances of the insulation materials made from wood (Bečvářová et al., 2018) and from miscanthus (McCalmont et al., 2017) would have particularly benefitted. However, in addition to direct land use change (dLUC), the cultivation of the feedstock can also induce indirect land use change (iLUC). Both dLUC and iLUC can have highly influential effects on environmental performances, especially when considering an increased future demand for bio-based products. In fact, it was shown, for example, in a study evaluating different agricultural substrates for biogas production, that dLUC can decrease the total GWP up to 50% while iLUC can increase it by between approximately 16% and 31% (Lask et al., 2020). To minimize adverse effects from dLUC and iLUC, cultivation on marginal land should be prioritized where less or no competition with food crops is expected (Lewandowski et al., 2016). Second, bio-based insulations avoid fossil carbon emissions via substitution of nonrenewable materials. However, corresponding displacement factors, which represent the calculated amount of fossil carbon saved, vary greatly in the literature as they depend, among other things, on the feedstock scenario (Breton et al., 2018).

A further assumption was the definition of the same system boundary. This was found to increase the comparability among the insulation materials under study. However, in particular, the raw material system of the insulations could not be modeled under entirely equal conditions. For the bio-based materials, this is especially true of wood cultivation, since silvicultural practices differ substantially from agricultural ones, for example, in terms of timescales.

For the LCIA, all 18 impact categories from the indicator approach ReCiPe were included (Huijbregts et al., 2016). This contributed to a holistic environmental performance comparison between the bio-based insulations, as is generally recommended (Bürger et al., 2017). In this context, the sensitivity analysis performed in this study revealed that the impact categories FPM, FE, GW, LU, ME, MRS, OFH, OFT, SOD, TA, and WC were of particularly high importance in the assessment of the environmental performance of bio-based insulations from agricultural crops. This is based on the fact that in all these impact categories, a yield decrease of 15% led to a relative impact change exceeding 15% for all agricultural crops (Table S1). By contrast, for the wood fiber insulation, a yield decrease of 15% led to an effect greater than 15% for LU only.

A number of limitations were found with regard to the assumptions made in the life cycle stages. In the agricultural cultivation systems, yield levels have a strong influence on the environmental performance. However, yields can vary widely as they are subject to significant uncertainties and depend on numerous factors including precipitation, soil conditions, temperature, fertilization, and pesticide regimes (Beus et al., 2019; Schmidt et al., 2004; Section 4.2.1). The manufacturing systems of hemp fiber, flax, and miscanthus were based on an upscaled pilot processing line from Gusovius and Pecenka (2008) because fiber processing lines in other LCAs provide only limited data on many processes (Beus et al., 2019). Data for the wood fiber manufacture were taken from IBU (2016); however, few published LCA data exist on the subprocesses

required for wood fiberizing, which limits the detailed identification of the processes responsible for the largest impacts (Skinner et al., 2016). In this study, the use phase was represented by the mass of insulation material required to fulfill the FU, in line with Schmidt et al. (2003), who also performed an extensive comparative LCA of insulation materials. This approach was found to be appropriate to increase comparability during this life cycle stage. In the present study, a discount rate of zero did not affect the outcome. However, in other contexts, including a discount rate can be an essential component of an LCC approach (Swarr et al., 2011).

The end-of-life scenario of the insulations was assumed to be incineration. In Germany, there are considerable cost differences between waste incineration plants (Statista, 2018). Despite containing flame retardants providing fire resistance class E, incineration is regarded a possible scenario for insulation materials (EFRA, 2004) and finds application in other LCA studies (Ardente et al., 2008; Lopez Hurtado et al., 2016). However, incorporating the substitution credit into the overall environmental and economic performance introduces uncertainties, as future incineration efficiencies, costs, and the substituted heat and electricity mix remain unknown. Nevertheless, incineration, including energy recovery, is considered a probable future scenario for insulation materials, since experience with material recycling of bio-based insulations is still limited (Krauß, 2014). Primary studies performed in this field, for example, Małaszkiewicz and Sztukowska (2018), have shown the feasibility of reusing wood fiber from waste wood products, for example, as lignocellulosic aggregates in bio-based concrete. An alternative end-of-life scenario for bio-based insulations is disposal to landfill, as was assumed by Usubharatana and Phungrassami (2019). However, if insulations are properly installed and removed, the focus should be placed on alternative end-of-life scenarios that involve the reuse of either energy or the entire material. In this regard, material reuse could play an important role since the storing of carbon in buildings performs a crucial function in decelerating the detrimental effects of global climate change (Pittau et al., 2019).

4.2 | Decisive performance drivers along the life cycle

Per FU, the two most environmentally friendly insulation materials were found to be wood fiber and miscanthus. The latter also incurs the least costs along the life cycle of all bio-based materials assessed. There are various reasons for this superior performance, and these are discussed in the following section.

4.2.1 | Cultivation system

First, the cultivation stage plays a decisive role, as it is a substantial hotspot for environmental impacts along the life cycle of bio-based insulation materials. Miscanthus' good performance is, on the one hand, due to its high biomass output. This has a strong influence on the subsequent results, as all impacts of the cultivation systems are divided by the yield (Meyer et al., 2018). In a central European setting, miscanthus can exceed the yield of hemp by approximately 200% and that of flax by around 300% (Beus et al., 2019; Schmidt et al., 2004; Wagner et al., 2017). On the other hand, its perennial nature is pivotal, as this reduces the number of agricultural management activities required per year. Wood is also likely to have a low environmental burden during the cultivation system because fertilizer and pesticide applications are not required, and these are responsible for most impacts in the agricultural systems.

However, the assumed yield levels of all feedstocks still underlie significant uncertainties. The sensitivity analysis performed shows that changing yield levels to 70% or 130% results in substantial relative impact changes for all feedstocks, especially in the category LU (+43% LU for 70% yield, and −23% LU for 130% yield; Table S1). Considerable changes were also found for FE, ME, and SOD for all agriculturally sourced materials. Decreasing the yield to 70% resulted in relative impact increases exceeding 40% for hemp fiber and flax, but not for miscanthus. This, together with the results of the other impact categories in the sensitivity analysis, indicates a considerably greater sensitivity toward differing yield levels in hemp and flax than in the perennial crop. These results highlight the more environmentally benign performance of miscanthus compared to the annual crops and thus reinforce the argument for an increased utilization of its biomass for material applications. In contrast to the currently dominant energetic use of miscanthus, its material use could also provide more stable market options (Lewandowski et al., 2016; Moll et al., 2020).

The costs of the agricultural cultivation systems amount to 171.47 € t^{-1} for hemp and 206.49 € t^{-1} for flax production, according to Beus and Piotrowski (2017) and Beus et al. (2019). The costs of the miscanthus cultivation system are significantly lower at 44.08 € t^{-1}. Together with its high yield, this underlines the above argument for the material use of miscanthus and its role as a promising biomass feedstock in a future European bioeconomy (Lewandowski, 2016). The costs of the silvicultural system amount to 121.37 € t^{-1}, with the land rent being divided by a timber volume of 336 m^3 ha^{-1} (BMEL, 2014). However, the different wood species and forest management regimes, including regional characteristics, remain factors which hamper definite LCIs (Cardellini et al., 2018; González-García et al., 2014) and thus specific LCCs on silvicultural systems.

4.2.2 | Manufacturing method and additives

Second, there are decisive performance drivers within the manufacturing system. One is the composition of the electricity sources fueling the processing lines. Major sources of electricity demand during manufacture were found to be the drying of wood and heating process to blend wood fiber with additives, and the chipping or decortication processes, which are in turn mainly influenced by moisture content of the biomass (Bitra et al., 2009; Liu et al., 2016). Thus, drier harvest conditions and species with inherently lower water content induce less environmental impacts during subsequent processing. In addition to the electricity composition, performance-enhancing additives are another decisive factor influencing the environmental performance of the manufacturing system of bio-based insulations. Although the assessed wood fiber insulation contained only 1.3% blended PE and PP fiber and only 2.4% aluminum sulfate, both additives contribute significantly to the environmental performance of the product. This tendency has also been found in other LCAs on bio-based insulations, such as by Zampori et al. (2013). In that study, a polyester fiber, which makes up 15% of the mass of a hemp insulation, contributes 61% of the total CO_2 eq. of the product from "cradle to gate" (excluding biogenic CO_2 sequestration). Schmidt et al. (2004) reported a comparable significance of the additive in a flax insulation. Here, a polyester binder, which also constitutes 15% of the material mass, had a larger environmental burden from "cradle to grave" than the entire flax cultivation itself. For a blended miscanthus insulation product containing 14% PP, Uihlein et al. (2008) reported the additive's contribution to be 29.4% of the overall environmental impact. To potentially reduce this environmental hotspot in bio-based insulations, bio-based binders (Viel et al., 2019) and bio-based flame retardants (El Hage et al., 2019; Zheng et al., 2019) have recently been investigated as substitutes for fossil-based additives. However, their contribution to the environmental impact still needs to be assessed. Regardless of whether fossil or bio-based, the omission of additives in the manufacture of bio-based insulations substantially improves their environmental performance during this life cycle stage, as shown in this study for flax and miscanthus insulations.

4.2.3 | Mode of installation

Third, the use phase acts as a decisive performance driver with regard to economic costs. In this study, two different modes of installation were modeled, based on the method proposed by the producer of the material. For hemp fiber, miscanthus insulation, EPS, and stone wool, a manual installation was modeled, while for wood fiber and flax, a mechanical blow-in installation was assumed. It was found that,

per FU, manual installation is about 4.5 times more expensive than blow-in installation, mainly due to the higher time spent and thus wage costs incurred. Yet, regardless of the installation method applied, the environmental burden was found to be negligible; therefore, mechanical installation is to be recommended. In either case, the environmental impacts saved through installation of insulation products can be more than 100 times greater than all burdens induced throughout their life cycle, due to avoided heat, and thus energy losses compared to no installation at all (Schmidt et al., 2003). This highlights the crucial role played by the general application of insulation materials. In other LCC studies, the installation mode and corresponding employment costs are not accounted for. Instead, costs are mostly given for the situation ex-factory (i.e., from factory gate), and these can differ substantially between studies. For instance, for an Italian setting, Lazzarin et al. (2008) indicated costs of 5.37 € m^{-2} for stone wool, 27.30 € m^{-2} for a flax insulation, and 4.86 € m^{-2} for EPS, using an FU of 1 W m^{-2} K^{-1}. Yet, when the FU is set to equal that used in this study, that is, 0.24 W m^{-2} K^{-1}, the costs from Lazzarin et al. (2008) are approximately four times higher and thus all substantially higher than the ex-factory costs presented here. A similar cost range for insulation materials was also found by Klingler et al. (2018), using an FU of 0.2 W m^{-2} K^{-1}. For a central European setting, the authors stated EPS costs of 10.61–17.00 € m^{-2}, a difference of 160% between the lower and upper figure. Similarly, they gave a cost range of 23.18–38.00 € m^{-2} for stone wool, a difference of 164% between lower and upper limits. For wood fiber, the authors stated costs of 22 € m^{-2}, which represents a difference of 184% to the ex-factory costs indicated in the present study. Together, this emphasizes the high level of uncertainty associated with the LCC results for insulations, which mainly depend on the FU, and also stem from the lack of information on manufacturing processes and their energy demands, properties of the insulation materials (i.e., density and thermal conductivity), and the discount rate used for future processes. The latter was omitted from this study for all costs resulting from the use phase and end-of-life system. Instead, a steady state (i.e., the application of current conditions) was assumed with no discount rate (Swarr et al., 2011), since future costs for the corresponding processes are unknown.

4.2.4 | End-of-life scenario

Finally, the end-of-life system constitutes a crucial factor that can significantly influence the overall environmental and economic performance of bio-based insulation materials. This influence depends on the end-of-life scenario chosen. In this study, the scenario was comparable to that chosen by Uihlein et al. (2008) and consisted of disposal by incineration while applying a "cut-off" for energy recovery from all

insulations except stone wool to be consistent with the allocation approach applied. This plausible scenario was selected for the central European setting since it represents the less environmentally burdening alternative compared to landfill disposal. One reason for this is that landfill disposal provokes increases in GWP due to methane formation from the decaying biogenic carbon (Schmidt et al., 2004). This trend was also found by Ortiz et al. (2009) for construction materials, while the authors highlight that a material recycling is the most environmentally friendly treatment with respect to climate impacts.

Generally, the credit for the substituted heat and electricity mix can substantially improve the environmental performance of the bio-based and EPS insulations. Thus, the end-of-life scenario chosen can easily change the overall environmental and economic outcome of an assessment. However, one decisive factor influencing the impacts of this life cycle stage can again be seen in nonrenewable additives. The fossil-based EPS had substantial environmental impacts in the categories FRS, GW, HCT, and WC, where it proved to be the least environmentally friendly insulation under study (Figures 3 and 4). At the same time, EPS had the lowest economic costs from incineration (Table 3); the considerably lower mass required per FU led to comparatively low disposal costs. However, these disposal costs for EPS fail to take into account the substantial external costs incurred through the GWP of the combustion of its fossil matter. This increases the pressure for its substitution in the context of incentivizing the use of climate-friendly building materials (Bürger et al., 2017), especially since EPS is currently the insulation material most used for renovation purposes in Europe (Pittau et al., 2019). By contrast, bio-based products often bear externalities with regard to, for example, eutrophication, as was also found in this study. Any externality, regardless which, should ideally be internalized in LCC calculations of the products; however, methods used for their calculation differ widely and are controversially debated (Nguyen et al., 2016).

5 | CONCLUSIONS

The use of LCA and LCC to compare the environmental and economic performance of different bio-based insulations in order to identify the best is difficult, as respective studies frequently lack comparability. The present study aimed to alleviate this problem, first by measuring the thermal conductivity of four bio-based insulations under ceteris paribus conditions, second by using the same FU and system boundary for all materials under assessment, and third by applying the same LCIA method. The results show that, compared to their bio-based counterparts and the nonrenewable references examined, wood fiber and miscanthus insulation are the most

environmentally friendly materials, with miscanthus insulation and EPS accounting for the least costs along their life cycle. A hotspot analysis of the entire life cycle of the bio-based insulations revealed that, for the most part, cultivation of the feedstock causes the largest environmental impact. By contrast, the manufacturing system and installation were found to be hotspots for economic costs. It is concluded that, overall, the bio-based insulations under study have a more environmentally benign performance than the two nonrenewable counterparts examined, EPS and stone wool. In addition, the utilization of miscanthus as insulation material could offer an environmentally and economically promising higher value product alternative to help facilitate the currently sluggish market integration of its biomass. Finally, it is recommended that, firstly, future environmental impact assessments of bio-based insulations include preceding thermal conductivity measurements, and secondly, that future LCA studies consider a material reuse of insulations, as this would further contribute to mitigating the detrimental effects of climate change.

CONFLICT OF INTEREST
The authors declare that they have no conflicts of interest.

AUTHOR CONTRIBUTIONS
Maximilian Schulte performed the LCA modeling and led the writing process. Iris Lewandowski made substantial contributions to conception and design of the study. Ralf Pude made significant contributions to primary data acquisition and thus the creation of the Life Cycle Inventory. Moritz Wagner made substantial contributions to conception of the study and interpretation of data.

DATA AVAILABILITY STATEMENT
The data that support the findings of this study are available in the supplementary material of this article.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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