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Life cycle assessment of perennial cultivation systems: Advancing applicability and comprehensiveness

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Abstract

With the development of a European bioeconomy, the use of biogenic resources, including lignocellulosic biomass, is likely to increase. Resource-efficient perennial cultivation systems in particular are considered promising sources of sustainably produced biomass to meet the growing demand. They require fewer agricultural procedures than annual systems, as tillage and application of plant protection agents are only necessary during the establishment phase. Perennial systems can contribute to an increase in soil carbon sequestration and be productive on marginal land unsuitable for the cultivation of typical crops. In Europe, the C4 grass miscanthus is the most prominent and best researched perennial crop for lignocellulosic biomass production. Recently, wild plant mixtures have been suggested as a more diverse alternative perennial system.

Perennial cultivation systems have already been the subject of multiple sustainability assessments, with life cycle assessment (LCA) being the method most commonly used. This method aims to provide a holistic depiction of the environmental performance of a product or service. However, two challenges are usually encountered. First, results of agricultural LCAs very much depend on site- and management-specific characteristics. Parameters such as biomass yield, quantity of fertiliser applied and carbon sequestered can vary considerably, impairing the general applicability of the method and related results. Second, most of these studies focus on greenhouse gas emissions only. Land use impacts on biodiversity are commonly neglected, casting doubt on the comprehensiveness that LCA is trying to achieve.

This thesis aims to advance the applicability and comprehensiveness of LCA of perennial cultivation systems. For this purpose, it focuses on three aspects relevant to the assessment of such systems, each of which was addressed by a dedicated research question. These are: 1) How can the conducting and application of LCAs of perennial cultivations systems be simplified? 2) Which methodological approaches are best suited for the consideration of carbon sequestration and storage in LCAs of perennial cultivation systems? 3) How can land use impacts of perennial cultivation systems on biodiversity best be incorporated into the LCA framework?

These questions were answered by applying the LCA method to perennial cultivation systems in three case studies, using specific approaches for the inclusion of sensitivity analysis and the evaluation of carbon sequestration and storage. In addition, information on the biodiversity impacts of perennial crop cultivation was collated by means of a meta-analysis which compared species richness and abundance in annual and perennial crop cultivation systems.

The life cycle inventory phase forms the core of any LCA and encompasses the collection and quantification of inputs and outputs associated with a product system. Depending on the inherent complexity and variability of the system, it can be quite intricate. Thus, the conducting of an LCA can be substantially simplified by focusing on a few relevant inputs and outputs only. In this thesis a global sensitivity analysis was used to identify the most important inventory parameters in the greenhouse gas assessment of miscanthus cultivation: carbon sequestration, biomass yield, length of the cultivation period, nitrogen and potassium fertiliser application, and the distance over which the harvested biomass is transported. Focusing on these inventory parameters, a simplified model was developed. It allows farmers and SME active in miscanthus-based value chains easy access to customised LCA results. The outcome underlines the importance of global sensitivity analyses and simplified models in advancing the applicability of LCAs of agricultural systems.

This thesis includes a detailed analysis of the relevance of carbon sequestration and storage in the sustainability assessment of perennial cultivation systems. It was found that the quantity and in particular the permanence of carbon sequestered through the cultivation of perennial crops are critical for their favourability in terms of global warming impacts. Two alternative methodological approaches for the quantification of carbon sequestered were tested within two of the case studies – a simple carbon model and an allometric approach. In addition, the handling of the uncertain permanence of the carbon storage was reflected upon. The approaches were compared with regard to their suitability for use by typical LCA practitioners. It was concluded that allometric models should be used for the quantification of carbon sequestered and the corresponding amount accounted for as delayed emissions according to the International Reference Life Cycle Data System (ILCD) handbook. This combination provides a manageable and transparent approach for the accounting of benefits from carbon sequestration and storage, and also prevents their overestimation.

Established impact assessment methods such as ReCiPe2016 suggest characterisation factors for the incorporation of land use impacts on biodiversity into LCA. These characterisation factors use relative species richness as an indicator and assume a higher species richness in perennial than annual cultivation systems. This thesis includes a critical review of these characterisation factors, drawing on the results of the meta-analysis comparison of species richness in annual arable crops and perennial rhizomatous grasses. The meta-study did not confirm a higher number of species in perennial rhizomatous grasses than in annual arable crops. Based on these findings, it was concluded that LCA studies on perennial cultivation systems need to be cautious in their application of the land use characterisation factors suggested in present-day impact assessment methods. Criticisms of the approach include the application of one single characterisation factor for diverse perennial cultivation systems such as wild plant mixtures and miscanthus and the sole focus on species richness. In future, LCA research should focus on context-specific adjustment options for land use characterisation factors to ensure an adequate representation of biodiversity impacts in agricultural LCAs. Finally, the current focus on species richness in biodiversity impact assessment needs to be reassessed – phylogenetic diversity would be a promising alternative in this context.

The conclusions drawn and recommendations derived in this thesis can, in general, also be applied to other types of agricultural production systems, and thus support the wider application of LCA in decision support for sustainable development.

Zusammenfassung

Mit der fortschreitenden Entwicklung einer europäischen Bioökonomie wird die Nutzung biogener Ressourcen, wie beispielsweise von lignocellulose-haltiger Biomasse, zunehmen. Besonders mehrjährige Anbausysteme werden als vielversprechende Quellen betrachtet, die zur Bereitstellung nachhaltig produzierter Biomasse beitragen können. Diese Systeme nutzen Ressourcen sehr effizient und benötigen weniger Kulturmaßnahmen als einjährige Anbausysteme. Maßnahmen, wie Bodenbearbeitung oder Ausbringung von Pflanzenschutzmitteln sind lediglich während der Etablierungsphase notwendig. Grundsätzlich können mehrjährige Anbausysteme zu einer verstärkten Kohlenstoffsequestrierung im Boden beitragen und auch auf marginalem Land angebaut werden, welches für den Anbau klassischer Feldfrüchte nicht geeignet ist. Das mehrjährige Gras Miscanthus ist das bekannteste und meist untersuchteste mehrjährige Anbausystem für die Bereitstellung lignocellulose-haltiger Biomasse in Europa. In den letzten Jahren wurden zunehmend auch mehrjährige Wildpflanzenmischungen als alternative mehrjährige Systeme vorgeschlagen.

Mehrjährige Anbausysteme wurden im Rahmen zahlreicher Studien bereits Nachhaltigkeitsbewertungen unterzogen. Meist wird hierfür die Methode der Ökobilanzierung (LCA) verwendet. Diese zielt auf eine ganzheitliche Untersuchung und Darstellung der Umweltauswirkungen eines Produkts oder einer Dienstleistung ab. In diesen Studien treten oftmals zwei Schwierigkeiten auf: Einerseits hängen die Resultate von agrarischen LCAs stark von Standort- und Management-spezifischen Charakteristika ab. Parameter wie der Biomasseertrag, die Menge der eingesetzten Düngemittel sowie des sequestrierten Kohlenstoffs variieren beträchtlich. Dies erschwert die allgemeine Anwendbarkeit der LCA sowie der Nutzung der Resultate. Anderseits beschränken sich die Studien zumeist auf die Untersuchung der Treibhausgasemissionen. Durch Landnutzung bedingte Biodiversitätsauswirkungen werden oftmals vernachlässigt, wodurch die Ganzheitlichkeit des Ansatzes in Frage gestellt wird.

Ziel dieser Arbeit ist es, die Anwendbarkeit und Ganzheitlichkeit von LCAs mehrjähriger Anbausysteme zu fördern. Hierzu wurde das Augenmerk auf drei relevante Aspekte der Bewertung dieser Systeme gelegt, die im Rahmen einer Forschungsfrage adressiert wurden: 1) Wie kann die Durchführung und Anwendung von LCA mehrjähriger Anbausystemen vereinfacht werden? 2) Welche methodischen Herangehensweisen eignen sich für die Betrachtung von Kohlenstoffsequestrierung und –speicherung in LCAs mehrjähriger Anbausysteme? 3) Welche Herangehensweisen eignen sich für die Abbildung landnutzungsbedingter Biodiversitätsauswirkungen in LCAs mehrjähriger Anbausysteme?

Um diese Fragen zu beantworten, wurde die Methode der Ökobilanzierung im Rahmen dreier Fallstudien auf mehrjährige Anbausysteme angewandt. Dabei wurden verschiedene Herangehensweisen zur Durchführung von Sensitivitätsanalysen und der Bewertung von Kohlenstoffsequestrierung und -speicherung genutzt. Zusätzlich wurden Informationen über Biodiversitätsauswirkungen mehrjähriger Anbausysteme zusammengefasst. Hierzu wurde eine Meta-Analyse durchgeführt, in welcher Artenreichtum und Abundanz in ein- und mehrjährigen Anbausystemen verglichen wurde.

Die Sachbilanz (LCI) bildet den Kern einer jeden LCA und umfasst die Zusammenstellung und Quantifizierung von Inputs und Outputs eines Produktsystems. In Abhängigkeit der Komplexität und Variabilität des Systems kann diese aufwendig sein. Durch die Fokussierung auf wenige wesentliche Inputs und Outputs kann die Durchführung einer LCA daher stark vereinfacht werden. In dieser Arbeit wurden mithilfe einer globalen Sensitivitätsanalyse die wichtigsten Parameter für die Erstellung eines Treibhausgas-Assessments des Miscanthusanbaus identifiziert: Kohlenstoffsequestrierung, Biomasseertrag, Dauer der Anbauperiode, Stickstoff- und Kaliumgabe und die Transportdistanz des Ernteguts. Basierend auf diesen Parametern wurde ein vereinfachtes Modell entwickelt. Landwirte sowie kleine und mittlere Unternehmen, die Teil von Miscanthus-basierten Wertschöpfungsketten sind, bekommen somit einen einfachen Zugang zu individuell anpassbaren LCA Resultaten. Diese Resultate unterstreichen die Bedeutung von globalen Sensitivitätsanalysen und einfachen Modellen für eine verbesserte Anwendbarkeit von agrarischen LCAs.

Die Bedeutung von Kohlenstoffsequestrierung und –speicherung für die Nachhaltigkeitsbewertung von mehrjährigen Anbausystemen wurde in dieser Arbeit detailliert analysiert. Es wurde gezeigt, dass Quantität und vor allem Dauerhaftigkeit der Kohlenstoffspeicherung während des Anbaus mehrjähriger Pflanzen zentrale Faktoren für die Vorzüglichkeit dieser Systeme in Bezug auf die Auswirkungen auf die globale Erwärmung sind. Zwei methodische Herangehensweisen zur Quantifizierung der Kohlenstoffspeicherung wurden im Rahmen zweier Fallstudien getestet – ein einfaches Kohlenstoffmodell sowie eine allometrische Abschätzung. Ergänzend wurde der Umgang mit einer fraglichen Dauerhaftigkeit der Kohlenstoffspeicherung kritisch reflektiert. Die Herangehensweisen wurden im Hinblick auf ihre Eignung für die Nutzung durch typische LCA-Anwender verglichen. Es wurde empfohlen, allometrische Modelle für die Quantifizierung der Kohlenstoffspeicherung heranzuziehen und die resultierende Kohlenstoffmenge als zeitlich verzögerte Emission entsprechend des International Reference Life Cycle Data System (ILCD) Handbuchs zu erfassen. Diese Kombination stellt ein handhabbares und transparentes Vorgehen für die Betrachtung von Vorteilen aus der Kohlenstoffsequestrierung und -speicherung dar und verhindert deren Überbewertung.

Etablierte Wirkungsabschätzungsmethoden (LCIA-Methoden) wie ReCiPe2016 beinhalten Charakterisierungsfaktoren für die Berücksichtigung landnutzungsbedingter Biodiversitätsauswirkungen. Diese Charakterisierungsfaktoren nutzen den relativen Artenreichtum einer Landnutzung als Indikator und gehen von einem höheren Maß an Artenreichtum in mehrjährigen als in einjährigen Anbausystemen aus. Mithilfe der Ergebnisse der Meta-Analyse, die den Artenreichtum in einjährigen Ackerkulturen mit denen in mehrjährigen rhizombildenden Gräsern verglich, wurden diese Charakterisierungsfaktoren hinterfragt. In der Meta-Studie konnten für die mehrjährigen Anbausysteme keine signifikant höheren Artenzahlen nachgewiesen werden. Basierend auf diesen Ergebnissen wird empfohlen, die in den etablierten LCIA-Methoden vorgeschlagenen Charakterisierungsfaktoren für die Bewertung mehrjähriger Anbausysteme nur vorsichtig zu nutzen. Die Nutzung eines einzigen Charakterisierungsfaktors für diverse mehrjährige Anbausysteme wie Miscanthus und Wildpflanzenmischungen sowie der starke Fokus auf den Indikator Artenreichtum stellen Defizite dar. Zukünftige Forschungsarbeiten in diesem Bereich sollten auf eine kontext-abhängige Anpassung der Charakterisierungsfaktoren hinwirken, um eine adäquate Darstellung der Biodiversitätsauswirkungen in agrarischen LCAs zu ermöglichen. Abgesehen hiervon sollte der starke Fokus auf die Verwendung des Artenreichtums als Biodiversitätsindikator überdacht werden – die phylogenetische Diversität stellt hier einen vielversprechenden Ansatz dar.

Die aus dieser Arbeit hervorgegangenen Schlussfolgerungen und Empfehlungen für mehrjährige Anbausysteme können im Allgemeinen auch auf andere agrarische Produktsysteme übertragen werden. Somit können sie zu einer weiterführenden Anwendung von LCA für die Unterstützung von Entscheidungen im Sinne einer nachhaltigen Entwicklung beitragen.

Chapter 1

1.0 General Introduction

The Earth system provides food, raw materials and clean air. Human life, as we know it today, depends on these services and it is in the own interest of humankind to ensure the continuity of these functions. This will only be possible, if the Earth system remains in a resilient state, meaning it is able to adapt to changing conditions in the long run. To date, this resilience is interfered and disturbed by humankind's actions (Steffen et al., 2015).

For this reason, ten planetary boundaries have been proposed and detailed in 2015 (Steffen et al., 2015). The concept extends on Rockström et al. 2009. It emphasises processes that are critical for the Earth's resilience on a global scale and reveals their current status. This includes processes such as *climate change*, the conservation of *biosphere integrity*, *land-system change*, *freshwater use*, and changes in *biochemical flows*. It was attempted to quantify these global processes and indicate carrying capacities that shall not be exceeded in order to ensure the resilience of the Earth system. For four of them – *climate change, biosphere integrity, biochemical flows* and *land system change* – the boundaries have already been surpassed. Climate change and biosphere integrity are intertwined with all the other processes. Even if only one of them is substantially changed, it can individually drive the Earth system out of its current stable state. Thus, both are considered *core planetary boundaries* which require distinguished attention (Steffen et al., 2015).

Climate change describes the shifts in climate patterns, which result from the increase of the average temperatures on Earth. Within the planetary boundary concept, changes in the process are monitored using atmospheric CO_2 concentrations (in parts per million: ppm) as an indicator. While the corresponding planetary boundary was set at 350 ppm CO_2, an annual average concentration of 407 ppm was reported for 2018 (NOAA, 2021). Major drivers of the exceedance of the boundary are human activities comprising the combustion of fossil fuels and deforestation (Rosenbaum, Hauschild, et al., 2018).

Biosphere integrity is measured taking the global species extinction rate as a metric. It serves as a surrogate for the loss of genetic diversity, which is fundamental for the potential of Earth's biosphere to continuously adapt and thus succeed in the long run. Depending on the estimate, the extinction rate is currently exceeding the planetary boundary by a factor of ten to 100 (Steffen et al., 2015). Anthropogenic land use is the key driver here and has decisive influence on global species extinction (Maxwell, Fuller, Brooks, & Watson, 2016; Newbold et al., 2020).

Given their relevance as *core planetary boundaries*, it is of outstanding significance to take actions in order to return both of these processes towards a safe operating space. The bioeconomy has been suggested as one course of action by governments at regional (MLR & MLU, 2019), national (BMBF & BMEL, 2020) and supranational level (European Commission, 2018).

1.1 Bioeconomy

The conference on *New perspectives on the knowledge-based bio-economy* held by the European Commission in 2005 (European Commission, 2005) marks the starting point for the development of a European bioeconomy. The concept has gained substantial attention since then and undergone a dynamic evolution, indicated by the release of the European Union's bioeconomy strategy in 2012

(European Commission, 2012) and its update in 2018 (European Commission, 2018). The former defined the concept as encompassing “the production of renewable biological resources and the conversion of these resources into value added products, such as food, feed, bio-based products and bioenergy.” (European Commission, 2012, p. 9). According to the latter, the bioeconomy “includes and interlinks: land and marine ecosystems and the services they provide; all primary production sectors that use and produce biological resources (agriculture, forestry, fisheries and aquaculture); and all economic and industrial sectors that use biological resources and processes to produce food, feed, bio-based products, energy and services” (European Commission, 2018, p. 4).

With the bioeconomy, the European Union intends to foster the transition to a more sustainable future. As substantiated in the strategies, the bioeconomy shall strengthen the European Union’s competitiveness and reduce the dependency on non-renewable resources. At the same time, it should also support the Union in “mitigating and adapting to climate change” (European Commission, 2018, p. 9). These objectives should be achieved while “ensuring food and nutrition security” (European Commission, 2018, p. 8) as well as sustainably managing natural resources.

1.2 Biomass for a bioeconomy

As can be seen from the definition, the use of biological resources is a central pillar of the bioeconomy and essential to achieve the aforementioned objectives. Biological resources encompass all kind of organic material derived from animals, plants, micro-organism or waste (European Commission, 2018; Lewandowski, 2015). In the following, it will be referred to as biomass.

Almost two thirds (on a mass basis) of the biomass available in the European Union is derived from the agricultural sector. The forestry sector contributes approximately 30%, while the remainder is supplied by fishery and aquaculture (Gurría et al., 2020). Biomass has a range of applications, including food, feed, biomaterials and bioenergy. In 2018, approximately half of the biomass produced in and imported to Europe was used for food and feed, while roughly a fifth was used for bioenergy and materials application, each (Gurría et al., 2020). The quantitative use of biomass for the production of biomaterials and the generation of bioenergy is anticipated to increase with the advancing transition to a bioeconomy. In fact, the European Union is actively promoting and incentivising the use of biomass.

Its use for the production of bio-based and biodegradable materials can for instance replace plastics as suggested by the European Environmental Agency and emphasised in the bioeconomy strategy (European Commission, 2018; European Environment Agency, 2018). Standards, labels and certifications for a range of bio-based products including lubricants, bio-polymers and sanitary products have been developed and proposed, for instance the EU Ecolabel (Regulation (EC) No 66/2010).

However, the prime example for the promotion of the use of biomass is the Renewable Energy Directive (RED), which was established in 2009 (RED I; (Directive 2009/28/EC) and recast in 2018 (RED II; (Directive (EU) 2018/2001). The directive requires member states to increase the share of renewable energies in their final energy consumption (RED I: 20% in 2020; RED II: 32% in 2030) and has previously driven the demand for biofuels in Europe (Scarlat, Dallemand, Monforti-Ferrario, & Nita, 2015).

RED I and II set binding targets of 10 and 14% renewable energy used in the transport sector by 2020 and 2030, respectively. A large share of this target size was meant to be supplied using biomass but it also includes energy from non-biomass such as solar or wind, and biomass-based energy. In the first phase of RED I most of the biomass demand was supplied by first generation resources such as maize and rapeseed. First generation biofuels are questioned with regard to their environmental sustainability and a potential competition with the global food supply. In addition, they partly fail to fulfil the revised greenhouse gas mitigation targets (Humpenöder, Schaldach, Cikovani, & Schebek, 2013). For this reason, RED II set a cap on the share of conventional biofuels and defined a sub-target for the use of biofuels from advanced biomass resources (Directive (EU) 2018/2001). Advanced biomass resources include a broad range of non-edible feedstock such as waste from municipalities or biomass processing (e.g., empty palm fruit bunches, grape marcs) as well as algal biomass and non-food cellulosic material. The latter refers to biomass mainly consisting of cellulose and hemicellulose, having a lower lignin content than forest biomass and woody energy crops. Accordingly, the group of non-food cellulosic material includes harvest residues such as straw, stover and shells as well as grassy energy crops such as switchgrass (*Panicum virgatum*) and miscanthus (Directive (EU) 2018/2001).

1.3 Perennial cultivation systems

Switchgrass and miscanthus are expected to play a major role in meeting the future demand for advanced biomass resources (Lewandowski, 2016). Both are lignocellulosic perennial crops, which once established, grow for several years and are harvested annually. In practice, two major groups of perennial crops can be distinguished: woody plants, including short rotation coppice such as poplar, and perennial grasses, including switchgrass and miscanthus (Ledo et al., 2020). In general, the integration of perennial cultivation systems is considered favourable for the agricultural system as a whole. This is due to a number of beneficial characteristics, including environmental and socio-economic aspects. From an environmental perspective this includes carbon sequestration, a reduction in both nutrient leaching and soil erosion as well as the anticipated provision of habitats for a number of species. Socio-economic aspects include reduced management requirements and stable biomass yields during the cultivation period as well as versatile utilisation options (Lewandowski, 2016; McCalmont et al., 2017). Using miscanthus as an example, these characteristics will be introduced in detail in the following paragraph.

Miscanthus

The perennial grass miscanthus stems from South-East Asia. For the last 30 years it has been the subject of substantial research efforts in Europe (Clifton-Brown et al., 2017; Lewandowski, Clifton-Brown, Scurlock, & Huisman, 2000). Presently, *Miscanthus x giganteus* is the only commercially cultivated species. It is a sterile hybrid that is propagated via rhizomes (Lewandowski et al., 2016). After the planting of rhizomes and an establishment phase of two years, miscanthus plantations can be harvested annually, delivering stable long-term biomass yields (Gauder, Graeff-Hönninger, Lewandowski, & Claupein, 2012; Larsen, Jørgensen, Kjeldsen, & Lærke, 2014). Commercially, miscanthus is mainly used for heat and power co-generation. For this purpose, it is harvested brown in early spring, when moisture contents are low. In general, reported biomass yields for brown harvest range from 10 to 20 t DM ha^{-1} yr^{-1} in temperate regions of Europe (Witzel & Finger, 2016). Even yields of up to 25 t DM ha^{-1} yr^{-1} can be encountered occasionally (Lewandowski et al., 2000).

During the entire cultivation period the soil is covered, which reduces the risk of soil erosion and is beneficial for carbon sequestration (Harris, Spake, & Taylor, 2015). It has been reported that miscanthus cultivation on arable land increases the soil carbon content by 0.7-2.2 t carbon ha^{-1} yr^{-1} (McCalmont et al., 2017). This is due to the substantial carbon input from the degradation of leaf litter and the substantial below-ground biomass including roots and rhizomes (Ledo, Heathcote, Hastings, Smith, & Hillier, 2018).

With the end of the vegetation and the senescence of the crop, miscanthus effectively relocates nutrients, in particular nitrogen, from the above-ground biomass into the below-ground system (Cadoux, Riche, Yates, & Machet, 2012). These nutrients can be remobilised in the following vegetation period, ensuring efficient nutrient use and reducing nitrate leaching from the decay of above-ground biomass.

In addition to the above mentioned aspects, miscanthus is considered beneficial for biodiversity in agricultural landscapes, particularly when compared with typical annual crops (Immerzeel, Verweij, van der Hilst, & Faaij, 2014; Werling et al., 2014). For instance, soil organisms such as earthworms can take advantage from the extended soil rest in comparison with annual crops. Other animals such as deer and hares can benefit from the late harvest in early spring, as the standing miscanthus crops can provide shelter and habitat (Dauber, Jones, & Stout, 2010).

From a socio-economic perspective, miscanthus seems promising as it requires few management operations considering the entire cultivation period of 20 years. Tillage is only required prior to establishment and after the cultivation period (Lewandowski, 2016). As established miscanthus effectively suppresses weeds, plant protection is only necessary during the establishment phase. Further plant protection measures are usually not required. Due to the efficient nutrient mobilisation between above- and below-ground biomass, fertilisation requirements are comparatively low and in commercial practice mainly limited to the application of potassium and phosphorus (McCalmont et al., 2017).

A major target in miscanthus breeding is tolerance to cold, drought stress and saline conditions (Lewandowski et al., 2016). These breeding efforts aim to increase miscanthus' ability to grow on marginal or contaminated land. Marginal lands are characterised by biophysical conditions which prevent an economically viable production of conventional crops (Cossel et al., 2019). Farmers could generate value from such land by cultivating miscanthus and taking advantage of the wide usability of its biomass. Miscanthus biomass can be used for bioenergy production (e.g. direct combustion for heat and power generation (Iqbal et al., 2017), anaerobic digestion to biogas (Kiesel & Lewandowski, 2017), production of cellulosic ethanol (van der Weijde et al., 2013) and material applications such as insulation material (Schulte, Lewandowski, Pude, & Wagner, 2021).

Wild plant mixtures (WPM)

As mentioned above, perennial cultivation systems are considered beneficial for biodiversity in agricultural landscapes (Werling et al., 2014). However, crops such as miscanthus and switchgrass are usually grown in monocultures, which is not optimal for biodiversity. For this reason, diverse perennial cropping systems have been suggested. An example for this are perennial wild plant mixtures (WPM), which contain mixes of diverse flowering plant species including annual, biennial and perennial ones. Due to the plant diversity and the presence of flowers, WPM support a higher level of biodiversity than typical perennial crops such as miscanthus (Cossel, 2020). Similar to miscanthus and other perennial crops, management efforts are mainly to be undertaken during the establishment period. During the cultivation period itself, which generally amounts to 5 years although it could be potentially extended, only few operations besides harvest are required (Cossel, 2020). WPM provide lignocellulosic biomass which can be widely used. Most of the commercially grown WPM are currently used for biogas production (Cossel, Pereira, & Lewandowski, 2021).

The previous paragraphs indicated that perennial cultivation systems are a promising source for meeting the biomass demand of a growing bioeconomy. Miscanthus is a prime example in this regard due to its productivity, resource-use efficiency and ability to grow on all kinds of land. Although WPM might not be as productive as miscanthus, they offer additional ecological benefits by supporting pollinators. For these reasons, this thesis uses WPM and miscanthus as examples for perennial cultivation systems in a European context.

1.4 Life cycle assessment (LCA)

In line with the bioeconomy's goal of supporting the transition to a sustainable future, a socio-economic and environmentally benign biomass production and supply has to be ensured (Lewandowski, 2015; Pfau, Hagens, Dankbaar, & Smits, 2014). This requires a systemic approach, which considers all processes and activities required to provide a product as well as all associated environmental impacts. In terms of the environment, this means that climate change mitigation is ensured while pressure on other environmental aspects such as biodiversity is avoided (Hauschild, Rosenbaum, & Olsen, 2018). Scientifically-founded monitoring is required to ensure this when, for instance, comparing divergent feedstock or product options.

Life cycle assessment (LCA) is a tool that fulfils these requirements: It takes a life cycle perspective, covers a broad range of environmental concerns and describes them in quantitative terms (Bjørn, Owsianiak, Molin, & Laurent, 2018). It is the preferred tool for environmental assessment in academia and industry as it is widely recognised and standardised. The International Organization for

Standardization (ISO) defines LCA as a technique assessing resources used and potential environmental impacts associated with the life cycle of a product (including, raw material acquisition, production, use stages and waste management) (ISO, 2006a).

The ISO norms 14040 and 14044 are fundamental in this regard and define the structure that any LCA shall follow (ISO, 2006a, 2006b). This includes four phases, namely 1) goal and scope definition, 2) inventory analysis, 3) impact assessment and 4) interpretation. Each of these is characterised by specific elements, which will be briefly introduced in the following.

The first phase sets the studies' goals clarifying the question(s) to be answered as well as the target audience. In addition, it delineates the studies' scope stating the functional unit and outlining the product system under investigation. In the second phase, information related to the investigated product system is collected and processed into a life cycle inventory (LCI). This includes all kind of flows such as products and wastes, but also exchanges with the environment like resource extraction and emissions. In the third phase, (life cycle) impact assessment (LCIA), the LCI information is translated into potential environmental impacts using models from environmental science. Finally, the results from the impact assessment are interpreted in the fourth phase in view of the studies' goal as stated in the first phase. (Hauschild, 2018)

In the following, the *life cycle inventory* (LCI) analysis and *life cycle impact assessment* (LCIA) are scrutinized with regard to the specific characteristics of agricultural LCAs, which is due to the present thesis' focus on perennial cultivation systems.

1.4.1 Life cycle inventory (LCI) of perennial cultivation systems

During the LCI, inputs and outputs required within a product's life cycle are compiled and quantified (ISO, 2006b). As such, it is the core of any LCA. Given the number of physical flows required for, e.g., the manufacture of a certain product, this phase can be quite complex and time-consuming (Bjørn, Moltesen, et al., 2018). This applies in particular to agricultural products, for which inventories require the consideration of emissions that are associated with agricultural activities but cannot easily be tracked. This includes, amongst others, emissions of nitrogen (nitrate, nitrous oxide, ammonia, nitrogen oxides) and phosphorus (phosphorus, phosphate) due to the application of nitrogen and phosphorus fertilisers but also the release of toxic substances to the environment caused by the application of plant protection agents. These emissions are usually estimated using default emission factors or estimations models (Peter, Fiore, Hagemann, Nendel, & Xiloyannis, 2016). The major challenge in agricultural LCAs however, is the inventory's substantial variation due to context-dependent conditions such as climate, soil type as well as agricultural practices (Goglio et al., 2015). In view of perennial crops, this complexity is even further increased due to the duration of the cultivation period (up to 20 years) and the consideration of carbon sequestration associated with their cultivation (Ledo et al., 2018; Ledo et al., 2020). Carbon sequestration is highly variable owing to site-specific parameters (Rowe et al., 2016) and has previously shown to be critical for the carbon footprint of perennial crop production (Lask, Wagner, Trindade, & Lewandowski, 2019; Sanscartier et al., 2014).

1.4.2 Life cycle impact assessment (LCIA)

The main purpose of the LCIA phase is to relate the in- and output information derived in the LCI phase to potential environmental impacts. This is based on models that estimate an emission's effect on the environment using cause-effect chains (Rosenbaum, Hauschild, et al., 2018). For this, elementary flows derived in the LCI, are classified considering their potential environmental effects. For instance, greenhouse gas emissions such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are summarised in a specific *impact category*. The individual emissions are then characterised using factors (*characterisation factors*) which describe each flow's relative contribution to the environmental concern, as derived from environmental models. As an example, characterisation factors of 1 kg CO_2eq kg^{-1}, 30 kg CO_2eq kg^{-1}, and 265 kg CO_2eq kg^{-1} for CO_2, fossil CH_4 and N_2O, respectively have been proposed by the Intergovernmental Panel on Climate Change (IPCC) (Myhre et al., 2013). The values are then summed up to derive the *midpoint (impact) indicator* (e.g., global warming potential or climate change, given in kg CO_2eq kg^{-1}) for the impact category under consideration (Rosenbaum, Hauschild, et al., 2018). Midpoint indicators can be further translated into endpoint indicators (using midpoint-to-endpoint characterisation factors), which indicate how human health, ecosystem quality or natural resources are potentially affected by the physical flows described. Global warming impacts can thus be expressed in terms of potential impact on human health (e.g., in disability-adjusted life years) and the ecosystem (e.g., in species lost) (Huijbregts et al., 2016). Besides climate change, a broad range of environmental impacts, including aquatic eutrophication, acidification and more can be assessed using LCIA methods such as Impact World+ and ReCiPe 2016 (Bulle et al., 2019; Huijbregts et al., 2016). The latter for instance, includes midpoint and midpoint-to-endpoint characterisation factors for 18 midpoint and three endpoint categories.

In practice, LCAs of bioeconomic systems, such as perennial cultivation systems, assess only a selection of midpoint impact categories (Wagner & Lewandowski, 2017). Global warming is the impact category most widely analysed. This is due to the relevance of climate change in the public perception and the bioeconomy's objective of mitigating climate change. In addition, global warming is a comparatively well-defined impact category with an agreed indicator (kg CO_2eq) (Rosenbaum, Hauschild, et al., 2018).

Other environmental impacts are not as easily measured. For instance, biodiversity impacts are largely excluded from present LCAs of annual and perennial cultivation systems (Gabel, Meier, Köpke, & Stolze, 2016). This is despite the fact that biodiversity is critical to the well-being of humankind due to its relevance for the functioning of the planet's ecosystems, as emphasised in the planetary boundaries concept (Steffen et al., 2015).

The Convention of Biological Diversity (CBD) defines biodiversity as "the variability among living organisms from all sources including, inter alia, terrestrial, marine, and other aquatic ecosystems and the complexes of which they are part; this includes diversity within species, between species and of ecosystems" (UN, 1992, p. 3). It is thus considered a complex, multifaceted concept, which cannot easily be operationalised (Gabel et al., 2016). Globally, biodiversity is endangered by a range of man-made drivers, including habitat loss due to land use, climate change, excessive release of nutrients, overexploitation, and the spread of invasive species (Winter, Pflugmacher, Berger, & Finkbeiner, 2018). For this reason, the mentioned drivers and the associated impacts need monitoring in sustainability assessments such as LCA. Impact assessment methods usually cover three of these drivers, namely global warming, nutrient release (eutrophication), and land use (Helin, Holma, & Soimakallio, 2014). Land use, in particular, is considered the most important driver of global biodiversity losses, mainly for agricultural production systems (Maxwell et al., 2016; Newbold et al., 2020). Thus, the incorporation of land use impacts on biodiversity is a focal point of LCIA research (Curran et al., 2016) and is especially relevant for agricultural LCAs (Gabel et al., 2016). Consequently, this thesis focuses on the impacts on biodiversity associated with the use of land for perennial cultivation systems such as miscanthus and WPM.

1.4.3 LCA in practice

In today's practice, LCAs are used in a wide range of applications. The major rationale is to support decision-making processes of governments, industries and consumers (Owsianiak, Bjørn, Laurent, Molin, & Ryberg, 2018). The European Union has a strong record of using LCA-based approaches in policy implantation and regulation. The Renewable Energy Directive (RED; (Directive 2009/28/EC) & (Directive (EU) 2018/2001)) has been accompanied by a carbon footprint methodology from its beginning. This was required as the greenhouse gas mitigation potential of renewable energy had to be proven over their life cycle in order for the fuels to be eligible under the RED regulations. Beyond the energy sector, the European Union encourages the use of LCA approaches to assess and communicate the environmental performance of all kind of products in a consistent way. This takes place under the umbrella of the product environmental footprint (PEF) method, which presents an advancement of previous standardisation approaches like environmental product declarations (EPD) (Finkbeiner, 2014b; Galatola & Pant, 2014; Zampori & Pant, 2019). In industry, LCAs are typically used in product development in order to identify environmental hotspots. However, the approach is increasingly used in marketing to communicate environmental benefits of a product in comparison with potential competitors (Owsianiak et al., 2018).

Due to the complexity and variability described above, this can be challenging in the context of agricultural value chains. This applies in particular to perennial cultivation systems and perennial crop-based value chains, which are mainly driven by small and medium-sized enterprises (SME). SME struggle with the application and conduct of LCAs for several reasons, including the cost and complexity of the method (Kurczewski, 2014). Close collaboration with LCA experts could help to overcome these issues (Zackrisson, Rocha, Christiansen, & Jarnehammar, 2008). However, appropriately skilled personnel in the companies' workforces is usually lacking (Kurczewski, 2014). For this reason, simplification of the LCA method's conduct and use is a promising and necessary step forward in order to promote the widespread use of LCA (Zamagni, Masoni, Buttol, Raggi, & Buonamici, 2012).

1.5 Objectives of the thesis

As shown above, perennial cultivation systems hold substantial potential to contribute to meeting the demand for sustainably produced biomass that will be required for a bioeconomy. For this reason, perennial crop-based value chains have been the subject of manifold environmental sustainability assessments using the life cycle assessment framework. However, two substantial challenges are regularly encountered in these assessments:

First, LCAs of agricultural systems are characterised by considerable complexity. This is mainly due to the high degree of system variability caused by management- and site-dependent factors in perennial cultivation systems. Results of greenhouse gas assessments in particular are strongly determined by the level of carbon storage which depends on site-specific conditions and, at the same time, on methodological assumptions and approaches. For these reasons, conclusions drawn from an assessment within one context cannot easily be applied to another context. Although it is possible for LCA practitioners to generate a fully context-dependent LCA, this is rarely done given the complexity and resource-intensity. By contrast, for farmers who cultivate perennial crops and small companies that process them, it is largely impractical as they lack the necessary LCA know-how. Together, this limits a broader application of LCA for perennial cultivation systems.

Second, the integration of perennial cultivation systems into agricultural landscapes of temperate climates is commonly associated with positive biodiversity impacts, usually related to an extended soil rest and habitat provision. In combination with carbon sequestration, this is one of the strongest arguments for a wider deployment of perennial cultivation systems in agricultural landscapes. Existing LCAs tend to overlook the impacts on biodiversity associated with land use on account of the scarce knowledge on the actual effects and the imperfect implementation and incorporation of land use impacts on biodiversity in LCIA methods.

With these challenges in mind, this thesis aims to advance the applicability and comprehensiveness of life cycle assessments of perennial cultivation systems in order to facilitate their use in environmental sustainability management. More specifically, it aims to:

- establish a methodological approach to simplify the conducting and application of LCAs of perennial cultivation systems, while embracing management- and site-specific variability,
- improve comprehensiveness by supporting the incorporation of land use impacts of perennial cultivation systems on carbon levels and biodiversity.

Based on these objectives, three research questions were derived.

1) How can the conducting and application of LCAs of perennial cultivation systems be simplified?
2) Which methodological approaches are best suited for considering carbon sequestration and storage in LCAs of perennial cultivation systems?
3) How can land use impacts of perennial cultivation systems on biodiversity best be incorporated into the LCA framework?

These research questions are addressed within four publications, each of which is included as a separate chapter in the thesis. **Chapter 2** is dedicated to research question 1 and proposes a simplified model for estimating the greenhouse gas emissions associated with the cultivation of miscanthus in European conditions using only six determining parameters.

In line with research question 2, **Chapters 3** and **4** present different methodological approaches for the handling of carbon sequestration and storage associated with the cultivation of wild plant mixtures and miscanthus. Research question 3 is addressed in Chapters 4 and 5. In **Chapter 4**, an operational LCA approach the assessment of land use impacts on biodiversity is tested. In **Chapter 5,** species richness and abundance in miscanthus and common annual arable crops are compared. The results of this comparison are used in Chapter 6.3 to critically reflect on the applied approach for assessing the biodiversity impact of agricultural land use.

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Chapter 2

2.0 A parsimonious model for calculating the greenhouse gas emissions of miscanthus cultivation using current commercial practice in the UK

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ORIGINAL RESEARCH

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A parsimonious model for calculating the greenhouse gas emissions of miscanthus cultivation using current commercial practice in the United Kingdom

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Abstract

Life cycle assessment (LCA) is a widely recognized tool for the assessment of the potential environmental impacts associated with the life cycle of a product or service. The environmental impact category most commonly quantified in LCAs is global warming potential, a measure of greenhouse gas (GHG) emissions. For agricultural products such as miscanthus, the creation of an inventory can be labour-intensive and is context-specific. This impairs the transfer of results to comparable but not necessarily similar situations. Farmers and small- and medium-sized enterprises cannot easily dedicate resources for this purpose (in particular when using marginal land) and often lack the expertise to do so. Simplified LCA models could offer a promising solution to this problem. They are simplified versions of more complex models that require only a few critical parameters to calculate representative results. This study develops such a model for the computation of GHG emissions associated with commercial miscanthus cultivation. The model focuses on rhizome-based propagation and the indirect harvesting method (cutting to swath, swathing, baling). A parametric life cycle inventory (LCI) was established and used to identify the most influential parameters by means of a global sensitivity analysis (GSA). A simplified model for calculating GHG emissions associated with miscanthus cultivation was developed by fixing input parameters with a low relevance at their median impact values. Six of 38 parameters were identified as relevant parameters: soil carbon sequestration, harvestable yield, duration of cultivation period, quantities of nitrogen and potassium fertilizer applied, and distance between field and customer. The simplified model allows practitioners an easy assessment of the GHG emissions associated with the production and supply of miscanthus. It thus provides a wider audience facilitated access to LCA knowledge and promotes its use as a management and reporting tool in bio-based industries.

KEYWORDS

global warming potential, greenhouse gas emissions, LCA, miscanthus, simplified LCA, sustainability

1 | INTRODUCTION

Miscanthus is considered a promising crop for the supply of biomass, in particular lignocellulose, to help meet the growing global demand. The plant uses resources, water and nutrients, efficiently and can grow in a wide range of environments, including marginal sites (Clifton-Brown et al., 2017; Lewandowski et al., 2016; van der Weijde et al., 2013). In recent years, industry's interest in miscanthus has grown due to its possible usage in a range of applications. It has been shown that miscanthus biomass can be used for the production of insulation material and chemicals, and can also serve as a bioenergy feedstock (Lask, Martínez Guajardo, et al., 2020; Lask, Rukavina, et al., 2020; Moll et al., 2020; Wagner et al., 2017). Each of these application pathways has been considered in environmental sustainability assessments, which commonly rely on the life cycle assessment (LCA) methodology (Kiesel et al., 2017; Lask et al., 2019; Wagner & Lewandowski, 2017; Wagner et al., 2019).

LCA is a widely recognized tool for the assessment of potential environmental impacts associated with the life cycle of a product or service. This is achieved by inventorying exchanges between the system under study and the environment (ISO, 2006a, 2006b). For agricultural products such as miscanthus, the creation of an inventory can be labour-intensive and is context-specific. This is due, firstly, to biophysical conditions that shape the cultivation system (e.g. with respect to yield and soil carbon sequestration) and secondly to a wide range of crop management approaches that may or may not be taken (e.g. with respect to the fertilizer amount applied). Each change in a parameter may affect the overall LCA result.

The environmental impact category most widely quantified in LCAs is global warming potential, which is a measure of greenhouse gas (GHG) emissions. Several studies, usually focusing on the physical flows related to the production system (attributional LCA; Finnveden et al., 2009), have assessed the GWP impacts associated with miscanthus production. In attributional assessments, estimates range between 58 and 170 g CO_{2eq} (kg dry matter)$^{-1}$, neglecting credits from direct land use change (dLUC; soil carbon sequestration). The lower-end value stems from the ecoinvent dataset for miscanthus production in Germany (Nemecek, 2020). The higher-end values are derived from a Canadian case study, which reports figures for five distinct scenarios with GHG estimates between 100 and 170 g CO_{2eq} (kg dry matter)$^{-1}$ (Sanscartier et al., 2014). Within this range, 113 and 156 CO_{2eq} (kg dry matter)$^{-1}$ were reported as comparisons for miscanthus production in Germany and the United Kingdom (Lask et al., 2019). Although the variation in results is substantial, it can mainly be explained by differences in the assumed system characteristics. Soil carbon sequestration, biomass yield, duration of the cultivation period and nitrogen fertilization have previously been identified as parameters that strongly influence the life cycle GHG emissions of miscanthus cultivation (Sanscartier et al., 2014). The values of these parameters (and also others) can vary substantially on account of site- and management-specific diversity. This impairs the transfer of results to comparable but not necessarily similar situations and makes performing LCAs complex and resource-consuming (Finnveden et al., 2009; Zamagni et al., 2012). Farmers and small- and medium-sized enterprises cannot easily dedicate resources to this task (in particular when using marginal land) and often lack the expertise to do so, and this hinders the wider application of LCA as a management and reporting tool in bio-based industries.

Simplified LCA models could offer a promising solution to this problem. They are reduced versions of more complex LCA models that require only a few critical parameters to calculate representative results (Beemsterboer et al., 2020). They incorporate knowledge from LCA experts and allow non-experts easy and fast computation of LCA results. The present study develops a simplified model for the calculation of GHG emissions associated with commercial miscanthus cultivation in Europe. For this purpose, a parametric life cycle inventory (LCI) model is developed based on the experience of practitioners. It is used to identify those parameters that account for the major sources of variation in the GHG emissions and then to set up the simplified model.

2 | MATERIALS AND METHODS

2.1 | Goal and scope

This study aims to develop a simplified parametric model for the computation of GHG emissions related to commercial miscanthus cultivation. The model follows the LCA framework as standardized in ISO (2006a, 2006b) and therefore accounts for all impacts associated with crop production and subsequent transport of biomass by truck (Figure 1). It is intended to be applicable to situations in which miscanthus is grown on arable land (marginal or non-marginal) in Europe for commercial purposes. It focuses on rhizome-based propagation and the indirect harvesting method (cutting to swath,

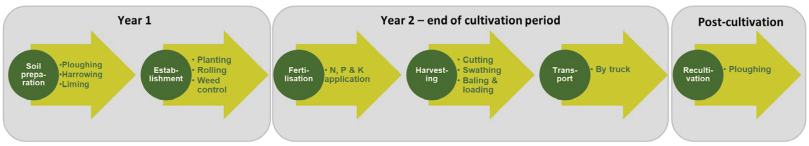

FIGURE 1 Product system, for which a simplified model for the computation of GHG emissions is developed

swathing, baling). The identification of critical parameters influencing the GHG emissions of miscanthus production is a prerequisite for the development of the simplified model and is thus considered a sub-goal of the study.

The calculations are based on a unit of 1 kg dry matter (DM; moisture content <15%) of miscanthus harvested in spring, baled and transported by truck. In line with the study's goal, the only impact category considered was climate change. Characterization factors were derived from the impact method IPCC, 2013 (100 years; IPCC, 2013). Data for the foreground processes were determined in close collaboration with practitioners experienced in commercial miscanthus cultivation (J. Kam, personal communication, October 2020). The few remaining gaps were filled using information from the literature. Background data were derived from ecoinvent 3.7 cut-off (Wernet et al., 2016).

2.2 | Basic calculations, global sensitivity analysis and simplified models

All calculations were performed using the LCA software Brightway2 (Mutel, 2017) and the library lca-algebraic (https://github.com/oie-mines-paristech/lca_algebraic). Lca-algebraic is a library that provides a layer above Brightway2 and implements a functional definition of models. It permits quick LCA computation, fast Monte Carlo simulations and advanced statistical analyses. This also includes global sensitivity analyses (GSAs), which are required to set up simple parametric models. In the present study, the following steps were conducted to define a simple model for computing life cycle GHG emissions of miscanthus production. First, a parametric LCI was defined. It contained information on the probability distributions of the parameters included. Second, the probabilistic distribution of the GHG emissions associated with miscanthus cultivation was calculated, using the assumed input variables. For this, a Monte Carlo simulation with 10,000 runs was conducted. Each run considered a characteristic set of values for the individual parameters. Third, a variance-based GSA was performed and Sobol indices derived for each of the parameters included. These indicate the parameters' influence on the total variation in the resulting GHG emissions. Finally, a simple parametric model was established using the information on the Sobol indices and by fixing input parameters with low relevance at their median impact values. The corresponding script is available from the main author's GitHub page (https://github.com/janlask/simplifiedmiscanthusLCA).

Each of the steps is described individually in the sections below. The development of the parametric inventory is explained in the sub-section 'Life Cycle Inventory and Parametrization'. The probabilistic distribution of the GHG emissions and Sobol indices is presented graphically in the results section followed by the simplified model developed.

2.3 | Life cycle inventory and parametrization

In line with the scope of the study, the inventory included all processes required for miscanthus production: land preparation (liming, ploughing and harrowing), establishment of the crop (rhizome planting incl. production of planting material, rolling and weed management), fertilization, harvest-related activities (cutting to swath, swathing, baling, bale loading and transport by truck) and recultivation (preparation of land for subsequent use). For each of the agricultural operations, standard ecoinvent datasets were taken, with the diesel consumption adapted according to information from practitioners. All required agricultural inputs (fertilizers, herbicides, etc.) and associated field emissions were accounted for. This includes emissions from lime application (CO_2), nitrogen fertilization (direct and indirect N_2O emissions) and decay of non-harvested biomass (leaves and stubble) (N_2O). As the simplified model is intended to be representative of average European cultivation conditions, market datasets were selected for background processes, where possible. This means that, for example, for fertilizers, a substrate mix representative of the European market for inorganic fertilizers was taken. The influence of this assumption was tested by specifying the fertilizer type (calcium ammonium nitrate and potassium chloride) in a sensitivity analysis (specified fertilizer model).

Present-day commercial miscanthus production relies on rhizome-based, vegetative propagation, which is costly and impedes faster uptake of the crop by industry and farmers. Seed-based plugs from new miscanthus hybrids could overcome these hurdles and are on the verge of market introduction (Clifton-Brown et al., 2017). For this reason, the default model using rhizome-based cultivation was complemented by a sensitivity analysis assessing plug-based miscanthus production. Inventory data for the rhizome production were taken from the ecoinvent database (Wernet et al., 2016), while inventory data for the seed plug production were compiled based on insights from commercial seed plug production (J. Kam, personal communication, October 2020; inventory provided in Table S2).

Following the initial creation of the inventory, influential parameters were collated for all process steps (see Figure S1) and incorporated into the model. In collaboration with experts from research and practice, each parameter was characterized using one of the following options: linear, triangular or normal distribution, or fixed values. The distributions were further defined by providing descriptive information (default/mean, min/max and standard deviation) representative of commercial-scale cultivation. The outcome is summarized in Table 1, where the parameters are classified in four groups: diesel consumption, biomass-related, management-related and transport

TABLE 1 Parameters – descriptions and distributions

Phase	Parameter	Description	Unit	Default	Min	Max	Std	Distribution	Main reference
Biomass	*yield harv*	Average annual harvestable DM yield after establishment	kg DM ha^{-1} $year^{-1}$	14,000	9500	23,500		Triangle[c]	McCalmont et al. (2017); Witzel and Finger (2016)
	mass rhizomes	Mass of rhizomes	kg $piece^{-1}$	0.0750	0.0500	0.100		Triangle[c]	[d]
	carbon seq	Carbon sequestered	kg C ha^{-1} $year^{-1}$	0	(0)	(2200)		Fixed[a]	McCalmont et al. (2017)
	dmshare_stubbles	Proportion of DM peak yield remaining on field after harvest	%	0.04	0.02	0.06		Triangle[c]	[d]
	dmshare_leaves	Proportion of DM peak yield lost as leaves		0.24	0. 20	0.29		Triangle[c]	
	RS	Ratio of below- and above-ground biomass		0.80	0.70	1.50		Triangle[c]	IPCC (2006)
	N_stubbles	Nitrogen content of stubbles	kg N $(kg\ DM)^{-1}$	0.0020	0.0015	0.0025		Triangle[c]	[d]
	N_leaves	Nitrogen content of leaves lost		0.0070	0.0060	0.0080		Triangle[c]	
	N_bgr	Nitrogen content of below-ground biomass		0.0100	0.0060	0.0140		Triangle[c]	
	mass rhizomes	Mass of rhizomes	kg $piece^{-1}$	0.0750	0.0600	0.1000		Triangle[c]	
Diesel consumption in agricultural operations	*diesel ploughing*	Diesel consumption for ploughing	kg ha^{-1}	16.62	12.61	20.97		Triangle[c]	[d]
	diesel harrowing	Diesel consumption for harrowing		14.78	10.75	19.76		Triangle[c]	
	diesel_rhizome planting	Diesel consumption for rhizome planting		13.57	11.13	16.56		Triangle[c]	
	diesel rolling	Diesel consumption for rolling		2.15	1.75	2.66		Triangle[c]	
	diesel appplantprot	Diesel consumption for application of plant protection agents		2.11	1.50	2.66		Triangle[c]	
	diesel fertilisation	Diesel consumption for broadcasting of fertilizer		0.20	0.15	0.25		Triangle[c]	
	diesel combine	Diesel consumption for combine harvester		9.03	7.56	10.5		Triangle[c]	
	diesel swathing	Diesel consumption for swathing		2.94	2.46	3.42		Triangle[c]	
	diesel baling	Diesel consumption for baling	kg $bale^{-1}$	0.84	0.60	1.10		Triangle[c]	
	diesel baleload	Diesel consumption for bale loading		0.42	0.35	0.49		Triangle[c]	
	diesel recultivation	Diesel consumption for ploughing	kg ha^{-1}	16.62	12.61	20.97		Triangle[c]	

(Continues)

TABLE 1 (Continued)

Phase	Parameter	Description	Unit	Default	Min	Max	Std	Distribution	Main reference
Management	*cult per*	Duration of cultivation period	years	20			3	Normal[b]	
	num rhizomes	Number of rhizomes for planting	pieces ha^{-1}	15000				Fixed[a]	
	bale m	Bale weight	kg bale^{-1}	615				Fixed[a]	
	freq Nfertilisation	Nitrogen fertilization per cultivation period		0	0	0.5		Triangle[c]	[d] and McCalmont et al. (2017)
	freq Kfertilisation	Potassium fertilization per cultivation period		1	0.75	1		Triangle[c]	
	freq Pfertilisation	Phosphorus fertilization per cultivation period		0.25	0.25	0.5		Triangle[c]	
	N fert	Total nitrogen fertilizer applied	kg N (cult. period)$^{-1}$	0	0	600		Triangle[c]	
	K fert	Total potassium fertilizer applied	kg K_2O (cult. period)$^{-1}$	2380	1700	3500		Triangle[c]	
	P fert	Total phosphorus fertilizer applied	kg P_2O_5 (cult. period)$^{-1}$	280	200	350		Triangle[c]	
	lime app	Liming	Yes (1) – No (0)	1				Fixed[a]	
	lime fert	Lime applied	kg (cult. period)$^{-1}$	2160	1500	2500		Triangle[c]	
	herb_app	Number of herbicide applications		3	1.5	5		Triangle[c]	[d]
	amount_glyphosate	Avg. amount of glyphosate applied per herbicide application	kg active substance (average application)$^{-1}$	0.3240	0.2916	0.3564		Triangle[c]	
	amount_pendimethalin	Avg. amount of pendimethalin applied per herbicide application		0.2610	0.2349	0.2871		Triangle[c]	
	amount_phenoxy	Avg. amount of phenoxy-compound applied per herbicide application		0.3300	0.2970	0.3630		Triangle[c]	
	amount_pesticide_unspec	Avg. amount of unspecified pesticide applied per herbicide application		0.0110	0.0099	0.0121		Triangle[c]	

(Continues)

TABLE 1 (Continued)

Phase	Parameter	Description	Unit	Default	Min	Max	Std	Distribution	Main reference
Transport distances	*distance rhizomesplugs*	Distance between rhizome/seed plug nursery and farm	km	800	20	2000		Triangle[c]	[d]
	distance rhizomesplugsfield	Distance between farm and field		2	0.5	10		Triangle[c]	
	distance cust	Distance between field and customer or collection point		60			18	Normal[b]	
	distance generic input	Distance for input materials, if not further specified		150				Fixed[a]	Authors' estimate
Emission factors	*EF_1*	Direct N_2O from application of nitrogen fertilizer		0.0100				Fixed[a]	IPCC (2019)
	Frac_GASF	Fraction of N lost through volatilization (NH_3 and NO_x)		0.1100				Fixed[a]	
	EF_4	Indirect N_2O from N volatilized and redeposited		0.0100				Fixed[a]	
	Frac_LEACH	Fraction of N lost through leaching (NO_3)		0.2400				Fixed[a]	
	EF_5	Indirect N_2O from nitrate leached		0.0110				Fixed[a]	
	EF_limestone	CO_2 from liming		0.1200				Fixed[a]	

[a]Fixed: default value taken.
[b]Normal: default is taken as mean.
[c]Triangle: default represents highest probability.
[d]Expert estimate.

distances. Some of these parameters are selected for more detailed explanation below, on account of their sheer number (diesel consumption) or relevance for the study's results (see Section 4).

Parameters for diesel consumption in the agricultural operations are provided in Table 1 and were defined considering mean values and ranges provided by practitioners (J. Kam, personal communication, October 2020). These parameters were used to adjust standard ecoinvent datasets of agricultural operations in terms of the diesel input and associated emissions.

Biomass-related parameters include, among others, the estimated miscanthus dry matter yield (fully established crop) and potential carbon sequestration. Given the possible cultivation on marginal as well as on non-marginal land, a wide yield range (min = 9500 kg DM ha^{-1}, max = 23,000 kg DM ha^{-1}) with a mean of 14,000 kg DM ha^{-1} was considered. The mean value was calculated as the average of published data on yields in Europe (McCalmont et al., 2017; Witzel & Finger, 2016), while minimum and maximum values were based on expert estimates. A triangular distribution was selected, as mean yields are more likely and yields below the minimum value do not seem viable from a commercial perspective. It was assumed that the plantations reach 50% of their full yield potential in the second year and 100% from the third year onwards. During its cultivation, miscanthus can substantially increase the amount of carbon in the soil through the turnover of leaves, harvest residues and below-ground biomass (rhizomes and roots). In addition, the absence of tillage operations during the cultivation period can retard the mineralization of carbon (Chimento et al., 2016; McCalmont et al., 2017). The extent of these effects and their accounting in LCA greatly depends on site-specific characteristics as well as on methodological choices of LCA practitioners (Goglio et al., 2015). As seen in preliminary analyses performed in the context of the present study, the parameter carbon sequestration can easily dominate the variation in GHG emissions associated with miscanthus cultivation. This is due to the broad range of values that could be taken, ranging from 0 to 2.2 t C ha^{-1} $year^{-1}$ (McCalmont et al., 2017). The inclusion of this parameter impeded further assessments. For this reason, carbon sequestration was not included as a variable in the GSA. However, as it is a critical parameter, it was subsequently incorporated into an extended simplified model (see Section 4.3).

Management-related parameters include those variables that can be actively influenced by farmers, for example, the application of fertilizers and duration of the cultivation period. LCA studies commonly assume cultivation periods of 20 years for miscanthus. The effect of extended cultivation periods is rarely assessed, although the maximum lifetime of miscanthus plantations can be up to 25 years (Lewandowski et al., 2003). In practice, however, farmers occasionally decide to terminate the miscanthus cultivation earlier. For these reasons, we described the duration of the cultivation period (*cult_per*) using a normal distribution, given a mean of 20 years and a standard deviation of 3 years. Two parameters were defined for the application of each fertilizer. This includes one parameter detailing the application frequency (*freq_Nfertilisation*, *freq_Pfertilisation* and *freq_Kfertilisation*) and one for the total fertilizer amount applied per hectare during the cultivation period (*N_fert*, *P_fert* and *K_fert*). Academic cultivation trials commonly consider moderate nitrogen fertilization (e.g. 60 kg N ha^{-1} $year^{-1}$, see for instance Kiesel et al., 2017). This is despite the fact that contradicting effects of nitrogen application on miscanthus biomass yield have been reported and in commercial practice, no nitrogen is commonly applied (J. Kam, personal communication, October 2020 and McCalmont et al., 2017). In line with the study's objective of representing commercial miscanthus cultivation, a triangular distribution with a maximum probability at 0 kg nitrogen per hectare was defined. By default, lime application was assumed. The quantities of fertilizers and lime applied were estimated using first-hand experience from commercial miscanthus cultivation (J. Kam, personal communication, October 2020).

The group *transport distances* comprises parameters describing all truck transport distances involved. Analyses performed in preparation of the current study indicated transport distance of the harvested biomass as the only parameter that substantially contributes to the variation of results in the model. The parameter (*dist_cust*) was defined according to commercially relevant information (J. Kam, personal communication, October 2020).

Parameters describing emission factors were fixed at IPCC default values, as is standard LCA practice (IPCC, 2019). For the specified fertilizer model, more specific emission factors were taken using calcium ammonium nitrate as nitrogen source.

3 | RESULTS

3.1 | Probabilistic distribution of the life cycle GHG emissions and Sobol indices

The probabilistic distribution of the GHG emissions associated with the rhizome-based production of 1 kg miscanthus DM is presented in Figure 2. For each of the 10,000 runs, the parameter values varied within the range indicated in Table 1 (*carbon_seq* fixed to 0). The distribution is characterized by a mean (μ) of 87.1 and a median of 85.0 g CO_{2eq} (kg DM)$^{-1}$, and a 5th and 95th percentile of 66.2 and 115.0 g CO_{2eq} (kg DM)$^{-1}$ respectively.

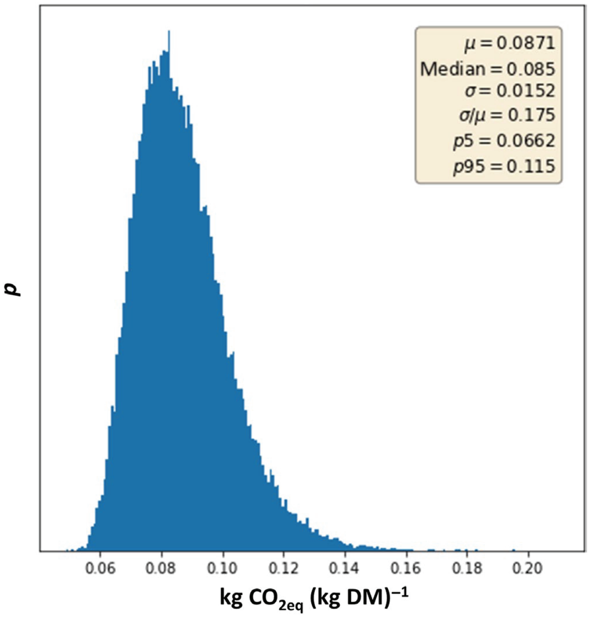

FIGURE 2 Probabilistic distribution of greenhouse gas emissions for parameter distributions as given in Table 1 [kg CO_{2eq} (kg DM)$^{-1}$]

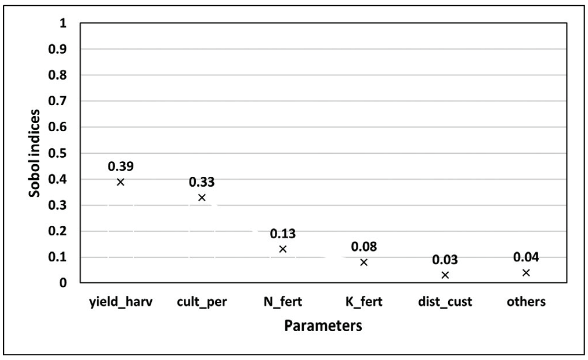

FIGURE 3 Sobol indices; yield_harv: biomass yield when established [kg DM ha^{-1}], cult_per: duration of cultivation period [years], N_fert: amount of N applied [kg N cult. per^{-1}], K_fert: amount of K_2O applied [kg K_2O cult. per^{-1}], dist_cust: biomass transport distance by truck [km]

As mentioned above, Sobol indices indicate how strongly an individual parameter influences the variation of the overall result. Indices were calculated for all (non-fixed) parameters. Figure 3 presents the contribution to variation of each of these parameter. The most important ones (in descending order) are *yield_harv*, *cult_per*, *N_fert*, *K_fert* and *dist_cust*, which together account for more than 96% of the total variation. From these results, it becomes clear that the biomass yield is the most influential parameter in the model, followed by the duration of the cultivation period and the amount of nitrogen and potassium fertilizer applied. In addition, the distance between field and final customer or storage site should not be neglected. As described in Section 2.3, it was observed in preliminary analyses that the parameter *carbon_seq* dominated the overall variability due to its substantial variation (see also Figure S2). To enable the analysis of other parameters, it was excluded from the GSA.

3.2 | Simplified model

Based on the results of the GSAs and the Sobol indices, a simplified model was created by fixing less influential parameters at their median impact value. The resulting equation is given below:

$$\text{GHG emissions}_{\text{miscanthus}}\left[\frac{\text{kg CO}_{2\text{eq}}}{\text{kg DM}}\right] = \frac{3.09\text{K}_{\text{fert}} + 10.4\text{N}_{\text{fert}} + 83.3\text{cult}_{\text{per}} + 2310}{\text{yield}_{\text{harv}}\left(\text{cult}_{\text{per}} - 1.5\right)} + \frac{0.0713}{\left(\text{cult}_{\text{per}} - 1.5\right)} + 0.000162\text{dist}_{\text{cust}} + 0.0224, \qquad (1)$$

where Kf_{ert} is the total potassium fertilizer applied [kg K_2O (cult. period)$^{-1}$]; N_{fert} is the total nitrogen fertilizer applied [kg N (cult. period)$^{-1}$]; $cult_{per}$ is the duration of cultivation period [years]; $yield_{harv}$ is the harvestable DM yield when established [kg DM ha^{-1}]; $dist_{cust}$ is the distance between field and customer or collection point [km].

3.3 | Extending the simplified model to account for carbon sequestration

In the previous calculations, the parameter *carbon_seq* was fixed and thus not considered in the GSA due to its substantial impact on the results. Nevertheless, carbon accumulation through miscanthus cultivation is an important and relevant parameter. For this reason, a term was added to the simplified model, which enables the integration of the carbon amount sequestered over the cultivation period ($carbon_{seq} * \frac{44}{12} * cult_{per}$). This results in an adjusted simplified model:

$$\text{GHG emissions}_{\text{miscanthus}}\left[\frac{\text{kg CO}_{2\text{eq}}}{\text{kg DM}}\right] = \frac{3.09\text{K}_{\text{fert}} + 10.4\text{N}_{\text{fert}} + 83.3\text{cult}_{\text{per}} - \left(\text{carbon}_{\text{seq}} * \frac{44}{12} * \text{cult}_{\text{per}}\right) + 2310}{\text{yield}_{\text{harv}}\left(\text{cult}_{\text{per}} - 1.5\right)} + \frac{0.0713}{\left(\text{cult}_{\text{per}} - 1.5\right)} + 0.000162\text{dist}_{\text{cust}} + 0.0224, \qquad (2)$$

where Kf_{ert} is the total potassium fertilizer applied [kg K_2O (cult. period)$^{-1}$]; N_{fert} is the total nitrogen fertilizer applied [kg N (cult. period)$^{-1}$]; $cult_{per}$ is the duration of cultivation period [years]; $yield_{harv}$ is the harvestable DM yield when established [kg DM ha^{-1}]; $dist_{cust}$ is the distance between field and customer or collection point [km]; $carbon_{seq}$ is the carbon sequestered due to miscanthus cultivation [kg C ha^{-1} $year^{-1}$].

4 | DISCUSSION

In this study, a simplified model for calculating the GHG emissions associated with miscanthus cultivation and supply was developed. It facilitates quick assessments by LCA practitioners and miscanthus farmers. As a prerequisite, those inventory parameters were identified that explain the major share of variation in the results of GHG emission calculations.

The most important is the *amount of carbon sequestered* during the cultivation period (*carbon_seq*). This can vary widely and greatly depends not only on site-specific conditions (McCalmont et al., 2017) but also on the methodological approach selected for carbon accounting (Goglio et al., 2015) and the (assumed) permanence of the carbon sequestered (Lask, Martínez Guajardo, et al., 2020; Lask, Rukavina, et al., 2020). Due to the high variability in the possible values that this parameter can take, it is left to the user which value to select. Parameter values could be selected from the range provided in Table 1 (based on McCalmont et al., 2017) or using allometric models, as suggested in Ledo et al. (2018). In any case, the permanence of the carbon storage has to be critically reflected upon. For precautious users of the model, it is recommended to consider only a temporary carbon storage during the cultivation period. This can be done using the approach suggested in the ILCD handbook (ILCD, 2010) and avoids overestimation of the benefits from carbon storage.

If carbon sequestration is not taken into account, five remaining parameters explain more than 96% of the variation in the results. The most influential of this group is *dry matter yield*, which is in line with previous publications (Meyer et al., 2017; Sanscartier et al., 2014). Actual yields vary substantially due to genotypic as well as climatic and soil variations. This applies in particular to cultivation on marginal land which is strongly advocated for miscanthus. If yield decreases are expected to occur over the cultivation period, the parameter *yield_harv* has to be set in a way that reflects the average yield over the cultivation period after the establishment period.

The second most influential parameter was identified as the *duration of the cultivation period*. The relevance of this parameter in LCA studies on miscanthus cultivation and other perennial crops has been emphasized before (Hastings et al., 2017; Hastings, Clifton-Brown, Wattenbach, Mitchel, Stampfl, et al., 2009; Hastings Clifton-Brown, Wattenbach, Mitchell, & Smith, 2009; Ledo et al., 2020; McCalmont et al., 2017). With an increase in duration of the cultivation period, the emissions associated with the provision of planting material and establishment can be distributed over a higher biomass output. Consequently, the GHG intensity per kilogram miscanthus dry matter decreases. This aspect is relevant for miscanthus crop management in practice. While the annual decline is substantial during the first 11 years, the effect diminishes in the following years and levels out from year 15 onwards (Figure S3). Thus, we conclude that miscanthus, once established, should be cultivated for at least 10, ideally 15 years. LCA studies on miscanthus-based value chains usually use the default of 20 years. Given previous and present results, we highlight the importance of reflecting on the sensitivity of miscanthus LCA results in regard of this parameter.

Unsurprisingly, the amounts of *potassium and mainly nitrogen fertilizer applied* influence the GHG emissions associated with miscanthus cultivation substantially. Nitrogen fertilization, in particular, has previously been identified as an important parameter (Sanscartier et al., 2014). However, the present study also reveals a substantial influence of the potassium fertilizer applied. This is mainly due to the market datasets representing the average European market fertilizer mix that were selected to give the simplified model wider validity. NPK fertilizer accounts for a high share of the market mix for inorganic potassium fertilizer (Symeonidis, 2020), which results in substantially higher impacts than when using specific fertilizers. Taking potassium chloride, a common potassium fertilizer in the European context, results in a two-third reduction of GHG emissions per kilogram K_2O supplied. A shift in fertilizer providers from market processes to, for example, calcium ammonium nitrate and potassium chloride, would affect the model outcome accordingly, reducing the overall impacts (see Figure S4).

In the simplified model, the four parameters mentioned above were complemented by one describing the *distance between field and customer*. This is particularly relevant for marginal sites, as these can also be marginal in an economic sense due to their remote location and/or poor accessibility. Together, these five identified parameters, complemented with the one on carbon sequestration, explain the major share of the variation in the results for GHG emissions associated with miscanthus cultivation.

Other parameters were less relevant and not included as variables in the simplified model. The impact associated with these was included using the median GHG emissions derived from the range of the input variables given in Table 1. This applies, for instance, to the diesel consumptions of the agricultural procedures. For the ranges assumed, these parameters did not have a substantial influence on the variation in impact results for the system as a whole.

On the whole, the suggested simplified model can be used to estimate the GHG emissions associated with miscanthus cultivation. For a comparison with previously published assessments, the simplified model was run with the parameters given in two publications. Table 2 shows the results as given in the publication along with the ones derived using the simplified model. The comparison reveals that the

TABLE 2 Comparison of impact results from previously published assessment with results derived using the simplified model (excluding carbon sequestration)

Reference	Study	Sanscartier et al. (2014)					Nemecek (2020)	Lask et al. (2019)	
	Scenario	A	B	C	D	E	—	GER	UK
Parameters	Life span [years]	20	20	15	15	15	20	20	20
	Yield [kg DM ha^{-1}]	11100	10000	10000	8400	9500	17000	15316	9745
	N [kg N (cult. per)$^{-1}$]	1200	1600	1200	900	1200	850	1200	1200
	P [kg P_2O_5 (cult. per)$^{-1}$]	240	220	165	135	150	850	600	600
	K [kg K_2O (cult. per)$^{-1}$]	2100	1900	1425	1185	1350	2023	2400	2400
	Transport distance [km]	95	95	95	185	95	2	48	83
g CO_{2eq} (kg DM)$^{-1}$ acc. to …	Reference (without dLUC)	129.20	149.33	167.79	167.79	182.89	58.00	109.49	153.99
	Simplified model	153.38	184.82	194.50	203.87	200.66	87.20	118.28	172.11
	Simplified model (spec. fert.)	130.31	160.99	169.65	179.15	175.71	72.76	99.34	142.51

suggested simplified model provides a satisfactory estimation of the GHG emissions associated with miscanthus cultivation (Table 2). However, it seems that the generic model results in a conservative estimation when compared with the results from the reference studies. Comparing the results with the results of the ecoinvent dataset (Nemecek, 2020), a more substantial deviation can be observed. The simplified model results in higher impacts, which can be attributed to substantially lower N2O flows in the ecoinvent dataset. This probably results from an omission of nitrogen losses via the degradation of leaves before harvest. For the other comparisons given in Table 2, the simplified model can deliver meaningful estimates for the assessment of impact variations through yield fluctuations and differences in management practices.

It should be emphasized that the simplified model can only be considered a representation of the current commercial miscanthus cultivation and changes in management need to be continuously monitored and incorporated into the model. An example of a change already taking place that could require adaptions to the model is related to the planting material. Seed-based plugs are at the threshold of achieving market readiness and their provision clearly differs from rhizome propagation. Possible discrepancies between GHG emissions arising from miscanthus establishment based on rhizomes and seed-based plugs were analysed in a sensitivity analysis. The screening indicated that impacts are almost twice as high for plugs as for rhizomes mainly due to the heat and light required to raise the plugs in the greenhouse. However, when taken over the entire cultivation period and total DM yield, differences between the two cultivation systems are relatively small, as the production of the planting material has only a minor impact (see Figures S4 and S5).

The major outcome of this study is the establishment of a simplified parametric model—an effort, which so far is only rarely undertaken in LCA practice and research. It allows practitioners simple calculation of GHG emissions associated with the production and supply of miscanthus, including marginal sites. Using this model would enable LCA know-how to be more easily shared among a wider audience. A simplified model with only a few variables would give non-LCA experts a better idea of central parameters to be used as leverage points in the optimization of their systems. In addition, simplified parametric models hold substantial opportunities for LCA research as they enable automated and faster computation of potential environmental impacts.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available in the supplementary material of this article and at the online repository: https://github.com/janlask/simplifiedmiscanthusLCA.

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SUPPORTING INFORMATION

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

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Chapter 3

3.0 Comparative environmental and economic life cycle assessment of biogas production from perennial wild plant mixtures and maize (*Zea mays* L.) in southwest Germany

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Comparative environmental and economic life cycle assessment of biogas production from perennial wild plant mixtures and maize (*Zea mays* L.) in southwest Germany

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Abstract

Maize silage is the main biogas co-substrate in Germany, but its use is often questioned due to negative environmental impacts. Perennial wild plant mixtures (WPM) are increasingly considered alternatives, as these extensive systems improve soil quality and enhance agrobiodiversity. Methane yields per hectare however do not match those of maize. This study examined whether the potential advantages of replacing maize with WPM for biogas production are counteracted by lower yields and associated effects. Life cycle assessment and life cycle cost assessment were used to compare the environmental and economic performance of electricity generation from WPM in two establishment procedures, 'standard' (WPM E1) and 'under maize' (WPM E2). These metrics were benchmarked against those of maize. The production of 1 kWh electricity was chosen as functional unit. The life cycle inventory of the agricultural phase was based on multi-annual field trials in southwest Germany. Both WPM E1 and E2 had lower marine eutrophication and global warming potentials than maize. The GWP favourability was however sensitive to the assumptions made with regard to the amount and fate of carbon sequestered in the soil. WPM E1 performed less favourable than WPM E2. This was mainly due to lower yields, which could, in turn, result in potential indirect land use impacts. These impacts may outweigh the carbon sequestration benefits of WPM cultivation. Maize performed best in terms of economic costs, freshwater eutrophication, terrestrial acidification, fine particulate matter and ozone formation. We conclude that the widespread deployment of WPM systems on productive agricultural land should only take place if permanent soil carbon sequestration can be ensured. In either case, WPM cultivation could be a valid alternative for bioenergy buffers and marginal land where competitive yields of common crops cannot be guaranteed, but which could accommodate low-input cultivation systems.

KEYWORDS

agrobiodiversity, alternative substrates, bioenergy, biogas, environmental performance, LCA, LCC, maize, perennial cropping system, wild plant mixtures

1 | INTRODUCTION

Electricity production based on biogas can contribute to greenhouse gas mitigation (Kiesel, Wagner, & Lewandowski, 2017; Scarlat, Dallemand, & Fahl, 2018; Wagner et al., 2019). Germany has seen a large deployment of biogas production during the last two decades and has become the world's largest producer with an installed electrical capacity of 4.8 GW in 2018. Energy crops, of which 69% are maize (*Zea mays* L.) silage, accounted for more than half of the substrates in 2016 (FNR, 2019). The wide cultivation of maize in Germany is often questioned due to potential negative impacts on the environment, including the risk of soil erosion and compaction, nitrate leaching and high pesticide use (Herrmann, 2013; Kiesel et al., 2017).

For this reason, an extensification of biogas production is increasingly being investigated and alternative biogas substrates sought (Herrmann, Idler, & Heiermann, 2016; von Cossel, Möhring, Kiesel, & Lewandowski, 2018). Perennial crops such as miscanthus and *Silphium perfoliatum* are considered particularly promising alternative biogas substrates (Kiesel & Lewandowski, 2017; Mayer et al., 2014; von Cossel, Wagner, et al., 2019). Wild plant mixtures (WPM) have also been suggested as a potential future biogas substrate production system (BSPS) and have recently become the subject of research interest (Carlsson, Mårtensson, Prade, Svensson, & Jensen, 2017; Vollrath et al., 2012; Vollrath, Werner, Degenbeck, & Marzini, 2016; von Cossel & Lewandowski, 2016; von Cossel, Steberl, et al., 2019). WPM are perennial polycultures, consisting of wild, flower-rich plant species of annual, biennial and perennial nature. The dynamic cultivation systems are characterized by changing compositions over the cultivation period: Annual plant species are predominant in the first cultivation year, biennial species in the second and perennial plants in the following years. The perennial nature provides almost constant soil coverage, preventing soil erosion and contributing to soil carbon sequestration and improved soil quality (Emmerling, 2014; Emmerling, Schmidt, Ruf, Francken-Welz, & Thielen, 2017). The seed mixtures include legume species such as lucerne (*Medicago sativa* L.), white and yellow melilot (*Melilotus* spp.), to ensure atmospheric nitrogen (N) fixation, thus reducing the amount of fertilizer needed (von Cossel & Lewandowski, 2016). The main reason for encouraging WPM cultivation as a BSPS is its potential contribution to the protection and preservation of agrobiodiversity. Their perennial nature can provide food and habitats for wild/field animal species. Insects, in particular pollinators, benefit from the species mix of the plant stands (von Cossel & Lewandowski, 2016). These systems could thus contribute to an extensification of the present maize-dominated biogas substrate production, while enhancing a range of ecosystem functions on a local scale (Carlsson et al., 2017; von Cossel, Steberl, et al., 2019).

In addition to environmental advantages, it has occasionally been claimed that WPM cultivation also has economic benefits. For example, it has been suggested that perennial cropping systems can provide biomass more economically than annual systems, as less management is required (Lewandowski, 2016; Lewandowski et al., 2016). However, WPM tend to have lower yields than comparable maize systems, due to both lower biomass and specific methane yields (Friedrichs, 2013; Vollrath et al., 2016; von Cossel & Lewandowski, 2016; von Cossel, Steberl, et al., 2019). For this reason, approaches that combine the cultivation of maize and WPM are currently being investigated. The idea is to establish biennial and perennial WPM species under maize, in order to make use of the high maize yield in the first year and combine this with the beneficial characteristics of WPM in the following years. It has been demonstrated that this combination does not significantly reduce maize performance and can improve the long-term performance of WPM cultivation for biogas production (von Cossel, Steberl, et al., 2019). Despite these efforts, hectare yields do not match the output levels of conventional biogas crops over the cultivation period.

A more holistic assessment of WPM systems (WPM pure and WPM established under maize) for the extensification of biogas substrate production require a comparison with the conventional maize system. This study examines whether the advantages of an extensified biogas cropping system mentioned above are counteracted by lower yields, higher land requirements and the associated direct (e.g. soil carbon sequestration) and also potentially indirect effects. This question is relevant for both the environmental and economic dimension and has so far not been investigated. The study aims to quantify the environmental impacts and economic costs of electricity generated in a biogas plant based on three different feedstocks: maize only, WPM only and a combination of both. For this purpose, a comparative life cycle assessment (LCA) and a life cycle costing (LCC) are conducted. The results of the LCA are used for a comparison with a fossil reference to assess the greenhouse gas mitigation potential of the biogas systems analysed. This is an important motivation factor for investments in biogas technologies. If extensified BSPS, such as WPM, are intended for large-scale deployment, their contribution to achieving this goal needs to be ensured.

The first section introduces the methodological approach, including a description of the system to be analysed as well as fundamental assumptions. The second section presents the results of the sustainability assessment. In the final section, conclusions are drawn from these results, taking potential limitations and trade-offs into consideration.

2 | MATERIALS AND METHODS

2.1 | Goal and scope

This study conducted a comprehensive comparative environmental and economic assessment of the production of electricity from biogas. The BSPS analysed were mono-cropped maize (Maize) as reference crop, and two alternative systems with perennial features. Both alternatives were based on WPM but differed in the establishment approach: a standard WPM establishment procedure (WPM E1) and the establishment of WPM under maize, which served as a nurse crop (WPM E2; as described in von Cossel, Steberl, et al., 2019). All life cycle stages, namely the agricultural production of the systems mentioned, anaerobic digestion, and heat and electricity generation, were considered in the assessment. The alternative systems were compared based on a functional unit of 1 kWh electricity produced on a farm in southwest Germany, ready to be fed into the national grid. Marginal processes were used for modelling in line with Weidema, Pizzol, Schmidt, and Thoma (2018), who argue that this approach is fundamental for comparative assessments. As the arable land in southwest Germany is limited, an increase in biogas substrate production is likely to replace existing production systems. Since the global demand for agricultural products is still increasing, the replaced production is likely to be relocated elsewhere. Such implications need to be acknowledged in sustainability assessments of bioenergy cropping systems (Agostini, Giuntoli, Marelli, & Amaducci, 2019). The potential impacts of these considerations were investigated in a sensitivity analysis.

2.2 | Methods

This study is in accordance with the International Organisation for Standardization (ISO) framework (ISO 14040; ISO 2006). Data for the life cycle inventory of the BSPS (agricultural procedures and inputs, biomass yields and properties) were taken from a field trial performed at the University of Hohenheim between 2014 and 2018. Although the use of primary data limits the global significance of the results, it enables a more reliable comparison of the two biogas feedstocks under the given conditions. Biogas batch tests were performed to determine the potential methane yields, as described in von Cossel, Steberl, et al. (2019). For the anaerobic digestion and the heat and power generation, literature data were used. Background data and data on the fossil reference were taken from the ecoinvent database v3.5 consequential (Wernet et al., 2016). Modelling and impact calculations were performed using openLCA 1.8.0. The impact assessment methodology ReCiPe 2016 v1.1 was chosen due to its European focus and its up-to-date nature (Huijbregts et al., 2017). The following impact categories were selected because of their relevance in agricultural systems and biogas production: global warming potential (GWP), freshwater eutrophication (FE), marine eutrophication (ME), terrestrial acidification (TA), fine particulate matter formation (PM) and ozone formation (OF). The economic assessment followed the Code of Practice for environmental Life-Cycle Costing (Swarr et al., 2011) and was primarily based on data from the KTBL database (KTBL, 2019). The economic assessment included all costs that occur over the cultivation period of the substrates and takes a discount rate of 6% into account. For the biogas plant, all investment, operation and capital costs were considered, assuming a lifetime of 20 years.

2.3 | System boundaries

Figure 1 shows the schematic representation of the assessed system. The temporal boundaries ranged from the establishment of the biogas substrates, through final harvest and clearing of the fields to a state where a new crop could be planted. A total cultivation period of 5 years was investigated due to data availability. The maximum cultivation period of WPM can exceed 5 years (von Cossel, Steberl, et al., 2019). All agricultural inputs, such as herbicides and seeds, and management procedures were included. It was assumed that the annually harvested biomass was transported by tractor to a nearby biogas plant, where it is ensiled and fermented in an anaerobic digester. The biogas produced is combusted in a combined heat and power plant to generate electricity. The fermentation residues, which contain considerable amounts of nutrients, are returned to the field and used as fertilizer. The heat produced during the biogas combustion is used to meet the heat requirements of the biogas plant.

2.4 | Life cycle inventory

2.4.1 | Agricultural systems

Data for the substrate cultivation were taken from a field trial conducted at the University of Hohenheim, southwest Germany. In this field trial, different WPM establishment procedures were tested for their long-term methane yields over a 5-year cultivation period (2014–2018; von Cossel, Steberl, et al., 2019). Although further biogas cropping systems were assessed in this trial, the present study focused on a single WPM (BG90; Saaten-Zeller, 2016), as it performed best in terms of dry matter (DM) and methane yield. The standard establishment procedure (WPM E1) and the simultaneous sowing of maize and WPM (WPM E2) were evaluated. In addition, mono-cropped maize was included as a reference.

For all three BSPS, soil preparation included ploughing and harrowing, followed by sowing. In 'Maize' and 'WPM E2', the maize was sown at a density of 90,000 seeds per

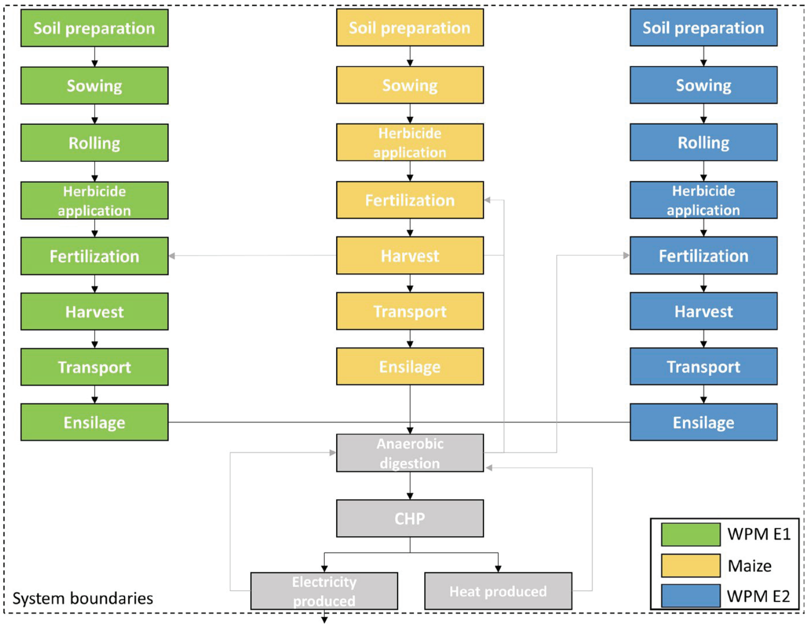

FIGURE 1 Process tree and system boundaries of the comparison of three biogas substrate production systems: monocropped maize (Maize), standard wild plant mixture (WPM) establishment (WPM E1), and WPM establishment under maize as nurse crop (WPM E2; as described in von Cossel & Lewandowski, 2016; von Cossel, Steberl, et al., 2019)

TABLE 1 Summary of agricultural inputs. Yearly average over 5-year cultivation period

	Maize	WPM E1	WPM E2	Unit
Seeds[a]				
Maize	90	—	18	1,000 seeds × ha^{-1} × a^{-1}
WPM	—	2	2	kg × ha^{-1} × a^{-1}
Stomp Aqua[b]	2.00	0.40	0.40	l × ha^{-1} × a^{-1}
Spektrum[b]	1.00	0.20	0.20	l × ha^{-1} × a^{-1}
MaisTer power[b]	1.00	0.20	0.20	l × ha^{-1} × a^{-1}
Laudis[b]	1.70	0.34	0.34	l × ha^{-1} × a^{-1}
Buctril[b]	0.35	0.07	0.07	l × ha^{-1} × a^{-1}
Nitrogen[a]	90.0	77.0	77.0	kg N × ha^{-1} × a^{-1}
Phosphorus[a]	17.6	17.6	17.6	kg P_2O_5 × ha^{-1} × a^{-1}
Potassium[a]	35.2	35.2	35.2	kg K_2O × ha^{-1} × a^{-1}

[a]von Cossel, Steberl, et al. (2019).
[b]Kiesel et al. (2017).

hectare. For 'WPM E2', 10 kg WPM seeds were sown immediately afterwards. In 'WPM E1', 10 kg seeds of WPM were sown alone (Table 1). The WPM systems included the additional step of rolling to ensure sufficient soil contact of the small seeds. In the field trial, weed management was conducted by hand. As this is unrealistic in commercial plantations, this study assumed the application of pesticides by field sprayers. The respective pesticides amounts were taken from a comparable field trial also performed at the University of Hohenheim (Kiesel et al., 2017; Table 1). In line with this study, two plant protection procedures were considered during the establishment period. The application of plant protection agents was assumed to take place in the first year only in WPM, as perennial plants effectively suppress weeds once established. In maize, an annual application was assumed.

The harvest of the biogas substrates was assumed to be conducted by maize choppers and self-loading trailers, as is typical for silage production (Wernet et al., 2016). In the establishment year 2014, all biogas substrates were harvested in October. From the second year onwards, harvest in the WPM systems took place in August, while the harvest date for maize remained in October. In the maize BSPS, harvest was followed by stubble ploughing each year. For WPM, it was assumed that stubble ploughing took place after the last harvest in 2018 only to prepare the field for further cultivation. The agricultural procedures for the silage production are summarized in Table 2.

For this study, the average annual yields of the field trial cultivation period (2014–2018) were taken. These are given in Table 3, together with average methane yield and content of the biogas substrates.

TABLE 2 Summary of agricultural procedures. Yearly average over 5-year cultivation period

	Maize	WPM E1	WPM E2	Unit
Ploughing	0.2	0.2	0.2	procedures × ha^{-1} × a^{-1}
Stubble ploughing	0.8	0.2	0.2	procedures × ha^{-1} × a^{-1}
Rotary harrowing	2.0	0.4	0.4	procedures × ha^{-1} × a^{-1}
Sowing	1.0	0.2	0.4	procedures × ha^{-1} × a^{-1}
Rolling	0.0	0.2	0.2	procedures × ha^{-1} × a^{-1}
Herbicide spraying	2.0	0.4	0.4	procedures × ha^{-1} × a^{-1}
Fertilizing	1.0	1.0	1.0	procedures × ha^{-1} × a^{-1}
Harvesting	1.0	1.0	1.0	procedures × ha^{-1} × a^{-1}
Ensiling	1.0	1.0	1.0	procedures × ha^{-1} × a^{-1}

TABLE 3 Average yield data according to von Cossel, Steberl, et al. (2019)

	Maize	WPM E1	WPM E2	Unit
DM yield	20.10	11.28	15.32	t × ha^{-1} × a^{-1}
FM yield	56.53	29.10	34.71	t × ha^{-1} × a^{-1}
CH_4 yield	6,376.38	2,694.63	3,800.60	m^3 × ha^{-1} × a^{-1}
CH_4 content	52.83	53.55	53.14	%

It was assumed that the biomass from all cultivation systems is transported to a nearby (distance 10 km) farm by tractor. There, the biomass is first ensiled and then fed into the digester by means of a wheel loader (Wernet et al., 2016). These steps are assumed to be accompanied by a DM loss of 5%.

Fertilization is vital for biomass production and was conducted in the field trial by means of mineral fertilizers, and nutrient quantities applied are given in Table 1. As in practice, nutrients are usually supplied through fermentation residues, this procedure was assumed for the assessment. 100% of the potassium and phosphorus in the digestate were considered plant available, while it was only 60% for nitrogen. The difference between the plant available nutrients in the digestate (after spreading) and the amounts applied in the field trial was assumed to be covered by mineral fertilizers. The environmental impacts associated with the application of fermentation residues and mineral fertilizers were estimated using the following models. Nitrate emissions were calculated by the SALCA-NO3 model taking into consideration the monthly balance of N mineralization, N uptake by the crops, the risk of nitrate leaching from fertilizer application and the intensity of soil tillage (Richner et al., 2014). Ammonia emissions from the spreading of fermentation residues were calculated assuming a volatilization of 20% of the plant available nitrogen in the digestate (IPCC, 2006). Direct N_2O and NO emissions were based on IPCC (2006) and include indirect N_2O from harvest residues (IPCC, 2006). Phosphorus and phosphate emissions were calculated according to Nemecek and Kägi (2007). It was assumed that the pesticides applied are released to agricultural soil (Wernet et al., 2016).

The cultivation of perennial plants on arable land can constitute a change of land use. Potential effects, such as changes in soil organic carbon content (SOC), need to be considered in bioenergy assessments (Agostini et al., 2019). Due to the unknown fate of the carbon stored in the soil, it is recommended that CO_2 removal through a change in SOC is shown separately in the results graphs (ISO 2018). The estimates of SOC changes used in this study were based on the Roth C model (Coleman et al., 1997). An initial field soil carbon content of 59 t carbon (C)/ha was assumed, which is representative of the regional conditions (Jacobs et al., 2018). The amounts of C added to the soil during the cultivation period (crop residues, root biomass) were estimated using data on carbon allocation in plants from a recent report by Jacobs et al. (2018). The considered carbon allocation factors are reported in Data S1. It was assumed that the biomass left on the field at the end of the cultivation period in 2018 remained there and was accounted as carbon input in the RothC model. In addition, the C input from fermentation residues was considered in all three systems. The calculation of the fermentation residues' carbon content is described in Section 2.4.2. Main assumptions and detailed information on the SOC modelling are provided in Data S1.

2.4.2 | Anaerobic digestion and electricity generation

The biomass yield data were complemented by data from laboratory analyses on the potential biogas yield and composition (CH_4:CO_2 ratio) of each substrate (Table 2). It was assumed that the substrates are anaerobically digested in a biogas plant with an electrical output of 500 kWh. Data for the infrastructure and related impacts of the anaerobic digestion plant were taken from the ecoinvent database (Wernet et al., 2016). Furthermore, it was assumed that the biogas produced is combusted in a combined heat and power generation unit with an electrical efficiency of 40% and a thermal efficiency of 43%. Emissions associated with the biogas combustion were modelled in accordance with ecoinvent standard processes (Wernet et al., 2016). It was assumed that 18% of the total heat and 10% of the total electricity production are used internally to cover the plant's heat and electricity demand. The residual heat was assumed lost, although it could be used to supply nearby buildings. For this reason, additional results for the use of 50% of the remaining heat are reported in Data S1.

As mentioned above, it was assumed that fermentation residues are recycled to the fields. The mass of the residues was calculated as the mass of the substrate minus the mass of biogas (CO_2 and CH_4) produced (Wernet et al., 2016). As explained above, it was assumed that phosphorus and potassium contents are unchanged by the fermentation process, but that only 60% of the nitrogen in the residues is available for plants (Giuntoli, Agostini, Edwards, & Marelli, 2017). Emissions from digestate storage were calculated according to EMEP/EEA (2013) for NH_3, NO and N and according to IPPC (2006) for N_2O. Emissions from the biogas combustion, including methane, N_2O and NO_x, were modelled according to Wernet et al. (2016). Methane emissions from the biogas plant, including all production steps, were assumed as 2% of the total biogas production.

2.4.3 | Fossil reference system

The marginal German electricity mix (Lauf, Memmler, & Schneider, 2019) was taken as fossil reference using standard ecoinvent inventory data (Wernet et al., 2016). In addition, a comparison with the European fossil fuel comparator is reported within a sensitivity analysis.

2.4.4 | Life cycle costing

The total production costs of the electricity production, including biogas substrate production, anaerobic digestion, and the heat and power cogeneration, were calculated based on the LCC methodology (Swarr et al., 2011). All costs occurring over the 5-year cultivation period were accounted for. The system boundaries were the same as in the environmental assessment. For each substrate scenario, the following costs were taken into consideration: land rent, CAP contribution, costs of machinery, diesel, agricultural inputs (e.g. pesticides) and labour costs. For the land costs, average land rents for the studied region in southwest Germany of 270 €/ha were taken (Statistisches Landesamt Baden-Württemberg, 2017). Costs of machinery and diesel were based on the KTBL database (KTBL, 2019), while cost data for pesticides were taken from typical consumer sources. Labour costs of 17 €/hr were assumed as a representative average estimate for the German agricultural sector (Wagner et al., 2019).

For the biomass conversion process, costs for the construction of the biogas plant and the cogeneration unit were included, assuming an electrical plant output of 500 kWh. Maintenance and operating costs for both units were based on Leible, Kälber, Kappler, Oechsner, and Mönch-Tegeder (2015). Average labour costs of 17 € were assumed, similar to those for the substrate production. To include a temporal dimension, a discount rate of 6% was applied to discount all costs to their present value. Data S1 reports assumed costs, including corresponding references, as well as results of additional sensitivity analyses regarding the discount rate, heat use and land prices.

2.4.5 | Sensitivity analysis

To assess the influence of certain important parameters on the final results of the environmental assessment, three sensitivity analyses were performed. Methane losses at combined heat and power plants vary widely (Liebetrau, Reinelt, Agostini, & Linke, 2017). To assess the sensitivity of the results with regard to this aspect, the first sensitivity analysis assumed increased methane losses. The second scenario assessed the result's sensitivity to the fate of the carbon sequestrated in the cultivated soil. In the third sensitivity analysis, potential impacts of indirect land use change (iLUC) were assessed. Considering the required arable land per functional unit as well as the relative net primary productivity of this land, potential impacts from land use changes on a global level were assessed in line with Schmidt, Weidema, and Brandão (2015). Due to the major relevance of iLUC for GWP, the sensitivity analysis focused on this impact category only. Although the results will also be sensitive to other aspects, such as the choice of by-product and ammonia emissions from application of fermentation residues, these were not considered here as they do not influence the conclusions with respect to the ranking of the biogas substrates.

3 | RESULTS

3.1 | Life cycle impact assessment

The following sections present the potential environmental impacts of the electricity production scenarios assessed. All results are given per kWh of electricity produced, ready to be fed into the national grid. For each category, impacts were attributed to the following contributors: substrate production (silage production), related changes in soil carbon contents (dLUC), biomass transport, biogas and electricity production (biogas plant). In each category, the main drivers are identified and the alternative substrate scenarios are compared. In addition, the results of each impact category are compared to the fossil reference.

3.1.1 | Global warming potential

The GWP results emphasize the importance of biogas production for the overall impacts. This phase represented the major contributor of impacts in all substrate scenarios (Figure 2a). Related emissions can mainly be attributed to methane emissions. Overall, WPM E2 had the lowest net impacts (78 g CO_{2eq}/kWh), followed by WPM E1 (183 g CO_{2eq}/kWh) and maize (236 g CO_{2eq}/kWh). Both WPM systems had higher impacts in the substrate provision phase than the maize system. The GWP of this phase was dominated by N_2O emissions from the nitrogen application and CO_2 emissions from the

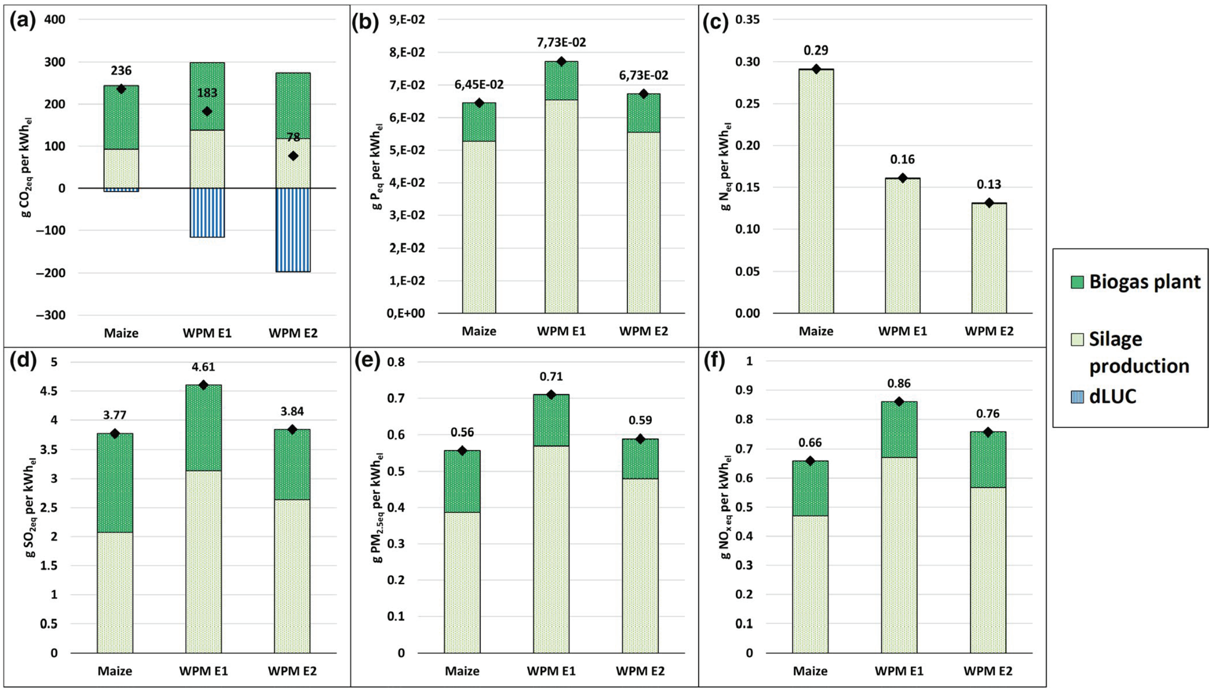

FIGURE 2 Comparison of life cycle impact assessment results of electricity production from biogas using three different substrates: Maize, wild plant mixture (WPM E1) and WPM established under maize (WPM E2). (a) Global warming potential (GPW) in g CO_{2eq}/kWh; (b) freshwater eutrophication (FE) in g P_{eq}/kWh; (c) marine eutrophication (ME) in g N_{eq}/kWh; (d) terrestrial acidification (TA) in g SO_{2eq}/kWh; (e) fine particulate matter formation (PM) in g $PM_{2.5eq}$/kWh; (f) ozone formation (OF) in g NO_{xeq}/kWh

harvesting procedures in all systems. For WPM E1 and E2, these additional impacts were counteracted by the carbon sequestration in the cultivated soil (dLUC), underlining the importance of dLUC considerations. It should be noted that the high dLUC contribution per functional unit in the WPM-based BSPS is also due to their lower productivity and thus higher land use (2.3 times the area of maize for WPM E1 and 1.5 times for WPM E2), which gives more room for soil carbon sequestration.

3.1.2 | Eutrophication potential (freshwater and marine)

Both freshwater (FE) and ME impacts were dominated by the substrate production (Figure 2b,c). This was more pronounced for ME (Figure 2c); for FE, some impacts were also related to biogas plant procedures (Figure 2b). Impacts from silage production were mainly due to phosphorus and nitrogen emissions from the application of fermentation residues and mineral fertilizers. Maize and WPM E2 had similar FE impacts, while those of WPM E1 were higher (Figure 2b). This was mainly due to the fact that the models for the phosphorus and phosphate emissions were predominantly based on the area considered, which was higher for the WPM E1 system. For ME, the WPM-based BSPS (WPM E1 and E2) were more favourable than maize (Figure 2c), mainly due to reduced nitrate losses over the cultivation period. This resulted from the constant vegetation cover, which prevents leaching during the growth period.

3.1.3 | Terrestrial acidification

Biogas and substrate production contributed an almost equally high share of the total acidification potential. Most of the impacts could be attributed to ammonia emissions associated with digestate storage and the application of fermentation residues. The lower land use efficiency of WPME E1 led to higher ammonia emissions and consequently higher impacts in the silage production than for WPM E2 and maize.

3.1.4 | Fine particulate matter formation and ozone formation

As with TA, the results for PM were dominated by the secondary aerosol ammonia. Particulate matter emitted by agricultural machinery contributed some additional PM impacts. Most of the OF impacts were due to the emission of nitrogen oxides from these machines. In sum, more than two-thirds of both the PM and OF impacts could be traced back to the agricultural stage.

Comparison of biogas options with fossil reference

Figure 3 presents the comparison of the environmental impacts of the biogas systems assessed with those of the marginal German electricity mix. The biogas systems were clearly able to contribute to the mitigation of global warming impacts under the conditions and assumptions of the study. The GWP of maize, WPM E1 and WPM E2 were 24%, 19% and 8% those of the marginal electricity mix. Similarly, the bioenergy systems resulted in lower impacts for FE and OF than the fossil reference, which had substantial impacts from lignite mining and combustion. By contrast, the fossil systems fared better in the other impact categories assessed (ME, TA and PMF). Taking all impact categories together, the biogas alternatives were clearly more favourable in terms of GHG emission mitigation, FE and OF, but these reductions come at the expense of additional burdens in three other impact categories.

3.2 | Life cycle costing

Figure 4 presents the total costs of electricity production in the biogas systems assessed. In general, the results reflect the yield differences between the biogas substrate systems. Maize was the cheapest option at 0.138 €/kWh, followed by combined maize and WPM cultivation at 0.143 € (WPM E2). The standard WPM system (WPM E1) was the most expensive at 0.156 € (Figure 4). The costs of biomass supply accounted for less than of the total costs in the WPM systems, with harvesting procedures having the most influence. This was also due to the lower specific methane yield of WPM. In addition, land costs were responsible for almost a quarter of the biomass supply costs for WPM systems, while for maize, these accounted for less than 20%, harvesting and pesticide application being the major cost driver. Accordingly, higher land costs would have a much stronger effect on WPM systems due to their less efficient use of agricultural land.

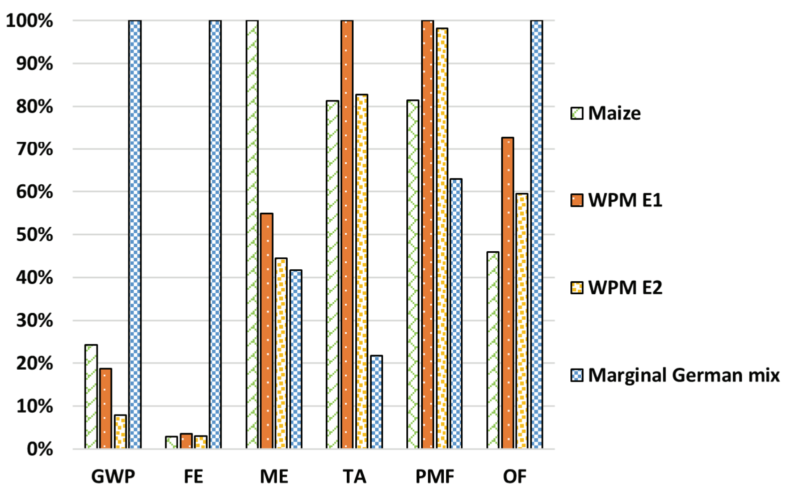

FIGURE 3 Comparison of life cycle impact assessment results of electricity production from biogas (using three different substrates: Maize, WPM E1, and WPM E2) with fossil references (marginal German electricity mix). For each category, the maximum impact is set to 100% and the impacts of the alternatives presented in relation

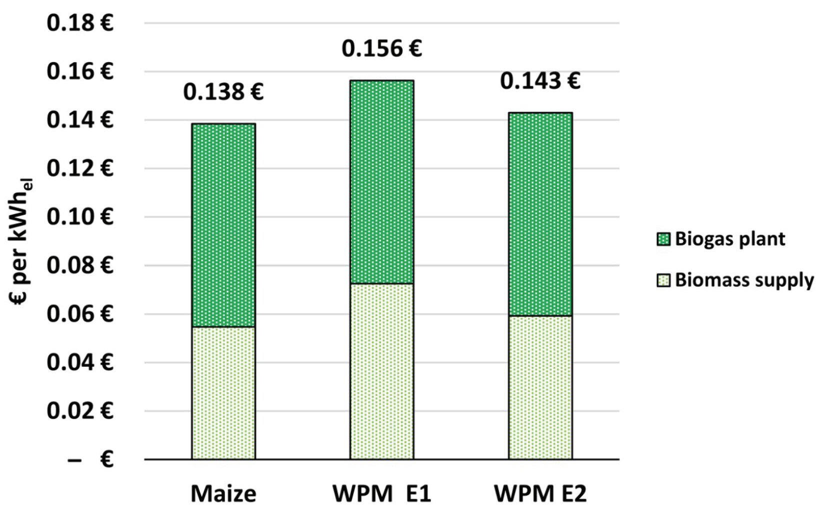

FIGURE 4 Life cycle costs of production of 1 kWh electricity to be fed into the grid, using biogas substrates from three different systems: maize (Maize), a stand-alone wild plant mixture cultivation (WPM E1), and a combination of maize under-sown with biennial and perennial wild plant species (WPM E2)

3.3 | Scenario analysis

Three sensitivity analyses were performed varying (a) methane emissions at the biogas plant, (b) the duration of carbon storage and (c) the potential influence of iLUC considerations.

3.3.1 | Methane emissions at the biogas plant

In the baseline scenario, methane emissions at the biogas plant were assumed as 2% of the total production. According to Liebetrau et al. (2017), methane emissions can vary immensely and thus strongly affect the comparison of the biogas alternatives with electricity production from fossil resources. An increased methane loss of 4% led to a surge of the total GWP by 46%, 59% and 134% for maize, WPM E1 and WPM E2, respectively (Figure 5). Despite these increased emissions, the biogas systems still performed more favourably in terms of GWP than the fossil references—GHG emissions were at least 65% lower than for the marginal German electricity mix. However, it needs to be noted that under the high-methane-emission scenario only WPM E2 would achieve a 70% reduction based on the fossil fuel comparator as suggested by the EU (SWD258, 2014).

Duration of soil carbon sequestration

In the results presented above, the GWP favourability of WPM E1 and E2 in comparison with maize very much depended on the soil carbon sequestration. The assessment was based on the assumption that the carbon sequestered during the cultivation period would be indefinitely withdrawn from the atmosphere. In reality however, the fate of the carbon would strongly depend on the consecutive cultivation techniques and systems. For this reason, it was tested how the length of the carbon storage influences the overall GWP results. This was achieved by considering soil carbon sequestration as a temporary storage only. Storage periods between one and 100 years were tested by the introduction of correction flows for delayed emissions, as suggested in ILCD (2010). Figure 6 depicts the potential GWP impacts of the maize, WPM E1 and WPM E2 systems plotted against the length of time the carbon is stored in the soil. Based on this assessment, it can be concluded that the full amount of carbon needs to remain in the soil for at least 51 years for WPM E1 and 17 years for WPM E2 to ensure lower GWP impacts than maize (Figure 6).

Effects of potential indirect land use changes

It has been shown that iLUC can strongly influence impact assessment results of agricultural product systems. The potential additional impacts for GWP when these effects were included in our study are shown in Figure 7. The iLUC impacts accounted for a substantial share of the overall GWP results. They were quantified in accordance with Schmidt and Muños (2014) and included the transformation of land not used as cropland (34.5%) and the intensification of land already in use (65.5%). Impacts of the intensification comprised the additional consumption of nitrogen fertilizers and the associated N_2O emissions according to IPCC (2006). As the results were highly sensitive to whether iLUC impacts occur or not, this aspect is crucial for decision support. For WPM E1, the inclusion of iLUC effects results in substantially higher impacts than for the more productive biogas substrate maize. This is due to the extensive land use (maize: 0.372 m^2/kWh; WPM E1: 0.892 m^2/kWh) and thus greater

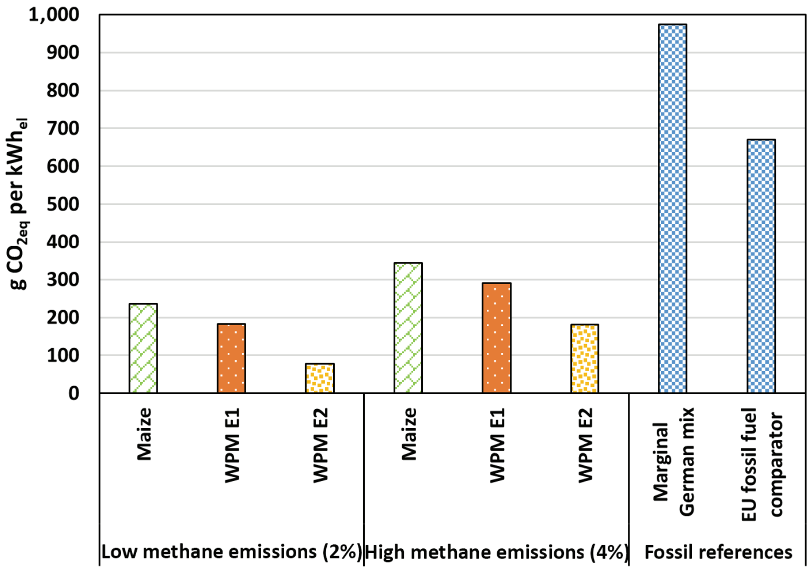

FIGURE 5 Global warming potential (GWP) of generation of 1 kWh produced from maize, wild plant mixtures (WPM), and two fossil references: Marginal German electricity mix (Lauf et al., 2019) and European Union's fossil fuel comparator (SWD258, 2014). For biomass systems, low methane emissions (2% of methane production; low methane emissions) as well as higher methane emissions (4% of methane production; high methane emissions) were compared

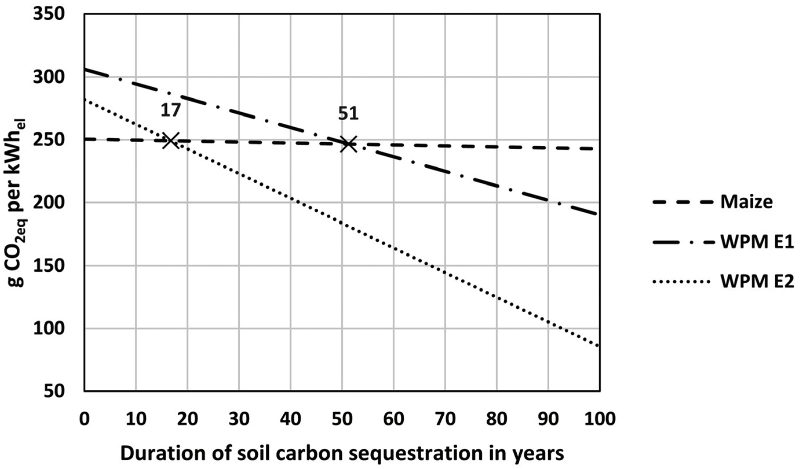

FIGURE 6 Comparison of global warming potential (GWP) of Maize, WPM E1 and WPM E2 systems depending on duration of soil carbon sequestration after end of cultivation. The ILCD correction flow of −0.01 kg CO_{2eq}*year was applied (ILCD, 2010)

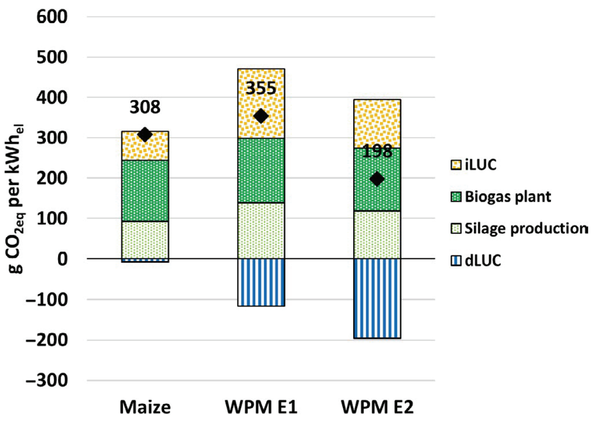

FIGURE 7 Comparison of the global warming potential of electricity production from biogas using three different substrates (in kg CO_{2eq}/kWh electricity to be fed into the grid), factoring in potential impacts from indirect land use change (in accordance with Schmidt & Muños, 2014). Biogas substrates are maize (Maize), wild plant species (WPM E1), and a combination of both (maize under-sown with biennial and perennial wild plant species; WPM E2)

land use transformation and intensification of WPM E1. If the potential iLUC impacts are factored into the extent assumed, they could outweigh the carbon sequestration related to the direct LUC. However, even under this assumption, WPM E2 would result in a lower GWP than maize and WPM E1.

4 | DISCUSSION

This study assessed the environmental and economic performance of a potential path for the extensification of biogas production by the replacement of the conventional biogas substrate maize (*Z. mays* L.) with perennial WPM. Two establishment approaches for WPM were assessed: (a) a standard procedure for WPM alone (WPM E1) and (b) WPM undersown with the use of cover crop maize (WPM E2). First, the results of the environmental assessment are critically reviewed with respect to the main contributors and sensitivity considerations. Second, potential methodological shortcomings are discussed, which may affect the decision between the extensive WPM system and maize for use as biogas substrate. Finally, the economic perspective is included to identify the conditions under which WPM cultivation could be a promising alternative to conventional biogas substrates.

4.1 | Environmental performance

The results of the environmental analysis reveal that both the maize and the two WPM systems have a better performance than the fossil reference in the impact category GWP. This observation is in line with other studies on alternative biogas substrates (e.g. Kiesel et al., 2017; Wagner et al., 2019). Judging from the results of the baseline scenario alone, WPM E2 appears to be a better option in the assessed categories as it represents a good combination of the advantages of the mono-cropped maize and WPM systems. The results emphasize the importance of both biomass and methane yields in all impact categories, and this is also one of the reasons why WPM E2 is more favourable than WPM E1 in the assessed impact categories. This finding is in line with similar studies on bioenergy, which highlight the importance of biomass yield (Lask, Wagner, Trindade, & Lewandowski, 2019; Meyer, Wagner, & Lewandowski, 2017; Wagner et al., 2019). The advantages of perennial systems with regard to the nitrate leaching risk become apparent when considering ME. In this impact category, WPM E1 and WPM E2 have substantially fewer impacts than the maize system.

The importance of land use change factors needs to be emphasized in the comparison of the maize and WPM systems, as the cultivation of perennial crops can have a tremendous influence on the soil carbon stock. In general, including dLUC benefits the environmental performance of the WPM biogas systems in terms of GWP. Per se, environmental impacts from the substrate production of WPM E1 and E2 are higher per functional unit than for maize, but these effects are outweighed by soil carbon sequestration. It should be noted that soil carbon dynamics very much depend on site-specific characteristics (e.g. initial SOC content, clay content, former land use) and on the modelling approach (Harris, Spake, & Taylor, 2015). The present study relied on RothC simulations, which use site-specific data on climate, soil features and carbon input. For these reasons, it is considered a more reliable approach than, for instance, the default emission factors proposed in the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) commonly applied in LCA (Peter, Fiore, Hagemann, Nendel, & Xiloyannis, 2016). Calibration of the model using SOC data representative of multi-annual WPM cultivation could further improve its reliability. In addition, the fate of the sequestered carbon is uncertain; it could be released with the subsequent land use and thus become GWP relevant again. As shown in the scenario analysis (Figure 6), the soil carbon sequestration would need to be ensured over at least 51 and 17 years, respectively, to render WPM E1 and WPM E2 valid alternatives to maize in terms of greenhouse mitigation. However, WPM cultivation systems could be particularly advantageous if planted on degraded lands with low SOC, since they can effectively increase the local soil carbon content.

The sensitivity analysis on the methane emissions at the biogas plant showed their strong influence on the absolute GWP of the systems under consideration. It was shown that, even under the assumption of higher methane emissions, electricity produced from the biogas substrates analysed can contribute to GWP mitigation when replacing the marginal German electricity mix. The comparison with the EU fossil fuel comparator underlines however that the substrates should ideally not be digested alone but rather in combination with manure, as has previously been suggested (Agostini et al., 2015).

The consideration of iLUC impacts that may arise due to a substrate change is also of high relevance. The third sensitivity analysis emphasized the importance of acknowledging related risks in the decision-support scheme. Although these impacts are highly uncertain, the magnitude of their implications for GWP justifies their inclusion in the environmental assessment. Although modelling approaches for iLUC impacts vary widely, the iLUC results of this study are in line with other studies on agricultural produce such as wheat, maize silage, barley and milk products (Brinkman, Wicke, & Faaij, 2017; Chobtang, McLaren, Ledgard, & Donaghy, 2017; Gerssen-Gondelach, Wicke, & Faaij, 2017; Schmidt & Muños, 2014). The sensitivity analysis applied in this study focused primarily on GWP impacts. However, iLUC and related land-transforming activities could also have impacts on the soil and habitat quality of the affected areas, and this should be addressed in future research (Gerssen-Gondelach et al., 2017). The consideration of iLUC impacts in this study is based on the assumption that the increase in land demand is entirely met by transformation of land and intensification of arable land use. However, it should be noted that a decline in German biogas production is expected in the near future. This is due to the withdrawal of political incentives, which will eventually result in a lower demand for biogas substrates and thus arable land for this application. It could be argued that the higher amount of land required for the WPM cultivation in comparison to maize could be partially offset by a reduction in demand. However, if this does not materialize, additional effects need to be considered and these are reviewed in the last section of the discussion.

4.2 | Potential methodological shortcomings

Biodiversity conservation is one of the major arguments in favour of WPM cultivation as the local biodiversity could substantially benefit from the provision of habitat, breeding space and feed for open-land birds and small game (von Cossel, Steberl, et al., 2019). This is in line with the general recognition that perennial second-generation bioenergy crops tend to contribute positively—or at least less negatively—to biodiversity conservation than first-generation bioenergy crops such as maize (Immerzeel, Verweij, van der Hilst, & Faaij, 2014). At the same time, it should be noted that the expansion of arable land is the most pressing threat to biodiversity conservation on a global scale (Ceballos et al., 2015). In this study, the risks associated with iLUC were only assessed with respect to GWP. However, they also need to be acknowledged for biodiversity aspects. This is of particular relevance since iLUC effects are likely to take place in locations of higher biodiversity potential. It is important that local biodiversity conservation in European agricultural landscapes is not counteracted by global effects. Unfortunately, conventional LCA frameworks do not consider these aspects sufficiently. In recent years, a number of methods have been proposed (Winter, Lehmann, Finogenova, & Finkbeiner, 2017) but these are only rarely used and still need to be validated for perennial crops such as WPM. Similar to biodiversity, soil health and quality aspects—advantages of WPM systems—are commonly overlooked in LCA practice. In particular for WPM systems containing legumes, aspects such as potential soil quality improvement need to be considered.

4.3 | Economic aspects

It can be concluded from the economic analysis that, of the systems assessed, the conventional biogas substrate maize is the cheapest option for the farmer, who is ultimately the decision-maker. The lower costs for maize are mainly due to the high biomass productivity, which results in low farm-gate production costs of 25.65 € per t fresh matter, compared to 27.71 € for WPM E1 and 25.65 € for WPM E2. The figures for maize are substantially lower than those previously reported for silage production in northern Italy (Agostini et al., 2016), mainly due to differences in the costs taken into account for agricultural procedures. The WPM E1 and E2 costs are comparable to German figures for miscanthus (24.2 € per t fresh matter; Wagner et al., 2019). The total costs of electricity generation per kWh_{el} produced are 0.138 € for maize, 0.156 € for WPM E1 and 0.143 € for WPM E2, which corresponds to 38.45 €, 43.42 € and 39.72 € per GJ_{el}, respectively. For WPM systems in particular, land costs are the major contributor. The land rents assumed in this study are average values for the German federal state of Baden-Württemberg, but in typical biogas regions land costs are usually significantly higher than the average at 750 €/ha (Statistisches Landesamt Baden-Württemberg, 2017). The assumption of these higher values would widen the economic imbalance between the systems with the costs of the alternative systems exceeding maize (0.158 €) by 28% (WPM E1: 0.202 €) and 11% (WPM E2: 0.175 €; Figure S4). Consequently, a switch to WPM systems in these regions cannot be recommended to the farmer from an economic point of view, if no political incentives are in place and only direct costs are accounted for. This becomes even more apparent when the German legislation on renewable energy is considered. Plant operators receive up to 0.1690 €/kWh if their plant (>150 kW) was operational before 2017. For newer plants, a maximum value of 0.1488 €/kWh was set for 2017 with a yearly decrease of 1%. This compares to the 0.2500 €/kWh guaranteed in the first phase of the biogas-supporting schemes in Germany (Renewable Energy Sources Act, 2014).

4.4 | Wild plant mixtures: A sustainable, alternative biogas substrate? Conditions and requirements

The environmental assessment reveals the higher-yielding WPM E2 system as the more promising of the extensive systems in terms of the categories considered here. However, its economic performance is still lower than that of the maize system. Thus, agricultural practices for WPM cultivation need further optimization. The importance of finding alternative biogas substrates such as WPM is highlighted by the requirements of the German renewable energy legislation, which aims at a continuous decline in the share of maize in the overall feedstock composition. From 2021 on, the maximum proportion is set at 44% of maize silage, compared to 69% in 2016 (FNR, 2019). Further improvement of the WPM systems seems feasible, as these predominantly contain undomesticated species, which have not been the subject of breeding efforts (Kuhn, Zeller, Bretschneider-Herrmann, & Drenckhahn, 2014; Schmidt, Lemaigre, Delfosse, von Francken-Welz, & Emmerling, 2018). However, maize would still be the more economic biogas substrate. The maize yield (20.1 t DM/ha) in the present study is rather high, exceeding the regional average of 15.8 t DM/ha by more than 25% (Statistisches Landesamt Baden-Württemberg, 2020). When comparing the biogas substrates, it is worth noting that the WPM E1 yield (11.3 t DM/ha) exceeded yields of other WPM cultivation trials in southern Germany (8.9 t DM/ha) by a similar percentage (Vollrath et al., 2016). As the yields of both biogas substrates were on an equally high level, the conclusions regarding the feedstock comparison can be considered robust. In general, lower yields, in practice, would result in higher environmental impacts and costs than reported in this study. The maize system assessed in the here achieved these high DM yields under moderate N inputs of 90 kg/ha, resulting in lower environmental impacts for the maize silage than in comparable studies. For instance, Kiesel et al. (2017) reported DM yields of 18.9 t when applying 240 kg N/ha under similar conditions. Likewise, the ecoinvent process for the integrated production of silage maize in Switzerland (Wernet et al., 2016) suggests DM yields of 17.2 t can be achieved with fertilizer inputs of 98 kg N/ha.

Yields are also important in determining the amount of land required for the production systems under assessment, with the WPM systems requiring substantially larger areas. This could lead to additional greenhouse gas emissions as a result of a geographical shift of agricultural production, as reported in the iLUC scenario analysis. The scenario analysis showed that these effects could partially outweigh the reported carbon-related advantages of the WPM systems such as increased soil carbon accumulation. Accordingly, critical assessment of WPM cultivation, including the handling of potential iLUC effects, is crucial for it to be able to offer a favourable alternative to maize in terms of greenhouse gas mitigation.

A range of iLUC mitigation measures, such as the use of marginal land, have been discussed in the literature (Wicke et al., 2015). In general, marginal lands are characterized by a number of limitations that constrain agricultural productivity. The total marginal agricultural land area in Europe has been estimated at 646,833 km^2 (von Cossel et al., 2019). The cultivation of WPM on such land is considered feasible and would benefit from the substantial potential for adaptability, stress resistance and resilience achieved through the wide range of plant species in the seed mixtures. Once established, the deep rooting systems and perennial nature of WPM provide an advantage over maize and allow successful adaptation to the conditions characterizing marginal agricultural lands (Vollrath et al., 2012; von Cossel & Lewandowski, 2016). Degraded land could considerably benefit from the effective soil-carbon build-up in WPM cultivation systems, thus contributing to the capture of CO_2 from the atmosphere. WPM could likewise be used in strip cultivation or buffer systems, as already suggested for other perennial crops (Ferrarini, Serra, Almagro, Trevisan, & Amaducci, 2017). The perennial polycultures could also meet the requirements of ecological focus areas in the near future. In all these situations mentioned above, the agricultural system could benefit from the advantages of WPM cultivation, including carbon storage, biodiversity conservation, reduced nutrient leaching and other ecological services at landscape level.

The electricity generation from biogas produced from WPM resulted in lower impacts than for maize in two of the six impact categories assessed. The maize system seems more favourable from an economic perspective, which is a major decision criterion for farmers. It has been suggested that the goal of using WPM for biogas production is not to achieve the same level of productivity as maize, but to justify a change of feedstock through a combination of economic and environmental factors (Vollrath et al., 2012). Nevertheless, under current conditions, the adoption of WPM cultivation should only take place if long-term soil carbon sequestration can be ensured. In addition, large-scale adoption of stand-alone WPM cultivation should not take place on good quality land but should instead be reserved for low-yielding land. This recommendation needs to be emphasized where deployment would result in a geographical shift of agricultural production, as shown in the iLUC scenario analysis. The uncertainty in dLUC considerations and the potential occurrence of iLUC impacts could affect the relative ranking of the GWP results, underlining the importance of acknowledging these aspects in the assessment of agricultural extensification strategies. The adoption of WPM systems will most likely depend on the management of soil carbon storage as well as of the trade-offs between iLUC-associated risks and higher costs, which are ultimately related to the question of productivity. Agricultural biogas facilities in Germany usually use a substrate combination of energy crops and manure. WPM from non-cultivated, marginal agricultural land, ecological focus areas and buffer strips could complement substrate mixes, while enriching agricultural landscapes.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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SUPPORTING INFORMATION

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

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Chapter 4

4.0 Lignocellulosic ethanol production combined with CCS - A study of GHG reductions and potential environmental trade-offs

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Lignocellulosic ethanol production combined with CCS—A study of GHG reductions and potential environmental trade-offs

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Abstract

The combination of bioethanol production and carbon capture and storage technologies (BECCS) is considered an indispensable method for the achievement of the targets set by the Paris agreement. In Croatia, a first-of-its-kind biorefinery project is currently underway that aims to integrate a second-generation ethanol plant into an existing fossil refinery. The goal is to replace the fossil fuel production by second-generation ethanol production using miscanthus. In the ethanol fermentation, CO_2 is emitted in highly concentrated form and this can be directly compressed, injected and stored in exploited oil reservoirs. This study presents an assessment of the greenhouse gas (GHG) reduction potential of miscanthus ethanol produced in combination with CCS technology, based on data from the planning process of this biorefinery project. The GHG reduction potential is evaluated as part of a full environmental life cycle assessment. This is of particular relevance as a lignocellulosic ethanol industry is currently emerging in the European Union (EU) and LCAs of BECCS systems have, so far, often omitted environmental impacts other than GHG emissions. Overall, the ethanol to be produced in this planned biorefinery project would clearly achieve the EU's global warming potential (GWP) reduction target for biofuels. Depending on the accounting approach applied for the biological carbon storage, reduction potentials between 104% and 138% relative to the fossil comparator are likely. In addition, ethanol can reduce risks to resource availability. As such, the results generated from data based on the intended biorefinery project support the two major rationales for biofuel use. However, these reductions could come at the expense of human health and ecosystem quality impacts associated with the combustion of lignin and biogas. In order to prevent potential environmental trade-offs, it will be imperative to monitor and manage these emissions from residue combustion, as they represent significant drivers of the overall environmental impacts.

KEYWORDS

BECCS, bioenergy, carbon capture and storage, environmental impacts, ethanol, LCA, miscanthus

1 | INTRODUCTION

The Paris Agreement stipulates that global warming needs to be kept below 1.5°C above preindustrial levels. As the transportation sector is a major contributor to global greenhouse gas (GHG) emissions, new approaches are necessary to mitigate related negative impacts. Biofuels, in particular fuels from lignocellulosic biomass, are considered a valid option for reducing the sector's GHG emissions (Morales et al., 2015). However, due to the shortage of time for action, carbon capture and storage technologies (CCS) are often considered an indispensable complement to other mitigation efforts. The combination of bioethanol production and CCS (BECCS) is seen as a promising approach that could contribute to the removal of carbon dioxide (CO_2) from the atmosphere (Edwards & Celia, 2018). For a number of reasons, including land use aspects and competition with food production, the European Union favours the supply of advanced biofuels from feedstocks such as lignocellulose (Directive (EU), 2018/2001). Miscanthus is a promising lignocellulosic feedstock due to its high productivity and potential to grow on marginal land. Until now, miscanthus has not been used for commercial ethanol production and the environmental performance of miscanthus ethanol has only been analysed based on lab-scale experiments or techno-economic models (Boakye-Boaten et al., 2017; Lask et al., 2019). However, the potential for commercial ethanol production has been recently proven in precommercial refinery trials (e.g. as assessed within the EU-financed demonstration project GRACE [grant agreement no 745012]). In Croatia, a first-of-its-kind biorefinery project is currently under development. The aim is to integrate a miscanthus ethanol plant into an existing fossil oil refinery in close proximity to exploited oil reservoirs, which are suitable for carbon storage.

The fossil fuel production is to be replaced by ethanol production based on miscanthus. The ethanol fermentation generates a stable flow of CO_2 as off-gas. In contrast to CO_2 sources in other BECCS systems (mainly combustion processes), this is emitted in a highly concentrated form and can be directly compressed, injected and stored in the given depots. The feedstock for the refinery will be cultivated on unused arable land, which is abundantly available in the region surrounding the refinery. Ongoing commercial-scale field trials show the potential of these areas for miscanthus production and provide valuable information on agronomic operations. These practical experiences of feedstock cultivation as well as data from the planning and design phase of the refinery can be used for a representative assessment of the contribution of this low-input feedstock to the sustainable advancement of the transportation sector. This study presents an evaluation of the GHG reduction potential of miscanthus ethanol produced in combination with CCS technology. The GHG reduction potential is evaluated as part of a full environmental life cycle assessment, which compares ethanol's environmental performance with that of fossil petrol. This is of particular relevance as a lignocellulosic ethanol industry is currently emerging in Europe and LCAs of BECCS systems have, so far, often omitted environmental impacts other than GHG emissions (Gough et al., 2018; Sanchez et al., 2018).

2 | MATERIALS AND METHODS

2.1 | Goal and scope

This study has two objectives. First, it aims to determine the GHG reduction potential of miscanthus ethanol produced in a BECCS system in comparison with a fossil alternative. For this purpose, the GWP per mega joule (MJ) is evaluated and the results are benchmarked with the EU's RED2 reduction targets for liquid biofuels. In line with the RED2 calculation methodology, impacts from car manufacture and maintenance as well as road construction are not included in these results.

The second objective is the identification of a potential environmental burden shifting when replacing petrol by ethanol. This is achieved through an assessment of both the production of ethanol within a BECCS system and its use in a medium-sized passenger car. It is also performed to provide information on optimization potential in terms of environmental performance. A cradle-to-grave approach is taken and a comparison with petrol, the fossil reference, is included. The analysis is based on a functional unit of 1 km driven in a medium European passenger car (vehicle weight 1600 kg). In contrast to the assessment in line with the RED2 calculation methodology, car manufacture and maintenance as well as road construction and maintenance are included in these results.

2.2 | Methods

The environmental performance is assessed by conducting an LCA according to the ISO standards 14040 and 14044 (ISO 14040, 2006; ISO 14044, 2006), using the life cycle impact assessment method collection ReCiPe2016 v1.1 (Huijbregts et al., 2017). ReCiPe 2016 v1.1 is chosen as it is up-to-date and allows endpoint results to be derived. Results for all endpoints—damage to human health, ecosystems and resource availability—are presented and relevant midpoint indicator results are discussed. Relevant midpoint indicator categories are identified as those which contribute at least 80% of the total impacts at endpoint level. The ecoinvent database v3.5 cut-off is taken for background data and openLCA 1.9 for modelling and impact calculation (Wernet et al., 2016).

2.3 | System boundaries

A Croatian biorefinery project currently under development serves as a case study. The facility is to be incorporated into an existing oil refinery in Sisak. It is designed for an annual biomass intake of 243 kt (15% moisture content) and an expected production output of 55 kt ethanol. The location has two major advantages: First, the refinery will be in close proximity to depleted crude oil reservoirs, which can be used for the storage of CO_2. Experience with enhanced oil recovery ensures the feasibility of carbon sequestration in this area and provides reliable data for the assessment. Second, land for miscanthus cultivation is abundantly available in Croatia. In total, 2,149,080 ha of land can be utilized for agriculture. In 2018, only 69.1% of this area was cultivated, while 30.9% (663,435 ha) remained unused. Large sections of these areas had been used for agricultural production in the past but were abandoned during the war in the 1990s (Tomić, 2020). A substantial proportion of this unused land lies within a 75–100 km radius of Sisak. It has been estimated that around 60,000 ha of unused agricultural land is located in Sisačko–moslavačka, the county surrounding Sisak (Bilandžija et al., 2016). These lands hold substantial potential for the cultivation of energy crops without affecting other agricultural production structures.

Figure 1 presents the system assessed in the study. It includes preparation of the unused agricultural land, miscanthus cultivation, transport to and processing in the nearby refinery in Sisak, capture and sequestration of fermentation off-gas, product distribution and the use phase. A 20 year cultivation period for miscanthus, including the production and transport of inputs as well as agricultural procedures were considered. It was assumed that miscanthus reaches its full yield potential from the third cultivation year onwards. For the second cultivation year, only half the amount was taken. Input and output data for the processing stage are derived from the precommercial tests using miscanthus as a feedstock and basic refinery engineering. The modelling of the fossil reference system is based on an established dataset from ecoinvent v3.5 (Wernet et al., 2016).

2.4 | Life cycle inventory

2.4.1 | Land clearing

Unused agricultural land for miscanthus production is abundantly available in the region under study, as described in Section 2.3. This land has not been in use for at least 20 years and is densely vegetated with bushes, shrubs and small trees (above-ground biomass ~47 t FM ha^{-1}; 50% DM content; 50% carbon content in dry matter; INA d.d., personal communication, March 2020). For this reason, land clearing and biomass removal is required prior to the miscanthus establishment. 90% of the above-ground biomass is removed for bioenergy production. The remaining 10% decay, releasing carbon. The clearing activities require 130 L diesel per hectare (INA d.d., personal communication, March 2020). The impacts of the diesel consumption and the biomass decay are fully allocated to the miscanthus cultivation. The removed biomass is used for energy production. For this reason, the transportation and combustion impacts are not imputed to the miscanthus cultivation but allocated to the energy production.

2.4.2 | Agricultural system

A rhizome-based miscanthus establishment and cultivation is considered. Required agronomic operations include soil

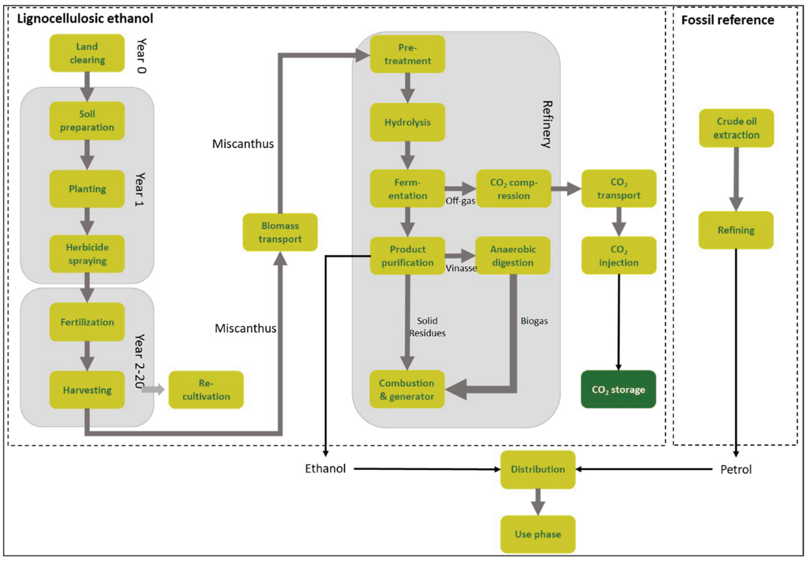

FIGURE 1 Analysed system and system boundaries

preparation, establishment, fertilization, harvest and recultivation of the plantation. Harvest includes mowing, swathing, baling and loading of the bales. Data for the harvesting procedures, e.g. quantities and diesel consumption, are derived from commercial miscanthus cultivation. Table 1 presents all procedures with corresponding frequencies over the cultivation period. Fertilizers and herbicides are summed up over the entire period including the establishment phase and divided by the total biomass yield over 20 years. A full biomass yield of 22 t DM ha^{-1} $year^{-1}$ is assumed. In addition, lower- (19 t DM ha^{-1} $year^{-1}$) and higher-yielding (25 t DM ha^{-1} $year^{-1}$) scenarios were tested. Experience from commercial miscanthus plantations indicates that approximately 4 t DM ha^{-1} remain on the field after harvest (Terravesta Ltd, personal communication, March 2020). For this reason, the harvestable biomass yield amounts to 7 (or 5.5/8.5) t DM ha^{-1} $year^{-1}$ for the second year and 18 (or 15/21) t DM ha^{-1} $year^{-1}$ for the subsequent years. Table 2 presents annual quantities and biomass properties. As nitrogen fertilization is not typically applied in commercial miscanthus plantations, only phosphorus and potassium fertilization was considered in this study. Quantities of phosphorus and potassium fertilizers as well as application of herbicides and other agricultural procedures are based on experiences of commercial-scale miscanthus cultivation (Terravesta Ltd, personal communication, March 2020). All input data are shown in Table 2. Field emissions associated with the miscanthus cultivation are modelled using established models. These are summarized in Table 3. Pesticides are assumed to be released to agricultural soil (Nemecek & Kägi, 2007). As the miscanthus cultivation takes place in a specific region, location-specific characterization factors (CF) for land use are considered as suggested in the ReCiPe2016 methodology (Huijbregts et al., 2017). The global CF for permanent crops is replaced by the ecoregion-specific CF given in Baan et al. (2013). The baled biomass is assumed to be transported from field to refinery by truck over a distance of 40 km.

TABLE 1 Agricultural procedures during a 20-year miscanthus cultivation period

Agricultural procedure	Number per cultivation period
Ploughing—prior to establishment	1
Rotary harrowing	1
Planting	1
Rolling	1
Herbicide spraying	5
Phosphorus and potassium fertilisation	20
Mowing	19
Swathing	19
Baling	19
Bale loading	19
Ploughing—final year	1

Miscanthus captures substantial amounts of carbon in the below-ground plant parts. From here on, this is referred to as *biological carbon storage* and is based on the amount of carbon which can be stored in the below-ground biomass of established *Miscanthus* × *giganteus*. An average ratio of 0.52 for the above-/below-ground biomass distribution was derived from the literature (Davey et al., 2017; Kahle et al., 2001; Richter et al., 2015). Assuming a biomass carbon content of 48%, the total below-ground carbon storage was calculated as 17.54, 20.31 and 23.08 t C ha^{-1}, respectively, for a full yield of 19, 22 and 25 t ha^{-1} (for detailed calculation, see Section 1.1 in Supplementary Material). As the fate of the soil carbon sequestered is uncertain, a conservative accounting approach is applied. In accordance with the ILCD handbook, a temporary carbon storage for the cultivation period is assumed (European Commission, 2010). Following this approach, the carbon is assumed to be completely released after the miscanthus cultivation. However, such flash emissions after miscanthus removal are unlikely (Mangold et al., 2019) and for this reason, indefinite biological carbon storage is considered as a maximal contrast in a sensitivity analysis.

TABLE 2 Main inputs of 20-year miscanthus cultivation period and average properties of harvested biomass

Inputs/outputs	Quantity	Unit
Rhizomes	15,000	pieces ha^{-1}
Phosphorus, in form of triple superphosphate—56 kg P_2O_5 ha^{-1} once in 4 years	14	kg P_2O_5 ha^{-1} $year^{-1}$
Potassium, in form of potassium chloride—119 kg K_2O ha^{-1} each year	119	kg K_2O ha^{-1} $year^{-1}$
Herbicides	11	L ha^{-1}
Dry matter (DM) harvested (full yield)	18,000	kg ha^{-1}
DM harvested (over cultivation period)	16,650	kg ha^{-1} $year^{-1}$
DM content	85	%

TABLE 3 Considered field emissions and their primary sources with references

Emission	Source	Reference
Nitrous oxide (N_2O)	Harvest residues	Bouwman et al. (2002); IPCC (2019)
Phosphorus/Phosphate (P, PO_4^{3-})	P fertiliser	Prasuhn (2006)
Heavy metals	Fertilisers/pesticides	Freiermuth (2006)

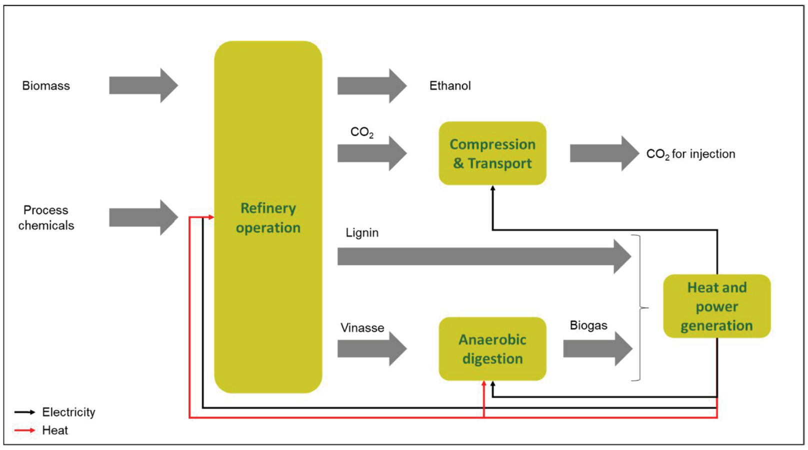

FIGURE 2 Process overview of the ethanol refinery including carbon dioxide compression, transport and injection

2.4.3 | Refinery

Chemical and nutrient inputs are derived from the planning phase of the lignocellulose refinery. A conversion efficiency of 22.6% is assumed for the analysis, meaning that 226 g of ethanol is produced per kilogram of dry miscanthus (INA d.d., personal communication, March 2020). Figure 2 presents an overview of the refinery processes.

Lignin and vinasse are by-products of ethanol production. Vinasse is anaerobically digested and the digestate sent to an on-site wastewater treatment (WWT) plant. Emissions from the WWT plant are estimated based on (Doka, 2009). The biogas is combusted alongside the lignin in an adjacent boiler for heat and power generation, covering the refinery's entire steam and electricity demand (including the electricity for the WWT as well as for CO_2 compression and transport). Carbon-, nitrogen- and sulphur-related components in the flue gas and ash are modelled in accordance with transfer coefficients for solid waste incineration (Doka, 2013). Flue gas cleaning is necessary to meet the European legal requirements for emissions of NO_x, SO_2 and particulates (see Table S3. Quicklime is applied for the desulphurization of the flue gas. The resulting calcium sulphate is assumed to be disposed of to landfill. NO_x reduction is achieved using ammonia as catalyst and particulate matter reduction by electrostatic precipitation. Non-carbon-, nitrogen- and sulphur-related emissions from the residue combustion are modelled using emission factors for wood chip combustion in a cogeneration unit as surrogate (Wernet et al., 2016). Presently, a recast of the European industry emission limits is anticipated. For this reason, lower legal limits were tested in a sensitivity analysis (see Table S3).

2.4.4 | Carbon capture and storage

The fermentation off-gas is highly concentrated CO_2 and is processed without further refinement. It is compressed to 30 bar, liquefied and transported to the injection facility via pipelines. Here, it is further compressed to 190 bar for injection into the former oil wells. Electricity requirements for compression and pumping are taken from real-life data in Croatia (0.30 kWh kg^{-1} CO_2 for compression and 0.15 kWh kg^{-1} CO_2 for injection; INA d.d., personal communication, March 2020). The electricity requirements for initial compression and transport are covered by the refinery's electricity generation. For injection, the Croatian electricity mix is considered (Wernet et al., 2016).

2.4.5 | Use phase

Information on the use phase, including the car manufacture and maintenance, road construction and maintenance as well as emissions from driving are based on established processes from the life cycle inventory database ecoinvent 3.5 with EURO5 standard (Wernet et al., 2016). For carbon dioxide and carbon monoxide, the source of the emissions is changed from fossil to biogenic/non-fossil. It should be noted that car manufacture and maintenance (of car and road) were only included in the full environmental assessment.

2.4.6 | Fossil reference system

The fossil reference system is based on standard ecoinvent processes for crude oil extraction and refining as well as the distribution of petrol and its use in a medium-sized European car with EURO5 standard (Wernet et al., 2016).

3 | RESULTS

3.1 | GWP reduction potential

European legislation defines mandatory GWP reduction targets for biofuels (European Parliament, Council of the

European Union, 2018). For lignocellulosic ethanol production facilities starting operation after January 2021, 65% reductions need to be achieved relative to the fossil fuel comparator (94 g $CO_{2\ eq}$ MJ^{-1}). Figure 3 presents the GWP of miscanthus ethanol from the assessed production for three biomass yield scenarios. In line with the REDII calculation methodology, the results are presented in g $CO_{2\ eq}$ MJ^{-1} and do not include impacts from car manufacture and maintenance or road construction. For all yield levels, lignocellulosic ethanol from miscanthus exceeded the reduction targets when combined with CO_2 capture and storage (CCS). The conservative assumption of temporary storage of the biologically captured CO_2 led to emission reductions of more than 104% relative to the fossil comparator, i.e. net-negative emissions are achieved. Indefinite storage gave substantially higher emission savings of approximately 138% (Figure 3). Detailed contribution analyses of the GWP results are provided in Figure S1.

3.2 | Endpoint indicator results

The following sections present the results for the three endpoints, *damage to human health*, *ecosystem quality* and *resource availability*, calculated for the baseline scenario of driving 1 km in a medium-sized European car running on either ethanol or petrol. A full biomass yield of 19, 22 and 25 t DM ha^{-1} (corresponding to 15, 18 and 21 t DM ha^{-1} harvestable biomass), temporary storage of carbon in the below-ground miscanthus plant parts over the cultivation period (17.5, 20.3 and 23.1 t C ha^{-1}) as well as capture and storage of CO_2 from the fermentation off-gas are considered.

3.2.1 | Damage to human health

Figure 4 presents impacts on human health associated with driving a medium-sized car over a distance of 1 km. Under the

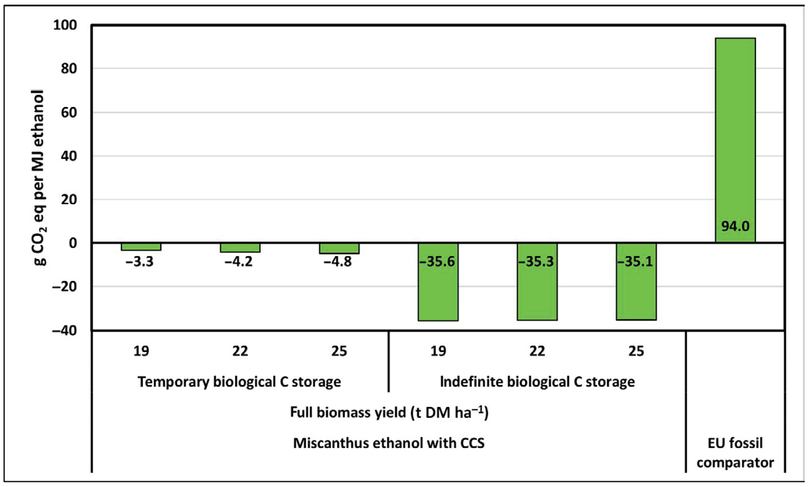

FIGURE 3 Global warming potential per MJ (in g CO_2 eq) of ethanol produced using two alternative approaches to biological carbon storage accounting: temporary and indefinite. Temporary biological C storage assumes that all carbon stored in the below-ground biomass is released after the cultivation period. Indefinite biological C storage assumes that all carbon remains indefinitely in the soil. Three yield levels are compared: 19, 22 and 25 t DM ha^{-1}, corresponding to harvestable biomass yields of 15, 18 and 21 t DM ha^{-1}

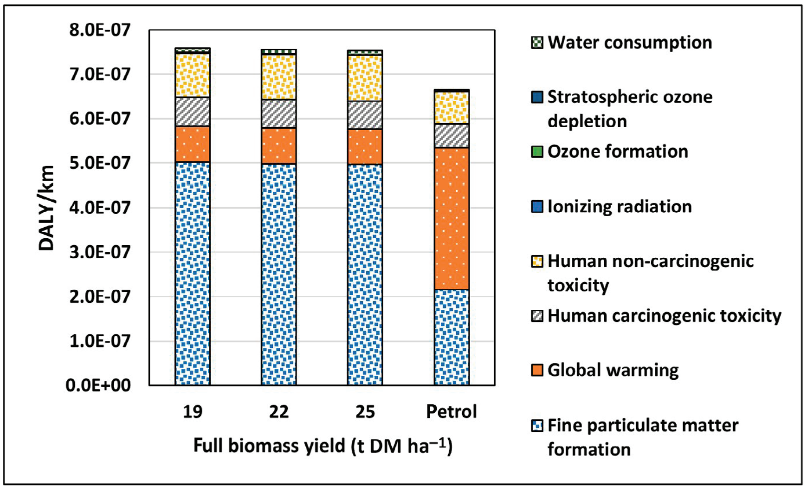

FIGURE 4 Human health impacts of driving a medium-sized car running on ethanol from miscanthus or petrol; given in disability-adjusted life years (DALY) per kilometre driven. Three yield levels are compared: 19, 22 and 25 t DM ha^{-1}, corresponding to harvestable biomass yields of 15, 18 and 21 t DM ha^{-1}

present assumptions and irrespective of the miscanthus yield, ethanol results in higher human health impacts than the fossil reference. These are dominated by fine particulate matter formation (FPMF), which accounts for 66% of total impacts and mainly stems from residue combustion and associated SO_2 and NO_x emissions. Further relevant impact categories include human non-carcinogenic toxicity (HNCT), GWP and human carcinogenic toxicity (HCT), contributing 13%, 11% and 8% respectively. Impacts from other categories (water consumption, stratospheric ozone depletion, ionizing radiation) are negligible, as the four categories FPMF, HNCT, GWP and HCT together account for 98% of the human health impacts. Major differences in impact characteristics between ethanol and petrol are observed for GWP and FPMF: GWP impacts are only a quarter of those of petrol, whereas FPMF impacts of ethanol are twice as high and mainly stem from the residue combustion in the refinery. Detailed contribution analyses are provided in Section 3.2.4.

3.2.2 | Damage to ecosystem quality

Ecosystem quality impacts for ethanol are slightly higher (<5%) than for petrol. Figure 5 presents potential ecosystem damage associated with the reference flows. Similar to human health impacts, variations in biomass yield influence the results only slightly. The most relevant midpoint impact categories are terrestrial acidification (TA), land use (LU), GWP and ozone formation (OF), contributing 33%, 24%, 17% and 14% respectively. In sum, these four categories account for 88% of the total ecosystem impacts. Major differences in the impact patterns between the alternatives are found for TA, LU and GWP. While ethanol results in substantially lower GWP, both TA and LU impacts are considerably higher than for the fossil reference. A detailed contribution analysis of the relevant impact categories is given in Section 3.2.4.

3.2.3 | Damage to resource availability

Figure 6 presents the damage to resource availability associated with the reference flows. The results indicate lower resource consumption for ethanol than for petrol. Mineral resource impacts are negligible for both alternatives, with the midpoint indicator *fossil resource scarcity* accounting for 97% (ethanol) and 99% (petrol) of total impacts. For ethanol, the largest proportion of impacts is associated with refinery chemicals (mainly ammonia) and diesel consumption in the biomass provision.

3.2.4 | Contribution analyses of most relevant impact categories

The most relevant impact categories for human health and ecosystem quality were identified from the results presented in Figures 4 and 5. These include FPMF, GWP, HNCT, LU, OF and TA. Figure 7 presents contribution analyses for each of these midpoint impact categories (for the medium-yield scenario: 22 t DM ha^{-1} full biomass yield/18 t DM ha^{-1} harvestable yield).

Five of the six categories, FPMF, GWP, HNCT, OF and TA, are only slightly influenced by impacts from the **miscanthus cultivation**, with contributions ranging from 14% for to −6.8% for HNCT. Negative values for the HNCT contributions result from the generic heavy metal balance, which indicates an export of heavy metals from the field through the harvested biomass. In contrast to the patterns for these five impact categories, miscanthus cultivation is a major driver of LU impacts, accounting for 27% of the total impacts. **Refinery operation** contributes to all examined impact categories, with contributions ranging from 12% for HNCT to 51% for LU and 69% for GWP. The LU impacts derive predominantly from the agricultural production of major refinery inputs. Although the

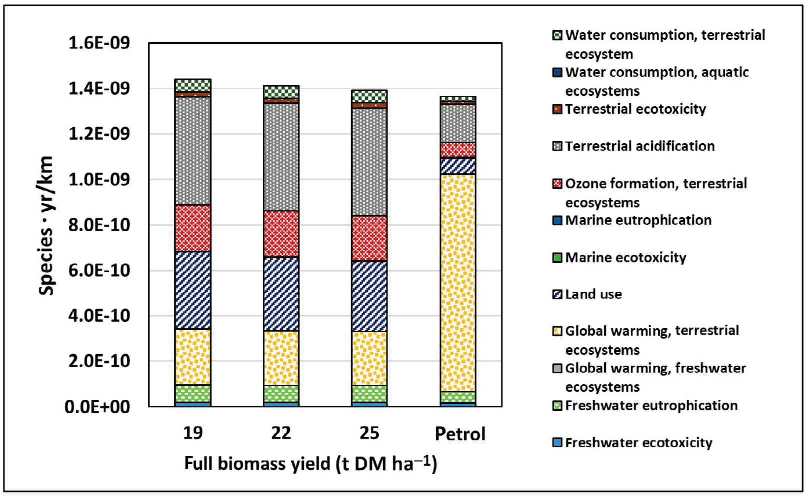

FIGURE 5 Damage to ecosystem quality resulting from driving a medium size car running on ethanol from miscanthus or petrol; given in species loss (species·year) per kilometre driven. Three yield levels are compared: 19, 22 and 25 t DM ha^{-1}, corresponding to harvestable biomass yields of 15, 18 and 21 t DM ha^{-1}

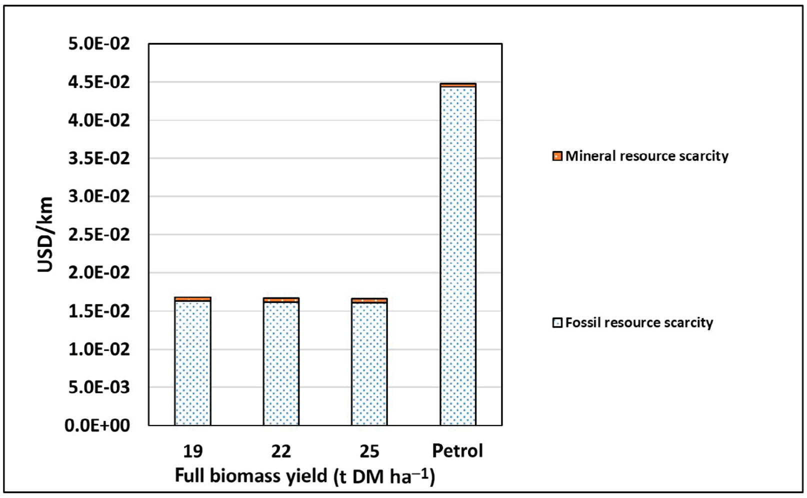

FIGURE 6 Damage to resource availability from driving a medium size car running on ethanol from miscanthus or petrol; given in USD (US dollar) per kilometre driven. Three yield levels are compared: 19, 22 and 25 t DM ha^{-1}, corresponding to harvestable biomass yields of 15, 18 and 21 t DM ha^{-1}

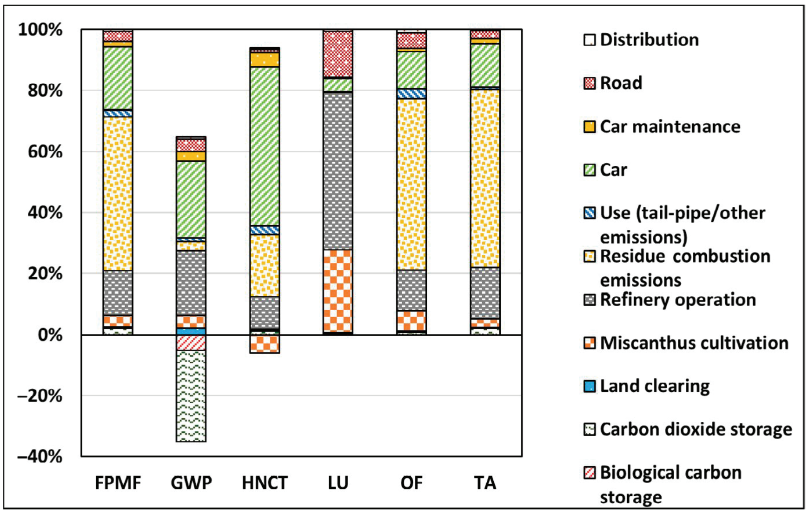

FIGURE 7 Contribution analysis for the six most relevant impact categories for driving a medium-sized car running on petrol (FMPF, fine particulate matter formation; GWP, global warming potential; HNCT, human non-carcinogenic toxicity; LU, land use; OF, ozone formation; TA, terrestrial acidification)

absolute area used for miscanthus cultivation is higher than for the provision of the refinery inputs, the impacts on species losses and ecosystem quality are lower due to the ecoregion-specific characterization factors for permanent crops. The GWP impacts are dominated by the upstream impacts of major refinery inputs as well as emissions from the wastewater treatment. These impacts are however counterbalanced by credits from the **CO_2 storage**, which represents the major individual contributor in the GWP category. **Residue combustion** and related emissions constitute the major driver in three of the six impact categories. This stage accounts for 50%, 56% and 58% of the impacts for FPMF, OF and TA, respectively, and is predominantly due to the emissions of NO_X, $PM_{<2.5}$, NH_3 and SO_2. Sulphur-containing refinery inputs are the major source of the latter. **Car production and maintenance** contribute substantially to all impact categories considered except land use. It is a major individual driver of the total impacts for both GWP and HNCT. It should however be emphasized that these impacts are less relevant for the ethanol-petrol comparison, as the absolute impacts per kilometre driven are identical for both alternatives.

3.3 | Accounting for soil carbon sequestration

The endpoint indicator results presented above are based on a conservative accounting approach for soil carbon storage. It assumes a temporary carbon storage, i.e. all the carbon is released at the end of the cultivation period. A sensitivity analysis considers indefinite storage and compares it to the default assumption. The endpoint indicator results for damage to ecosystem quality are presented in Figure 8. These indicate lower impacts for the indefinite-storage scenario. In this scenario, impacts on ecosystem quality are lower than for petrol. Similarly, total human

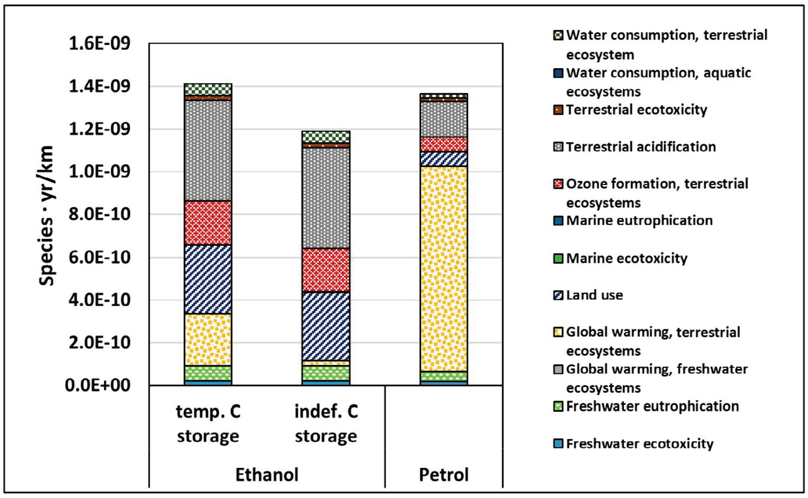

FIGURE 8 Ecosystem quality impacts of driving a medium size car running on ethanol from miscanthus or petrol; given in species loss (species·year) per kilometre driven. Comparison of soil carbon sequestration accounting: temporary and indefinite carbon storage

health impacts of ethanol are reduced but still exceed the ones for petrol (Figure S2).

4 | DISCUSSION

Ethanol production from miscanthus in combination with CCS can contribute substantially to the reduction of GHG emissions in the European transportation sector when replacing petrol. Irrespective of biological carbon storage and yield variations, the reduction targets set by the European Union can be achieved in the given context. Depending on the accounting approach for the biological carbon storage, reduction potentials between 104 and 138% are likely relative to the EU's fossil fuel comparator. Both accounting approaches taken in this study are extreme alternatives. Carbon stored in the below-ground biomass is neither fully and spontaneously released when the miscanthus cultivation ends, nor is the whole amount stored indefinitely. In practice, a reduction potential in between these worst- and best-case scenarios is to be expected. Whatever the actual case, net-negative emissions are likely.

In addition, it was shown that ethanol use affects resource availability to a smaller extent than petrol. Accordingly, the major rationale for biofuel use—mitigation of climate change impacts and fossil resource dependence—can be effectively achieved. However, it should be noted that these positive effects could potentially be accompanied by environmental trade-offs, as indicated by the endpoint indicator results in Figures 4 and 5. Under the baseline conditions and assumptions, ethanol indicates higher potential impacts on human health and ecosystem quality than the fossil alternative petrol. Given a relative difference of less than 15% (human health impacts) and 5% (ecosystem quality) between the ethanol and the petrol scenarios, these trade-offs are not significant. However, the results clearly emphasize the need to actively assess and manage potential risks. In order to support this, the contribution of the major life cycle stages (primary production, processing, CCS and use phase) and associated uncertainties are discussed below. In line with the study's objective, optimization potentials are also identified.

4.1 | Feedstock production and supply

The impact contributions of feedstock production and supply, including land clearing and miscanthus cultivation, are relatively small. In addition, the analyses revealed that the endpoint results and related conclusions are consistent for the considered yields ranging between 19 and 25 t DM ha^{-1} (harvestable yield; 15–21 t DM ha^{-1}). Irrespective of the yield level, miscanthus cultivation has inherent land use impacts. These constitute a major driver of the impacts on ecosystem quality and are substantially higher than for petrol. This is a trade-off typical for land-based bioenergy production. It should be emphasized that the land use impacts are due to assumptions for potential species losses related to the cultivation of perennial (permanent) crops in Europe. However, biodiversity impacts vary strongly depending on context and location (Elshout et al., 2014). The actual situation in Croatia may not be ideally represented by the applied characterization factors as the area in question is currently experiencing a rapid spread of *Amorpha fruticosa* L., an invasive plant originating from North America (Krpan et al., 2014). Established miscanthus cultivations effectively suppress the growth of weeds. For this reason, large-scale cultivation of miscanthus is considered one option to reduce the proliferation of this invasive species and support local species diversity (Cossel et al., 2019).

As shown in Figure 8, the endpoint results and associated conclusions are strongly influenced by the assumed storage duration. The assumption of indefinite carbon storage reduces the total human health impacts of ethanol to below the level of petrol. Although an indefinite storage of the whole carbon amount is unlikely, this scenario emphasizes the need to ensure biological carbon sequestration over long periods. Regardless of the accounting approach, it should be noted that a relatively conservative approach was taken for the estimation of the biological carbon storage potential. Only carbon stored in the below-ground biomass was considered and further soil carbon dynamics were neglected. Approximately 1 t C ha^{-1} $year^{-1}$ was assumed to be stored, which is in the lower range of estimates for miscanthus cultivation. Soil carbon sequestration potentials between 0.7 and 2.2 t C ha^{-1} $year^{-1}$ have been previously reported for cultivation on arable land (McCalmont et al., 2017). However, the area intended for the miscanthus cultivation has not been in use for the last 20 years and has had a constant vegetation cover in this period. Thus, a strong soil carbon accumulation exceeding the carbon stored in the below-ground biomass seems questionable. Nevertheless, even without net soil carbon accumulation, ethanol achieves a substantial GHG reduction potential. The cultivation of the area could potentially even result in emissions following the initial soil preparation. Nevertheless, the utilization of this previously unused land has further implications. Bioenergy production is often related to the displacement of existing agricultural production systems. This displacement is referred to as indirect land use change (iLUC) and is associated with substantial environmental impacts (Schmidt, 2015). These effects and additional environmental impacts can be precluded under the present conditions as no replacement of agricultural production occurs.

4.2 | Biomass processing and supporting activities

Impacts from biomass processing and supporting activities are mostly related to the combustion of residues (biogas and lignin) and to the refinery operation itself. Impacts from the latter are mainly relevant for GWP and LU and are due to upstream activities and wastewater treatment. Almost a quarter of the GWP impacts are due to the use of chemicals required for the refinery operation. The contribution analysis (Figure 7) and the endpoint results (Figures 4 and 5) show that the combustion of residues and associated emissions are the major source of impacts on the damage level. This applies to the indicators *ecosystem quality* (via terrestrial acidification) and *human health* (via fine particulate matter formation). For human health in particular, it should be emphasized that the emissions related to the refinery operations will occur in an area of low population density (Sisak-Moslavina: 32.95 inhabitants km^{-2}; Croatian average: 72.03 inhabitants km^{-2}; Croatian Bureau of Statistics, 2020) and emissions from residue combustion will be released through a tall flue gas stack. In combination, this may reduce the actual human exposure and corresponding human health impacts. In addition, it should be noted that the assumed amounts of the corresponding emissions, mainly SO_2, NO_x and particulate matter, are based on the current European legal limits for gaseous emissions and can be considered worst-case scenarios. In practice, additional emission reduction could be achieved through technical solutions. SO_2 and NO_x concentrations in the flue gas could be further reduced by the use of supplementary lime and ammonia. Although these supplements may come at the expense of a few additional impacts, they could improve the overall environmental performance of the ethanol production (see Figures S3 and S4). The additional emission reduction could be tackled by the refinery operator through the management and process design of the plant. It needs to be highlighted that commercialization of lignocellulosic ethanol production has just begun and improvements in biorefining technologies can be expected (Field et al., 2020). In previous sustainability assessments of bioethanol projects, environmental impacts of residue combustion have often been overseen. This is probably due to the fact that GWP is the most commonly investigated impact category (Morales et al., 2015). The results of this study however underline the importance of considering related emissions and impacts in order to optimize the entire design in terms of environmental performance.

4.3 | CCS

The significance of the carbon capture and storage stage is underlined by the fact that it substantially reduces the climate change impacts and thus the associated damage to human health and ecosystem quality. For this reason, its feasibility—in technical and economic terms—needs to be considered. An annual refinery output of 55 kt ethanol corresponds to approximately 52 kt of CO_2 per year, which amounts to 1560 kt over an assumed refinery lifetime of 30 years. Clearly, this poses the question of whether CCS can be delivered at sufficient scale. The cavities provide adequate volume to store the annual CO_2 production for several hundred years (INA d.d., personal communication, March 2020). From an economic perspective, the question is what incentive there is to store CO_2 when it can easily be emitted into the air. In this specific case, the infrastructure for carbon storage is already in place and only a few additional expenses are required. A price of CO_2-emission certificates of approximately 25 € t^{-1} CO_2 would suffice to cover these supplementary costs (INA d.d., personal communication, March 2020). Following this line of argument, CO_2 storage could be considered a by-product of the ethanol production, which would require changes in the allocation approach. However, even assuming an economic allocation based on current prices of fuel ethanol and emission certificates, the ethanol production would achieve the European Union's reduction targets.

4.4 | Use phase

The use phase includes the contributors *car production and maintenance* as well as *road construction* and *exhaust and non-exhaust emissions*. Apart from land use, all relevant impact categories are substantially influenced by this life cycle stage. Major impacts derive from car production and maintenance, while exhaust and non-exhaust emissions are comparatively unimportant. The present study assumed a medium car size (1600 kg). During the previous decade, the average size of passenger cars in Europe has substantially increased (CCFA, 2020). The assumption of a larger car would marginally influence the results in all impact categories. However, it would not affect the comparison between ethanol and petrol, as the same impacts are assumed for the use phase (except for the biogenic and fossil origin of the exhaust emissions).

Overall, we conclude that the ethanol produced within this biorefinery project, which combines lignocellulosic ethanol production and CCS, can clearly achieve the European Union's GWP reduction target for liquid biofuels. In addition, ethanol can reduce the risks related to resource availability. In order to prevent potential trade-offs with respect to human health and ecosystem quality, it will be imperative to monitor and manage in particular the emissions from residue combustion, which are a significant driver of the overall environmental impacts.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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SUPPORTING INFORMATION

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

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Chapter 5

5.0 Perennial rhizomatous grasses: Can they really increase species richness and abundance in arable land? – A meta-analysis

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ORIGINAL RESEARCH

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Perennial rhizomatous grasses: Can they really increase species richness and abundance in arable land?—A meta-analysis

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Abstract

Perennial rhizomatous grasses (PRG), such as miscanthus and switchgrass, are considered promising lignocellulosic feedstocks. Their cultivation is expected to experience a significant increase in the near future, as it offers a wide range of benefits. For instance, when PRG replace typical annual crops, positive biodiversity impacts are usually anticipated. However, to date, there is no solid, statistically strong evidence for this hypothesis. This study aims to evaluate its validity through a meta-analysis based on an extensive systematic literature review of research comparing biodiversity attributes in PRG and common annual crops. Dynamics of species richness and abundance in response to PRG cultivation were quantitatively evaluated drawing on 220 paired comparisons from 25 studies. This includes data on five taxonomic groups—arthropods, birds, earthworms, mammals and plants—and three PRG—miscanthus, switchgrass and reed canary grass. The results indicate that biodiversity tends to be higher in PRG cultivations relative to the reference crops, but the initial hypothesis of significantly beneficial impacts could not be confirmed. Trends were specific to the individual taxonomic groups: significantly higher biodiversity was found for plants and small mammals. Positive but insignificant trends were observed for arthropods and birds, while earthworm response was neutral and insignificant. More substantial conclusions could not be drawn, which is mainly due to the low number of studies conducting biodiversity assessments in PRG cultivations that included a comparison with annual crops. In addition, a detailed analysis of the observed responses was impaired by poor reporting of the parameters influencing biodiversity in the studies reviewed, such as planting and crop density, as well as yields. For this reason, we conclude with a call for improved data reporting in biodiversity assessments of PRG cultivations and detail requirements for future biodiversity research.

KEYWORDS

abundance, biodiversity, meta-analysis, miscanthus, perennial biomass crop, reed canary grass, species richness, switchgrass

Jan Lask and Elena Magenau should be considered joint first author.

1 | INTRODUCTION

Perennial biomass crops are considered a promising resource to meet the growing demand for biomass in a developing global bioeconomy. Perennial rhizomatous grasses (PRG) such as miscanthus and switchgrass (*Panicum virgatum* L.) are receiving increasing attention from industry due to their versatile applications and high-yield potentials. The cultivation of these crops is expected to experience a significant increase in the near future, as they have numerous benefits. For instance, they can provide rewarding yields in a wide range of climatic and soil conditions, including marginal agricultural land (Clifton-Brown et al., 2017; Lewandowski et al., 2016). Their fertilizer and pesticide demand is low compared to annual crops due to efficient nutrient recycling and the absence of major pests (Kiesel, Wagner, & Lewandowski, 2017; van der Weijde et al., 2013). Previous research has demonstrated economic and ecological advantages of PRG cultivation (Kiesel et al., 2017; McCalmont et al., 2017; Wagner et al., 2018), in particular when integrated along field margins and on marginal lands (Ferrarini, Serra, Almagro, Trevisan, & Amaducci, 2017; Manning, Taylor, & Hanley, 2015). It is concluded that these crops could be produced sustainably without affecting global food supply and even decrease pressure on planetary boundaries such as climate change and other biogeochemical processes of the Earth system (Steffen et al., 2015).

The functioning of ecosystems and the provision of related services depends strongly on biodiversity and is endangered by species losses at local and wider scales (Gamfeldt & Roger, 2017). In general, intensive agricultural production is associated with negative effects on biodiversity (Ceballos et al., 2015; Flohre et al., 2011). Lower impacts are usually reported for lignocellulosic second-generation than for first-generation bioenergy crops. Research commonly indicates that PRG cultivation substantially improves agro-biodiversity at the field scale, if replacing typical annual crops such as maize and wheat (Dauber, Jones, & Stout, 2010; Dauber & Miyake, 2016; Immerzeel, Verweij, van der Hilst, & Faaij, 2014). Cultivation periods of up to 20 years which ensure extended soil rest, harvest in late winter or early spring as well as low input requirements are considered conducive to species richness and abundance, features commonly regarded as biodiversity attributes (Dauber et al., 2010). This assumption is usually based on studies which focus on single taxonomic groups (e.g. plants, mammals) and species (e.g. hares, butterflies; Haughton et al., 2016; Petrovan, Dixie, Yapp, & Wheeler, 2017; Semere & Slater, 2007a) and a small number of reviews which qualitatively examined effects of PRG cultivation on species richness and abundance (Dauber et al., 2010; Immerzeel et al., 2014). However, a few studies also indicate neutral (Bellamy et al., 2009; Clapham & Slater, 2008; Felten & Emmerling, 2011; Semere & Slater, 2007a; Stanley & Stout, 2013) or even negative effects on individual taxa (Briones, Elias, Grant, & McNamara, 2019; van der Hilst et al., 2012; Williams & Feest, 2019). The literature commonly suggests and expects positive biodiversity effects for the replacement of annual cropping systems with PRG cultivation. However, solid, statistically strong evidence for this is still lacking. The present study aims to evaluate the validity of this hypothesis through a meta-analysis of available data. The dynamics of species richness and abundance in response to PRG cultivation were quantitatively assessed as this provides an objective mean of testing the potential effects of PRG cultivation on biodiversity. This aids a better understanding of the biodiversity changes associated with a switch from classic arable crops to PRG cultivation and improves the interpretation of existing biodiversity assessments.

A meta-analysis was conducted based on an extensive systematic literature review of studies comparing biodiversity components in PRG and common annual crops. It drew on 220 paired comparisons from 25 publications analysing the response of species richness, abundance and diversity indices. This was done for five taxonomic groups—arthropods, birds, earthworms, mammals and plants—which have a predominant role in biodiversity assessments globally. Based on the assumptions from previous research, in particular the qualitative syntheses, we hypothesized significantly increased biodiversity for PRG cultivation when replacing annual arable crops.

2 | MATERIALS AND METHODS

2.1 | Data collection

Data were collected using the literature databases, Google Scholar (https://scholar.google.de/) and Scopus (https://www.scopus.com/). We identified potentially relevant journal articles, dissertations and master theses using 10 search terms, which combined keywords for common PRG crops with biodiversity key terms and five taxonomic groups. The search terms are given in Appendix 1. For the analysis, we selected only data from studies which were based on field experiments (not, e.g., pot experiments), compared PRG and annual arable crops in similar environments and investigated at least one of the biodiversity attributes 'species richness' (number of different species), 'abundance' (number of individuals) and 'diversity indices' (combination of species number and evenness of their abundance e.g. Shannon-Wiener and Simpson).

In total, 25 studies were selected from the initial set of 1,874 studies (2,259 prior to duplicate removal), which resulted from the search-term-based literature research. These are presented in Table 1. From the selected studies, we collected data on means, standard errors/deviation, and sample size for the biodiversity attributes species richness, abundance and diversity indices. If available, information on site and plantation characteristics was also considered and coded

TABLE 1 Studies included in meta-analysis

No.	Study	Arthropods	Birds	Earthworms	Mammals	Plants
1	Bellamy et al. (2009)	x	x	x		x
2	Berkley et al. (2018)	x				x
3	Blank et al. (2014)		x			
4	Bourke et al. (2014)	x				x
5	Bright et al. (2013)		x			
6	Briones et al. (2019)			x		
7	Chauvat, Perez, Hedde, and Lamy (2014)	x				
8	Clapham (2011) and Clapham and Slater (2008)		x		x	
9	Emmerling (2014)			x		
10	Feledyn-Szewczyk, Matyka, et al. (2019) and Feledyn-Szewczyk, Radzikowski, et al. (2019)			x		x
11	Felten and Emmerling (2011)			x		
12	Harrison and Berenbaum (2013)	x				
13	Hedde, van Oort, Renouf, Thénard, and Lamy (2013) and Hedde, van Oort, Boudon, Abonnel, and Lamy (2013)	x		x		
14	Helms, Ijelu, Wills, Landis, & Haddad (2020)	x				
15	Heyer, Deter, Eckstädt, and Reinicke (2018)	x				
16	Kaczmarek et al. (2018)		x			
17	Kempski (2013)	x				
18	Korpela, Hyvönen, Lindgren, and Kuussaari (2013)	x				
19	Sage et al. (2010)		x			
20	Schwer (2011)				x	
21	Stanley and Stout (2013)	x				x
22	Vepsäläinen (2010)		x			
23	Ward and Ward (2001)	x				
24	Werling et al. (2014)	x	x			x
25	Williams and Feest (2019)	x				
	Total per taxonomic group	14	8	6	2	6

as moderators to consider differences between studies. Data were taken from text and tables in the main manuscript or supplementary material. Additionally, values were extracted from figures using the GetData Graph Digitizer version 2.26 (http://getdata-graph-digitizer.com/).

2.2 | Data description

Overall, 25 studies published between 2001 and 2020 were considered. All of them assessed biodiversity attributes in Europe and the United States and focused mainly on the perennials miscanthus and switchgrass. While miscanthus was mostly studied in Europe, switchgrass was the predominant research object in the United States. Reed canary grass featured in only three studies. Maize and wheat were the annual crops mainly used as reference, irrespective of the location.

The studies assessed five taxonomic groups—arthropods, birds, earthworms, mammals and plants. Comparisons of arthropod abundance in PRG and annual arable crops were contained in 14 of the studies, making this the most widely investigated taxonomic group under consideration. More than half of the selected studies were published in 2013 and 2014.

The PRG cultivation data consisted mainly of miscanthus (75%) and switchgrass (22%), with the remaining data relating to reed canary grass. Substantial differences in collection approaches were found between studies. Two groups were distinguished: First, the collection of non-ground-dwelling arthropods, which were trapped by sweep net sampling, pan traps, bucket traps and sticky cards. Second, the collection of ground-dwelling arthropods, trapped by pitfall traps and soil cores.

Abundance and richness of birds was reported in eight studies published between 2006 and 2015. Four studies alone were conducted in the United Kingdom. The remaining include one from Finland and Poland and two from the United States. Due to the predominantly European focus, most of the comparisons included had miscanthus (69%) as PRG. Similarly, earthworm biodiversity was assessed in six studies from Germany, France, the United Kingdom and Poland. Despite all being European, these studies, which were published between 2009 and 2019, included data on both miscanthus and switchgrass. Small mammal populations in PRG and annual crops were compared in only two studies, one from the United Kingdom comparing miscanthus and reed canary grass (Clapham, 2011) and the other from the United States focusing on switchgrass (Schwer, 2011). Data on plant species richness and abundance were reported in six studies, mainly with miscanthus as PRG. These studies were published between 2009 and 2019 and were located in Ireland, Poland, the United Kingdom and the United States.

2.3 | Data analysis

The effect of PRG cultivation on the biodiversity attributes 'species richness', 'abundance' and 'diversity indices' was quantitatively evaluated in accordance with Fletcher et al. (2011) and Hedges, Gurevitch, and Curtis (1999). We calculated response ratios (RR) for each comparison pair using Equation (1):

$$\mathrm{RR} = \ln\left(\frac{\bar{x}_{\mathrm{PRG}}}{\bar{x}_{\mathrm{ara}}}\right), \tag{1}$$

where $\bar{x}_{\mathrm{PRG}}$ and $\bar{x}_{\mathrm{ara}}$ denote means for each biodiversity attribute for PRG and annual arable crops respectively. The five taxonomic, as well as different sampling methods and years within a study, were each treated as separate comparison pairs.

Not all studies indicated standard errors and deviations. For this reason, the variance and weighting factors of the study-specific RRs were based on the number of locations (Hamman, Pappalardo, Bence, Peacor, & Osenberg, 2018; Núñez-Regueiro, Siddiqui, & Fletcher, 2019). The weighting factor W was calculated by Equation (2):

$$W = \left[\frac{(N_{\mathrm{PRG}} * N_{\mathrm{ara}})}{(N_{\mathrm{PRG}} + N_{\mathrm{ara}})}\right], \tag{2}$$

where N_{PRG} and N_{ara} are the number of locations with PRG cultivation and annual arable crops respectively (Núñez-Regueiro et al., 2019). The mean weighted response ratio (RR_{++}) was calculated from the RRs of individual pairwise comparisons between PRG and the reference, as given in Equation (3):

$$\mathrm{RR}_{++} = \frac{\int_{i=1}^{m}\int_{j=1}^{k} W_{ij}\mathrm{RR}_{ij}}{\int_{i=1}^{m}\int_{j=1}^{k} W_{ij}} \tag{3}$$

The standard error of RR_{++} was estimated according to Equation (4):

$$SE\left(\mathrm{RR}_{++}\right) = \sqrt{\frac{1}{\int_{i=1}^{m}\int_{j=1}^{k} W_{ij}}}. \tag{4}$$

The meta-analysis was conducted on two levels. First, for all data and second, separately for each taxonomic group. A multilevel random-effects model was fitted to account for the nonindependence of effect sizes due to the nested data structure (Bender, Contreras, & Fahrig, 1998; Konstantopoulos, 2011; Viechtbauer, 2010). The random-effects model assumes that studies are using distinct research methods and differ in their characteristics of response. Z-tests with a significance level of $p \le .05$ were conducted to test the significance of the differences between PRG and annual crops. Heterogeneity of variance was analysed with the I^2 statistic, which describes the deviation between study results (Higgins & Thompson, 2002). We then tested the effect of different moderators including year, country, PRG type, annual arable crop type, and age group of PRG. Data were analysed by the metafor package (Viechtbauer, 2010) in the program R (R Core Team, 2019). The resulting data were displayed using the R package ggplot2 (Wickham, 2016).

3 | RESULTS

The following section presents the results for the responses of the biodiversity attributes species richness, abundance and diversity indices on PRG production. A positive response was found for the pooled taxonomic groups ($\mathrm{RR}_{++} = 0.31$; $SE = 0.18$; $p = .08$), indicating beneficial biodiversity impacts of PRG in comparison with annual arable crops (Figure 2). The response strength varied significantly with the type of biodiversity attributes. A significant response ($\mathrm{RR}_{++} = 0.40$; $SE = 0.20$; $p = .05$) was observed for abundance while richness and diversity indices showed positive, but nonsignificant, trends (Figure 1). Tendencies and significant responses are presented per taxonomic group and biodiversity attribute below.

For arthropods, the analysis presented a clear trend (Figure 1). Species richness, abundance and diversity indices showed higher figures in PRG than in annual arable crops. Although figures for non-ground-dwelling arthropods were lower than for ground-dwelling arthropods, RR were still positive, but differences between the groups were insignificant. For this reason, the results for arthropods are presented as a single value in Figure 1.

No significant responses were observed for birds. RR for abundance and diversity indices were close to zero (Figure 1), indicating similar biodiversity figures for PRG and annual crop cultivation. Only the RR of species richness was slightly higher than zero, but still insignificant. Earthworm abundance and species richness showed no significant difference between PRG cultivation and annual crops. The RR of species richness was close to zero across all studies. Abundance, including earthworm biomass and number of individuals, was slightly below zero with a decrease in biomass and an increase in number of individuals. Due to lack of data, no response was calculated for diversity indices. All RR for small mammals and plants, except plants indices, were consistently higher for PRG cultivation than for annual crops.

Data were heterogeneous for arthropods abundance and plant indices. When data of all taxonomic groups were pooled, positive RR_{++} were observed for the individual PRG. Only for switchgrass a significant positive response was observed. In contrast, reference crop type (e.g. wheat, maize) had no effect. For plant diversity indices, only three comparison pairs were analysed. The age of the PRG cultivation

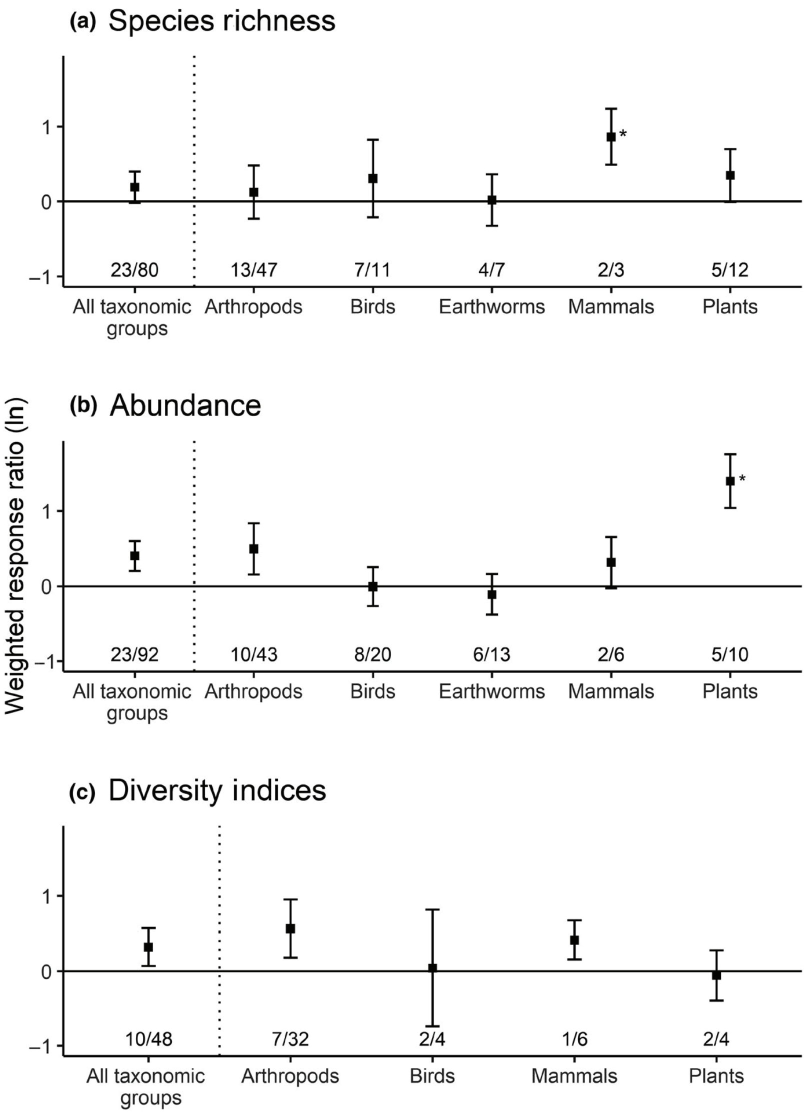

FIGURE 1 Weighted response ratio of biodiversity attributes: (a) species richness, (b) abundance and (c) diversity indices for the comparison of perennial rhizomatous grasses (PRG) and annual arable crops. A response ratio above zero indicates a positive response to PRG cultivation. Bars indicate standard errors, *statistically significant ($p \leq .05$) response. Number of studies and paired comparisons considered given below

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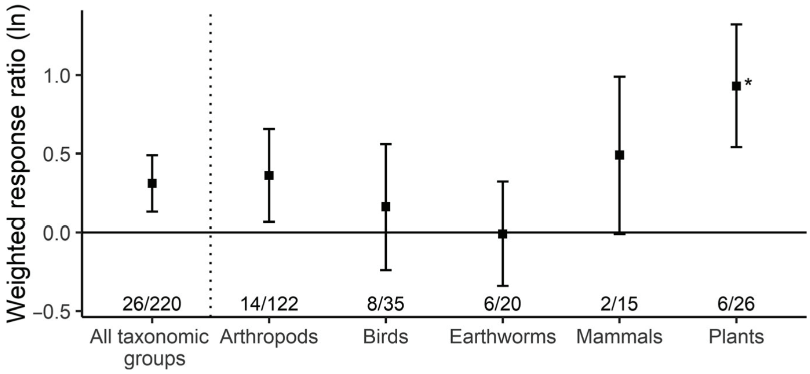

FIGURE 2 Cumulated response of biodiversity impacts to cultivation of perennial rhizomatous grasses (PRG). A response ratio above zero indicates a positive response associated with PRG cultivation. Bars indicate standard errors, *statistically significant ($p \leq .05$) response. Number of studies and paired comparisons considered given below

under consideration also influenced the RR, significant higher biodiversity was found for cultivation ages between 3 and 6 years. For older stands, results indicated negative, but insignificant impacts. In addition, RR varied significantly with the taxonomic group, as shown in Figure 1.

Overall, RR varied slightly between the three biodiversity attributes. Abundance of PRG crops showed a positive trend compared to annual crops, while the response patterns of species richness and diversity indices were less clear. Differences between the biodiversity attributes were however insignificant except for small mammals and plants. With the attribute data pooled, significantly positive responses were observed for plants (Figure 2).

4 | DISCUSSION

This study collates quantitative data from a range of publications reporting on the comparison of biodiversity in PRG cultivation and common annual arable crops. It was hypothesized that PRG cultivation promotes a higher level of biodiversity, which can be quantified through the attributes 'species richness', 'abundance' and their combination in 'diversity indices'.

This initial hypothesis could not be confirmed. However, the results of the meta-analysis indicated that biodiversity tends to be higher (without statistical significance) in PRG cultivations relative to the reference situation. These trends are in line with results from previous research qualitatively assessing biodiversity impacts of PRG cultivation (Dauber et al., 2015; Immerzeel et al., 2014). The strength of the trends also varies between the taxonomic groups (partially significantly). For instance, abundance and species richness in small mammals and plants clearly benefitted from PRG cultivation, while earthworm biodiversity attributes showed no or even negative effects. Although the detected trends were consistent across the taxonomic groups, only a few of them were significant. Effects on species richness and abundance did not differ significantly between the considered PRGs (mainly miscanthus and switchgrass).

Unfortunately, it was not possible to draw more substantial conclusions due to the generally low number of studies conducting biodiversity assessments in PRG cultivations. In addition, such studies often do not provide information on the biodiversity status in annual crops which could be used as a reference (e.g. Robertson, Landis, Sillett, Loomis, & Rice, 2013; Semere & Slater, 2007b). Due to the site-specific nature of biodiversity, this information is however imperative to be able to assess and compare the actual impact of PRG cultivation. For this reason, 42 studies were rejected and only 25 studies were finally found eligible in accordance with the selection criteria.

In addition to site-specific aspects, biodiversity attributes are influenced by numerous factors related to the crop and its management (e.g. plant age, planting density, etc.). It was not possible to assess the impact of these factors on the response of the biodiversity attributes to PRG cultivation on arable land using the selected studies. Essential information for response interpretation is often absent or given in non-standardized form. This is a general concern in the biodiversity assessment of agricultural systems and has been previously criticized (Brown & Matthews, 2016; Gotelli & Colwell, 2001). The following section presents factors that can potentially influence biodiversity attributes in PRG cultivation but are not systematically reported in assessments, thus impeding a thorough analysis of studies on biodiversity in PRGs. We have classified these into three categories:

1. biomass yield, crop density and phenotype
2. landscape context
3. temporal issues.

The first category is related to information on biomass yield, which predominantly depends on climate and soil conditions but also on factors including planting density, crop establishment status, plant age and genotypic variation. As has been previously shown, these attributes strongly influence biodiversity potential in second-generation biomass crops and PRGs in particular (Dauber et al., 2015; Núñez-Regueiro et al., 2019). Biomass productivity is directly related to crop/canopy cover and the associated light interception. These factors are, however, negatively correlated with the abundance and richness of plant species in PRG cultivations (Bekewe, Castillo, & Rivera, 2019). As the canopy/crop cover increases over the years after establishment, plant species richness and abundance usually also decrease (Holguin et al., 2010). This highlights the importance of considering the entire life cycle of PRG cultivation when assessing species richness and abundance of plants. The majority of studies included in our assessment evaluated established PRG cultivations, potentially resulting in an underestimation of the benefits of PRG cultivation for plant biodiversity. In addition, planting density and crop establishment status should be assessed in order to enable comparisons of plant biodiversity in PRG cultivation. This is of particular importance, as noncrop vegetation in plantations can indirectly affect other organisms by serving as a food source and/or habitat. For instance, arthropod biodiversity is commonly interrelated with plant abundance and species richness. It has been found that species richness and abundance of ground beetles, butterflies and spiders are negatively correlated with yields and reduction of the noncrop vegetation (Dauber et al., 2015; Semere & Slater, 2007a). This emphasizes the importance of reporting data on PRG cultivation status, including phenotype and genotype (crop density/canopy cover) and could explain variation in values given in studies on arthropods in PRG cultivation, at least to a certain extent. In addition, it should be emphasized that most of the approaches for the quantification of biodiversity rely purely on species richness and abundance, while aspects such as rarity and endangerment are rarely considered.

Similar to arthropod and plant biodiversity, bird abundance appears to be related to the PRG phenotype, in particular plant height. It has been reported that, due to the provision of shelter and nesting sites, birds benefit from PRG cultivation in the first years after establishment in intensive farmland (Bellamy et al., 2009). However, for switchgrass, it has also been reported that bird abundance reaches a maximum at a crop height of 0.5–0.6 m (and a biomass yield of 3–4 t/ha, Blank, Sample, Williams, & Turner, 2014) and then decreases with increasing crop height. A negative correlation between bird abundance and miscanthus crop height was also reported by Bellamy et al. (2009). In contrast, Bright et al. (2013) did not find a significant correlation. Bird species richness is also affected by the PRG cultivation status. Typical field species such as corn bunting, skylark and starling generally prefer younger, poorly established PRG cultivations and avoid older, dense and well-established plantations. The latter are however, usually a preferred habitat for woodland species (Bellamy et al., 2009; Clapham, 2011; Kaczmarek, Mizera, & Tryjanowski, 2018). In the United Kingdom, PRG are preferred by woodland species (Bellamy et al., 2009; Clapham, 2011). However, in summer, more farmland bird were identified in PRG than in annuals crops by Bellamy et al. (2009) and in Poland, farmland species dominated in PRG (Kaczmarek et al., 2018). These aspects are often not addressed in enough detail in biodiversity assessments, and this can result in a change in species composition being overseen. Small mammals constitute the only group which clearly profit from denser biomass stands. It has been previously reported that vegetative cover is an important characteristic of habitat quality for small mammals. This is mainly due to its functions of predator protection and provision of nesting opportunities (Clapham and Slater, 2008).

When summarizing these first aspects, it should be emphasized that biodiversity assessments of PRG cultivations require more detailed information on the specific PRG setup in order to give clear indications. While the age of plants is reported in most studies, crop establishment success and crop/canopy density as well as yields are only rarely reported, despite their importance in evaluating the biodiversity attributes measured.

The second category of factors is mainly determined by aspects relating to the surrounding environment and the integration of PRG cultivation into the landscape. Biodiversity potentials vary depending on landscape, and this is also one reason why not all the five taxonomic groups assessed respond in a consistent way across locations and studies. For instance, it has been previously shown that the probability of observing grassland bird species declines with an increasing share of forest land cover (Robertson, Doran, Loomis, Robertson, & Schemske, 2011; Werling et al., 2014). In addition, the way in which PRG cultivation is integrated into the landscape affects habitat quality. Field size is an important parameter influencing biodiversity attributes in PRG cultivation. For example, the number of grassland birds has been found to be negatively correlated with field size, as dense PRG monocultures do not constitute a suitable habitat (Norment, Ardizzone, & Hartman, 1999). Similarly, miscanthus is not a food source for small mammals and these cannot thrive in areas densely planted with miscanthus. However, as a well-dosed complement to an agricultural landscape, miscanthus cultivation may provide biodiversity benefits by increasing refuge areas for mammals such as brown hares (Petrovan et al., 2017). Switchgrass seeds in comparison could also provide a food source for small mammals (Briones, Homyack, Miller, & Kalcounis-Rueppell, 2013). The integration of PRG as landscape elements, for example, the cultivation along field margins, could provide habitat and forage for birds and small mammals, resulting in high species richness in field edges (Clapham, 2011). These beyond-field impacts are commonly overseen in typical biodiversity assessments of PRG cultivation. The typical focus on species number often results in neglect of habitat specialists and endangered species in biodiversity evaluation. Taken together, this puts the focus on the concept of landscape moderation with the major goal of increasing crop heterogeneity in agricultural landscapes (Landis, 2017; Sirami et al., 2019; Tscharntke et al., 2012).

In addition to the two categories mentioned above, the influence of temporal issues, for example, seasonality, is commonly neglected in biodiversity reporting. This is despite the fact that evidence for seasonal changes has been observed in assessments on birds. Seen over the year, bird abundance is higher in poorly established than in well-established stands. However, well-established stands reveal higher bird abundance during the winter (Gardiner et al., 2010). Harvest dates can also be responsible for variation in biodiversity impact assessments. Miscanthus can be harvested in autumn or spring. The difference in harvest date has a direct influence on biodiversity, since an autumn harvest completely removes the winter cover for small mammals and birds. An early harvest can also result in a reduction of organic substance recycling and a reduced soil carbon input. It has been hypothesized that performing an autumn harvest over several consecutive years reduces both abundance and biomass of earthworm communities in miscanthus in comparison to a winter/spring harvest (Ruf & Emmerling, 2017).

The previous paragraphs outlined adjustments and further recording requirements for future biodiversity assessments of PRG cultivation. In addition, it should be emphasized that other relevant taxonomic groups are so far underrepresented in PRG biodiversity research. Our work provides a quantitative overview of potential PRG biodiversity impacts. We conclude that biodiversity can, in general, benefit from the replacement of annual crops by PRG, but this could not be proven statistically, due to data gaps in the PRG biodiversity impact assessments. These gaps include the neglect of entire taxonomic groups such as amphibians, but also the fact that management practices and plant-related data are only rarely reported. It should also be noted that biodiversity impacts of PRG cultivation and associated ecosystem services are dependent on the location relative to other habitats. We conclude that, in order to exploit the full potential of biodiversity assessments in PRG cultivation, these need to include a wider range of parameters.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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SUPPORTING INFORMATION

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

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APPENDIX 1

Search strings

1. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (biodiversity OR “species diversity” OR “species abundance” OR “species richness”)
2. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (invertebrate OR vertebrate OR arthropods)
3. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (Bird OR skylark OR “meadow pipit” OR “lap wing” OR aves)
4. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (Insect* OR Pollinat* OR Coleoptera OR Beetle OR Carabidae OR Chrysomelida OR Syrphidae OR Hoverflies OR Diptera OR Lepidoptera OR Butterflies)
5. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (Hymenoptera OR Bee OR Apoidea OR Hemiptera OR Thysanoptera OR Dermaptera OR Neuroptera OR Psocoptera OR Orthoptera)
6. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (Spider OR Araneida OR Arachnida)
7. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (Phytodiversity OR “plant diversity” OR weed OR “segetal flora”)
   a. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (Phytodiversity OR weed OR “segetal flora”)
   b. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (“plant diversity” OR weed OR “segetal flora”)
   c. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (Phytodiversity OR “plant diversity” OR “segetal flora”)
   d. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (“segetal flora”)
8. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (“Soil biodiversity” OR “soil diversity” OR Lumbricidae OR earthworm OR “Soil organism” OR “soil microbiology” OR bacteria OR Archaea)
9. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (Mammal* OR “Microtus” OR vole OR Rat OR Rattus OR “Micromys” OR mouse OR Lepus OR Hare)
   a. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR phalaris OR “Arundo donax” OR (“Bioenergy crop” AND perennial)) AND (mammal* OR “Microtus” OR vole OR rat OR rattus OR “Micromys” OR mouse OR lepus OR hare))
   b. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR phalaris OR “Arundo donax” OR (“Biomass crop” AND perennial)) AND (mammal* OR “Microtus” OR vole OR rat OR rattus OR “Micromys” OR mouse OR lepus OR hare))
10. (miscanthus OR switchgrass OR “Panicum virgatum” OR “Reed canary grass” OR Phalaris OR “Arundo donax”) AND (Amphibia OR Lissamphibia OR Mollusc* OR Gastropoda OR snail)

Chapter 6

6.0 General Discussion

In the following, the results presented in Chapters 2 to 5 are discussed with regard to the thesis' overall aim of simplifying the conducting and improving the comprehensiveness of LCA of perennial cultivation systems. The structure of the discussion aligns with the three research questions raised in the general introduction.

Research question 1 focuses on approaches for simplifying the conduct of LCAs of perennial cultivation systems. The life cycle inventory (LCI) is the core of an LCA. Reducing its complexity is a valid option for simplified LCAs of all types of product systems. In Chapter 3, a corresponding approach, including a variance-based global sensitivity analysis and the proposal of a simplified model for the calculation of greenhouse gas emission of miscanthus cultivation has been suggested. In Chapter 6.1, this approach and possible further applications are reflected upon.

The handling of carbon sequestration and storage in LCAs of perennial cultivation systems is the key aspect of **research question 2**. The inclusion of carbon sequestration is highly beneficial for the carbon footprint of perennial cultivation systems as evidenced in Chapter 3 and 4. However, LCA practitioners encounter several challenges when aiming for its inclusion One challenge is the quantification of carbon stored due to the perennial cultivation system and the other is the permanence of the carbon storage. The treatment of these critical aspects is discussed in Chapter 6.2 and a recommendation for LCA practitioners in the field of perennial cultivation systems is elaborated.

Research question 3 addresses the incorporation of land use impacts of perennial cultivation systems on biodiversity. In Chapter 6.3 current approaches for biodiversity assessments in LCA are discussed drawing on insights from Chapter 4 and 5. In Chapter 4, biodiversity land use impacts were included in an LCA on biofuels using an operational approach that distinguishes species richness in annual and perennial cultivation systems. The approach used in this study is critically reviewed considering the species richness information derived in Chapter 5. Chapter 6.3 concludes with a discussion on the meaningfulness of the use of species richness as a biodiversity indicator within the LCA framework.

6.1 Reducing complexity in the conducting and use of LCAs of perennial cultivation systems

According to the ISO standard 14040 (ISO, 2006b), the life cycle inventory (LCI) phase includes the collection and quantification of in- and outputs required in a product's life cycle or the life cycle stages under investigation. Essentially, it details the flows that enter and leave the system, in order to ensure the fulfilment of the product system's function. The collection of all this information can become complex, especially given the comprehensiveness and level of detail LCA is aiming for.

Inventories of agricultural production systems are characterised by a high variability due to site- and management-specific variation. This variability impedes the widespread use of LCA for agricultural systems. However, the wider use of LCA for these value chains could be facilitated by a reduction in complexity of the LCI phase. Usually, only a few inventory flows dominate the variation in the overall impact results (Saltelli et al., 2007b). As shown in Chapter 2 (Lask, Kam, et al., 2021), carbon sequestration is the most critical parameter in the greenhouse gas assessment of perennial cultivation systems, followed by biomass yield, duration of the cultivation period, fertiliser quantity applied and distance to customer. These parameters are the major drivers of result variability and are thus referred to as *key parameters*. Their identification and use in simplified models is critical for reducing complexity in LCA of perennial cultivation systems.

The following section is divided into two sub-sections, each dealing with a certain aspect related to the reduction of complexity in life cycle inventories. Section 6.1.1 discusses methodological approaches and procedures for identifying key parameters. Section 6.1.2 suggests how key parameters can feed simplified LCA models and how such models can be applied to enhance the use of LCA by, amongst others, farmers and small and medium-sized enterprises (SME).

6.1.1 Identifying key parameters in LCI of perennial cultivation systems

From a methodological stance, the identification of key parameters is similar to the analysis of parameter uncertainty in LCA. This is due to the fact, that both approaches aim to assess, where a small change in a LCI parameter value results in a substantial deviation in the overall impact results. Correspondingly, approaches for sensitivity analysis can be undertaken for key parameter identification and parameter uncertainty analysis.

A range of methods is available to perform sensitivity analysis in the LCA framework. In practice, the one-at-a-time approach is the most widely used one (e.g., as applied in Chapter 4), thanks to its simplicity for practitioners. The fundamental idea is to change one input parameter of the model while keeping all the others constant and then repeating the procedure for parameters of interest. For this reason, it is also referred to as local sensitivity analysis. Depending on the number and variability of the parameters, this can require considerable effort and might not allow exploring the entire range of possible parameter values. These shortcomings can be overcome by a global sensitivity analysis which is increasingly used in LCA practice (e.g., in Wolf et al. (2017) and Groen, Bokkers, Heijungs, and Boer (2017)). The Sobol or variance-based sensitivity method is the most dominant in this respect. It quantifies the variance in the result caused by a parameter and thus, the parameter's individual importance. In contrast to a local sensitivity analysis, the Sobol method accounts for the effect of all parameters included at the same time and helps to assess the influence of a collection of parameters. This allows the LCA practitioner to gain a more holistic understanding of the model and its defining parameters, which can support a more holistic understanding of the investigated product system (Groen, 2016; Saltelli et al., 2007a). Due to its probabilistic nature, it requires fully quantified parameters (i.e. characterised by probability distribution functions) and typically involves Monte Carlo methods for uncertainty propagation. Parameter identification and quantification are critical steps in a variance-based sensitivity analysis. In the following, both steps are briefly introduced and critical aspects in particular with respect to perennial crops will be reflected upon.

Parameter identification

In the first step of *parameter identification*, parameters that explain and characterise the life cycle inventory are determined. There are mainly two critical aspects in this procedure that require attention: first, the (in-)comprehensiveness of the parameter selection and second, correlations between parameters (Saltelli et al., 2007b).

The first one is the comprehensiveness of the parameter selection that has to be reflected upon. Despite LCA's objective of being a holistic approach that incorporates all impacts associated with a product system comprehensively, a life cycle inventory is a simplification of reality. It reflects and is based on a deliberate selection of aspects and structures representing the world. The simplified miscanthus cultivation model suggested in Chapter 2 for instance, contains information on management parameters and related flows of miscanthus cultivation. However, it does not include indirect land use effects, which, due to their considerable impacts could strongly influence the results. This could be evidenced in the assessment comparing wild plant mixtures and maize cultivation for biogas production in Chapter 3 (Lask, Martínez Guajardo, et al., 2020), where it became a substantial determinant after inclusion. The non-consideration in the development of the simplified model is due to the attributional approach taken in the miscanthus assessment in Chapter 2. In product-based LCAs, the attributional modelling is the standard approach, which is also in line with international standards such as the ILCD handbook (ILCD, 2010) or the environmental product declaration's product category rules (EPD International AB, 2020). An alignment with these is of particular relevance when an LCA model is developed for SME or farmers who want to communicate the results. In addition, the inclusion of indirect land use changes is highly controversial amongst LCA practitioners due to a number of interrelated reasons: First, it is questioned, if indirect effects, that are not physically connected to a product system should be included in LCAs at all (Finkbeiner, 2014a). This is a question of how the system is delimited (per se, a normative choice) and is essentially defined by the (social) responsibility paradigm assumed. Weidema, Pizzol, Schmidt, and Thoma (2018) distinguish three paradigms: 1) value chain, 2) supply chain (both being attributional) as well as 3) consequential responsibility. They argue in favour of the consequential paradigm as it ensures the inclusion of all consequences associated with an action, while the value and supply chain paradigms tend to result in an exclusion of consequences. Although critical to further the mainstreaming of LCA, a consensus on this topic has not been reached within the LCA community.

Second, even if LCA experts agreed upon the inclusion of impacts from indirect land use change, the quantification of related environmental impacts remains challenging. The impact can vary by several orders of magnitude, which introduces substantial uncertainty to the assessment (Finkbeiner, 2014a). In particular, in land use-intense product systems such as agricultural production, this variability has the potential to strongly affect the conclusions of the assessment. Correspondingly, indirect land use changes and related impacts would become a key parameter in the assessment of miscanthus cultivation, if included. This example highlights the role of reflecting on the in- and exclusion of inventory parameters for the outcome of key parameter identification and the development of simplified models. For this reason, it is concluded that the in- and exclusion has to be justified and needs to be in line with the goal and scope of the study. In Chapter 4 for instance, iLUC is not relevant as the studied miscanthus cultivation occurs on land not used for agricultural production. This might not necessarily apply to other circumstances.

The second critical aspect regarding parameter identification is the recognition of dependencies between parameters. This is essential as variance-based sensitivity analyses, by default, assume parameters to be independent, which means that they can be sampled from the corresponding distribution without consideration of the sampling of others (Saltelli et al., 2007b). However, in agricultural systems correlations can occur. A typical example is the correlation between nitrogen fertiliser application and biomass yields. Commonly and only within a defined range, a higher biomass yield is expected with higher nitrogen application rates (Lassaletta, Billen, Grizzetti, Anglade, & Garnier, 2014). The neglect of such correlations may result in incorrect conclusions with respect to the relevance of individual parameters (Groen & Heijungs, 2017). Nonetheless, nitrogen and yield correlations are only rarely considered in LCA practice (e.g., in Wagner, Kamp, Graeff-Hönninger, & Lewandowski (2019)). The parameters are generally treated as independent from each other. This approach was also taken for developing the simplified model in Chapter 2. This is because the correlation is not clearly established for miscanthus, as contradicting effects of nitrogen application on biomass yields have been reported (McCalmont et al., 2017). Nevertheless, future efforts and updates of the simplified model should focus much stronger on the treatment of such dependencies and work on the incorporation of correlations (if applicable). As a first step in this direction, it is recommended to evaluate the relevance of potential correlations (e.g., between nitrogen fertilisation and yield). This could be achieved by testing the effect of the inclusion of potential correlations on the result's variance, using the analytical approach suggested by (Groen & Heijungs, 2017). In a second step, relevant correlations could be incorporated into the model using covariance matrices Groen & Heijungs (2017). This would require clearly established correlations between the parameters, which, are not yet available for miscanthus yields and nitrogen fertilisation.

Parameter quantification

The quantification of parameters needs the assignment of probability distributions to each identified parameter in order to describe the range of values a parameter could potentially take in reality. Probability distributions can be characterised by a range of possibility functions, each indicating a certain tendency described by mean or median and a dispersion value (e.g. minimum and maxima, standard deviation) (Rosenbaum, Georgiadis, & Fantke, 2018). Examples include linear, (log-)normal, triangular and others. In LCA practice, the selection is usually limited to a certain set implemented in the used LCA software. This is also one of the reasons, why the assessment of key parameters in Chapter 2 was solely based on normal and triangular distributions. Although included in the software, linear distributions were disregarded as they did not fit the intended distribution of the identified parameters. Naturally, the selection of a distribution is a deliberate choice, which has to be critically reviewed since it can potentially exert considerable influence on the outcome of the assessment. A fitting probability function could be approximated using a representative sample set of parameters based on literature or expert estimates (Huijbregts, Gilijamse, Ragas, & Reijnders, 2003). It is recommended for LCA practitioners to select the source in line with the goal and scope of the study. In Chapter 2, values were mainly derived from expert estimates, i.e. miscanthus cultivators. This choice reflected the scope of the study to represent commercial conditions, which are not well represented in scientific literature. An example in this respect is the strong discrepancy between nitrogen fertilisation rates as reported by experts from practice and the ones reported in field trial-based scientific publications. In situations where neither small data sets from literature nor expert estimates are available, the pedigree approach is increasingly used in LCA (e.g. within the ecoinvent database (Wernet et al., 2016)). It enables the estimation of probability functions based on data quality indicators such as representativeness and age (Frischknecht et al., 2005). However, its' compatibility with Monte Carlo-based uncertainty propagation has been recently questioned (Heijungs, 2020) and should thus be avoided for variance-based global sensitivity analyses.

6.1.2 Using key parameters in simplified models – potentials and limitations

Variance-based sensitivity analyses provide information on each parameter's contribution to the overall result variation given as first-order derivatives (Sobol indices). These help to identify 1) those parameters that can be fixed at a value within their range of variation without affecting the output variance and 2) those parameters crucial for setting up a simplified model. The selection of parameters included in a simplified model depends on the modeller's choice on the ratio of output variability to be explained by the model (Saltelli et al., 2007b).

When developing a simplified agricultural model dedicated for the usage by farmers or SME, the present study recommends focusing, as far as possible, on parameters for which information is easily accessible. This includes parameters like as biomass yield and management parameters such as fertiliser quantities. Clearly, this is not always possible. For instance, carbon sequestration is an important parameter in the case of perennial cultivation systems but is more challenging to quantify due to site-, crop- and methodological issues (Ledo et al., 2018). The treatment of this parameter in simplified and mainstreamed assessment is critical and further elaborated in Section 6.2.

The generic nature of a simplified model has to be distinctly emphasised and kept in mind when interpreting results. A generic scope will always result in inaccuracies. This is evidenced in the sensitivity analysis in Chapter 2, which emphasises the importance of the selection of the fertiliser type. It is apparent that simplified models, although suitable for deriving estimates and screening activities, cannot replace specific assessments. Thus, it is recommended that developers of simplified models transparently communicate a model's limitations to its users. In addition, developers are encouraged to critically reflect about in- and exclusion of model parameters as well as about potential dependencies between parameters.

Nevertheless, it is indisputable that simplified models can reduce the effort associated with conducting and applying LCAs (Beemsterboer, Baumann, & Wallbaum, 2020). Practitioners such as farmers and SME might benefit strongly from the simplified access to LCA models and results. They can apply these in two ways: First, it allows them to easily calculate customised LCA results, which will be of increasing relevance given the growing importance of LCA tools such as the European Commission’s product environmental footprint. In future, simplified models of agricultural systems can be coupled with existing mandatory field records. This will allow farmers to derive and communicate verified information on environmental impacts associated with the biomass production without additional effort. Second, simplified models can be helpful to get to know leverage points for environmental optimisation. As previously shown, the duration of the cultivation period is one of the key determinants, influencing the greenhouse gas emissions associated with miscanthus cultivation. This information could rise awareness among farmers to grow perennial crops for longer periods, but at least for 15 years, to optimise the impacts per kg DM. This example shows, how simplified models could broadcast LCA know-how and indicate leverage points for environmental optimisation. For both application options, it will be critical that LCA practitioners and researchers take the lead in setting up the corresponding simplified model.

LCA practitioners and researchers can also benefit from the use of simplified models and, in particular, global sensitivity analysis. The identification of relevant parameters improves the understanding of LCA results and supports verifying the validity of derived conclusions. In addition, it can simplify the data collection process as it helps to prioritise focus and efforts on a few critical model parameters. Moreover, the use of simplified models could be extended in terms of the impact categories. Although Chapter 2 performed the assessment of key parameters only for global warming potential, further impact categories can be easily taken into consideration, if the corresponding inventory data is available. Unfortunately, global sensitivity analysis is not yet a common practice in LCA, as typical LCA software do not offer the corresponding functionalities. The implementation of global sensitivity techniques (as in Brightway2 (Mutel, 2017)) could further advance their use and support LCA experts in creating simplified models, which in turn will help advance the use of LCA information by practitioners.

6.2 Treating carbon sequestration and storage in LCAs for perennial cultivation systems

Carbon sequestration and storage refers to the withdrawal of carbon as carbon dioxide from the atmosphere and its storage for a given time in different stocks, including biomass, soil or even geological storages. Perennial crops can contribute to carbon sequestration and storage in two ways: First, substantial amounts of carbon can be stored in the plant biomass (mainly rhizomes and roots). Second, considerable amounts of carbon can be added to the soil via root exudates and the decomposition of root and above-ground litter (Ledo et al., 2020). In addition, the absence of soil disturbances through annual tillage operations stabilises soil carbon, thus reducing associated emissions (Ledo et al., 2018).

Due to its importance in terms of greenhouse gas mitigation (as seen in Chapter 3 (Lask, Martínez Guajardo, et al., 2020) and 4 (Lask, Rukavina, et al., 2021)), carbon sequestration and storage associated with the cultivation of perennial crops is a major focus in sustainability assessments of perennial cultivation systems. Irrespective of the perennial crop under investigation, two major concerns arise when dealing with carbon sequestration and storage in LCAs of perennial-crop based value chains. The first one concerns the quantification of carbon sequestered during the cultivation period of perennials. The second concern evolves around the uncertainty related to the permanence of the carbon storage (Ledo et al., 2018). In the following section, options for treating carbon sequestration and storage in LCAs of perennial cultivation systems are discussed. The section concludes with practical recommendations for LCA practitioners who usually have limited expertise in soil carbon modelling.

6.2.1 Quantification of carbon sequestered

Ideally, LCAs rely on primary data. This means, greenhouse gas assessments of perennial cultivation systems would use empirical data on carbon changes associated with the cultivation (Goglio et al., 2015). Unfortunately, empirical data are neither abundantly available nor site-generically applicable. For this reason, perennial crop LCAs commonly use literature values instead. LCAs on miscanthus-based value chains usually estimate soil carbon changes ranging between 0.7 and 2.2 t C ha^{-1} yr^{-1} as reported by McCalmont et al. (2017) for the cultivation on arable land. This range has also been used for the parameterisation of the miscanthus model in Chapter 2. It is questionable if carbon changes in this range can be expected everywhere due to the dependence on various factors, including climate, soil conditions and land-use history (Rowe et al., 2016). For this reason, approaches for the estimation of accumulated carbon due to agricultural land uses are required. In literature three classes of approaches are distinguished: the estimation via emission factors, modelling by means of simple carbon models and complex models integrating soil carbon and plant growth models (Goglio et al., 2015).

Combined soil carbon and plant growth models are widely used in agricultural research, as they can provide accurate results. Given their excessive data (e.g. daily meteorological data, photosynthesis rate, etc.) and computational requirements, these models are considered too complex for the application by LCA practitioners and thus for the integration into decision-support tools such as simplified models (Goglio et al., 2015; Ledo et al., 2018). Consequentially, these approaches are not further considered here.

The estimation of soil carbon changes due to agricultural activities via emission factors is most widely applied in LCAs. The European Commission (ILCD, 2010) has suggested an approach for estimating emission factors that enable the consideration of a range of land use changes (e.g. from annual crop or set-aside land to perennials). Default values for the native soil carbon levels, given prevalent climatic and soil conditions, are adjusted considering land use (annual, perennial, etc.) and management (tillage, fertilisation, etc.) factors. The difference between the native or previous and adjusted state is subsequently used to estimate the change in the carbon stock due to the intended land use. It is a simple approach providing standardised estimates of the potential carbon losses and gains. However, it offers only few specifity in regard of the crop considered, as only a single value is indicated for perennial crops. This is despite the fact that soil carbon storage can vary substantially between perennial crop types and locations depending on the soil conditions (Ledo et al., 2018; Ledo et al., 2020).

Simple carbon models, such as RothC (Coleman et al., 1997), are an alternative solution and occasionally used in agricultural LCAs. These models simulate the soil carbon dynamics using information on the carbon inputs to soil due to a certain land management scheme along with data on temperature, water and clay content. Thus, simple carbon models deliver site-dependent and better estimates than could be achieved using emission factors (Goglio et al., 2015; Peter et al., 2016). Although, these models are simpler than the ones including crop growth models, they require expertise that might be beyond an average LCA practitioner. This could be experienced in Chapter 3, where substantial assumptions on biomass and soil characteristics had to be taken. In particular for studies, where a large geographical range is considered (e.g. in Chapter 4), assessments were challenging.

The above mentioned approaches aim to detail the changes in soil carbon due to an agricultural activity. Allometric approaches (as applied in Ledo et al. (2018) and Chapter 4) do not estimate changes in soil carbon but characterise the carbon accumulation in the plant biomass over time. This includes above-ground parts and below-ground parts, which are in particular important for assessments of perennial crops. The quantity of carbon accumulated is estimated by drawing on information on harvestable yield (as provided by the model user), relations between crop fractions (e.g. below- in relation to above-ground biomass), carbon contents, (root) senescence ratios and decomposition rates (Ledo et al., 2018). Apparently, the required parameters have to be defined crop-specifically. For miscanthus, this is possible, and has been done previously, as literature is available (e.g., in Ledo et al. (2018)). For newer cultivation systems such as wild plant mixtures this might be more challenging but will be increasingly possible when these systems are more thoroughly researched. If data is available, carbon quantities sequestered in association with the cultivation of perennial crops, can be estimated by LCA practitioner with relative ease. In contrast with IPCC emission factors, this is possible in a crop- and yield-specific manner. Clearly, the carbon accumulated in the biomass during the cultivation period must not be confused with changes in soil carbon. However, the most considerable part of the carbon sequestered during the cultivation of perennial crops is stored in the plants below-ground organs, while only a minor fraction is due to changes in soil carbon (Dohleman, Heaton, Arundale, & Long, 2012; Martani et al., 2021). Thus, allometric models enable the consideration of the major share of carbon sequestered in perennial cultivation systems. For this reason, it is recommended here to use allometric models (e.g., in Ledo et al. (2018)) for LCA of perennial crops.

6.2.2 Permanence and duration of carbon storage in perennial crop cultivation

The quantification of the amount of carbon sequestered during the cultivation period provides a first crucial information for greenhouse gas assessments. However, the benefit of any carbon sequestration in terms of greenhouse gas mitigation depends greatly on the permanence of the storage, which, ideally, is ensured infinitely. In the case of carbon sequestered in soil or biomass fractions, this permanence is uncertain, as losses are possible and even likely. The amount and duration of carbon remaining stored, depends mainly on the subsequent land use and the given soil characteristics as well as on stability of crop residues (Ledo et al., 2018; Rowe, Keith, Elias, & McNamara, 2020). Contradicting results have been reported on the long term soil carbon effects of miscanthus cultivation on arable land. For miscanthus, net carbon increases (Dufossé, Drewer, Gabrielle, & Drouet, 2014) as well as net carbon losses (Rowe et al., 2020) after the reversion to arable land have been reported. For this reason, a socially responsible and precautious LCA practitioner or farmer should not consider an infinite storage.

In contrast to infinite carbon storage, finite carbon storage implies that, e.g. carbon dioxide is withdrawn from the atmosphere but re-emitted later. For this reason, temporary carbon storage is also referred to as delayed emission. From a perspective of inter-generational equity, it does not matter, if an emission occurs today or some when in the future. Nevertheless, it is argued that temporary storage and delaying emissions can be beneficial (Brandão et al., 2013; Dornburg & Marland, 2008; Fearnside, 2008). This is due to the continuing rise in atmospheric greenhouse gas concentration and global temperature, which might contribute to the exceedance of tipping points in the global climate system. Even the short-term delay of emissions might be beneficial to prevent immediate passing of tipping points and thus provide time for more sustainable solutions (Jørgensen & Hauschild, 2013; Jørgensen, Hauschild, & Nielsen, 2015; Lenton et al., 2019).

Several relevant LCA standards suggest approaches for considering temporary carbon storage (Jørgensen & Hauschild, 2013). Approaches were suggested for instance in PAS2050:2008 (BSI, 2008) and the ILCD handbook (ILCD, 2010), which apply credits for delayed emissions (e.g., ILCD: -0.01 times years delayed times emission in kg CO_{2eq}). Although PAS2050 and the ILCD handbook suggest slightly different factors for calculating credits, the difference in the relative impact of an emission delayed, is comparatively small. The selection of one or the other approach does not substantially influence the results of an assessment (Brandão et al., 2013).

Both approaches are based on the same critical assumption: a 100-year accounting period. The accounting period defines the time horizon after which, impacts are neglected. The selection of a 100-year timeframe in greenhouse gas assessment is an arbitrary and mainly policy-driven (non-scientific) definition. Essentially, its use implies the assumption that atmospheric CO_2 concentrations have returned to pre-industrial levels in 100 years. Following this, humankind did not have to fear negative impacts associated with the emissions then. Given this framework, a delayed emission, occurring in 50 years from now, had a lower integrated radiative forcing than an emission today. It becomes apparent from this, that the benefits of temporary carbon storage and delayed emissions, are based on deliberate assumptions and there is a risk of overestimating benefits (Jørgensen & Hauschild, 2013). Nevertheless, they should be accounted for given the urgency for preventing the reaching of climate tipping points.

Based on the above reflections on possible approaches for the treatment of carbon sequestration and storage, the following is recommended for LCA of perennial cultivation systems: The quantity of carbon sequestered and stored can be estimated using allometric models. These models are comparatively simple, provide crop- and yield-specific estimates, and could even be integrated in simplified models in future.

LCA practitioners should consider the carbon sequestered due to the cultivation as a delayed emission, assuming that the entire amount is released after the cultivation period. Clearly, this is a worst case scenario, as not all the carbon will be released directly after the end of the cultivation period. Nevertheless, this conservative approach shall be taken to reduce the risk of overestimation of benefits derived from carbon storage associated with cultivation of perennial crops. Delayed emissions should be accounted for following the approach suggested in the ILCD handbook (ILCD, 2010), which is recommended due to its simplicity. This is favourable for inexperienced LCA practitioners, as it allows simple and quick calculation of benefits from delaying emissions.

6.3 Incorporating land use impacts on biodiversity into the LCA framework

As outlined in the introduction, biodiversity is fundamental for keeping the Earth system in a resilient state. Globally, biodiversity loss is driven by several factors including climate change and habitat loss. The latter results mainly from anthropogenic land use (Maxwell et al., 2016), which needs to be considered in the LCA framework (Curran et al., 2016). For this reason, the following chapter focuses exclusively on biodiversity impacts caused by land use.

6.3.1 Land use and biodiversity in LCA

In LCA, all types of land use impacts, e.g. on soil quality and biodiversity, are usually described using a standardised framework suggested and promoted by the United Nations Environment Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) (Koellner et al., 2013). For a given indicator, the framework allows LCA practitioners to quantitatively describe a change in quality due to a certain land use (incl. the occupation and transformation phase). For this, the value of a quality indicator for the land use in question is compared with the indicator value for a reference state, also considering the time and area affected. The difference is then used to derive characterisation factors for land use impacts. Depending on the environmental impact in question, an appropriate quality indicator has to be selected (Koellner et al., 2013). This indicator has to be easy to measure and to communicate (Curran et al., 2011).

In view of the inherent complexity of biodiversity, it is acknowledged that a simplification is required for the consideration in life cycle impact assessment (LCIA) (GGLCIA 2016). Consequently, a wide range of indicators has been developed and suggested (see, i.a., Baan, Alkemade, & Koellner (2013), Chaudhary, Verones, Baan, & Hellweg (2015), Jeanneret, Baumgartner, Freiermuth Knuchel, Koch, & Gaillard (2014), Lindner, Fehrenbach, Winter, Bloemer, & Knuepffer (2019), Michelsen (2008), Schmidt (2008), Souza et al. (2013)). In these approaches, biodiversity impact assessment is commonly reduced to a single or maximum a few metrics. Usually, they cover biodiversity at species and population levels, which are represented using the indicators *species richness*, *abundance* and (*diversity) indices*. *Species richness* indicates the number of species present, while neglecting the number of individuals of each species. The number of individuals is considered in *abundance* measures. Diversity *indices* combine the information on richness and abundance. Due to its simplicity and data availability, *species richness* is the most widely applied indicator in LCA biodiversity assessment approaches (Teixeira et al., 2016). Given the focus on species richness, impacts are usually quantified using a measure of the amount of species that are disappearing due to a certain land use occupying an area for a certain time. Correspondingly, the most common metric in LCA biodiversity assessments is the (partially) disappeared species fraction (PDF) (Crenna, Marques, La Notte, & Sala, 2020).

Due to a lack of consensus on how to assess biodiversity impacts associated with land use, these are not considered in many LCA studies (Winter et al., 2018). Nevertheless, there are established impact assessment methods, which incorporate land use impacts on biodiversity in their framework. The impact assessment method collection ReCiPe 2016 (Huijbregts et al., 2016) is an example for an operational approach. It is available in LCA software and used by LCA practitioners (Crenna et al., 2020).

6.3.2 Operational approaches for biodiversity land use impact assessment

ReCiPe 2016 (Huijbregts et al., 2016) is an established impact assessment method and widely used by LCA practitioners (see for instance, Bussa, Zollfrank, & Röder (2020), Schulte et al. (2021), Wagner et al. (2019)). Characterisation factors for land use impacts on biodiversity are based on the relative loss of terrestrial species due to a certain land use considering consequences from land transformation, occupation and relaxation (Huijbregts et al., 2016). Data on relative species losses are mainly based on Baan et al. (2013), who compared the species richness in a certain anthropogenic land use, e.g., cultivation of annual or perennial crops, with the one in a situation where no land use occurred (potential natural vegetation). The relative species loss was calculated for a number of land uses, considering four species groups – plants, mammals, birds and invertebrates (mainly arthropods) – which were considered as proxies for the total species loss. To derive midpoint characterisation factor as used in ReCiPe 2016 (Huijbregts et al., 2016), the relative species loss of the land use under investigation (e.g. a perennial cultivation system) is divided by the relative species loss caused by the land use of annual crops on a global average.

Table 1 presents the values of the relative species loss ($S_{rel\ loss}$) for annual and perennial crops (referred to as *permanent* in the publication), as suggested in Baan et al. (2013) as well as the corresponding land use characterisation factors in ReCiPe 2016 (Huijbregts et al., 2016). The relative species loss in perennial cultivation systems was found to be lower than in annual ones for all taxonomic groups and sub-groups except for birds (Baan et al., 2013), delivering overall midpoint characterisation factors of 1 and 0.7 for annual crop and permanent crop cultivation, respectively. Clearly, this is based on taxon-specific relative species losses on a global level, thus potentially neglecting variability of natural vegetation and fauna.

As suggested in the ReCiPe 2016 report on characterisation (Huijbregts et al., 2016), more specific characterisation factors could be derived taking biome-specific relative species loss data as provided in (Baan et al., 2013). For permanent crop cultivation in Europe (Biome 4 – temperate broadleaf forest), they suggest a relative species loss of 0.02 (median) which equals a characterisation factor of 0.033 crop equivalents. Accordingly, the biome-specific characterisation factors for perennial crops were substantially lower than for the global average. In summary, the characterisation factors at a global and biome level give the impression that the cultivation of perennials is substantially less detrimental to biodiversity than the cultivation of annuals.

Table 1 Relative species loss in annual and perennial land uses (based on Baan et al. (2013)) and corresponding midpoint characterisation factors as suggested in ReCiPe 2016 (Huijbregts et al., 2016).

<table>
<tr><th></th><th colspan="2">annual</th><th colspan="2">perennial</th></tr>
<tr><th>Taxonomic group</th><th>$S_{rel\ loss}$*</th><th>CF**</th><th>$S_{rel\ loss}$*</th><th>CF**</th></tr>
<tr><td>All</td><td>0.60</td><td>1.00</td><td>0.42</td><td>0.70</td></tr>
<tr><td>Arthropods</td><td>0.65</td><td rowspan="6">–</td><td>0.56</td><td rowspan="6">–</td></tr>
<tr><td>Other invertebrates</td><td>0.79</td><td>0.44</td></tr>
<tr><td>All vertebrates</td><td>0.50</td><td>0.39</td></tr>
<tr><td>Birds</td><td>0.53</td><td>0.62</td></tr>
<tr><td>Other vertebrates</td><td>0.45</td><td>0.27</td></tr>
<tr><td>All plants</td><td>0.56</td><td>0.38</td></tr>
</table>

This however could not be confirmed for perennial rhizomatous grasses in Chapter 5 (Lask, Magenau, et al., 2020). The meta-analysis did not indicate significantly higher species richness (i.e. lower relative species losses) in the perennial cultivation systems when compared with annual systems. Similar, Elshout, van Zelm, Karuppiah, Laurenzi, & Huijbregts (2014) could not confirm this neither. Against this background, the biome-specific characterisation factor for the cultivation of perennial crops in Europe, has to be critically reflected upon. In general, assessments at biome level might not be detailed enough to capture significant differences in local species distributions. A major issue besides this concern is the fact that all species are treated equally, irrespective of their specific threat level or endemism.

These shortcomings are partially overcome in the approach suggested by Chaudhary et al., 2015. It uses the countryside species area relation and provides characterisation factors for 804 terrestrial ecoregions of the world, each considering five taxonomic groups (mammals, birds, reptiles, amphibians, vascular plants). Six land use classes were included: intensive forestry, extensive forestry, annual crops, permanent crops, pasture and urban. The approach was meanwhile recommended by the UNEP/SETAC Life Cycle Initiative for biodiversity assessment in LCA (Milà i Canals et al., 2016). Nevertheless, it was criticised as it does not include sufficient land use classes and does not allow a differentiation of management regimes in agriculture or forestry. For this reason, Chaudhary & Brooks (2018) further advanced the method and suggested characterisation factors for three land use management intensity levels for the given 804 ecoregions.

Despite the method's advancement, LCA practitioners who want to include biodiversity land use impacts into LCAs of perennial cultivation systems currently encounter issues in practice. Presently, the use of the characterisation factors suggested in Chaudhary et al. (2015) requires additional effort as they are not yet implemented in established LCA software by default. This results, i.a., from the fact that it is not used in any of the established and ready-made impact assessment collections, which is mainly due to an incompatibility of the indicator results. Biodiversity impacts in Chaudhary et al. (2015) and Chaudhary and Brooks (2018) account for irreversible disappearance of species. Thus, the characterisation factors indicate the amount of species that are lost forever, while approaches relying on Baan et al. (2013) (such as ReCiPe 2016) consider temporarily disappeared species. For this reason, the characterisation factors suggested by Chaudhary and Brooks (2018) are not compatible with the ecosystem quality indicators of established impact assessment method collections such as ReCiPe 2016 (Huijbregts et al., 2016) and Impact World+ (Bulle et al., 2019). Further research will have to be dedicated to the harmonisation of these.

6.3.3 Advancing biodiversity land use impact assessment in LCA

LCA biodiversity impact assessments are not yet ideal. This can be depicted using an example from miscanthus cultivation: Bird species that commonly prefer open fields do not show a high affinity to established plantations of perennial rhizomatous grasses. These are however, preferred by woodland species that would not find a habitat in agricultural landscapes (Bellamy et al., 2009; Clapham, 2011; Kaczmarek, Mizera, & Tryjanowski, 2019). Essentially, perennial crop cultivation could create an additional niche in agricultural landscapes, which might be beneficial for an additional species. This change in the species composition however is not detectable using only species richness as indicator.

In addition, it shows that a net benefit for biodiversity depends greatly on dynamics at the landscape level. This is also in line with the environmental heterogeneity hypothesis (Palmer, 2007), which postulates that an increase in environmental heterogeneity goes along with an increase in biodiversity. This emphasises that the biodiversity benefits of a perennial cultivation system such as miscanthus, depend heavily on the extent of heterogeneity that is added to the landscape (Tscharntke et al., 2012).

These concerns, the strong focus on species richness and the neglect of contextual information, are regularly emphasised in regard to biodiversity impact assessments in agricultural LCAs (Gabel et al., 2016). For this reason, they will be addressed in the following.

Focus on species richness and alternatives

Many of the approaches assessing land use impacts on biodiversity rely on species richness as an indicator for biodiversity (Winter et al., 2018). It is a surrogate for which data are comparatively abundant (Baan et al., 2013) and is an indicator that is easy to communicate. However, its' use as a single indicator needs to be critically questioned given the complexity of biodiversity with its diverse organisational levels (genetic resources, species, population, ecosystems) and attributes (composition, function, structure) (Teixeira et al., 2016).

This highlights the necessity to reflect on alternative indicators for biodiversity assessments in LCA. Suggested alternatives include, i.a. ecological scarcity (Michelsen, 2008), naturalness (Brentrup, Küsters, Lammel, & Kuhlmann, 2002), biodiversity potentials (Jeanneret et al., 2014; Lindner, 2015) and the ones focusing on functional diversity (Souza et al., 2013).

Only a few of these indicators have been applied in actual case studies and none of them is implemented in a major LCIA method. A major issue impairing their use is the lack of corresponding data which are required for the setup and implementation. For instance, it is challenging to collect data on a species' contribution to a certain ecological function and how this interaction might be influenced by the presence of other species (Vrasdonk, 2020). This issue impairs the use of functional diversity indicators although they have repeatedly been suggested as more appropriate than species richness (e.g., (Curran et al., 2011; Souza et al., 2013).

The focus on functional diversity follows from the assumption that an ecosystem's stability and resilience essentially depends on its ability to maintain certain ecological functions despite environmental disturbances (McCann, 2000). Following this assumption, certain taxonomic groups are uniform in terms of their ecological function, which means that individual taxonomic groups within a functional cluster are redundant and their function can be performed by another species from the group (Geeta et al., 2014). Accordingly, a certain ecological function will be kept until the last species from the functional group has disappeared.

This approach, however, neglects the intrinsic value of biodiversity as well as the importance of genetic resources in regard of a community's or ecosystem's evolutionary adaptability (Geeta et al., 2014). Adaptability, and thus the protection of evolutionary potential, is the core of resilience and essentially forms the basis of biodiversity, which is why genetic information becomes increasingly pivotal for biodiversity assessments (Chaudhary, Pourfaraj, & Mooers, 2018; Curran et al., 2016).

Phylogenetic diversity is suggested as a promising indicator for the incorporation of genetic information in LCA biodiversity assessments (Chaudhary et al., 2018; Maier, Lindner, & Francisco, 2019). It indicates the evolutionary proximity of organisms based on the phylogenetic tree. Closely related species share more features and thus exhibit closer proximity than distantly related ones, which is indicated by the branch length in the phylogenetic tree (Chaudhary et al., 2018; Faith, 2008).

The inclusion of phylogenetic information into LCIA frameworks enriches the results of the corresponding biodiversity assessments. It would clearly expand the information provided by the number of species as it implicitly informs about their relation from which, in general, conclusions with regard to their functional diversity can be drawn as phylogenetically related species tend to fulfil similar ecosystem functions (Cadotte, 2013; Mace, Gittleman, & Purvis, 2003). Apparently, the wider application of phylogenetic information in LCA biodiversity assessments is rather a vision for the near future than a quick fix. However, first steps in this direction are undertaken (Chaudhary et al., 2018) and the widespread use of genetic information in biodiversity assessments seems possible in the midterm. This is mainly due to the global efforts in mass sampling, sequencing and analyses of genetic information from the environment, including information from all kind of sources (Bohmann et al., 2014). This enables the assessment of different types of ecosystems and the inclusion of further groups of organisms (Baird & Hajibabaei, 2012). So far, microorganisms have been widely neglected in biodiversity assessments although they account for a substantial amount of the global species diversity (Nee, 2004). The availability of global datasets is a prerequisite for this but will then allow LCA experts to derive more meaningful characterisation factors for biodiversity assessments based on phylogenetic information.

Neglect of context, management and crop-specific effects

The biodiversity impact of a given land use depends very much on contextual conditions (Gabel et al., 2016; Scherr & McNeely, 2008). The number and heterogeneity of habitats as well as their connectivity are critical for the biodiversity value of a certain landscape (Benton, Vickery, & Wilson, 2003; Katayama et al., 2014). A crop such as miscanthus, which is not widely established in European agricultural landscapes, would likely add a new type of habitat to its surrounding landscape. The net benefit however would decrease with an increasing share of the cultivated area. Similarly, management schemes, for instance, the planting density can influence the biodiversity value of a certain land use option. For perennial rhizomatous grasses, it was shown that the abundance of plants other than the crop is negatively correlated with the planting density and biomass yield (Dauber et al., 2015). Cultivation system-specific characteristics can also be critical to the biodiversity value, which can be emphasised by comparing miscanthus and wild plant mixtures. Although both are perennial systems, miscanthus cultivations usually remain a monoculture, while wild plant mixtures are more diverse by design and even provide a food source for pollinators (Cossel, 2020).

These examples highlight that the variability in the biodiversity value due to contextual conditions including interactions at the landscape level, crop management and crop characteristics needs to be reflected in assessments of agricultural production systems (Maier et al., 2019; Teixeira et al., 2016). Nonetheless, so far only a few methods allow LCA practitioners to derive specific characterisation factors for land use impacts considering the contextual conditions. If at all, they provide values for qualitative intensity levels. As previously introduced, Chaudhary and Brooks (2018) consider three intensity classes – minimum, light, and intense – for five land use classes (managed forests, plantations, pasture, cropland, urban). These classes are broadly differentiated based on characteristics such as field size, amount of fertiliser and pesticide, as well as the quantity of tillage operations. It can be judged from the presence of only three classes for all cropland-based systems that a reasonable differentiation of cultivation systems is not possible. Perennial cultivation systems such as miscanthus and wild plant mixtures would fall into the minimum intensity class according to the suggested classification irrespective of obvious differences in crop characteristics.

This highlights the need for approaches that enable a more specific quantification of management-specific parameters. So far, only few LCA biodiversity assessment methods feature approaches in this regard. These include for instance (Jeanneret et al., 2014), which is a site-specific approach that serves the quantification of agricultural management activities such as tillage operations and herbicide application on biodiversity in Switzerland. Due to its geographical focus and complexity it is not applicable for a wide range of LCA applications.

Further alternatives are increasingly suggested for the use in agricultural LCA. Some of them aim to quantify the qualitative change in biodiversity status due to a certain agricultural management scheme through assessing a selection of parameters (e.g., Maier et al. (2019) and Lindner et al. (2019)). The indicator for qualitative change is then related to an indicator which provides information on the inherent biodiversity value of the geography in which the land use occurs.

Following this approach, Maier et al. (2019) suggest the integration of a land use intensity index in the calculation of a biodiversity metric. The index summarises the relative influence of management parameters on biodiversity and is used to adjust the biodiversity metric for a specific land use. A selection of management parameter has been suggested for cropland based on literature information. This parameter selection includes fertiliser application, irrigation, pesticide application, mechanization (tillage), mixed cropping, and the presence of native vegetation (Maier et al., 2019). Lindner (2015) has suggested a related approach which relies on expert interviews and estimates. Taking a wider perspective, also biodiversity impacts related to crop-specific characteristics and landscape effects can be included in the assessment by developing and aggregation of contribution functions. The approach is based on a loose understanding of the biodiversity concept. This can be evidenced by the indicator selection, a universal biodiversity potential (Lindner, 2015), which prevents the inclusion of the approach in existing impact assessment frameworks.

The fundamental idea of adjusting a given biodiversity quality indicator using a collection of parameters is promising. It will help to improve the representativeness of biodiversity impact assessment in perennial crop LCAs as it enables the integration of contextual information. First steps in this direction have already been taken. Using the approach suggested by Lindner (2015), parameters that define the biodiversity potential of perennial crop cultivation in Europe were identified and ranked for their relevance by means of expert interviews (Annen, 2021). This includes management parameters such as field size, tillage and pesticide application, which are similar to the ones suggested by Maier et al. (2019), but also crop-specific characteristics such as the presence of inflorescence (Annen, 2021). In addition, landscape heterogeneity was identified and ranked as the most decisive parameter in defining the biodiversity value, which is in line with the environmental heterogeneity hypothesis (Palmer, 2007). For each of these parameters, an indicator has been agreed with the experts. For instance, heterogeneity at the landscape scale is described by the number of dissimilar crops adjacent.

In a second step, the individual contributions of the identified parameters have to be clearly described. For instance, the optimal field size for a perennial cultivation system from a biodiversity potential perspective has to be defined. This will require considerable research effort, in particular given the number of parameters. For this reason, the focus should be on the higher ranked parameters such as landscape heterogeneity. Parameters for which biodiversity impacts are covered by other LCA impact categories, are of lower priority. This applies for instance to the application of fertilisers, as eutrophication is commonly assessed in agricultural LCAs and the related impacts on biodiversity could be integrated via an endpoint assessment.

6.3.3 Recommendations for applying and advancing biodiversity land use impact assessment in LCA

Overall, assessment approaches and frameworks for land use impacts on biodiversity in LCA exist. Some of them are operational and can support LCA practitioners in the biodiversity land use impact assessment in agricultural LCAs. This includes operational approaches such as implemented in ReCiPe 2016 (Huijbregts et al., 2016) and even newer approaches such as Chaudhary and Brooks (2018). For now, LCA practitioners can use characterisation factors suggested in the operational methods. However, it is recommended to practitioners to critically reflect on them and refrain from biome-specific values for permanent crops as suggested in ReCiPe 2016 (Huijbregts et al., 2016).

In general, species richness should only be considered an intermediate indicator solution. Phylogenetic data should play a crucial role in the development of future land use characterisation factors, as this type of information emphasises the importance of genetic resources and serves as a surrogate of functional diversity. Nevertheless, even characterisation factors based on this approach will suffer from inaccuracies if contextual effects are not considered in the assessment. For this reason, it is recommended to advance research on biodiversity impact assessment in the direction of adapting existing land use characterisation factors using defined parameter sets.

6.4 Conclusion

LCA is the preferred tool for assessing the environmental performance of products or services. It is based on science, relies on life cycle thinking and accounts for a number of environmental issues. However, this ambition is a challenge and an unfulfilled promise at the same time. First, it comes at the cost of complexity, which limits its use by a wide range of users from practice. Second, the comprehensiveness in regard to the environmental issues covered is not fulfilled as can be evidenced by the widespread neglect of land use impacts on biodiversity in LCAs of perennial cultivation systems. These concerns have to be addressed, to increase LCA's relevance as an environmental decision support tool. This thesis investigated and gave recommendations on how to advance the applicability and comprehensiveness of LCAs of perennial cultivation systems.

Applicability can be improved by reducing the complexity of LCAs of perennial crops. It was found that global sensitivity analyses are key in simplifying the compilation of inventory data. They help to distinguish between determining and non-critical LCI parameters and, in this way, reduce the effort required for data collection. In addition, global sensitivity analyses facilitate the development of simplified models, which lower the barriers of LCA application by farmers and SME active in perennial crop-based value chains. For these reasons, the use of global sensitivity analyses and the development of simplified LCA models should be promoted and extended to further agricultural production systems.

In terms of comprehensiveness, carbon sequestration and storage as well as land use impacts on biodiversity have to be considered in a reliable sustainability assessment. To facilitate a wider and transparent inclusion of carbon changes associated with perennial crop cultivation, this thesis concluded the following: The carbon sequestered due to the cultivation of perennials shall be quantified following allometric models and accounted as a delayed emission according to the ILCD handbook. In combination, this approach reduces the risk of overestimating the benefits from carbon sequestration and storage. It allows LCA practitioners to account for this aspect with relative ease and thus ensures its pertinent inclusion in LCA practice.

When assessing land use impacts on biodiversity, LCA practitioners focusing on perennial crop cultivation need to be careful and critical about characterisation factors used in current LCIA methods such as ReCiPe 2016. The characterisation factor for perennial crops seems overly optimistic when compared with available data on species richness in annual crops and perennial grasses such as miscanthus. In addition, the current approaches neglect the importance of landscape-, management- and crop-specific aspects in the evaluation of land use impacts on biodiversity. Considering these aspects, future research should facilitate the adjustment of pre-existing characterisation factors to ensure a reasonable representation of biodiversity in agricultural LCAs. In the long run, the presently prevailing indicator species richness is to be replaced by phylogenetic diversity, which provides richer information on genetic resources and the functional relevance of organisms.

The recommendations above were mainly derived for LCAs of perennial cultivation systems. Nevertheless, they also apply to other types of agricultural production systems. Due to their general relevance, the outcomes of this thesis can contribute to further advancement of the applicability and comprehensiveness of agricultural LCAs, which will help to increase the tool's relevance in environmental management and decision support. This is imperative given the challenge to preserve the Earth system as a resilient, safe operating space.

6.5 References

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8.0 Curriculum vitae

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PROFESSIONAL EXPERIENCE

12.2017 –
Doctoral Student – University of Hohenheim, Institute of Crops Science, Department of Biobased Resources in the Bioeconomy, Stuttgart
- PhD thesis: „Life Cycle Assessment of perennial cultivation systems: Advancing applicability and comprehensiveness"
- Lectures in Life Cycle Assessment
- Supervision of Bachelor's and Master's theses
- Work package leader in BBI Demonstration Project "GRACE – Growing Advanced Industrial Crops on Marginal Lands for Biorefineries"

05.2016 –
Freelance Editor – BIOPRO Baden-Württemberg GmbH, Stuttgart

01.2015 – 03.2015
Internship – Max-Planck Institute of Immunobiology and Epigenetics, Freiburg

09.2014 – 12.2014
Internship – Staatliches Weinbauinstitut (State Viticultural Institute), Freiburg

EDUCATION

10.2015 – 04.2018
M.Sc. Bioeconomy – University of Hohenheim
- Master's thesis: „Life Cycle Assessment of ethanol production from miscanthus: A comparison of production pathways at two European sites"

10.2011 – 02.2015
B.Sc. Biological Sciences – University of Konstanz
- Bachelor's thesis: „Funktionelle Charakterisierung artifiziell gesplitteter Versionen der *S. cerevisiae* ERAD E3-Ligase Doa10"

09.2002 – 07.2010
Friedrich-Schiller Gymnasium Fellbach
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CONFERENCE CONTRIBUTIONS

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Poster presentation at SETAC Europe 30th Annual Meeting, Dublin
- Lask, J; Magenau, E; Lewandowski, I; Wagner, M: „Assessing biodiversity impacts of ecological intensification activities – a case study in southwest Germany"

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Oral presentation at 15th Ökobilanzwerkstatt, Stuttgart
- Lask, J: „Implications of technology advancement and CCS in 2nd generation ethanol production"

05.2019
Oral presentation at 27th European Biomass Conference & Exhibition, Lisbon
- Lask, J; Rukavina, S; Ištuk, M; Wagner, M; Lewandowski, I: „Combining lignocellulosic ethanol production and carbon storage – Towards net-negative emission bioethanol"

PUBLICATIONS

Lask, J., Kam, J., Weik, J., Kiesel, A., Wagner, M., & Lewandowski, I. (2021). A parsimonious model for calculating the greenhouse gas emissions of miscanthus cultivation using current commercial practice in the UK. *Global Change Biology. Bioenergy*. (Accepted for publication).

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Jan Lask

Stuttgart, 26th April 2021

9.0 Affidavit

Declaration in lieu of an oath on independent work

according to Sec. 18(3) sentence 5 of the University of Hohenheim's Doctoral Regulations for the Faculties of Agricultural Sciences, Natural Sciences, and Business, Economics and Social Sciences

1. The dissertation submitted on the topic

Life Cycle Assessment of perennial cultivation systems: Advancing applicability and comprehensiveness

is work done independently by me.

2. I only used the sources and aids listed and did not make use of any impermissible assistance from third parties. In particular, I marked all content taken word-for-word or paraphrased from other works.

3. I did not use the assistance of a commercial doctoral placement or advising agency.

4. I am aware of the importance of the declaration in lieu of oath and the criminal consequences of false or incomplete declarations in lieu of oath.

I confirm that the declaration above is correct. I declare in lieu of oath that I have declared only the truth to the best of my knowledge and have not omitted anything.

Stuttgart, 26th April 2021

Place, Date

Signature