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POLICIES AND SUPPORT IN RELATION TO LCA

Evaluating the Circular Footprint Formula (CFF) in the environmental assessment of mono-material carpets recycling: a case study approach

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Abstract

Purpose The purpose of this study is to apply and evaluate the Circular Footprint Formula (CFF), as part of the European Commission's Product Environmental Footprint (PEF) framework, through an assessment of the environmental performance of thermomechanically recycled mono-material polyester carpets. Unlike multi-material carpets that are often incinerated at the end of their life, mono-material designs facilitate fiber-to-fiber recycling - a key priority for the European Union in reducing the large environmental burdens of the textile industry.

Methods The environmental assessment is structured around the CFF as recycling allocation method. In addition, a comparison is made with the 100:0 approach as a benchmark, also known as the cut-off or recycled content method, as recommended in the Environmental Product Declaration (EPD). Different potential values for two main parameters within the CFF were considered: factor A, reflecting the market dynamics of supply and demand for recycled materials, and factor B, distributing the burdens and credits associated with energy recovery. Both allocation methods were applied to the carpet case study, which considered three possible scenarios: recycling of the monomaterial carpet (MMC), incineration of the conventional multi-material carpet, and incineration of the MMC. The impact assessment follows the indicators defined by PEF framework.

Results and discussion The findings indicate that recycling of MMCs generally results in lower environmental impacts than direct incineration across most impact categories, thereby aligning with the principles of a circular economy. Importantly, incinerating MMCs without prior recycling can lead to more severe environmental burdens than the incineration of conventional carpets, as demonstrated through a comparative product system-level analysis. The results also suggest that factor A has a significantly greater influence on environmental outcomes at individual product level than factor B, with changes in A, for example, leading to impact differences of up to 30% for Climate Change. Additionally, yarn thickness (titer) plays an important role, as a shift from very fine to coarse yarns altered the relative environmental performance of the product systems.

Conclusions Methodological enhancements are recommended to improve the accuracy of environmental impact assessments for textile recycling scenarios when using the CFF within the PEF framework. The high sensitivity to factor A emphasizes the need for a scientifically robust selection process. Moreover, the substantial influence of yarn thickness underscores the importance of developing more detailed Environmental Footprint (EF) datasets to achieve more consistent and reliable sustainability evaluations in the textile sector.

Keywords Product Environmental Footprint (PEF) · Circular Footprint Formula (CFF) · Polyester · Thermomechanical recycling · Monomaterial design · Textiles · Carpets

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1 Introduction

Increasing concerns about climate change, resource depletion, and global pollution have created a greater need to quantify the environmental impacts of products and services. This is based on the principle that effective management requires accurate measurement.

Life Cycle Assessment or LCA is a widely recognized holistic scientific method to calculate and evaluate the potential environmental impacts associated with a product, process, or service throughout its life cycle (European Commission (2013a). It is standardized internationally through the ISO 14040 series (ISO 14040 2006; ISO 14044 (2006). The latter is however not a certification standard but rather a guideline standard that specifies the principles and framework for conducting an LCA. The standard is quite open, allowing for choices in certain areas, such as allocation methods or impact characterization factors. Finkbeiner (2014) describes it as the 'constitution' of LCA, from which various spin-off standards and methodologies have emerged.

One of these LCA-based methodologies is the Product Environmental Footprint (PEF), developed by the European Commission who introduced the PEF method in 2013 as part of a Communication titled "Building the Single Market for Green Products" (European Commission 2013b; Finkbeiner 2014). The aim of PEF is to offer a standardized tool that enables assessment of the environmental impact of every product, process, or service using consistent and comparable criteria across the European Union. The PEF pilot phase concluded in 2018, marking the start of the transition phase, which was completed in 2025. Revisions and updates to the PEF methodology have continued beyond this period and remain ongoing as of 2026. The main objective of the transition phase, as well as the current revision process, is to further refine the existing PEF Category Rules (PEFCRs) and to strengthen the overall methodological framework (Ecochain 2025; Vieira 2020). PEFCRs help to shift the focus of the PEF study towards those aspects and parameters that matter the most for the product category under consideration (European Commission 2024a).

These developments are setting the stage for future policy initiatives and broader industry uptake, as the PEF could potentially become mandatory through regulatory instruments for certain sectors (Mordaschew and Tackenberg 2024). For instance, the PEFCR for rechargeable batteries already acts as basis for the carbon footprint declaration mandated by the battery regulation (European Commission 2024b).

PEF is often mentioned together with the Environmental Product Declaration or EPD. Several program operators exist, where the biggest one in terms of published EPDs is EPD International, founded in 1998 by the Swedish Environmental Protection Agency and industry. The system is now owned by EPD International AB and is operated in accordance with several standards, including ISO 14,025, ISO/TS 14,027, ISO 14,040, ISO 14,044, and ISO 14,067. For construction products, EPD international has adopted the European standard EN 15,804 (A1 and A2) as well as ISO 21,930 (International EPD System). It is important to be aware that EPDs have already become a well-established tool in the construction sector, where they play a crucial and relevant role (Durão et al. 2020). This in contrast to PEF, which is still in the development phase and has yet to see widespread application.

PEF and EPD are thus two major LCA-based methodologies with a shared goal: provide a single, consistent LCA toolbox with strict rules, unlike the ISO 14,040 core series that offers freedom of choice. However, PEF and EPD each have different content, resulting in significant differences in the assessment of environmental performance. One of the most significant differences is how they handle recycled materials and products. In LCA, recycling of material from one product to another presents an allocation issue, as the same material is used across multiple products. The method selected for modelling material recycling can have a large influence on the environmental impact evaluation of the different products along the recycling chain (Ekvall et al. 2020). The burdens and potential benefits of the involved materials and the corresponding recycling process can be divided in different ways between the supplier and user of recycled materials. There are various allocation methods in use to deal with this matter, but they can generally be categorized into three principal approaches: the 100:0 approach (also called cut-off or recycled content approach), the 0:100 approach (also called avoided burden or substitution approach) and the 50:50 approach (Corona et al. 2019), as explained later in the Materials and Methods section. EPD follows the rather straightforward 100:0 approach, which attributes the burden of virgin production to the first life cycle of the product, based on the 'polluter pays' principle. Additionally, EPDs align with EN 15804, in which a simplified form of system expansion is used, reporting the net environmental impacts of secondary functions separately under module D (Eberhardt et al. 2020a). The 0:100 approach considers the benefits of recycling and assigns them to first life cycle. In the 50:50 approach, the burdens and benefits are equally divided between the first and second life cycle.

The PEF builds on the 50:50 approach and extends it by introducing three factors: one that addresses material quality - to discourage downcycling; a second that reflects the market dynamics of supply and demand for recycled materials - to balance incentives for recycled content with those for end-of-life (EoL) recovery; and a third that aims to include the avoided burdens of possible disposal scenarios - to promote recycling (Obrecht et al. 2021; Zampori and Pant 2019). This is expressed by the Circular Footprint Formula (CFF). The CFF is quite complex and few case studies are available yet that illustrate and evaluate its applicability: Rickert and Ciroth (2020) for example applied the CFF on a hypothetical case study about intermediate paper products. Eberhardt et al. (2020b) used the formula to assess the environmental impacts of building components, whereas Hermansson et al. (2022) used it to examine the recycling process of carbon fiber composites. Farrapo et al. (2023) conducted a thorough analysis on briquettes made from recovered biomass, functioning as solid biofuel. Further, the 'Battery Pass consortium' evaluated a battery recycling scenario (Braunfels and Teuber 2023). The most recent study found is that of Hammar et al. (2024), focused on clothing made from chemically recycled cotton.

By fully applying the CFF in case studies across different application areas, specific issues emerge, demonstrating that a one-size-fits-all approach is not sufficient. The work by Farrapo et al. (2023) for example showed that in the context of solid biofuels, some critical aspects related to biological cycles are not addressed by the CFF, such as carbon and water balances. They also pointed out that for energy-based functional units, the section of the CFF related to energy recovery needs to be converted from a mass-based to an energy-based expression, to avoid double counting the material's lower heating value (LHV). The Battery Pass consortium study (Braunfels and Teuber 2023) identified problems with the predefined default parameter A, which the authors contend is not appropriate for battery applications and has a significant impact on the final results. Values for parameter A should be taken from Annex C (2022b); however, default values are missing for many materials and applications, a limitation that affected this paper's case study as well. Hermansson et al. (2022) also emphasized the considerable impact of factor A and its ineffectiveness when markets for secondary materials are not yet in place. In line with this insight, Eberhardt et al. (2020b) noted that using the CFF is questionable for products with a long lifespan, such as building components. Assigning impacts to potential future cycles of secondary product, sometimes over 100 years ahead, could result in industry 'greenwashing'.

This paper will explore the usability of the CFF by applying it to a textile recycling case, building further on the work of Hammar et al. (2024). The textile industry is a major polluter, causing significant environmental impacts due to its high energy, water, and chemical use, as well as the production of waste and microplastics (Leal Filho et al. 2022). Textile consumption in the EU for example ranks fourth in terms of climate change-related impacts, after food, housing, and mobility. It is also the third largest consumer of water and land, and the fifth largest in terms of raw material consumption. Successful closed-loop recycling of textile waste would mark a crucial step forward in global initiatives aimed at reducing the environmental burdens of the textile industry (Haslinger et al. 2019).

This aligns with the vision of the European Commission, as outlined in the EU Strategy for Sustainable and Circular Textiles published in March 2022. Among other points, it states that "The circular textiles ecosystem is thriving, driven by sufficient capacities for innovative fiber-to-fiber recycling, while the incineration and landfilling of textiles is reduced to the minimum." (European Commission 2022a).

In the context of promoting a more sustainable textile industry, this paper introduces a case study focused on carpet recycling, particularly the recycling of a mono-material carpet composed almost entirely of polyester (PET). This uniformity simplifies recycling and aligns well with the principles of a circular economy. In contrast, a conventional carpet consists of multiple layers and materials, where the complexity of separating these materials makes recycling nearly impossible. Typically, these conventional carpets end up being disposed through incineration with energy recovery. In contrast, a mono-material PET carpet can be recycled through a thermomechanical process, which involves re-melting thermoplastic waste materials (Stubbe et al. 2024).

The objective is not to perform a fully PEF-compliant LCA but rather to evaluate and demonstrate the applicability of the CFF in this fiber-to-fiber textile recycling scenario, with a specific focus on thermomechanically recycled PET. The recycling scenario of the mono-material carpet is also evaluated in comparison to that of the conventional alternative, which is a carpet composed of a mix of cotton and PET and discarded to incineration at the end of its life cycle.

To conduct this analysis, the general guidelines of the PEF as specified in Commission Recommendation 2021/2279 of December 15th, 2021, (Santos-Beneit et al. 2023) were used as basis. They were supplemented by the only available PEFCR relevant for textiles - specifically, the PEFCR Apparel and Footwear (Quantis 2025), which, at the time of writing this manuscript, was still under development and had only recently been approved by the European Commission. Furthermore, an approach following the EPD guidelines was included for comparison as a benchmark.

In terms of textiles, this case study, focused on the thermomechanical recycling of synthetic PET used in carpets, complements the study by Hammar et al. (2024), which focuses on the chemical recycling of natural cotton fibers used in clothing. Similar to all the previously mentioned studies (Braunfels and Teuber 2023; Farrapo et al. 2023; Rickert and Ciroth 2020), a thoroughly developed application of the CFF through a real case study can highlight specific needs and areas of focus, which could lead to improvements in methodology and practices that can be incorporated into the PEFCRs.

2 Materials and methods

2.1 The Circular Footprint Formula (CFF) – data inventory

As mentioned in the introduction, combining materials from different products can create allocation challenges, especially in recycling, which has been treated as a unique allocation issue requiring specific strategies since the early nineties (Toniolo et al. 2017). The following three strategies can be considered the main ones: the 100:0 approach, the 0:100 approach and the 50:50 approach.

In the 100:0 approach, also called cut-off or recycled content approach, the environmental burdens of virgin production are attributed to the primary product, i.e. the product of the first life cycle, receives, while the environmental burdens of recycling are assigned entirely to the secondary product, i.e. the product receiving the recycled material (Allacker et al. 2017; Obrecht et al. 2021). The primary product thus receives neither the credits nor burdens of recycling. This approach follows the 'polluter pays' principle and aligns with conventional business boundaries, making it relatively simple and straightforward to implement (Corona et al. 2019). It is also the method recommended by the EPD and will therefore be used in this study as benchmark for comparing the CFF results.

In the 0:100 approach, also called substitution, avoided burden or allocation to material losses approach (Allacker et al. 2017; Obrecht et al. 2021), the environmental burdens associated with the recycling process are assigned to the primary product. This first product also receives a credit for the amount of virgin material production avoided through recycling. The secondary product, in turn, carries the burden of the virgin material displaced in the first life. For instance, if the production of the primary product requires 1 kg of virgin material, and 0.8 kg of this material can be recycled for use in a secondary product, then only 0.2 kg of additional virgin material is needed for the secondary product. This 0.2 kg is credited as an avoided burden in the assessment of the primary product.

In closed-loop systems, the same type of virgin material is assumed for both the primary and secondary product. In open-loop systems, however, the recycled material may substitute a different virgin material (e.g. PET replacing cotton). Additionally, the secondary product bears the environmental impacts of its own EoL phase (Ekvall et al. 2020). In a less conventional reinterpretation of this approach, referred to as allocation to virgin material use by Ekvall et al. (2020), the environmental burdens associated with the EoL stage are attributed to the primary product, while the impacts related to the recycling process are assigned to the secondary product. When the EoL treatment involves incineration with energy recovery, credits for avoided energy production may also be allocated to the primary product.

In general, the 0:100 approach tends to promote design for recycling, while the 100:0 approach is more focused on incentivizing the use of recycled content. The 50:50 or equal-share approach combines both perspectives by evenly distributing the burdens of both virgin production and recycling, as well as the credits from avoided virgin material and energy production, between the primary and secondary products (Fig. 1).

Several variants of these methods have emerged, described in detail by Allacker et al. (2014; 2017) and Ekvall et al. (2020). One variant is the Circular Footprint Formula (CFF), developed in the PEF methodology (European Commission 2022a; Zampori and Pant 2019), which builds upon the 50:50 approach (Ekvall et al. 2020).

The CFF integrates a material component, an energy component, and a disposal component, see Eq. (1). Together, these three components form the inventory E_i of studied product i, covering all specific emissions and resource consumption per functional unit (where i = 1 represents the first or primary product, i = 2 the second product, and so on).

$$E_i = E_{i,material} + E_{i,energy} + E_{i,disposal} \tag{1}$$

Equation (2) to (4) represent each individual component:

$$\begin{aligned} E_{i,material} = {} & (1 - R_{i,1})^* E_v + R_{i,1} \\ & * \left(A^* E_{R,(i-1)} + (1 - A)^* E_v{}^* \frac{Q_i}{Q_p} \right) \\ & + (1 - A)^* R_{i,2} * \left(E_{R,i} - E^*_{v,i}{}^* \frac{Q_{(i+1)}}{Q_p} \right) \end{aligned} \tag{2}$$

$$\begin{aligned} E_{i,energy} = {} & (1 - B) * R_{i,3} * (E_{ER} - LHV * X_{ER,heat} \\ & * E_{SE,heat} - LHV * X_{ER,elec} * E_{SE,elec}) \end{aligned} \tag{3}$$

$$E_{i,disposal} = (1 - R_{i,2} - R_{i,3}) E_D \tag{4}$$

This complex formula accounts for the recycled content (R_1), the recycling rate (R_2), the portion of material directed to disposal—typically incineration with energy recovery (R_3), the material quality before and after each recycling process (Q), the balance between supply and demand for recycled materials (A), the allocation for energy recovery (B), as well as the life cycle inventories of virgin production (E_v), recycling (E_R), and disposal (E_{ER}, E_D). Furthermore, the life cycle inventories of substituted energy sources (E_{SE}) are included, utilizing the LHV of the material going

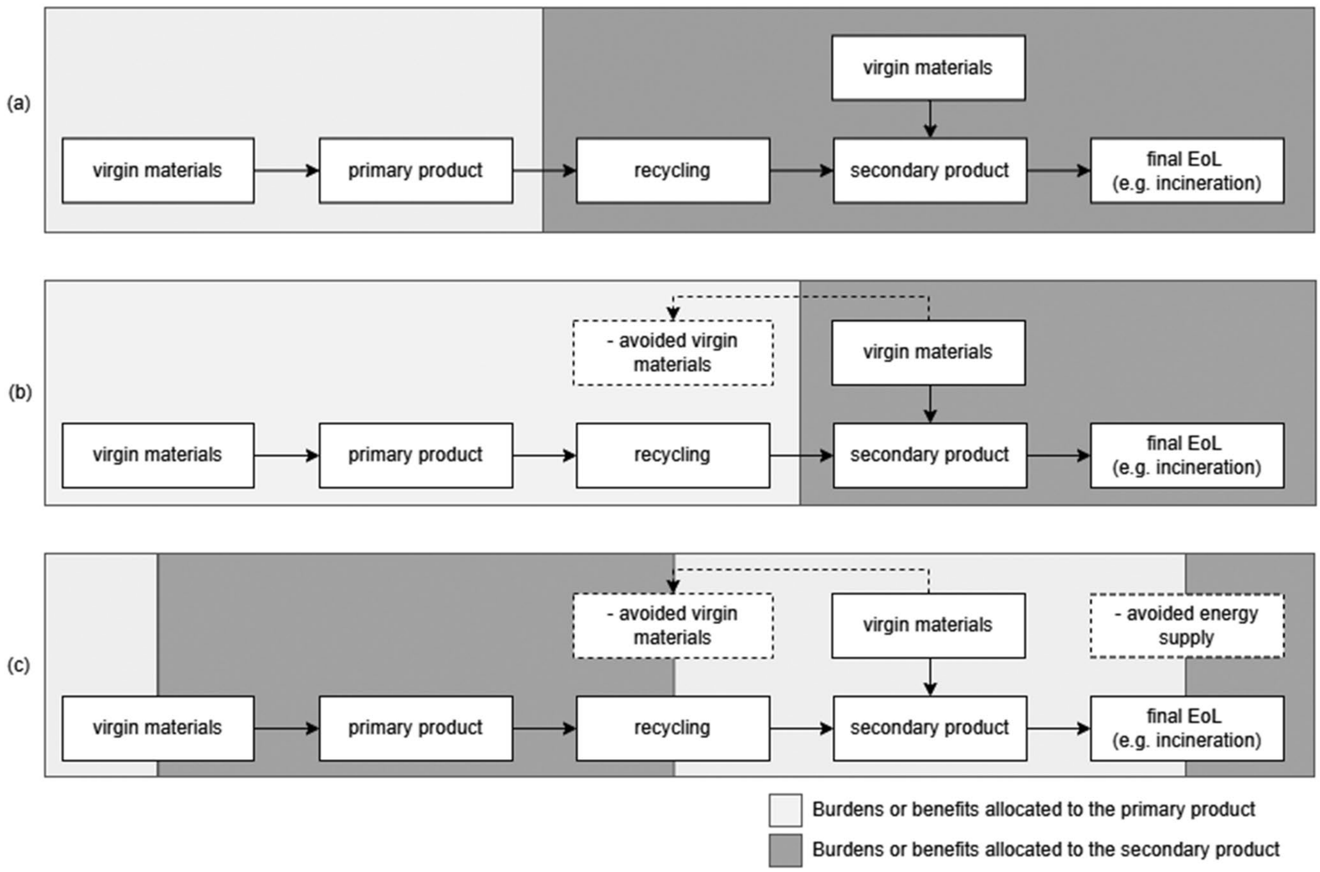

Fig. 1 Schematic representation of the (**a**) 100:0 approach, (**b**) 0:100 approach and (**c**) 50:50 approach. EoL = end-of-life. Based on the figure of Corona et al. (2019)

to incineration alongside the efficiency of the energy recovery process (X_{ER}) (Ekvall et al. 2020; Zampori and Pant 2019). A more detailed description of each parameter is included in Table 1.

Two parameters with major influence on the final results are factors A and B, as pointed out by several of the studies mentioned in the introduction (Braunfels and Teuber 2023; Hammar et al. 2024; Hermansson et al. 2022; Rickert and Ciroth 2020).

Factor A is designed to reflect market supply and demand by distributing the environmental burdens and credits of virgin production and recycling between the first product, which supplies the recycled material, and the second product, which receives the recycled material (Braunfels and Teuber 2023; Ekvall et al. 2021). An A-factor of 1 represent a 100:0 approach, meaning all credits are assigned to the recycled content. An A-factor of 0 represent a 0:100 approach, where credits are allocated to the recyclable materials at the end of their life cycle. In the PEF methodology, an A-factor range of 0.2 to 0.8 is used to ensure that both recycled content and EoL recyclability are considered (Zampori and Pant 2019). The lower limit of 0.2 suggests that the demand for recycled materials surpasses their supply, with market prices being similar to or the same as those for virgin materials (e.g. metals). In contrast, the upper limit of 0.8 indicates that the supply of recycled materials exceeds demand, resulting in market prices lower than those of virgin materials (e.g. wood from pallets). When the market situation is more balanced or uncertain, the A-value is set to 0.5 (e.g. PET from bottles). For default values specific to both products and materials, reference is made to Annex C (European Commission 2022b), which includes the following procedure: (i) check the availability of an application-specific A-value; (ii) if an application-specific value is not available, a material-specific value shall be used; (iii) if a material-specific value is not available, the user shall apply a value of 0.5.

However, as correctly pointed out in the Battery Pass Consortium study (Braunfels and Teuber 2023), neither values between those specified nor calculation rules for situation-specific values are included in Annex C (European Commission 2022b) or the PEF methodology in general. For our case study for example, 'carpet' as an application is not included in Annex C. While PET as material is listed with an A-value of 0.5, this value is linked to bottle applications, whereas recycled PET from fibers likely faces a different market reality due to the more complex recycling

Table 1 Individual parameters of the Circular Footprint Formula

Parameter	Clarification
$R_{i,1}$	The proportion of material in the input to the production that has been recycled from a previous system. This parameter represents the recycled content.
$R_{i,2}$	The proportion of the material in the product that will be recycled (or reused) in a subsequent system. $\boldsymbol{R_{i,2}}$ shall therefore take into account the inefficiencies in the collection and recycling (or reuse) processes and be measured at the output of the recycling plant. This parameter represents the recycling rate.
$R_{i,3}$	The proportion of the material in the product that is used for EoL energy recovery
Q_i	Quality of the ingoing secondary material, i.e. the quality of the recycled material at the point of substitution.
$Q_{(i+1)}$	Quality of the outgoing secondary material, i.e. the quality of the recyclable material at the point of substitution
Q_p	Quality of the virgin or primary material
A	Allocation factor of burdens and credits between supplier and user of recycled materials
B	Allocation factor of energy recovery processes: it applies both to burdens and credits
E_v	Specific emissions and resources consumed (per unit of analysis) arising from the acquisition and pre-processing of virgin material
E_v^*	Specific emissions and resources consumed (per functional unit) arising from the acquisition and pre-processing of virgin material assumed to be substituted by recyclable materials.
$E_{R,i}$	Specific emissions and resources consumed (per unit of analysis) arising from the recycling process, including collection, sorting and transportation
E_{ER}	Specific emissions and resources consumed (per functional unit) arising from the energy recovery process
E_D	Specific emissions and resources consumed (per unit of analysis) arising from disposal of waste material at the EoL of the analysed product
LHV	Lower Heating Value of the material in the product that is used for energy recovery.
$E_{SE,heat}$ $E_{SE,elec}$	Specific emissions and resources consumed (per unit of analysis) that would have arisen from the specific substituted energy source, heat and electricity respectively.
$X_{ER,heat}$ $X_{ER,elec}$	The efficiency of the energy recovery process for both heat and electricity

processes. The only textile application included is a T-shirt, for which a default A-value of 0.8 is assigned, regardless of the material it is made from. Based on this, there are already two possible values that could be used for our case study. Separating the material and application when determining the factor can therefore have a significant impact.

In the battery case study (Braunfels and Teuber 2023), the default A-value for metals in Annex C was adjusted to 0.8, along with a change to the R_2 value. These two adjustments together result in an 85% reduction in EoL credits and a doubling of the total environmental footprint. The considerable impact of parameter A was also demonstrated in the work of Rickert and Ciroth (2020): reducing its value from 0.8 to 0.2, while using a fixed value for parameter R_1, led to an increase in global warming impact of nearly 60%.

Just like the A-factor, the B-factor can also influence the results. This factor is defined as an allocation factor for energy recovery processes. The value is set to zero by default unless a different applicable value becomes available in Annex C (European Commission 2022b). Similar to factor A, the PEF methodology offers no guidelines for calculating or defining a value different from 0 (Braunfels and Teuber 2023). Ekvall et al. (2021) already provided criticism of this default B-factor. A B-value of 0 implies that the full net environmental benefit of energy recovery is accounted for, and this benefit is assigned to the products generating the waste. A higher B-value would give less incentive to energy recovery and more to material recycling. They suggested a new method for estimating factor B, which is based on the factors influencing investments in waste incineration plants, while considering the long-term impacts of directing waste to incineration. Using Swedish incineration data, a default B-value of 0.6 was established, rounded from 0.587. In countries without district heating, energy revenues are typically lower, so the gate fee must cover more of the incineration costs. As a result, factor B will be less than 0.6. Through a case study on renewable polymer, they demonstrated that a B-value of zero created an incorrect incentive for incineration, whereas the B-value of 0.6 provided a more balanced approach, nearly eliminating the risk of incorrect incentives for renewable polymers.

This work also analyses the sensitivity of the results to factors A and B In addition to the alternative values for factor A and the default value for factor B, the case study includes a country-specific B-factor calculated following the method proposed by Ekvall et al. (2021).

2.2 Impact assessment according to PEF

Finally, after compiling the total (life cycle) inventory (LCI) E_i of product i, the (life cycle) impact assessment (LCIA) should be conducted to evaluate the product's environmental performance, using the recommended PEF impact categories and models (Zampori and Pant 2019). The most recent impact assessment calculation framework available at the time of writing this paper was EF3.1.

The inventoried data is initially assigned to 16 impact categories in the classification step and then converted into environmental impacts using specific LCIA models in the characterization step. Next, these impact categories are normalized by dividing them by the total inventory of a reference unit, such as the entire world, in the normalization step. Finally, they are multiplied by a weighting factor to reflect their relative importance in the weighting step.

The weighted impact categories can subsequently be aggregated to generate the single overall score (Zampori and Pant 2019). While normalization, weighting, and aggregation of different impact categories are considered optional steps under ISO 14,044, the PEF method recommends calculating the Single Score as well (Sinkko et al. 2024).

However, the most recently approved version of the PEFCR for apparel and footwear (v3.1) does not yet allow the use of the Single Score in business-to-consumer (B2C) communication; its use is currently restricted to business-to-business (B2B) contexts. This restriction is due to the European Commission's ongoing update of the EF Single Score, which aims to incorporate additional impact areas such as microplastic shedding.

Positioning the impact categories and models of the PEF against those of the EPD reveals a major change since March 2022. From then on, the environmental impact indicators and models of EN15804:2012+A2:2019/AC:2021 were adopted: part of them mandatory, others optional. Some indicators must be accompanied by a disclaimer stating that "the results of this environmental impact indicator shall be used with care as the uncertainties of the results are high and as there is limited experience with the indicator. This acquisition ensures that the majority of impact indicators and models are now aligned with those of the PEF, whereas different ones were used in the past. Table 2 provides a concise overview of the indicators included in the PEF and the EPD. A more detailed comparative overview can be found in the Supplementary Information (SI).

Table 2 Impact indicators included in EPD and PEF. Indicators with an * include the disclaimer

Impact indicator	EPD	PEF
Global Warming Potential (GWP)	x	x
Acidification potential (AP)	x	x
Photochemical ozone creation potential (POCP)	x	x
Eutrophication potential (EP) – generic		
Eutrophication potential (EP) – freshwater	x	x
Eutrophication potential (EP) – marine	x	x
Eutrophication potential (EP) – terrestrial	x	x
Ozone depletion potential (ODP)	x	x
Abiotic depletion potential (ADP) - mineral resources	x (*)	x
Abiotic depletion potential (ADP) - fossil resources	x (*)	x
Water deprivation potential (WDP)	x (*)	x
Human toxicity, cancer		x
Human toxicity, non-cancer		x
Ecotoxicity, freshwater		x
Particulate matter		x
Ionising radiation, human health		x
Land use		x

2.3 Case study

2.3.1 Description of the system/goal & scope definition

This paper presents an in-depth examination of the CFF in the context of fiber-to-fiber textile recycling and compares it to the alternative 100:0 approach outlined in the EPD, used here as a benchmark. The textile case focuses on the recycling of a mono-material carpet and sets this scenario against the conventional approach of incinerating a multi-material carpet with energy recovery.

The considered life cycle perspective and system boundaries are set by the CFF. This covers the acquisition and pre-processing of raw materials, as well as the EoL stage, which involves either recycling or disposal. Not included are manufacturing processes - except for manufacturing losses accounted for in the CFF calculation - product distribution and storage, use, capital goods and infrastructure. An additional complexity regarding the system boundaries is that the CFF also considers the beneficial contribution of avoided impacts, whereas any avoided impact is excluded from the system boundaries in the 100:0 approach.

The starting point of the case study is a carpet production facility located in Belgium, producing both the mono-material carpet (MMC), made almost entirely of PET (>90%), and the conventional material carpet (CMC), made from different materials (cotton, PET and a latex-based coating).

At the end of their life, they typically undergo different EoL treatments: the MMC can be recycled into new PET material, while the CMC is incinerated for energy recovery. More and more initiatives, representing manufacturers and suppliers, are emerging to collect carpets separately, to either optimize the incineration process of conventional carpets or promote a suitable treatment for recyclable carpets. Although these collection platforms are still in their early stages, they represent the most likely path forward. In this case study, it was assumed that both the CMC and the MMC are collected through such a platform.

After collection, the MMC is sent to a testing facility in Austria that focuses on thermomechanical recycling with solid-state polymerization (SSP). During this process, the material is reprocessed into new PET granulate. SSP treatment is essential to restore the intrinsic viscosity

(IV), which typically decreases due to degradation during thermomechanical processing. To enable the application of the CFF, the recycling system must also include the conversion of this granulate into new raw materials for secondary products. Following SSP, the IV of the recycled granulate is reinstated to typical yarn extrusion values (0.6–0.7 ml/g), making it suitable for fiber melt spinning into new carpet fibers or, by extension, into other textile fibers. A yarn extrusion process utilizing this recycled granulate is currently being optimized in the Netherlands.

Transforming the granulate into new PET-based coating is a more complex challenge and is not yet underway. The long-term objective is to create a fully closed-loop system, enabling the production of both new yarns and coating from the recycled material entirely within Europe. In the context of the CFF, this process entails a single recycling loop, resulting in two products: the first product is the MMC produced from virgin materials, and the second product is a new MMC created using recycled materials derived from the first product. At the end of its lifecycle, the second product is sent to an optimized incineration process for energy recovery. The latter is based on the current situation with the expectation that multiple recycling loops will become feasible in the near future, provided the quality of the PET can be preserved to its fullest extent.

In this regard, a product system is modelled that includes both products: the first or primary MMC, which is recycled, and the second MMC, which is incinerated at the EoL stage. This approach is essentially equivalent to applying the CFF separately to each product, the first and the second, and summing the results. However, by adopting both a product-level and a system-level perspective, the analysis provides distinct insights and a more nuanced understanding of the outcomes. The functional unit is therefore defined as 2 × 1 m² in total, corresponding to 1 m² for each carpet within the system.

The same principle applies to the CMC scenario, which forms the basis of the second product system. This system, defined by the same functional unit, comprises two 1 m² CMC carpets. As the first product, 1 m² CMC, is not recycled after use, the second product, also 1 m² CMC, must be produced from virgin materials. Therefore, both products are identical in composition and EoL treatment.

A third hypothetical product system is also considered, where the first product, 1 m² MMC, is not recycled as it ideally should be, but instead is incinerated, similar to a CMC. As a result, a second product, 1 m² MMC, must be produced once more from virgin resources. This second product also undergoes incineration with energy recovery at the end of its life cycle. The three product systems and their corresponding material flows are illustrated schematically in Fig. 2.

2.3.2 Data inventory analysis

The calculation of the CFF for the three product systems requires the following life cycle inventory data, as described in the Materials & Methods section: E_v, E_v^*, E_R, E_D and E_{SE}. According to PEF Annex C, EF-compliant datasets have to be selected. Version 3.1 of this database is already available but cannot be used outside the context of an agreement with the European Commission as part of a PEFCR. Consequently, the Ecoinvent 3.11 database was used for this study.

i) Production of virgin raw materials (E_v)

To establish the inventory data for E_v, the product's Bill of Materials (BOM) serves as the primary reference. According to this BOM, the MMC comprises two types of PET yarns with thicknesses of 450 tex and 50 tex, along with a PET-based coating serving as the backing. The CMC consists of cotton yarns of 450 tex, cotton yarns of 50 tex, PET yarns of 500 tex, and a latex-based coating. The actual quantities of input materials required for manufacturing have to be retro-calculated through the loss rate (LR), as shown in Eq. (5), where n refers to the relevant processing step.

$$kg_{input_n} = kg_{output_n} + waste_n = kg_{output_n} * (1 - LR_n)^{-1} \tag{5}$$

The considered manufacturing process consists of several steps: weaving, applying a backing through a kiss-roll process (which involves either a latex-based or a PET-based coating) to provide anti-slip properties, then cutting the carpet to the required length, and finally finishing the sides and corners. The company estimated a total manufacturing loss rate of 3%. Based on this loss rate, the quantities of input materials required for both MMC and CMC were retroactively calculated and are presented in Table 3.

The PET yarns are obtained from a producer based in Western India near Mumbai, while the cotton yarns are supplied by a manufacturer located near Cairo, Egypt. In contrast, both the PET-based and latex-based coatings are sourced directly from the Belgian market.

To model PET yarns produced in India, the Ecoinvent database provides the dataset "*Fiber*, *polyester {IN} polyester fiber production*, *finished* | *Cut-off*, *U*." This flow was further refined by linking energy and water inputs to Western India locations wherever possible, rather than using generic global or Indian data. This level of geographical precision was also important for accurately quantifying the difference with the recycled yarns later, as re-extrusion takes place in the Netherlands. More details on the modification can be found in the SI.

For spun cotton yarns produced in Egypt, there are no suitable data models available in Ecoinvent or other SimaPro

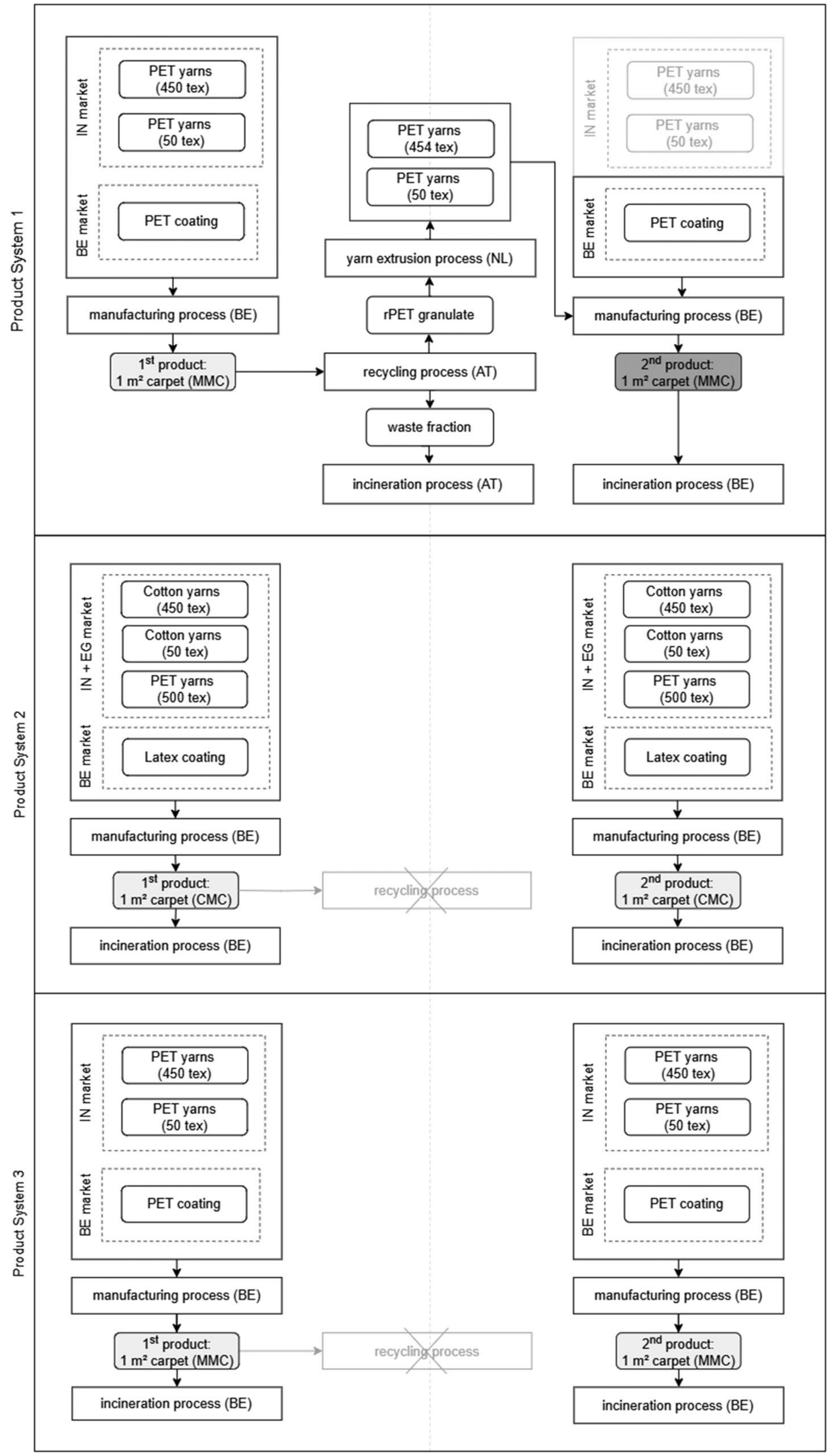

Fig. 2 Schematic overview of the considered product systems (BE = Belgium, AT = Austria, NL = the Netherlands)

Table 3 Life cycle inventory data and used Ecoinvent dataset models. All datasets refer to the cut-off system model. More details on modified and customised datasets are included in the SI. (IN = India, GLO = global, RER = Europe, CH = Switzerland, AT = Austria, BE = Belgium)

E	Inventory flow	MMC	CMC	unit	Ecoivent dataset
E_v	PET yarns 450 tex	1.12	-	kg	Polyester fiber production, finished {IN} - modified
	PET yarns 50 tex	0.11	-	kg	Polyester fiber production, finished {IN} – modified
	PET yarns 500 tex	-	0.22	kg	Polyester fiber production, finished {IN} - modified
	CO 450 tex	-	0.90	kg	See Supplementary Information
	CO 50 tex	-	0.11	kg	See Supplementary Information
	PET resin	0.20	-	kg	Market for PET, granulate, amorphous {GLO}
	Latex	-	0.11	kg	Market for latex {RER}
	Polyacrylic comp.	0.10	0.01	kg	Acrylic dispersion, in 58% solution state {RER}
	Surfactant	0.01	0.01	kg	Market for non-ionic surfactant {GLO}
	CaCO3 filler	-	0.19	kg	Market for calcium carbonate, precipitated {RER}
	Demi water	0.68	0.07	kg	Market for water, deionised {Europe excl. CH}
	Wastewater	7E-04	7E-05	m^3	Treatment of wastewater, average {Europe excl. CH}
E_R	SSP process	0.47	-	kwh	Market for electricity, medium voltage {AT}
	Extrusion process	1.24	-	kg	Polyester fiber production, finished {IN} - modified
E_D	Municipal incineration	1.50	1.50	kg	Treatment of municipal solid waste, incineration {BE}
	Plastics incineration	0.23	-	kg	Waste plastic, mixture, incineration {Europe excl. CH}
E_{SE}	Electricity	1.50	1.50	kwh	Market for electricity, medium voltage {BE}
	Heat	1.50	1.50	MJ	Market for heat, central/small-scale, natural gas {RER}

databases. This is mainly because none of the datasets take yarn thickness or titer into account, and also because the geographical location does not match, as the available data mainly represent India and Bangladesh. The importance of yarn thickness in LCA has already been highlighted by Van der Velden et al. (2014), as this parameter significantly impacts processing energy. While some EF-compliant datasets that account for yarn thickness have been developed, they are not yet available; see Results section for more details.

Therefore, a separate dataset was modelled for cotton yarns from Egypt. Based on the work of Van der Velden et al. (2014), the processing energy can be expressed as a function inversely proportional to the titer. To validate this approach, the energy consumption for cotton spinning at a Belgian company was analyzed and similarly expressed as a function of the titer, with further details provided in the SI. The results closely aligned with the simulated data from Van der Velden et al. (2014). For the energy consumption of cotton yarns in this case study, an average of both data sources was used. In the case of a 450 tex cotton yarn, this corresponds to approximately 0.24 kWh per kg, while for a yarn of 50 tex, it is about 2.16 kWh per kg. A second key input for the spinning process is water, used to maintain consistent humidity conditions. At the consulted Belgian company, this consumption amounts to 1 L per kilogram of yarn, which almost entirely evaporates during the process. Both energy consumption and water usage are modeled using Ecoinvent datasets linked to Egypt as the geographical context, see Table 3.

As for the coatings, approximate compositions were determined based on information from the suppliers. The PET-based dispersion contains 20% PET resin, 68.7% demineralized water, 10.3% polyacrylate-based components, and about 1% non-ionic surfactants. The latex-based coating is composed of 7.6% styrene-butadiene latex, 30% demineralized water, 61.5% calcium carbonate filler, 0.2% polyacrylate-based components and 0.4% non-ionic surfactants. The Ecoinvent datasets used to model these various components are listed in Table 3. When the coating is applied using a kiss-roll process, the water evaporates, leaving behind the primary material, which is either PET or latex, along with the other components. Only the residual dry mass is considered for further processing and recycling of the carpet.

ii) Recycling processes (E_R)

The recycling of the MMC is carried out by a testing facility in Austria, applying an integrated SSP process. This process includes steps of shredding, extruding and granulating. This process consumes approximately 0.35 kWh of energy per kg obtained material (Starlinger 2024), i.e. PET granulate, which was modelled through Austria's electricity grid mix. Furthermore, the material losses for a pure PET waste stream are estimated by the company to be approximately 4% (Vreriks 2020).

The MMC, weighing 1.5 kg per m^2, is made up of approximately 1.39 kg PET: 1.2 kg is derived from the yarn, while 0.19 kg comes from the coating. The remaining 0.11 kg includes the other components of the coating. Applying a 4% recycling loss rate to the 1.39 kg of pure PET results in an output of 1.34 kg. The total loss includes this 4% (0.06 kg) along with the 0.11 kg of other components, which are also removed during the recycling process. The next stage in the recycling process involves reprocessing the granulate into yarns via melt spinning, a process with an estimated maximum loss rate of 5% when utilizing recycled polymers. Reprocessing the granulate into new raw materials for a PET-coating is currently not yet feasible, but it remains a goal for the future. The preceding SSP process yields a sufficiently high output to fully replace virgin yarns. In summary, for 1.5 kg of waste input (corresponding with 1 m² carpet), 1.24 kg of PET yarn can be produced, indicating that the overall recycling process has a loss rate of 17.5%. A value of 82.5% is consequently used as recycling rate $R_{1,2}$. The residual 17.5% of material, which cannot be recovered, is directed to disposal via incineration with energy recovery. Further details on disposal processes are provided in the next section.

As mentioned earlier, a yarn extrusion process using recycled granulate is currently being optimized in the Netherlands. To model this, the Ecoinvent dataset "*Fiber*, *polyester {IN} | polyester fiber production*, *finished | Cut-off*, *U*." was modified again to reflect the Dutch geographical context. The yarn extrusion or melt spinning process is primarily driven by energy consumption. In the Ecoinvent dataset, this amounts to a total of 1.66 kWh per kg, which aligns quite well with other values found in the literature (1.7 kWh per kg for PET and nylon (Van der Velden et al. 2014) and our own measurements on a semi-industrial melt spinning pilot line (1.85 kWh per kg for nylon, 1.61 kWh per kg for PLA). Therefore, adopting an energy consumption of 1.66 kWh per kg for production in the Netherlands is considered acceptable. Further details regarding the modifications applied to the respective Ecoinvent dataset are provided in the SI.

Finally, the CFF requires that transportation is also accounted for in the EoL stage, which was not necessary for E_v and E_v^*. The transportation associated with the recycling scenario in product system 1 includes (i) from the final client to the municipal waste center, (ii) from the municipal waste center to the collection point, (iii) from the collection point to the recycling facility in Austria and (iv) from the recycling facility to the yarn extruder in the Netherlands. The relevant distances and transportation modes are listed in Table 4. For transportation type (i) and (ii), default values from the PEFCR Apparel and Footwear were applied. The first long distance transport, from the Belgian collection point - assumed to be centrally located in Brussels - to the recycler in Austria, was determined via Searates.com, as recommended by the PEFCR Apparel and Footwear. In the same way, the distance was determined between the recycler and the yarn extruder in the Netherlands (Amsterdam region), as well as between the yarn extruder and the carpet manufacturer (assumed to be centrally located in Brussels). To model these different transportation aspects, the following Ecoinvent datasets were used: '*Transport*, *freight*, *lorry*, *16–32 metric ton*, *diesel*, *EURO 6 {RER}|Cut-off*, *U (market for)*' and '*Transport*, *passenger*, *car*, *EURO 5*, *fleet average {RER}| Cut-off*, *U (market for)'*.

iii) Disposal processes (E_{ER}, E_D)

In this case study, E_{ER} equals E_D: incineration is the main disposal method in both Belgium and Austria, where the recycling process takes place. Landfill is only used in exceptional cases to dispose of treated, odourless residue from waste incineration plants (Stadt Wien 2023). According to the European Environment Agency, only 2% and 0.6% of municipal waste was landfilled in Austria and Belgium, respectively (European Environement Agency 2024). Disposal is therefore modeled using datasets for municipal solid waste incineration and mixed plastic waste incineration (incineration without energy recovery), as outlined in Table 3. The mixed plastic waste dataset is used for waste generated during the recycling process in Austria (product system 1), while the municipal solid waste dataset is applied to the incineration of the CMC and MMC at the end of their lifecycle (product systems 1, 2 and 3). Transportation related to the EoL phase of the CMC includes: (i) the transport from the final client to the municipal waste center, and (ii) from the municipal waste center to the incineration plant, both located in Belgium. For the MMC, transportation covers: (iii) the journey from the Austrian recycler to the local incineration plant. Default values for distance and transport mode from the PEFCR Apparel and Footwear were utilized for these generic transportation routes, see Table 4. As before, this transport was modelled using the dataset '*Transport*, *freight*, *lorry*, *16–32 metric ton*, *diesel*, *EURO 6 {RER}|Cut-off*, *U (market for)*'.

iv) Avoided virgin raw material production (E_v^*)

The avoided material production is only applicable to product system 1, which includes the recycling scenario. The

Table 4 Transport related inventory data (BE=Belgium, AT=Austria, NL=Netherlands)

Transport route	km	Transport mode
Final client (BE) to municipal waste center (BE)	1	Passenger car
Municipal waste center (BE) to incineration plant (BE)	30	Lorry (7.5–16 tonnes)
Municipal waste center (BE) to collection point (BE)	130	Lorry (7.5–16 tonnes)
Collection point (BE) to recycling facility (AT)	1026	Lorry (7.5–16 tonnes)
Recycling facility (AT) to yarn extruder (NL)	1075	Lorry (7.5–16 tonnes)
Yarn extruder (NL) to carpet manufacturer (BE)	303	Lorry (7.5–16 tonnes)
Recycling facility (AT) to incineration plant (AT)	30	Lorry (7.5–16 tonnes)

virgin raw materials that are expected to be entirely replaced by recyclable materials are the PET yarns, whose modelling has already been discussed earlier. The PET-based coating on the other hand must still be accounted for as a new virgin raw material in the production of a secondary carpet.

v) Avoided energy supply ($E_{SE,elec}, E_{SE,heat}$)

The avoided energy supply is relevant to all three product systems, as it is associated with the EoL incineration stage. $E_{SE,elec}$ and $E_{SE,heat}$ represent the conventional energy supply of electricity and heat, which is replaced by the energy recovered from waste incineration. The substituted electricity is modelled using Ecoinvent datasets for the electricity grid mix of the respective country, either Belgium or Austria. The substituted heat is modelled as thermal energy derived from natural gas in the European region. These datasets are included in Table 3.

vi) Other parameters

An overview of all the values used for the other CFF parameters is presented in Table 5. For all product systems, the recycled content $R_{1,1}$ of the first product is zero, which is also the default value in the PEF method. For product system 1, the recycling rate $R_{1,2}$ of the first product is 82.5%, as explained in the previous section. The remaining 17.5% waste, represented by $R_{1,3}$, goes to incineration for energy recovery. The secondary product has consequently a recycled content $R_{2,1}$ of 82.5%. Since only one recycling loop is considered, recycling rate $R_{2,2}$ is set zero and $R_{2,3}$, the waste fraction going to incineration, is 100%. In product systems 2 and 3, both products are directly sent to incineration after use, leading to an $R_{i,3}$ value of 100% and an $R_{i,2}$ value of 0.

To determine factors Q, A and B, Annex C of the PEF was consulted. For PET recycled via an SSP process, as applied in product system 1, the quality of the recycled material Q_2 is assumed to be equivalent to the quality of the virgin material Q_1, resulting in a ratio Q_2/Q_1 of 1. In product systems 2 and 3, no recycled material is used; therefore, the quality factor Q_i is equal to 1 for each product. Regarding factor A, as explained in the Materials & Methods section, there are two options based on Annex C: material-based (PET from bottles, with a value of 0.5) and application-based (textiles, i.e. T-shirt, with a value of 0.8), with the application-based option being the preferred choice according to the guidelines. However, since neither the application nor the material perfectly aligns with the product and material in our case study, the influence of both values will be assessed.

Table 5 Overview of other CFF parameters for each product system

Parameter	Product System 1	Product System 2	Product System 3
$R_{1,1}$	0	0	0
$R_{1,2}$	82.5%	0	0
$R_{1,3}$	17.5%	100%	100%
$R_{2,1}$	82.5%	0	0
$R_{2,2}$	0%	0	0
$R_{2,3}$	100%	100%	100%
Q_1	1		1
Q_2	1	1	1
LHV_{MM}	22.7 MJ/kg	21.9 MJ/kg	22.7 MJ/kg
$X_{ER,heat}$	25%	24%	25%
$X_{ER,elec}$	13%	12%	13%
A	0.5–0.8	0.5–0.8	0.5–0.8
B	0–0.6	0–0.6	0–0.6

The B-factor is used as an allocation factor of energy recovery processes and is set zero by default in the current PEF guidelines. The issues related to factor B have also been addressed in the Materials and Methods section: in the current PEF guidelines, B is set to zero, but Ekvall et al. (2021) propose a higher B-value of 0.6 to reduce the incentive for energy recovery. Since this B-factor is based on Swedish data, a country-specific B-factor for Belgium was calculated using the same method. The resulting value for Belgium was nearly identical to that of Sweden. Details of the calculation are provided in the SI. In this paper, both values (0 and 0.6) will be applied to assess their influence on the final results. They are applicable to each of the three product systems, as all scenarios involve incineration with energy recovery.

Other parameters in the CFF are the LHVs of the incinerated materials - in this case PET, cotton and latex - and the efficiencies $X_{ER,heat}$ and $X_{ER,elec}$ of the energy recovery process for both heat and electricity. Due to the compositional differences between the MMC and CMC, product systems 1 and 3 result in different LHV and X_{ER} values. According to Annex C, these values should be part of the EF-compliant dataset. However, since such datasets are not yet available for use, efficiency values were taken from the selected Ecoinvent dataset. The calculation details for individual efficiency values of heat and electricity are provided in the SI. LHV values for cotton and latex were obtained from Prasanga et al. (2021) and Ioelovich (2018), respectively. The LHV of PET was calculated as an average of various values from the literature; this calculation can also be found in the SI.

3 Results and discussion

This section analyzes three situations, each supported by an uncertainty assessment using Monte Carlo simulation with a 95% confidence interval, which is included in the SI. First, the environmental impacts of the three product systems are compared using the two allocation approaches considered

(CFF and 100:0) with fixed A- and B-values, to evaluate how the choice of allocation method influences system-level results. Second, the sensitivity of the results to changes in A- and B-values is analyzed, with particular attention to their impact on the distribution of environmental burdens among individual products within each system. Third, the influence of titer is assessed under a fixed A-B combination to determine its contribution to overall environmental performance.

The first situation is illustrated in Fig. 3, which shows the Climate Change impacts for the three product systems calculated using both the CFF and the 100:0 approach, together with the Single Score results derived from the CFF. For each product system, the impacts of the individual products are presented separately using stacked bar charts. As previously explained, the first and second products in Product Systems 2 and 3 are identical, due to the absence of recycling. In contrast, in Product System 1, the first product is an MMC produced from virgin materials, while the second product is an MMC produced from recycled materials from the first product.

A key finding of the Climate Change impact assessment is that the MMC recycling scenario (Product System 1) is more favorable than the scenarios without recycling (Product Systems 2 and 3). This result, which is consistent with circular economy principles and the waste hierarchy, is observed under both the CFF and 100:0 approach. The associated uncertainty analysis, reported in the SI, shows that Product System 3 exhibits higher Climate Change impacts than Product System 1 in 100% of the simulations for both approaches. For Product System 2, this holds in approximately 97% of the simulations under the CFF approach and in 92% under the 100:0 approach. This trend, however, is not consistent across all impact categories.

Under the 100:0 allocation approach, the recycling scenario of Product System 1 generally shows higher impacts than Product System 2 for ozone depletion and fossil resource use, occurring in nearly all simulations. For photochemical ozone formation, Product System 1 exhibits higher impacts in ±53% of the simulations. In contrast, for all other EPD impact categories, Product System 1 consistently shows the lowest impacts, with probabilities ranging from approximately 98% to 100%.

Under the CFF approach, Product System 1 likewise exceeds Product System 2 for ozone depletion and fossil resource use, and additionally for ionizing radiation (not included in the EPD) in all simulations. Product System 1 also exceeds Product System 3 for ionizing radiation in more than 99% of the simulations. For ecotoxicity, human toxicity, ozone formation, and water use, higher impacts for Product System 1 are observed in about 47% to 54% of the simulations. The first two of these categories are core PEF indicators but are optional in the EPD due to their high uncertainty, which is reflected in the results. For all remaining PEF impact categories, the recycling scenario (Product System 1) consistently shows the lowest impacts, with probabilities between 97% and 100%, except for land use, where this probability decreases to approximately 73% when comparing Product Systems 1 and 2. The aggregated Single Score, which combines all impact categories into a single metric, generally supports the conclusion that MMC recycling is environmentally preferable to the non-recycling scenarios. However, the associated uncertainty is substantially higher, as this outcome is observed in only about 53% to 55% of the simulations.

A comparison between Product Systems 2 and 3 indicates that, without recycling, the use of an MMC is not necessarily environmentally preferable to a CMC. For the Climate Change impact category, Product System 3 exhibits higher

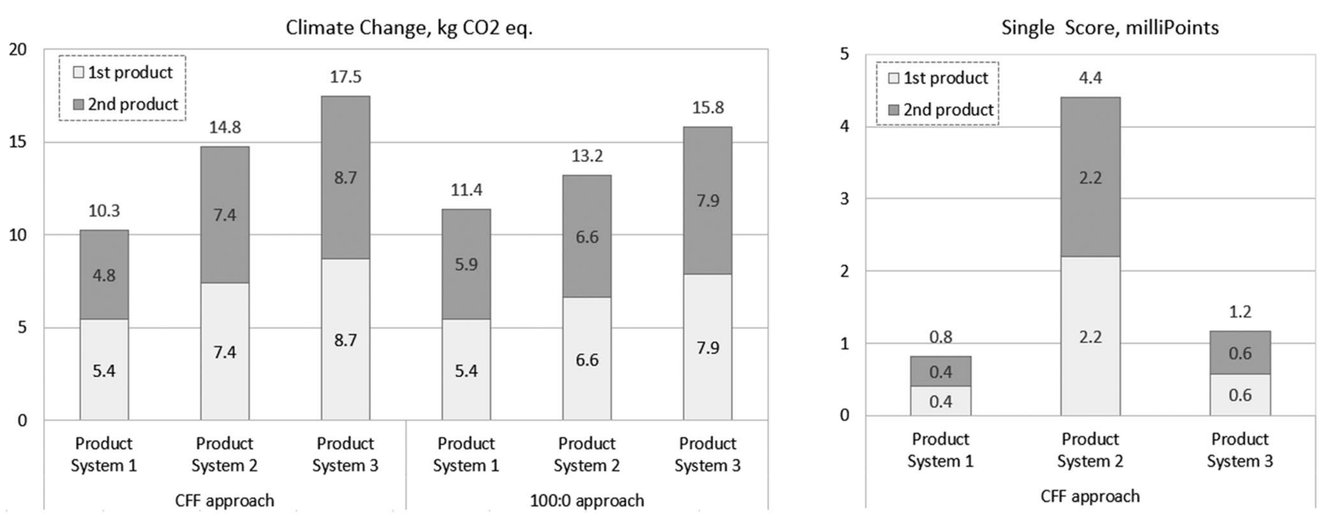

Fig. 3 Climate Change impacts and Single Score results per m^2 product (i.e. 2 m^2 per product system)

impacts, which is also the case for ozone depletion, photochemical ozone formation, and fossil resource use under both allocation approaches, as well as for ionizing radiation under the CFF approach only. These findings are supported by the uncertainty analysis, for approximately 93% to 100% of the simulations. The higher impacts observed for Product System 3 are primarily attributable to the virgin material production stage, which results in greater environmental burdens for virgin MMC compared to virgin CMC. This difference is mainly driven by compositional differences between the two materials. For land use, mineral resource use, and human toxicity, the probability that Product System 3 has the highest impacts is much lower, with this occurring in only about 30% to 50% of the simulations. This suggests a considerable overlap in impact results between the two product systems for these categories.

In summary, the environmental benefit of monomaterial carpets is realized only if they are effectively recycled. If incinerated without at least one recycling cycle, their impacts may exceed those of CMCs in several categories, including Climate Change. It should be noted that the manufacturing phase is excluded from the CFF calculation; however, as manufacturing is assumed identical for MMCs and CMCs, this omission is expected to have negligible influence on comparative results.

Up to this point, only the total impact of each Product System has been considered, with both the CFF approach and the 100:0 approach yielding consistent results and conclusions. However, this changes when the analysis is performed at the individual product level, particularly for the MMC recycling scenario in Product System 1. When the 100:0 calculation is applied to each product in this system, the (Climate Change) impact of the first product is higher than that of the second product, whereas under the CFF calculation, the impacts of both products are more equally distributed. This is, of course, a logical consequence of the A-factor and the underlying reasoning applied in the PEF, where the objective is to better balance burdens and credits between the first product and second (recycled) product based on market supply and demand. However, such a difference has a significant influence on the interpretation of conclusions on a product basis. From a policy perspective - for example, in the context of taxation on products with higher impacts - this difference could have substantial implications, especially when the products are produced by different companies.

Subsequently, the effect of factors A and B is examined in more detail, as shown in Fig. 4. As described in the Materials and Methods section, an alternative value of 0.8 is considered for factor A based on the broader application context, while the available values for factor B are 0 and 0.6. The results show that variations in factor A primarily influence differences at the individual product level (within Product System 1), whereas the effect of factor B is more limited and manifests mainly at the level of the overall product system.

For an A-factor value of 0.5, the two products yield very similar results for both the Climate Change impact category and the Single Score. Increasing the A-factor from 0.5 to 0.8 while keeping the B-factor at zero leads to percentage differences of 19% for the Climate Change impact and 40% for the Single Score between the two products in Product System 1. When the B-factor is set to 0.6, the same increase in the A-factor leads to a Climate Change impact difference of 29%, whereas the difference in the Single Score remains at approximately 40%. These results are consistent with those reported by Rickert and Ciroth (2020), who identified a strong influence of the A-factor on the final results. In their study, the B-factor was not evaluated, as it was held constant at zero. In the present case study, varying the B-factor while keeping the A-factor constant indicates that its influence on the results is comparatively small. Specifically, increasing the B-factor to 0.6 at an A-factor value of 0.5 results in a 6% difference in the Climate Change impact, while the Single Score changes by only 0.5%. At an A-factor value of 0.8, the same increase in the B-factor leads to a 4% difference in the Climate Change impact, with the Single Score again differing by approximately 0.5%.

This pattern is also reflected in the uncertainty analysis, see SI. Across all impact categories, changes in factor A lead to substantially larger absolute impact differences between product 1 and product 2 than changes in factor B, with probabilities ranging from 93% to 100%. The main exceptions are water use, ecotoxicity, human toxicity and the Single Score, for which the probability that variations in factor A result in larger impact differences than variations in factor B, lies around 50%. These latter three impact methods are associated with higher inherent uncertainty, as mentioned earlier.

Since factor A can thus have a substantial influence at the individual product level, it is important that this factor receives a more thorough scientific foundation than is currently the case. As mentioned earlier, there is no A-factor from Annex C that perfectly aligns with the material or application used in this case, allowing for flexibility in choice and, consequently, varying results.

In this analysis, only the two A-factors provided in Annex C were applied. However, considering the specific market conditions for rPET derived from carpets, the A-value for fiber-to-fiber recycling is likely to differ from that of rPET from bottles (0.5) due to distinct market dynamics and material characteristics. rPET from textiles, including carpets, remains scarce and highly specialized. Its supply is far lower than that of virgin PET or rPET from bottles, and the recycling process for textiles is more complex and costly

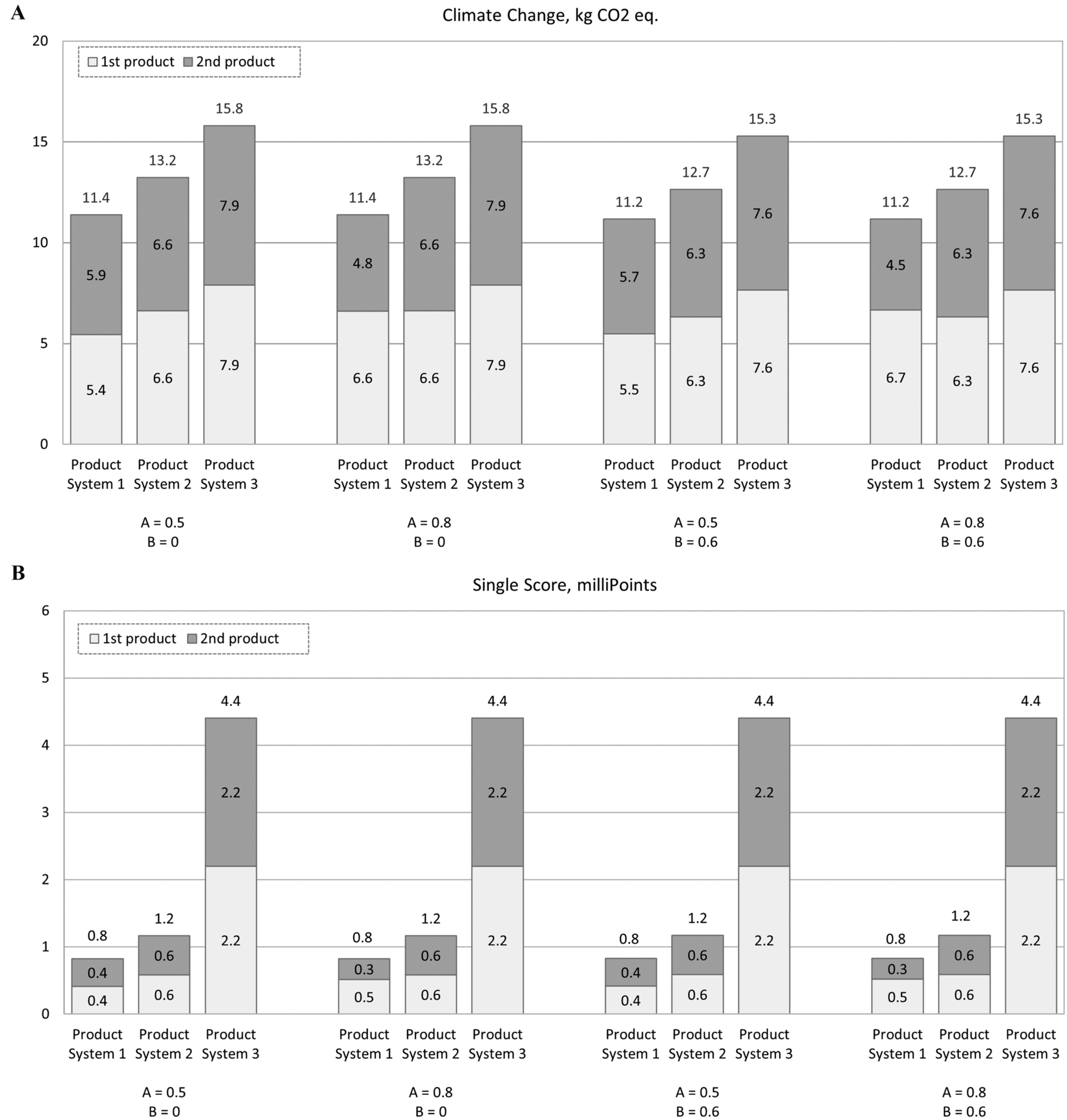

Fig. 4 Effect of factors **A** and **B** on Climate Change and Single Score impact results per m2 product (1 m²) and per product system (2 × 1 m²)

than bottle recycling. In addition, fiber-to-fiber recycling can lead to reduced material quality or altered properties, for example when SSP is not applied to restore intrinsic viscosity, limiting its direct substitution for virgin PET in certain applications. These factors point to an A-value closer to the lower bound (around 0.2), indicative of a market with limited supply and high demand. This demand is expected to grow further under forthcoming regulatory measures, notably those promoted by the EU Strategy for Sustainable and Circular Textiles, which aims to accelerate the development and implementation of true fiber-to-fiber recycling solutions. While the application-based A-factor of 0.8 for a T-shirt in Annex C may assume a large availability of waste textiles - which is indeed the case - the actual recovery of usable material from these textiles remains limited for the reasons outlined above. This could support the use of a

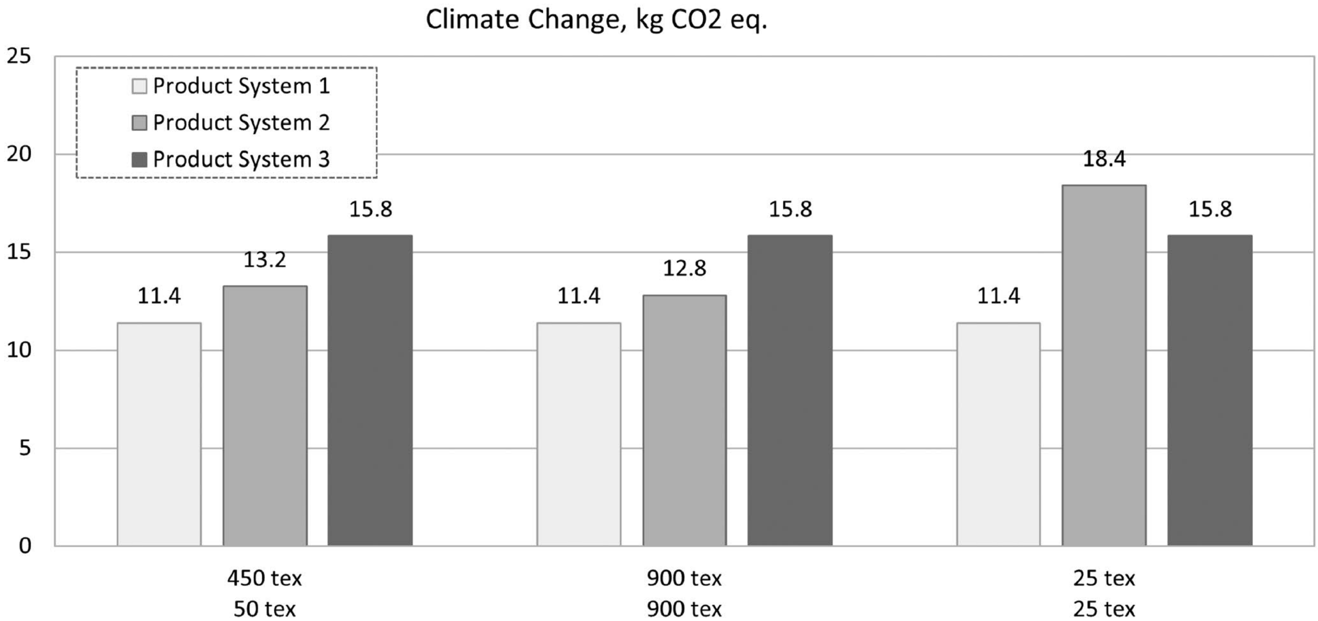

Fig. 5 Effect of titer on Climate Change impact results per m2 product (i.e. 2 m2 per product system)

lower A-value. To reflect this scenario, additional calculations using an A-factor of 0.2 have been included in the SI, providing a more complete overview of potential market conditions.

A more robust rationale for determining A-values, along with a more comprehensive list of possible A-values in the PEF guidelines, considering both material and application, seems to be essential. It is the combination of material and application that determines the market supply and demand, and, consequently, the A-factor. Viewing them separately when determining an A-value can, as demonstrated in this case study, result in significant variations in the outcomes.

Another parameter with an underestimated impact on the results is the yarn thickness or titer. In the considered case study, the energy consumption of the spinning process for the cotton yarns of the CMC was taken into account based on the yarn's titer. This refers to yarns of 450 tex and 50 tex. For comparison, the calculation was also performed for a scenario using coarser yarns (2×900 tex) and for a scenario using finer yarns (2×25 tex). When examining impact categories associated with energy consumption, such as Climate Change, it becomes clear that the titer can significantly influence the results, particularly in the case of fine yarns. Figure 5 presents the total Climate Change impact for each Product System. In the reference scenario, the CMC in Product System 2 consists of a combination of 450 tex and 50 tex yarns. In the comparative scenarios, it consists either of two 900 tex yarns (coarse yarns) or two 25 tex yarns (fine yarns).

In both the reference case and the scenario with coarse yarns, Product System 3 - direct incineration of the MMC without recycling - results in the highest environmental impact. However, when very fine yarns are used, the environmental impact of Product System 2 – direct incineration of the CMC - exceeds that of Product System 3. This finding is also confirmed by the complementary uncertainty analysis, see SI. As yarn fineness increases, energy consumption rises, leading to a greater environmental burden associated with virgin CMC production. Consequently, variations in yarn thickness can significantly alter the overall conclusion regarding Climate Change impacts.

To evaluate the extent to which PEF, and more specifically the PEFCR Apparel and Footwear, account for the titer, the current version of the EF3.1 database was examined. Although this database is not yet accessible for practical application, it is available for consultation. The processes related to yarn spinning in EF3.1 are defined for either 37 tex, a range of 6-118 tex, or an 'unspecified tex.' This represents an improvement over the Ecoinvent database, where processes for different titers are not distinguished. However, as demonstrated by the results above, this classification is still insufficient. For example, despite the significant differences in energy consumption between yarns of 6 tex and 118 tex, they are categorized under the same process. Additionally, no processes are available for yarns exceeding 118 tex. The variation in energy consumption of specific textile processes (e.g. spinning, weaving, knitting) as a function of the titer should therefore not be directly integrated into the CFF itself but should rather be introduced in the results using more appropriate datasets.

While the findings offer valuable insights into the applicability of the CFF for textile recycling, several limitations should be acknowledged to ensure proper interpretation of the results. One important limitation concerns the restricted system boundaries, which cover only raw material acquisition and end-of-life stages, while excluding manufacturing processes (except for material losses), product distribution, the use phase, and infrastructure. Although this approach is consistent with the CFF methodology, it limits the completeness of the life-cycle perspective when compared to a fully PEF-compliant assessment. Results are also based on specific geographical and market assumptions regarding transport routes and recycling conditions, making them dependent on the study's time, location, and technology. Further, the analysis assumes only one recycling loop for the mono-material carpet, whereas multiple loops could occur in practice and potentially influence the environmental benefits. Additionally, due to restricted access to EF-compliant datasets (EF3.1), the study relied on the Ecoinvent database, which may lead to differences compared to future PEF-compliant assessments. Finally, ongoing developments in the PEF methodology, such as the potential inclusion of microplastic impacts, may influence future assessments. At the time of writing, microplastic shedding only had to be reported as additional environmental information, with a disclaimer, in the PEFCR for apparel and footwear. Its future integration as an impact indicator could significantly affect single-score results for PET-based products.

4 Conclusions

This study evaluated the application of the PEF's CFF in the context of fiber-to-fiber recycling of mono-material PET carpets, comparing its outcomes with the conventional 100:0 approach used in Environmental Product Declarations. Through the case study, key methodological challenges were highlighted that need to be addressed in order to improve the reliability and applicability of the CFF in the sustainability assessment of textile applications.

In the evaluation of each Product System, the Climate Change impact assessment indicates that recycling of a MMC (Product System 1) is more beneficial than direct incineration of a CMC or an MMC without recycling (Product Systems 2 and 3). This conclusion is consistent with the principles of the circular economy. The result is further supported by the single score impact, which aggregates all individual impact categories into a single value. However, the single score is associated with a high level of uncertainty (with probabilities close to 50%). Similarly, high uncertainties of around 50% are observed for ecotoxicity, human toxicity, ozone formation, and water use. In addition, there are specific impact categories for which the recycling scenario results in higher impacts with a high probability, such as ozone depletion and fossil resource use.

Moreover, the environmental advantages of a MMC are realized only if it is effectively recycled through at least one cycle (i.e. Product System 1), as its direct incineration (Product System 2) can cause more pronounced negative environmental impacts, particularly in terms of climate change, compared to the direct incineration of a CMC (Product System 3). It is therefore of extra importance to ensure that products specifically designed for recycling are actually collected and processed according to their designated recycling route, and that they do not end up in the regular waste disposal system.

While the conclusions at system level obtained through the CFF approach and the 100:0 approach align, this is not the case at the individual product level. When the 100:0 allocation approach is applied to Product System 1, the first product exhibits higher environmental impacts compared to the second product. In contrast, the CFF method yields comparable results for both products when using an allocation factor A of 0.5. This difference is, of course, due to the A-factor and its market balance considerations. Such variations in results can significantly influence policy decisions, like taxing products with higher impacts.

Accordingly, the sensitivity of the CFF approach to its allocation factors (A and B) was investigated, with particular emphasis on factor A, which allocates environmental burdens and benefits between primary and secondary products. The effect of factor A is considerably larger than that of factor B, for which both the default value of 0 and a Belgium-specific value of 0.6 were considered.

Changes in the A-factor, while the B-factor is kept constant, can cause substantial differences in the impacts of individual products, for example reaching up to about 30% in the Climate Change category. In contrast, changes in the B-factor with a fixed A-factor have a smaller influence, with differences of up to 6% for Climate Change, as also confirmed by the uncertainty analysis. The effect of the B-factor appears to manifest more at the level of the overall product system rather than in product-specific results.

Since the A-factor has such a significant influence on individual product-level outcomes, the absence of a comprehensive set of default A-values for textile materials and applications leads to inconsistencies in the impact assessment, highlighting the need for further refinement in the PEF guidelines. A more scientifically grounded calculation method for the A-factor would facilitate the determination of new values for diverse combinations of materials and applications.

Furthermore, it has been demonstrated that the results can be strongly influenced not only by the A-factor but also by the yarn count (titer), which affects energy consumption in textile production processes such as spinning, weaving, and knitting. The titer is not a parameter within the CFF itself; rather, it relates to the influence of the recommended EF-compliant datasets used in the calculation formula. Recent updates to the EF3.1 database have made improvements by incorporating datasets with varying titer ranges. However, the available options remain limited, and a more comprehensive and detailed dataset is needed to more accurately account for yarn thickness.

In conclusion, with the potential future integration of the PEF into regulatory frameworks and policies in mind, further refinement of the CFF in relation to various textile-application related aspects (e.g., a more comprehensive set of A-factors, integration of titer influence in EF-compliant datasets) is crucial to ensure that environmental impact assessments remain accurate and comparable across different product categories. Furthermore, methodological transparency in methodology, e.g. with respect to determining the A-factor, is critical for maintaining credibility in sustainability reporting and supporting informed decision-making in the transition to a circular economy.

This study aims to advance sustainable practices in the textile industry by addressing the need for enhanced clarity and refinement in the CFF methodology, thereby contributing to its improvement for evaluating the environmental performance of recycled textile products. Future research could benefit from expanding case studies across various textile recycling contexts to further assess and enhance the reliability and application of the CFF, and by extension, the PEF.

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Data availability The authors declare that data supporting the findings of this study are available within the article and in the supplementary material.

Declarations

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References

Allacker K, Mathieux F, Manfredi S, Pelletier N, De Camillis C, Ardente F, Pant R (2014) Allocation solutions for secondary material production and end of life recovery: Proposals for product policy initiatives. Resour Conserv Recycl 88:1–12. https://doi.org/10.1016/j.resconrec.2014.03.016

Allacker K, Mathieux F, Pennington D, Pant R (2017) The search for an appropriate end-of-life formula for the purpose of the European Commission Environmental Footprint initiative. Int J Life Cycle Assess 22:1441–1458. https://doi.org/10.1007/s11367-016-1244-0

Braunfels AS, Teuber A (2023) Comparison of end-of-life allocation approaches: An analysis complementing the Battery Pass Rules for calculating the Carbon Footprint of the 'End-of-life and recycling' life cycle stage. Battery Pass Consortium. https://thebatterypass.eu/assets/images/content-guidance/pdf/2023_Battery_Passport_EOL_Analysis.pdf

Corona B, Shen L, Reike D, Rosales Carreón J, Worrell E (2019) Towards sustainable development through the circular economy—A review and critical assessment on current circularity metrics. Resour Conserv Recycl 151. https://doi.org/10.1016/j.resconrec.2019.104498

Durão V, Silvestre JD, Mateus R, de Brito J (2020) Assessment and communication of the environmental performance of construction products in Europe: Comparison between PEF and EN 15804 compliant EPD schemes. Resour Conserv Recycl 156. https://doi.org/10.1016/j.resconrec.2020.104703

Eberhardt MLC, van Stijn A, Rasmussen NF, Birkved M, Birgisdottir H (2020a) Development of a life cycle assessment allocation approach for circular economy in the built environment. Sustainability 12. https://doi.org/10.3390/su12229579

Eberhardt MLC, van Stijn A, Rasmussen FN, Birkved M, Birgisdottir H (2020b) Towards circular life cycle assessment for the built environment: a comparison of allocation approaches. IOP Conference Series: Earth and Environmental Science 588. https://doi.org/10.1088/1755-1315/588/3/032026

Ecochain (2025) Are you prepared for the product environmental footprint? https://ecochain.com/blog/are-you-prepared-for-the-product-environmental-footprint/. Accessed 21 May 2025

Ekvall T, Björklund A, Sandin G, Jelse K, Lagergren J, Rydberg M (2020) Modeling recycling in life cycle assessment. IVL Swedish Environmental Research Institute. https://www.lifecyclecenter.se/wp-content/uploads/2020_05_Modeling-recyling-in-life-cycle-assessment-1.pdf

Ekvall T, Gottfridsson M, Nellstrom M, Nilsson J, Rydberg M, Rydberg T (2021) Modelling incineration for more accurate comparisons to recycling in PEF and LCA. Waste Manag 136:153–161. https://doi.org/10.1016/j.wasman.2021.09.036

European Commission (2013a) Commission Recommendation 2013/179/EU on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations, Annex II, OJ L 124. https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2013:124:0001:0210:EN:PDF

European Commission (2013b) Building the Single Market for Green Products Facilitating better information on the environmental performance of products and organisations. Communication COM/2013/0196 final. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex:52013DC0196

European Commission (2022a) EU strategy for sustainable and circular textiles. https://environment.ec.europa.eu/publications/textiles-strategy_en

European Commission (2024a) PEFCR Guidance document - Guidance for the development of Product Environmental Footprint Category Rules (PEFCRs), version 6.3. https://eplca.jrc.ec.europa.eu/permalink/PEFCR_guidance_v6.3-2.pdf

European Commission (2024b) Supplementing Regulation (EU) 2023/1542 of the European Parliament and of the Council by establishing the methodology for the calculation and verification of the carbon footprint of electric vehicle batteries. https://eur-lex.europa.eu/eli/reg/2023/1542/oj/eng Accessed June 2025

European Environement Agency (2024) Municipal waste landfill rates in Europe by country. https://www.eea.europa.eu/en/analysis/maps-and-charts/municipal-waste-landfill-rates-in-2. Accessed May 2025

Europeropeaan Commission (2022b) Environmental Footprint reference packages Annex C. https://eplca.jrc.ec.europa.eu/LCDN/developerEF.html. Accessed May 2025

EU strategy for sustainable and circular textiles (2025) https://environment.ec.europa.eu/strategy/textiles-strategy_en. Accessed May

Farrapo AC, Matheus TT, Lagunes RM, Filleti R, Yamaji F, Lopes Silva DA (2023) The application of circular footprint formula in bioenergy/bioeconomy: challenges, case study, and comparison with life cycle assessment allocation methods. Sustainability 15. https://doi.org/10.3390/su15032339

Finkbeiner M (2014) Product environmental footprint—breakthrough or breakdown for policy implementation of life cycle assessment? Int J Life Cycle Assess 19:266–271. https://doi.org/10.1007/s11367-013-0678-x

Hammar T, Peñaloza D, Hanning A-C (2024) Life cycle assessment of a circular textile value chain: the case of a garment made from chemically recycled cotton. Int J Life Cycle Assess 29:1880–1898. https://doi.org/10.1007/s11367-024-02346-2

Haslinger S, Wang Y, Rissanen M, Lossa MB, Tanttu M, Ilen E, Määttänen M, Harlin A, Hummel M, Sixta H (2019) Recycling of vat and reactive dyed textile waste to new colored man-made cellulose fibers. Green Chem 21:5598–5610. https://doi.org/10.1039/c9gc02776a

Hermansson F, Ekvall T, Janssen M, Svanström M (2022) Allocation in recycling of composites - the case of life cycle assessment of products from carbon fiber composites. Int J Life Cycle Assess 27:419–432. https://doi.org/10.1007/s11367-022-02039-8

International EPD System The International EPD System (2025) https://www.environdec.com. Accessed May

Ioelovich M (2018) Energy potential of natural, synthetic polymers and waste materials - a review. Acad J Polym Sci 1. https://doi.org/10.19080/ajop.2018.01.555553

ISO 14040 (2006) Environmental management—life cycle assessment— principles and framework. Geneva, Switzerland

ISO 14044 (2006) Environmental management—life cycle assessment— requirements and guidelines. Geneva, Switzerland

Leal Filho W, Perry P, Heim H, Dinis MAP, Moda H, Ebhuoma E, Paço A (2022) An overview of the contribution of the textiles sector to climate change. Front Environ Sci 10. https://doi.org/10.3389/fenvs.2022.973102

Mordaschew V, Tackenberg S (2024) The Product Environmental Footprint – A Critical Review. Procedia Comput Sci 232:493–503. https://doi.org/10.1016/j.procs.2024.01.049

Obrecht TP, Jordan S, Legat A, Ruschi Mendes Saade M, Passer A (2021) An LCA methodolody for assessing the environmental impacts of building components before and after refurbishment. J Clean Prod 327. https://doi.org/10.1016/j.jclepro.2021.129527

Prasanga WC, Nawarathna A, Rathnasiri P (2021) Use of Cotton Apparel Waste as an Energy Source for Biomass Boilers: A Feasibility Study. Int J Des Nat Ecodyn 16:41–51. https://doi.org/10.18280/ijdne.160106

Quantis (2025) Product Environmental Footprint Category Rules: Apparel and Footwear, version 3.1. https://pefapparelandfootwear.eu/afw_pefcr_v3-1_final/

Rickert J, Ciroth A (2020) Application of the circular footprint formula (CFF) in product environmental footprint (PEF): Illustrated through a case study on intermediate paper products. GreenDelta GmbH. https://www.greendelta.com/wp-content/uploads/2020/10/2020-10-PEF-application-of-CFF.pdf

Santos-Beneit F, Chen LM, Bordel S, Frutos de la Flor R, Garcia-Depraect O, Lebrero R, Rodriguez-Vega S, Munoz R, Borner RA, Borner T (2023) Screening Enzymes That Can Depolymerize Commercial Biodegradable Polymers: Heterologous Expression of Fusarium solani Cutinase in Escherichia coli. Microorganisms 11. https://doi.org/10.3390/microorganisms11020328

Sinkko T, Amadei A, Venturelli S, Ardente F (2024) Exploring the environmental performance of alternative food packaging products in the European Union – Life cycle impacts of single-use and multiple-use packaging. Publications Office of the European Union. https://publications.jrc.ec.europa.eu/repository/handle/JRC136771

Stadt Wien (2023) Rautenweg landfill, Vienna‘s only municipal landfill. https://www.wien.gv.at/umwelt/ma48/service/publikationen/pdf/infoblatt-deponie-rautenweg-en.pdf. Accessed May 2025

Starlinger (2024) Recycling line recoSTAR PET art. https://www.starlinger.com/fileadmin/user_upload/recoSTAR_PET_art_24941_published_01_2024_6Seiter_geschuetzt.pdf

Stubbe B, Van Vrekhem S, Huysman S, Tilkin RG, De Schrijver I, Vanneste M (2024) White paper on textile fiber recycling technologies. Sustainability 16. https://doi.org/10.3390/su16020618

Toniolo S, Mazzi A, Pieretto C, Scipioni A (2017) Allocation strategies in comparative life cycle assessment for recycling: Considerations from case studies. Resour Conserv Recycl 117:249–261. https://doi.org/10.1016/j.resconrec.2016.10.011

Van der Velden NM, Patel MK, Vogtländer JG (2014) LCA benchmarking study on textiles made of cotton, polyester, nylon, acryl, or elastane. Int J Life Cycle Assess 19:331–356. https://doi.org/10.1007/s11367-013-0626-9

Vieira M (2020) Environmental footprint transition phase: what is happening now that the EF pilot phase is finished? https://pre-sustainability.com/articles/environmental-footprint-transition-phase-projects/. Accessed May 2025

Vreriks SG (2020) The environmental performance of the new circular mattress: A life cycle study. Dissertation, University of Twente

Zampori L, Pant R (2019) Suggestions for updating the Product Environmental Footprint (PEF) method. Joint Res Cent (JRC) 115959. https://publications.jrc.ec.europa.eu/repository/handle/JRC115959