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Multiple Goals for Biomass Residues in Circular Bioeconomies? Assessing Circularities and Carbon Footprints of Residue-Based Products

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Abstract

Biomass residues are often considered key in a reorientation towards circular bioeconomies, both by returning organic matter and nutrients to soils and by expanding the feedstock base for fossil-free products. Different indicators are available to assess progress towards circularity, but many available indicators and assessments seem to focus on product or material circularity, and lack in coverage of ecological or nutrient circularity. This study therefore applies both material and nutrient circularity indicators to two cases of residual biomass' valorisation: plastics production from wheat straw, and jet fuel production from animal by-products, in order to better understand the potential of the different types of indicators to assess the circularities of bio-based products.

Both the studied production systems achieve approximately 50% material circularity in the base case, but the scores are significantly lower when upstream processes such as cultivation and animal husbandry are included. In the plastics case, the nutrient circularity scores are consistently lower than material circularity scores. The contribution to circularity from composting and recycling of different streams can be interpreted differently following the different types of circularities and, in addition, considering the potential climate impact of different strategies. This study shows that a combination of methods and indicators can shed light on different types of circularities and goals, but also that a wider discussion on what circularity may entail for biomass and biomass residues, and how it can be measured, is needed to develop useful indicators for bio-based circularity and circular bioeconomies.

Keywords Biomass residues · Circularity · Circular bioeconomy · Carbon footprint · Valorisation

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Introduction

The valorisation of residual biomass is often mentioned as an essential part of circular bioeconomies [1–3], and the use of residual biomass streams can be considered necessary in the race to meet the future demand for food, feed, and bio-based products [4]. Both the concepts of circular economy and bioeconomy are to some extent multi-faceted [5], but in the context of this paper, the critical aspects of a circular bioeconomy can be seen as the replacement of fossil-based resources with biomass resources in a "*regenerative system in which resource input and waste*, *emission*, *and energy leakage are minimized by slowing*, *closing*, *and narrowing material and energy loops*" [6]. The concept of "circular economy" has, however, also been criticised for being both too broad and too vague [7, 8], and potentially conflicting visions of bioeconomy have been identified [9–11]. The limitations of these concepts are transmitted to the concept of circular bioeconomy, where the combination of circular economy and bioeconomy adds to the variety of possible definitions and interpretations [12]. It is therefore not evident what a circular bioeconomy might look like in practice, or what roles different types of residual biomass management can play in realising its intended goals.

Circularities for Biomass

In general, two types of circularities appear to be envisioned in the literature on circular bioeconomy. On the one hand, circularity referring to materials or products such as fine and specialty chemicals; namely basic chemicals, bioactive compounds, polymers, fuels, and building materials [13–15] appears centred around biorefinery approaches, optimal valorisation and cascading uses of biomass [13, 15–17]. As already mentioned, the valorisation of residual biomass is central in this context of providing biomass feedstock. Cascading use can refer to sequential use but also to value optimisation and making use of all parts of biomass [2], or as cascading in time, value, and function [18]. Extending the use phase of bio-based materials is considered an important aspect of circularity [19, 20], but biodegradability is also recognised as a limiting factor [21, 22].

On the other hand, the role of biomass circulation in preserving or restoring natural resources and ecosystem services is emphasised [23]. This type of ecological circularity or cycling could be defined as "*outputs from the use of renewable biomass which can re-enter biogeochemical cycles and contribute to new plant growth*" [24] and includes the restoration of land and soil fertility, and the return of biological nutrients to the biosphere [15, 19, 20]. Ecological circularity and material and product circularity should not be seen as separable phenomena [25], but as supplementary focus lenses for cycling through the earth's biogeochemical cycles by degradation on the one hand, and on cycling in product and material loops within the technosphere on the other [26]. With a focus on the recirculation of organic matter and nutrients, biomass and biobased materials' contribution to circularity depends on biodegradability, availability and fate of nutrients, and that hazardous or toxic substances are not present [19, 20]. Typical examples of practices that may advance this type of circularity include leaving crop residues and applying manure, compost and digestate to soil [4, 15]. Bos and Broeze [4] also mention the possibility to return waste products from valorisation processes to the soil to close the circle, which could, to some extent, compensate for the initial removal of e.g. crop residues [27]. The recycling of wood ash could be one such example, but where contamination through co-incineration may present an issue in regard to hazardous substances [28].

Circularity Indicators for Bio-Based Products

Despite the differences and ambiguities in available definitions and interpretations of the circular bioeconomy, a number of indicators for measuring product circularity have been suggested in both scientific literature and by the private sector [29]. They differ in scope and data requirements, among other things, and most of them focus on non-biological materials and their reuse and recycling [30, 31], making them less suitable to assess the circulation of bio-based materials. Jerome et al. [31] found only four methods which aimed to include bio-based materials. Navare et al. [19] found that out of the four criteria for bio-based circularity which were derived from a definition of circular economy, no circularity method considered more than one of these criteria in any depth. In their systematic review of circularity indicators for biobased products, Vural Gursel et al. [32] found that no single method sufficiently addressed the nine requirements listed in [20]. In this review, the authors call for development of indicators that address biomass quality and closing of nutrient cycles. A better understanding of the available indicators could also be reached by coupling the analyses with applications to case studies [31].

While not specifically tied to biobased fuels, chemicals and other similar products, nutrient cycling, and the assessment of it, is a topic within other areas of the scientific literature. For instance, Velasco-Muñoz et al. [33] identified 41 available indicators for measuring the circular economy in the agricultural sector and others have suggested new frameworks for assessing the nutrient circularity of food production and agricultural systems [34, 35]. In their recent work, Møller et al. [36] apply different circularity indicators to pig production which cover, among other aspects, nutrient circularity of nitrogen (N) and phosphorous (P). Lavallais and Dunn [37] calculate the N circularity in the context of different manure treatment scenarios, and while they focus on mass flows of N, they calculate circularity using the micro-level product indicator “material circularity indicator” (MCI). Similarly, Burggraaf et al. [38] apply the MCI to calculate both material circularity and circularity of N and P of a dairy farm. These approaches and indicators covering nutrient circularity could be relevant also to assess the circularity of other types of bio-based production systems, including the valorisation of different types of residual biomass to high-value products.

Objectives

There appears to be gap between the coverage of available product circularity indicators and the critical aspects of circularity for biomass which not only refers to material or product circularity, but also to nutrient or ecological circularity. Specifically, in their review of circularity indicators for bio-based products, Vural Gursel et al. [32] call for investigation into indicators that better address nutrient cycles. In parallel, circularity indicators for agricultural systems that focus specifically on different nutrient cycles have been developed. The investigation into such indicators for bio-based products might be especially relevant to the case of biomass residues as product feedstocks since these are often envisioned to have the role of returning organic matter and nutrients to soils. This study therefore applies both a common product circularity indicator and nutrient circularity indicators to two different bio-based products made from residual biomass feedstocks. The objective is to better understand both the circularities of these production systems, and the potential of the different indicators to assess the circularities of bio-based products. The two studied cases are low-density polyethylene (LDPE) production from wheat straw, and jet fuel production from animal by-products. In addition, the carbon footprints of the same production systems are calculated since a bearing principle and ambition for the circular bioeconomy is to reduce greenhouse gas (GHG) emissions and climate impact, and these aspects are not covered by circularity indicators [39, 40].

The next sections describe the methods applied to assess circularities and carbon footprint, followed by an introduction and description of the two studied cases.

Methods: Circularity Indicators and Carbon Footprints of Bio-Based Products

To assess material or product circularity, the material circularity indicator (MCI) is chosen. The MCI [41] is by no means a perfect indicator for bio-based circularity, but it covers several circularity strategies in several life-cycle phases [31], it has been included in several published reviews and analyses of circularity indicators [19, 40, 42–45], and appears to be the most commonly applied indicator for micro-level assessment [46]. The MCI was first presented in 2015 and was updated in 2019 to better include the biological cycle of a circular economy. The update introduced two principles to decide whether biological materials can be considered circular [41]: firstly, the sustainable sourcing of biological materials which does not compromise natural systems and even regenerates them, and secondly, the cycling and return of biologically accessible nutrients to natural environments.

In this context, it is relevant to mention the derivations made of the first version of the MCI from 2015 to fit bio-based products. Rocchi et al. [47] modify the MCI for agricultural systems, but with a narrow scope focusing on livestock production. Razza et al. [48] suggest the MCI-BB for bio-based and biodegradable products which is mostly analogous to the updated MCI. One difference between the two indicators is that the parameters of the MCI-BB have been redefined; recycled feedstock is, for instance, instead regarded as biological feedstock, while the updated MCI has separate parameters to describe recycled feedstock and sustainably sourced bio-based feedstocks. As the cases studied in this paper are biomass residues—which are considered biological feedstocks in the MCI-BB and recycled feedstocks in the MCI—both methods consider the same flows. The MCI is considered more general and well-known than the MCI-BB, and therefore this analysis departs from the MCI as defined in [41].

To assess nutrient circularity, two different approaches are used. The approach used by e.g [37, 38]. to apply the MCI to flows of specific nutrients, in this case N and P, is used alongside indicators describing the recycling of nutrients, the N and P recycling indexes, NRI and PRI, respectively. These recycling indexes are applied by e.g. [36]. to assess nutrient circularity and differ from the MCI in the sense that they do not consider N and P inputs to a production system, but only their recycling. Other available indicators include the consumption of fossil P fertiliser [49] and the partial nitrogen balance [50], but the selection made here is considered to sufficiently cover different aspects of nutrient circularity.

Material Circularity Indicator, MCI

The MCI itself is a value between 0 and 1, where 1 implies a perfectly circular system. It is calculated from three product characteristics: (i) the mass *V* of virgin raw material used in manufacture, (ii) the mass *W* of unrecoverable waste that is attributed to the product, and (iii) a utility factor *X* that accounts for the length and intensity of the product's use. For instance, the utility factor (*X*) considers life extension strategies. Since the cases studied in this paper do not intend to change the longevity or use pattern of products, the utility factor is assumed to have a value of 1, which corresponds to the industry average. Instead, the focus is on the mass of virgin raw material and unrecoverable waste. This assumption also means that MCI scores in this study take a value between 0.1 and 1 which is due to the definition of the indicator [41].

V (virgin raw materials) is calculated from the mass of products (M) (Eq. 1) and the share of feedstock which is reused (F_U), recycled (F_R), or is biological material from sustained production (F_S). Sustained production of biological materials is defined as "*the extraction of natural materials at volumes and employing practices which aim to maximise the regeneration of natural systems in the indigenous ecosystems by for example supporting the development of healthy soils*" [41]. A simplified interpretation is also given based on maintaining long-term productivity and natural capital. Due to the difficulties in deciding whether these requirements can be considered fulfilled, a default assumption that they cannot seems reasonable. As the biomass feedstocks considered in this study are by-products, they are instead categorised as recycled feedstocks (F_R).

$$V = M * (1 - F_U - F_R - F_S),\ 0 \leq\ V \leq\ M \tag{1}$$

The transformation of biomass into bio-based products often requires the use not only of energy, but of different process additives. An adaptation of the MCI is therefore made here to better cover the studies cases in that all material process inputs are considered, and not only the feedstock. Therefore, the parameters F_U and F_R are reinterpreted as the fraction of reused and recycled material inputs, respectively. However, as the use of MCI for assessing circular strategies for agricultural products and systems can be hindered by large mass flows of water [51], the calculations in this study exclude process waters and any water for irrigation.

Similarly, as for virgin materials, the amount of unrecoverable waste (W, Eq. 2) is calculated via the amount of waste going to energy recovery or landfill (W_0, Eq. 3) from the mass of products (M) and the share of total product mass which is reused (C_U), recycled (C_R), composted (C_C) or energy-recovered (C_E), with the addition of waste created in recycling processes for feedstock (W_F), and products at end of life (W_C). Energy recovery is in the equation, but for energy recovery to be included and thus counted towards circularity, six criteria must be met. These include that the material recovered must be biological material that comes from sustained production, and that landfill or energy recovery is the only possible alternative for the material (see Online Resource 1 for the full list). Note that for the cases studied in this paper, there is no material for which all criteria are met, and thus energy recovery is not included in the present calculations of the MCI. Based on the above parameters, a linear flow index (LFI) is calculated, and the MCI is essentially a combination of the LFI and the utility factor. The full equations for the MCI can be found in Online Resource 1.

$$W = W_0 + \frac{W_F + W_C}{2} \tag{2}$$

$$W_0 = M * (1 - C_U - C_R - C_C - C_E),\ 0 \leq\ W_0 \leq\ M \tag{3}$$

$$LFI = \frac{V + W}{2M + \frac{W_F + W_C}{2}} \tag{4}$$

Nutrient Circularity Indicators

The two approaches to nutrient circularity in this study are the N and P recycling indexes, NRI and PRI, and MCI calculations as described above but following only the mass flows of N and P, the N-MCI and P-MCI, respectively.

The NRI (Eq. 5) is defined as the fraction of total nitrogen added to the defined system which is from recycled sources [36]. Added nitrogen is categorised as either imported (N_I) or recycled (N_R). This study also introduces the PRI (Eq. 6) as analogous to the NRI but focusing on imported P (P_I) and recycled P (P_R).

$$NRI = \frac{N_R}{N_R + N_I} \tag{5}$$

$$PRI = \frac{P_R}{P_R + P_I}$$

The MCI focusing on mass flows of N, the N-MCI, was calculated by Lavallais and Dunn [37] for products made from waste nitrogen from manure. The calculation of N-MCI is in principle analogous to that of the MCI but focusing on mass flows of nitrogen rather than total mass flows. Lavallais and Dunn [37] do not, however, base the calculation of virgin inputs (V) on the mass of the final product (M) as instructed by [41] (Eq. 1). Instead, and similarly to Rufí-Salís et al. [51], they appear to define M as the total mass of inputs (N inputs for the N-MCI). Additionally, Rufí-Salís et al. [51] input V as the known mass flows of virgin inputs instead of calculating V as in Eq. 1.

Taken together, these alternative definitions of M and V appear to have two important implications. First, V can be greater than M (compare to Eq. 1), meaning that for production processes where a significant part of virgin inputs does not end up in product mass, the entirety of virgin inputs is still included in the MCI. Second, for the N-MCI, it is the total input of N that is the basis for the MCI, and not the content of N in the product mass. This allows for calculating the N-MCI for products that contain little to no N, but that stem from agricultural or other processes which require inputs of N. Such products are included in the cases of this paper, and therefore, the alternative definition of M as total mass of inputs is used in the calculations of N-MCI and P-MCI in this study. However, V is not input manually as done by [51], but calculated from M as defined in Eq. 1, and with the redefinition of F_U, F_R, and F_S, as fractions of the total N inputs to the system. The calculation of P-MCI is analogous to that of N-MCI, but focusing on mass flows of P.

Carbon Footprint

In addition to the assessment of circularity, carbon footprints are calculated to illustrate the potential climate impact of the studied products and production systems. The goal of this assessment is to better understand the climate impact of the production systems in the case studies, how the emissions of greenhouse gases are distributed among processes and raw materials, and how the assessment of material and nutrient circularity compares to a climate impact assessment. The carbon footprint calculations follow the ISO 14044 standard for life cycle assessment, LCA [52], but focus only on well-mixed greenhouse gases and their resulting climate impact. No changes to surrounding supporting systems, such as grid electricity generation, are considered – similarly to the approach of the MCI.

The study departs from different functional units in each of the two studied cases. A functional unit is defined as the "*quantified performance of a product system for use as a reference unit*" [52, 53]. The performance provided by the production systems in the studied cases include both the performance, or functions, delivered by the considered main products, LDPE plastics and jet fuels, as well as their co-products. To present results for only one product (or function) from a production system with several co- or by-products, it is necessary to apply subdivision of processes by allocation, or by considering the replacement of alternative products in markets by substitution [54]. Such method choices can typically impact results and conclusions [55, 56]. For the calculations of carbon footprint in this study, such method choices are therefore avoided in line with the ISO 14044 order of priority, and carbon footprints are subsequently presented for a basket of products as further introduced in the following sections. An exception to this general approach is made by allowing for system expansion by substitution to illustrate the climate impact of different possible uses of certain co-products (see description of case studies below).

The biomass residues considered as feedstocks in the present case studies can also be considered valuable output streams from the production systems that they originate from. There is no motivation for excluding the climate impact of such upstream systems in analyses when the value of biomass residues is acknowledged [57, 58]. Therefore, calculations including the upstream production systems that produce the residual biomass are also performed, as explained further in relation to the individual case studies. For comparability, the same expanded calculation approach is also applied to the circularity indicators described in the previous sections.

The data needed for the carbon footprints is the same type of material flow data that the MCI is based on. In addition, information regarding the climate impacts of different inputs to the production systems is gathered from other sources, including scientific publications, relevant grey literature, and databases. The inventoried LCA data can be found in full in Online Resource 2. The resulting GHG emissions are weighed into a climate impact result using the factors for global warming potential over 100 years, the GWP100 [59, 60].

Despite the focus on GHG emissions and climate impact in this study, a broader understanding of sustainability is necessary for successful implementation of a circular bioeconomy. Climate change mitigation is crucial, but environmental sustainability also includes avoiding burden shifting and addressing biodiversity [25], including not only climate change, but also land use change and loss of habitat, as examples [61]. Other sustainability perspectives include the effect on economic viability and social equity [20]. The assessment of carbon footprints can therefore be considered an initial enquiry to better understand productions systems and indicators, but not a sustainability assessment.

Case Studies

The two cases examined in this paper are based on the production of LDPE from wheat straw, and the production of jet and diesel fuels from animal by-products. The cases thus differ both in terms of the type of residual biomass used and the type of end product. While wheat straw can be returned to the soil without treatment, animal by-products must undergo processing. Jet and diesel fuels are produced for energetic purposes, while LDPE can substitute fossil-based plastic material. The production systems and the sources of data used to define them are introduced in the following sections.

LDPE Production From Wheat Straw

The production of LDPE from wheat straw (Fig. 1) is assumed to be located in Sweden. The wheat (*Triticum* L.) straw is harvested and subsequently fed into an ethanol plant which produces ethanol and lignin pellets as a by-product. The removal of straw and subsequently its content of nitrogen, phosphorous, and potassium from the field is compensated for by additional fertilisation. A limited fraction of the straw is removed with the assumption that the removal has no negative impacts on levels of soil organic carbon. Both the cultivation and biofuel plant processes are modelled according to previously published data [62]. The ethanol is converted first into ethylene and then into LDPE in adjacent processes, which

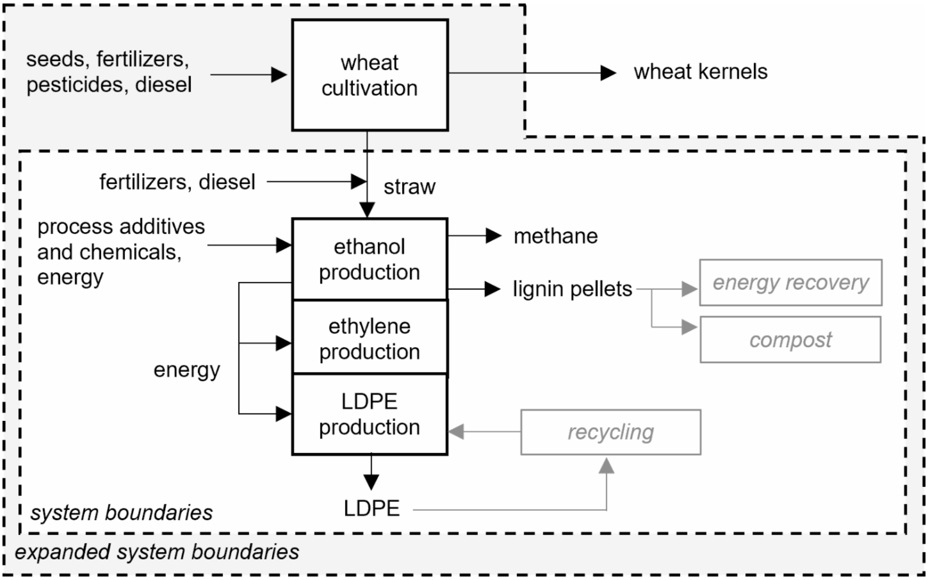

Fig. 1 LDPE (low density polyethylene) production from wheat straw. The figure shows the considered production system, including inputs, processes and products, with different system boundaries. Gray arrows and text represent the assumed uses of LDPE and of the co-product lignin pellets in alternative scenarios

allows for internally produced heat, electricity, and biogas from the ethanol plant to be used in ethylene and LDPE production. The ethylene and LDPE production processes are also based on published data [63]. The full inventory can be found in the Online Resource 2.

Two sets of system boundaries are used to delineate the system, as illustrated in Fig. 1. First, straw is assumed to enter the LDPE production system as a by-product biomass. The only processes included from the field are the fertilisation to compensate for nutrient removal by harvesting of straw, and the collection and transportation processes. As an alternative, the system boundaries are expanded to account also for the wheat cultivation phase which corresponds to a definition of straw as an agricultural co-product. In this case, 10% of the cultivation inputs such as fertilisers, seeds, and pesticides, are allocated to the straw, which is based on the economic value of straw in relation to wheat kernels (Online Resource 2). This allocation is made for the calculations of both circularity and carbon footprint.

Three alternative sub-cases are also investigated for the LDPE production, as indicated by the grey arrows in Fig. 1. In one case, lignin pellets are used for energy generation. In the other two, lignin pellets are composted and returned to soil, and LDPE is assumed to be recycled after use. When the lignin pellets are used for energy, no contribution is made to circularity, but the energy produced is assumed to replace Swedish district heating in the calculations of carbon footprint. When the lignin pellets are composted, they are assumed to consist of 1.3% N [64] and 0.37% P [65] by weight (dry matter). When composted, 40% of the P (assumption based on the fraction of labile P [65]) is assumed to be available to the local ecosystem. The availability of the added N is assumed to be limited in the short term [64], and as an approximation, none of the N content in straw is assumed to be available to the local ecosystem. In the sub-case where LDPE is assumed to be recycled, the recycling process is assumed to have a conversion efficiency of 82% with the remaining material ending up as losses [66]. A substitution ratio of 1:1 is assumed for the recycled LDPE and, consequently, the LDPE delivered in the recycling sub-case consists of 82% recycled LDPE and 18% virgin LDPE.

To enable the comparison of the three sub-cases, Table 1 illustrates how the co-products of the system and their performance are considered in the calculation of circularity indicators and carbon footprints, respectively. The circularity indicators are calculated for the full basket of co-products including LDPE (1 kg), methane (0.3 kg), and lignin pellets (2.1 kg),

Table 1 The products and functional units considered for calculations of the circularity indicators and carbon footprints, respectively. LDPE: low-density polyethylene, MBM: meat and bone meal

	Products	Products considered for circularity indicators	Functional unit for carbon footprint
LDPE production from wheat straw	1 kg LDPE	1 kg LDPE	1 kg LDPE
	0.3 kg methane	0.3 kg methane	0.3 kg methane
	2.1 kg lignin pellets	2.1 kg lignin pellets	
Jet fuel production from animal by-products	1 GJ jet fuel	1 GJ jet fuel	1 GJ jet fuel
	0.46 GJ diesel fuel	0.46 GJ diesel fuel	0.46 GJ diesel fuel
	2.8 kg naphtha	2.8 kg naphtha	2.8 kg naphtha
	40 kg MBM	40 kg MBM	
	16 kWh electricity		16 kWh electricity

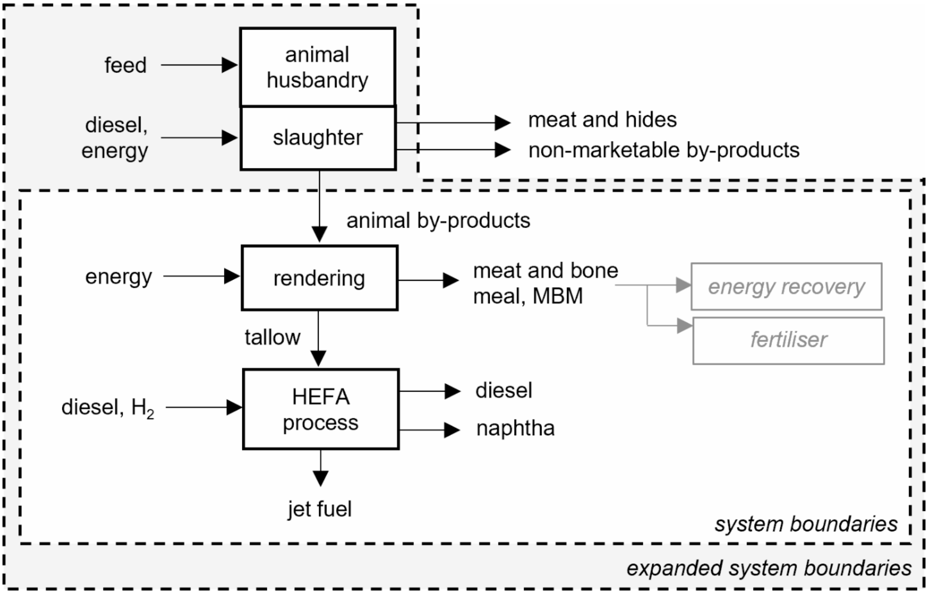

Fig. 2 HEFA (hydrotreated esters and fatty acids) fuel production from animal by-products. The figure shows the considered production system, including inputs and products, with different system boundaries. Gray arrows and text represent the assumed uses of the co-product meat and bone meal (MBM) in alternative scenarios

as there is no guidance for how to separate processes with more than one product outcome in the calculations [41]. For the calculation of the carbon footprint, the functional unit for which the system is assessed is 1 kg of LDPE and 0.3 kg of methane. This approach allows for considering the different uses of lignin pellets through system expansion with substitution, where the climate impact of the different products displaced by the pellets (Swedish district heating or mineral fertiliser) are included in the calculations. The different uses of the lignin pellets can thereby be assessed both in terms of material and nutrient circularity and climate impact.

Note that while the use phase of products, e.g. the conversion, retail and use of LDPE, is not included in the assessment, the fate of the products at end-of-life is included in the circularity indicators and therefore also in the calculations of carbon footprint. The combustion of the bio-based LDPE and methane is, however, assumed not to result in climate impact due to a simplified assumption of net carbon neutrality [67]. While such assumptions may be criticised, it is motivated in this study by the relatively short time-horizon of the considered biomass production systems.

HEFA Fuels From Animal By-Products

The case of HEFA (hydrotreated esters and fatty acids) jet fuel production from animal by-products concerns the rendering of animal by-products to produce tallow and meat and bone meal (MBM), and the subsequent processing of tallow in a HEFA process into jet fuel, diesel, and naphtha (Fig. 2). The production process is assumed to take place in Sweden, using animal by-products from Swedish beef cattle [68]. Data for the rendering process [69, 70] and for the hydrogenation or HEFA process of tallow [71] are taken from published studies that cover different geographic regions. For the calculations of carbon footprint, however, GHG emissions' data for Sweden are applied. The hydrogen needed for the HEFA process is assumed to be supplied through methane steam reforming of natural gas and using Swedish grid electricity for process energy. The full inventory can be found in Online Resource 2.

Two sets of system boundaries are used to delineate the system, as illustrated in Fig. 2. In the first alternative, the animal by-products enter the HEFA production system with 4% of the slaughterhouse processes allocated to them, based on the market value of different slaughterhouse products [70]. In the second alternative, the system boundaries are expanded to include the animal husbandry stage which corresponds to a definition of animal by-products as co-products. Therefore, 4% of the processes related to animal husbandry, including feed production, are also allocated to the animal by-products. These allocations are made for the calculations of both circularity and carbon footprint.

Two alternative scenarios are also assessed for the jet fuel production, as indicated by the grey arrows in Fig. 2. In the base case, MBM is incinerated with energy recovery. When the MBM is used for energy, no contribution is made to circularity, but the energy produced is assumed to replace Swedish district heating in the calculations of carbon footprint. In the second scenario, MBM is used as fertiliser and thus returned to soil. The MBM is assumed to consist of 8% N and 5% P by weight [72]. When used as fertiliser, 80% of the N content and 50% of the P content is assumed to have the potential to substitute mineral fertilisers [72], and thereby be available to the local ecosystem.

Similarly as for the LDPE production system, Table 1 illustrates how different co-products of the HEFA fuels production systems are considered in the calculation of circularity indicators and carbon footprint. The circularity indicators are calculated for the full basket of co-products including jet fuel (1 GJ), diesel fuel (0.46 GJ), naphtha (2.8 kg), and MBM (40 kg), as there is no guidance for separating processes with more than one product outcome in the calculations [41]. The mass-based circularity indicators do not include electricity as a product. For the calculation of the carbon footprint, the functional unit for which the system is assessed is 1 GJ jet fuel, 0.46 GJ diesel fuel, 2.8 kg naphtha, and 16 kWh electricity. This approach allows for considering the different uses of MBM through system expansion with substitution, where the climate impact of the different products displaced by MBM (Swedish district heating or mineral fertilisers) are included in the calculations. The different uses of the MBM can thereby be assessed both in terms of material and nutrient circularity and climate impact.

While the products included in the functional unit (Table 1) are mainly considered for energetic uses, the combustion of the bio-based materials are considered climate neutral, similarly as in the LDPE production case.

Results and Discussion

The results in terms of circularity scores and carbon footprints are presented and discussed in the following sections. First, the different results for the two individual case studies are presented, followed by common points for discussion.

LDPE From Wheat Straw

The different circularity and carbon footprint results for LDPE derived from wheat straw are summarised in Table 2, and circularity scores are further illustrated in Fig. 3. The circularity scores can be compared to a theoretical maximum of 1 (or 100%) for perfectly circular products or systems. The minimum level for the MCI indicators is instead 0.1 in this study.

Material Circularity

The material circularity of the LDPE production, which is assessed with the MCI, varies with the different assumed system boundaries and options for composting and recycling (Fig. 3). The MCI for the LDPE is 0.53 in the base case and increases to 0.80 with the composting of pellets or 0.64 with the recycling of LDPE. The composting of pellets thus provides a larger contribution to circularity than does recycling of LDPE with these calculations because the mass of pellets considered for composting is greater than the mass of LDPE considered for recycling. With the expanded system boundaries that include the wheat cultivation stage, the MCI is lowered significantly in all scenarios, and decreases from 0.53 to 0.21 in the base case.

Both the composting of pellets and the recycling of LDPE increases the circularity of the system, but the circularity scores depend on the applied system boundaries. The difference between the scores for each scenario can be explained by the difference in F_R, the fraction of feedstock from recycled sources. When the system boundaries are set at the collection of straw, straw is the main input to the LDPE production system in terms of mass. When the system boundaries are expanded to include the cultivation phase, however, mineral fertilisers are instead the dominating inputs in terms of mass. While straw is considered a recycled resource, mineral fertilisers are not. There is no change in the mass flows of pellets for composting and LDPE for recycling. These mass flows, however, make up a smaller fraction of the total mass flows of the system when cultivation is included, and therefore the circularity scores are lower when the system boundaries are expanded.

Table 2 Results for LDPE production from wheat straw: circularity scores and carbon footprints

	System boundaries and scenario					
	System boundaries: straw			System boundaries: expanded		
	Base case	Pellets composted	LDPE recycled	Base case	Pellets composted	LDPE recycled
MCI	0.53	0.80	0.64	0.21	0.48	0.31
MCI-N	0.28	0.28	no change	0.10	0.10	no change
NRI	0%	0%	no change	0%	0%	no change
MCI-P	0.21	0.31	no change	0.10	0.16	no change
PRI	0%	18%	no change	0%	12%	no change
Carbon footprint (kg CO_2-eq. per 1 kg LDPE with co-products)	0.80	1.4	0.47	1.1	1.7	0.53

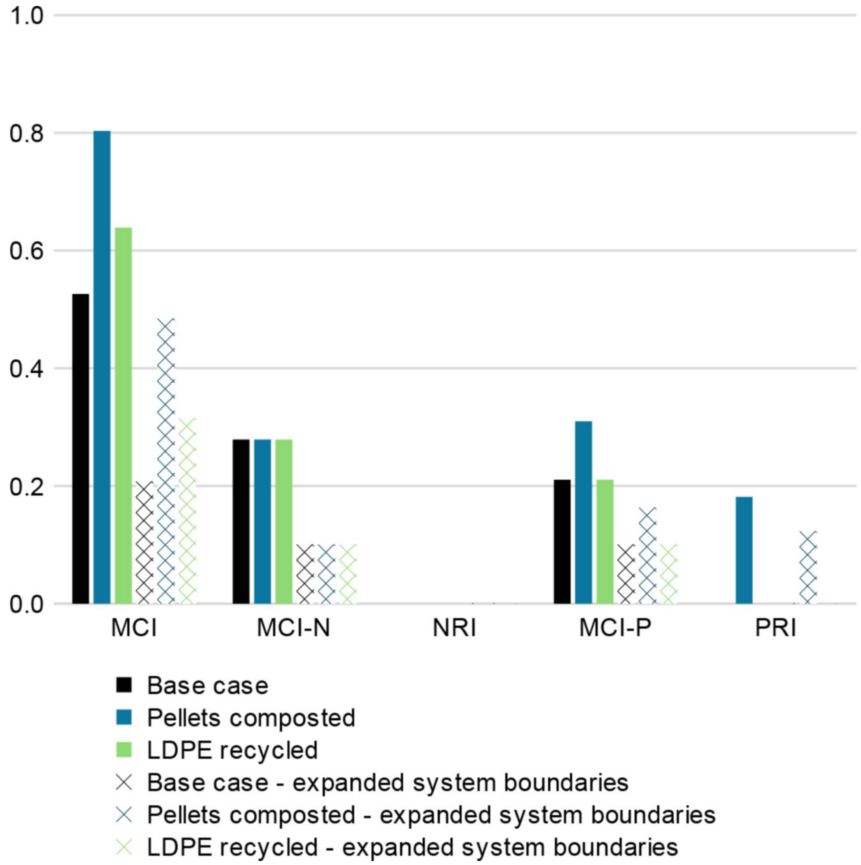

Fig. 3 Illustration of circularity scores for LDPE production from wheat straw using different circularity indicators. Solid bars show scores where the system boundaries depart from straw as feedstock. Dotted bars instead show scores where the cultivation of wheat is included inside the system boundaries

Nutrient Circularity

The nutrient circularity scores follow a similar pattern as the MCI when comparing the calculations that apply different system boundaries. The calculations that include the wheat cultivation, and hence the use of mineral fertilisers, result in MCI-N and MCI-P scores of 0.1, corresponding to linear production systems. This is because all the N and P added is from virgin resources, and no N or P is returned to ecosystems in the base case. When straw is considered the main input to the LDPE production, and being a recycled input, the circularity scores are instead 0.28 for MCI-N, and 0.21 for MCI-P. To clarify, this increase in the nutrient circularity scores is not due to an improvement of the studied production system, but a result of different choices of system boundaries that do, or do not, include the cultivation phase for straw. In addition, the allocation factor that was applied in the calculations including wheat cultivation, and that allowed for assessing the circularity of this system without including the entire wheat production with wheat products, also has the potential to affect these circularity scores.

Neither of the studied scenarios with composting and recycling improves the N circularity (Fig. 3). For the composting of pellets, the MCI-N and NRI scores are not improved as no N from the pellets was assumed to be available in the short term. For the recycling of LDPE, neither the N nor the P circularity scores are improved as the recycling does not result in any nutrients being returned to an ecosystem. For the P circularity, however, the composting of pellets increases the MCI-P from 0.21 to 0.31 and the PRI from 0 to 18%, or the MCI-P from 0.10 to 0.16 and the PRI from 0 to 12% when the wheat cultivation is included. Overall, the PRI scores follow a pattern similar to that of the MCI-P scores.

When the material circularity is compared to the nutrient circularity, the MCI-N and MCI-P scores are consistently lower than the MCI scores (Fig. 3). For instance, the composting of pellets results in an MCI score of 0.48 or 0.80, depending on the chosen system boundaries, while the MCI-N scores are 0.10 or 0.28, and the MCI-P scores are 0.16 or 0.31. The circularity of N and P can thus be considered lower than the circularity of mass in general.

Carbon Footprint

The carbon footprint of the LDPE production system also varies between the scenarios and with the applied system boundaries (Table 2). Here, it is the GHG substitution potential of the composted or recycled material that matters in the calculations, as compared to the mass of composted or recycled material in the MCI calculations. Similarly to the material and nutrient circularity scores, the inclusion of the cultivation phase increases the carbon footprint of the system, from 0.80 to 1.1 kg CO_2-eq. in the base case. Contrastingly to the circularity scores, however, the composting of pellets does not improve the carbon footprint of the system. This is because the GHG reduction potential from using pellets for energy, and substituting Swedish district heating, is assumed to be greater than the reduction potential from composting of pellets and substituting a relatively small amount of P fertiliser. In the scenario with LDPE recycling, the carbon footprint is lower than in the base case. The fact that one strategy is not necessarily beneficial from both the perspectives of circularity and climate impact is not surprising. While the circularity scores concern mass flows of the system, the carbon footprint may consider the function of a material on a wider market, and alternative products for delivering that same function. The combination of circularity scores and carbon footprints can therefore show potential trade-offs and synergies between goals of circularity and climate-change mitigation.

The absolute carbon footprint of the LDPE production system is dependent on different assumptions and varies from the base case to the scenarios including composting and recycling but appears to be lower than that of a fossil-based comparator. The carbon footprint of 1 kg LDPE and 0.3 kg of methane varies from 0.80 kg CO_2-eq. to 1.1 kg CO_2-eq. (depending on system boundaries), or as high as 1.4 to 1.7 kg CO_2-eq. when pellets are assumed to be composted and thus do not substitute Swedish district heating. An indicative fossil comparator may be created by combining the production and combustion of LDPE from fossil resources, approximately at 4.7 kg CO_2-eq. per kg, and natural gas, resulting in approximately 5.7 kg CO_2-eq. in total (see Online Resource 2). More detailed analyses should, however, be conducted to enhance comprehension of the potential for climate change mitigation.

HEFA Jet Fuel from Animal By-Products

The different circularity and carbon footprint results for jet fuel derived from animal by-products are summarised in Table 3, and circularity scores are further illustrated in Fig. 4. Just as in the previous case, a circularity score of 1 corresponds to a perfectly circular production system.

Material Circularity

The assumed system boundaries affect the MCI scores for the jet fuel production system. When the animal by-products are considered a recycled input to the system, the MCI score is 0.55. However, when the system boundaries are expanded to include the animal husbandry stage, the MCI score drops to 0.16 (to be compared to a theoretical linear production system with an MCI score of 0.1). This can be explained by the material flows that are considered the inputs to the system. First, when the animal by-products are considered the inputs to the system, and because they are considered recycled materials, the fraction of material inputs which are recycled (F_R) is 0.99 (calculations are available in Online Resource 2). When the system boundaries are expanded to include the animal husbandry, the feed is instead considered the input to the system. In the assumed production system, the feed is mainly from virgin and not recycled (or by-product) resources, and therefore, F_R is significantly lower – in this case 0.13. It is thus not the increase in sheer mass of the material inputs to the system that lowers the MCI score when the animal husbandry is included, but the decrease in F_R.

Similarly, the contribution to circularity from applying MBM as a fertiliser also varies with the assumed system boundaries, but the strategy to apply MBM as a fertiliser improves the overall circularity score of the HEFA fuels (Fig. 4). By applying MBM as a fertiliser, the MCI score can be improved from 0.55 to 0.78, or from 0.16 to 0.40 when the animal husbandry is included.

Nutrient Circularity

A similar pattern as with the MCI can be seen for nutrient circularity when the assumed system boundaries are alternated. An expansion of system boundaries to include the animal husbandry decreases the MCI-N and MCI-P scores from 0.55 to 0.10, due to the assumed

Table 3 Results for HEFA fuel production from animal by-products: circularity scores and carbon footprints

	System boundaries and scenario			
	System boundaries: animal by-products		System boundaries: expanded	
	Base case	MBM as fertiliser	Base case	MBM as fertiliser
MCI	0.55	0.78	0.16	0.40
MCI-N	0.55	0.89	0.10	0.10
NRI	0%	43%	0%	0.50%
MCI-P	0.55	0.76	0.10	0.11
PRI	0%	32%	0%	1.9%
Carbon footprint (kg CO_2-eq. per 1 GJ jet fuel with co-products)	22	16	419	413

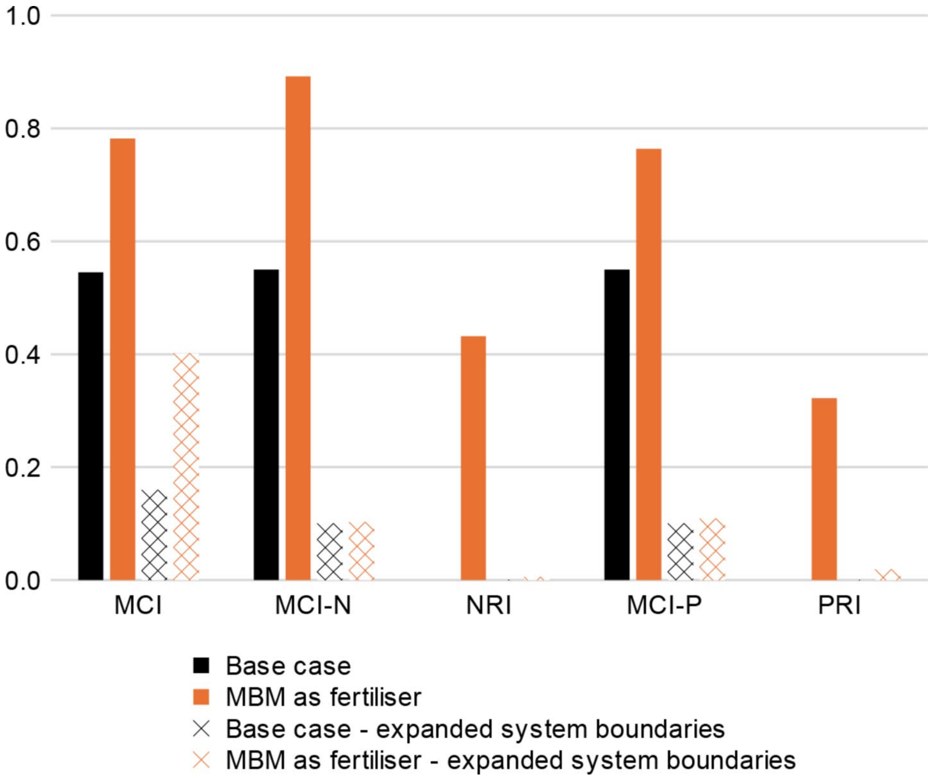

Fig. 4 Illustration of circularity scores for HEFA fuel production from animal by-products using different circularity indicators. Solid bars show scores where the system boundaries depart from animal by-products as feedstock. Dotted bars instead show scores where the animal husbandry is included inside the system boundaries

use of mineral fertilisers in the production of animal feed. The assumed system boundaries also affect the contribution to circularity from using MBM as a fertiliser (Fig. 4). When the animal husbandry is excluded, the use of MBM as a fertiliser results in relatively high circularity scores of 0.89 for MCI-N, and 0.76 for MCI-P. When the animal husbandry stage is included, the mass of N and P inputs to the system is larger, but the mass of N and P in the MBM used as fertiliser remains unchanged. This leads to lowered circularity scores: 0.10 for MCI-N and 0.11 for MCI-P, to be compared with the theoretical score of 0.1 for a perfectly linear production system.

The NRI and PRI indicators partly conflict and partly reflect the MCI-N and MCI-P scores. Unlike the MCI indicators, the NRI and PRI only consider the recycling of nutrients at end-of-life, but not the share of reused or recycled nutrient inputs. Since the base case does not include any recycling of N and P, the NRI and PRI scores therefore show perfect linearity (0%) for the HEFA production system in this case. Similarly to the MCI-N and MCI-P scores, the use of MBM as a fertiliser increases the NRI and PRI scores, but the significance of the contribution to circularity differs with the assumed system boundaries. When the animal husbandry is excluded, the NRI is 43% and the PRI is 32% when MBM is applied as a fertiliser. When the animal husbandry and thus the use of mineral fertilisers for feed production is included, however, the scores are instead 0.5% for the NRI and 1.9% for the PRI.

Unlike the case of LDPE production from straw, where nutrient circularity scores were consistently lower than material circularity scores, the comparison between material and nutrient circularity is not as clear for the HEFA production system. When the animal husbandry is not included, The MCI-N and MCI-P scores are approximately equal to, or higher than, the MCI scores for both the base case and the scenario including MBM as a fertiliser. When the system boundaries are expanded to include the animal husbandry, however, the nutrient circularity scores are lower than the material circularity scores. Here, the improvement of the MCI-N and MCI-P scores from using MBM as a fertiliser is barely visible.

Carbon Footprint

The strategy to use MBM as a fertiliser can potentially have a positive impact on both the circularities and the carbon footprint of HEFA products, but the size of the carbon footprint is highly sensitive to the assumed system boundaries. The carbon footprint is 22–16 kg CO_2-eq. when the animal husbandry is excluded and depending on the use of MBM as a fertiliser (Table 3). If the animal husbandry is instead included, this stage contributes the majority of GHG emissions and results in significantly higher carbon footprints of around 410 to 420 kg CO_2-eq. The animal husbandry is included in the production of animal by-products by use of an economic allocation factor of 4% to animal byproducts at the slaughterhouse. Both this factor, and the characteristics of the considered animal husbandry, are therefore highly sensitive parameters that could give rise to present and future variation to the carbon footprint of HEFA fuels. The present calculations indicate that the GHG mitigation potential of the HEFA production system compared to a comparable fossil-based production system at approximately 140 kg CO_2-eq. (see Online Resource 2 for full calculations), is highly sensitive to process designs, method choices and assumptions. Considering the extended system boundaries in further assessments of this and similar production systems using animal by-products are warranted, both regarding carbon footprints and circularities.

Discussion

Interpreting Nutrient Circularity and Availability

For the calculations of results reported in Tables 2 and 3, only the assumed plant available N and P contribute to the local ecosystem and, thus, to circularity as defined in the circularity indicators considering N and P. This could be considered in line with the intention of the Ellen MacArthur Foundation and ANSYS Granta [41, p.25] with the MCI in the sense that *"(...) the composting of biological materials may be treated as being up to 100% efficient depending upon the application of the resulting solid and liquid nutrients to specific ecosystems and the degree to which these are retained by those ecosystems*, *taking into account losses through leaching and run-off post-application.*" The nutrient circularity scores are however sensitive, and differently sensitive to the assumed nutrient availability factors. For instance, if all the P in MBM was to be considered for nutrient circularity, the use of MBM as a fertiliser results in an MCI-P score of 0.98 (as compared to 0.76) for the HEFA jet fuel production system, and a PRI score of 49% (compared to 32%) when the animal husbandry is excluded (for the full set of results, see Online Resource 2). When the animal husbandry is included, the change in the MCI-P score from changing the nutrient availability factor is barely visible, resulting in a score of 0.12 as compared to 0.11. Similarly, the PRI is 3.6% as compared to 1.9%.

This discussion raises the question of what a perfectly circular bio-based product or production system would look like. Consider a production system in which the total P or N content in the extracted biomass feedstock is recycled back to soil. The system could be considered perfectly circular, but it is not likely that all the returned N and P would be made available to the next generation of plants. Circularity indicators that only consider plant available nutrients would thus not reflect such perfect circularity. On the one hand, when leaching and other losses of nutrients are considered, both the value of recirculating different biobased materials to soils, and the potential environmental impacts related to such nutrient losses, can be considered. On the other hand, it may be that no bio-based production system can be considered perfectly circular using the definitions implied by the circularity indicators applied here. This is not necessarily an issue per se, but it affects how nutrient circularity scores should be interpreted. The existing definition of circularity for biobased products could therefore be further detailed to clarify what a perfectly circular biobased production system implies, and in which cases it is important to include factors describing e.g. plant availability in the calculation of nutrient circularity indicators.

Developing Better Indicators for Bio-Circularity

The reviewed literature seems to emphasise the essential ecological cycles as prioritised over the potential feedstock uses of biomass to support strong circular bioeconomies while circularity indicators for bio-based products appear, thus far, to focus more on product or material cycles. While circularity indicators such as the MCI may include the possibility to assess both product and ecological circularity, it provides no assistance in dealing with or interpreting potential goal conflicts. A possible way forward may be to assess both types of circularities separately by combining several indicators to make visible the interplay between different uses of residual biomass and different types and goals of circularity. Such a set of indicators may thus better reflect and measure bio-based circularity as it is currently understood.

Even with a varied set of indicators, additional guidance and clarification may also be needed in terms of defining system boundaries. The inclusion of the upstream processes of biomass residues, the cultivation of straw and the animal husbandry in the cases studied in this paper, has a significant impact on all circularity scores in this study. To some extent it also affects the studied carbon footprints, especially in the HEFA fuels production case. Here it may be possible to draw links to similar discussions on system boundaries in LCA and carbon footprints [57, 73, 74]. As a first step, the difficulty in assessing single products that are connected to other products in material flows and processes should be acknowledged and addressed in guidance for product-level circularity indicators. The inclusion of sensitivity analyses to identify and illustrate uncertainty and variability of results due to method choices, similarly to how it is practiced in the field of LCA, would also enhance circularity assessments that aim at increasing understanding and developing indicators for bio-based circularity.

Conclusions

Different types of circularities should be acknowledged for biomass, and residual biomass in particular. An aim for both ecological or nutrient, and product or material circularities may be necessary in a reorientation towards circular bioeconomies. It is, however, also necessary to acknowledge and clarify that these are different and may, in some cases, constitute conflicting goals. For instance, the studied case of LDPE production from wheat straw showed consequently lower scores for nutrient circularity than for material circularity, while scores were more even for the studied HEFA fuels production system. Focusing only on material circularity for biobased products can thus be misleading, and especially problematic for production systems that are based on residual biomass which is often considered to have the role of closing loops of nutrient and organic matter to support long-term soil health and productivity.

A combination of methods can shed light on this duality, but a wider discussion on what bio-based and nutrient circularity entails is, however, needed to facilitate the use and interpretation of circularity indicators. Overall, the circularity indicators applied in this study appear to be rough tools for assessments of circularity, partly lacking guidance on how to manage complex bio-based production systems with many co-products, and sensitive method choices. For instance, the circularity scores for the studied LDPE and HEFA fuels production systems were significantly lowered when the system boundaries for the residual biomass feedstocks were alternated. This variation, in combination with a comparison of carbon footprints, provided different and at times conflicting insights into potential improvements to the studied production systems. A continued discussion on what circularity may entail for biomass and biomass residues, and how it can be measured, is therefore needed to further develop useful indicators for bio-based circularity and circular bioeconomies.

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Data availability The datasets supporting the conclusions of this article are included within the article and its supplementary material.

Declarations

Competing Interests The author states that there is no conflict of interest.

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