Supplementary material to article titled "Exploring the Environmental Impact of Textile Recycling in Europe: A Consequential Life Cycle Assessment"

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1 Background to assumptions in Section 1 of the article

We assume that 6.5 million tonnes textiles are discarded in the EU today and in 2035, which is about 13.5 kg per person. This is based on estimates made by McKinsey & Company (2022) and Köhler et al. (2021). McKinsey & Company estimate that 7.0-7.5 million tonnes of textiles were discarded in the EU and Switzerland in 2020, with 85% coming from clothing or household textiles discarded by consumers. This includes textiles disposed of in the household waste, which are mostly incinerated or landfilled today, and textiles collected separately for sorting into reuse, recycling (textile-to-textile or, most common today, downcycling), incineration or landfilling. However, this figure does not account for textiles directly reused between customers or handed in to reuse platforms for direct reselling. Köhler et al. (2021) estimate, based on consumption data, that about 3.3 to 3.7 million tonnes of clothing and household waste were discarded to mixed waste streams, and that 1.7 to 2.1 million tonnes were separately collected in the EU-27 in 2020, totalling 5.0 to 5.8 million tonnes of clothing and household textiles being discarded annually. Assuming this represents 85% of all discarded textiles, the total discarded textiles range from 5.9 to 6.8 million tonnes, which is slightly lower than McKinsey & Company's estimate. The variation in numbers is likely due to the lack of centralized or uniform data collection on textile flows in the EU, reflecting a gap and uncertainty in the knowledge of textile waste flows (Walter 2023). Noteworthy, technical textiles are likely not included neither by McKinsey & Company (according to our interpretation of their numbers) or Köhler and colleagues. Among others, technical textiles include textiles used in the automotive, construction and agriculture sectors, which may constitute some of the textiles entering textile-to-textile recycling in 2035. Furthermore, the assumption that the same volumes of textiles will be discarded in 2035 as today, rests on the fact that the EU population is projected to remain as today, with growth until 2026 and then a decline

(Eurostat 2023a), and that the economic future, and its effects on consumption levels, of the EU is uncertain. In contrast, McKinsey & Company (2022) estimate that there will be a 20% growth of EU textile waste from 2020 to 2030. While the exact numbers for current and future discarded textile volumes are uncertain, they are not critical for this study's analysis or conclusions. What matters is the order-of-magnitude of the climate water-deprivation impact of the identified consequences, their relative importance, and the influence of uncertainties studied.

2 Equation parameters per studied change

2.1 Increased collection and sorting for recycling

The climate/water deprivation impact of collection and sorting in the EU today represent the average across all means and distances of collection, as well as the technologies of sorting. As this is a consequential LCA, we focus solely on the additional transports involved in collection compared to the business-as-usual scenario (incineration or landfilling). Similarly, when estimating the third consequence (reduced incineration and landfill), the transports to landfilling/incineration are not included.

The decarbonisation rate of the EU energy system corresponds to parameter A in the Monte-Carlo analysis, see Section 3.1.

To estimate the climate/water deprivation impact of collection and sorting, we used two sources:

- Lidfeldt et al. (2022): an LCA of four different recycling routes in Sweden, which includes data on collection of textile waste in southern Sweden, the subsequent transportation of these textiles for manual sorting in Lithuania, and a transport back to southern Sweden for automatic sorting. Collection and automatic sorting data are based on primary sources, whereas sorting data are based on a secondary source:
  Spathas (2017).
- Trzepacz et al. (2023): an LCA of the management of used textiles in Europe, based on primary data from several key collectors and manual sorting facilities in Europe. The data cover transports from collection points to sorting hubs, and the subsequent sorting, pressing, and baling.

The LCA model of Lidfeldt et al. (2022) yields, per tonne collected textiles:

- Climate impact results for collection and sorting of 0.190 and 0.116 tonnes CO_2 eq., respectively, and
- Water deprivation impact results for collection and sorting of 0.00112 and 0.00970 m^3 world eq., respectively.

Collection includes transport prior to manual sorting, while sorting includes transport between manual and automatic sorting, as well as the activities at both manual and automatic sorting facilities. The above results used in this report differ slightly from the original results due to two modifications we made to the LCA model: (i) the initial transport of textile waste from collection bins to the recycling centre was excluded, as a similar transport occurs if textile waste is incinerated or landfilled, and this consequential LCA focuses solely on additional processes; and (ii) smaller trucks were assumed, correcting an error in the original model, where LCI data on larger trucks were mistakenly applied. These adjustments were made possible as the authors of this report were also involved in study by Lidfeldt et al. (2022).

Using the results of Trzepacs et al. (2023), we estimated the climate impact for collection and sorting (activities at the sorting facility) to be 0.076 and 0.016 tonnes CO_2 eq., respectively, per tonne textiles being collected.

Considering the results of both Trzepacs et al. (2023) and Lidfeldt et al. (2022), we made a conservative estimate for the climate impact of collection and sorting by taking the highest numbers, i.e., those from Lidfeldt et al. (2022). This gives a total climate impact of about 0.31 tonnes CO_2 eq. per tonne collected textiles. Making a conservative estimate translates to a disadvantage for recycling. However, we deem the effect of this assumption to be negligible, considering that collection and sorting turned out to have a small contribution compared to the total climate-impact results (see Figure 3 in the main article).

For water-deprivation impact, we assumed the numbers from Lidfeldt et al. (2022), which totals about 0.011 m^3 world eq. per t textiles being collected.

2.2 More recycling processes

The proportion of recycling occurring outside the EU in 2035 corresponds to parameter C in the Monte Carlo analysis (see Section 3.3). Accordingly, the proportion of recycling occurring in the EU in 2035 is calculated as 100% minus parameter C.

The climate and water deprivation impacts of recycling processes in and outside the EU corresponds to parameters I and J, respectively, in the Monte Carlo analysis (see Section 3). These impacts are expressed per tonne of fibres leaving the recycling process. Therefore, the input of collected and sorted textiles are multiplied by the recycling yield, which corresponds to parameter E in the Monte Carlo analysis (see Section 3.6).

The decarbonisation rates of the EU and Asian energy systems correspond to parameters A and B, respectively, in the Monte-Carlo analysis (see Sections 3.1 and 3.2).

2.3 Reduced incineration and landfilling

The climate impact assessment method used does not include impact from incineration of fibres made of non-fossil resources (i.e., biobased fibres), and this impact is therefore not included in the above equation.

The proportion of fibres that are landfilled versus incinerated (i.e., the proportion of fibres landfilled out of the total fibres either landfilled or incinerated) in the EU by 2035 is represented as parameter G in the Monte Carlo analysis (see Section 3.8). From this parameter, we calculated the proportion of fibres incinerated out of those landfilled or incinerated.

The proportion of fossil fibres among those sent to recycling in the EU by 2035 is represented by parameter F in the Monte Carlo analysis (see Section 3.7). Using this parameter, we calculated the proportion of non-fossil fibres.

The replacement rate is parameter H in the Monte Carlo analysis (see Section 3.9). This parameter is included in the equation above because only when primary fibre production is replaced, does the quantity of textile fibres being incinerated or landfilled decrease (as all fibres produced eventually reach incineration or landfill, disregarding those lost in nature).

Table S1 provides the assumed climate and water-deprivation impacts of landfilling and incineration of fossil fibres and non-fossil fibres, respectively, along with the references used.

Table S1. Climate and water-deprivation impact of landfilling and incineration of fossil fibres and landfilling of non-fossil fibres.

Process	**Climate impact [t CO_2 eq./t fibres]**[1]	**Water-deprivation impact [m^3 world eq./t fibres]**	**Reference**[2]
Incinerating fossil fibres	2.0	0.022	Database: Ecoinvent 3.9. Dataset for incineration of polyethylene terephthalate in Switzerland.
Incinerating non-fossil fibres	*Assumed to be zero*	0.19	Database: Sphera Managed LCA Content (MLC) database, version 10.7.1.28. Dataset for textile (animal and plant based) in waste incineration plant (11.9% H2O content)
Landfilling fossil fibres	0.088	0.013	Database: Ecoinvent 3.9. Dataset for landfilling of polyethylene terephthalate in Switzerland.
Landfilling non-fossil fibres	0.99	0.00020	Database: Ecoinvent 3.9. Global dataset for landfilling of waste yarn and textile.

We exclude the impacts of transports from households to the incineration or landfill sites, as similar transports occur also in the collection of textile waste destined for sorting and recycling (i.e., transports from households to, e.g., textile collection bins), which are likewise excluded from the equation for the impacts of more collection and sorting for recycling (CI_1 and WDI_1 in Section 2.3.1 of the main article). This assumption is based on the premise that these transports remain unchanged when textile recycling in the EU increases.

It is important to note that such transports vary greatly across the EU, depending on how municipalities or regions organise their waste collection systems. However, if a study were conducted to assess the consequences of increased textile recycling within a specific municipality or region, it would be relevant to study the logistics in greater detail – from the point of textile disposal (e.g., households) to the recycling facilities. This is particularly important if changes in recycling affect reliance on car transports, as previous studies (e.g.,

[1] The higher climate impact of landfilling of non-fossil, compared to fossil fibres, is due to the decomposition of biogenic carbon into methane.
[2] The geographical scopes of the datasets in the table are Switzerland (CH) and "rest of the world" (RoW). Other geographical scopes were available in Ecoinvent 3.9, but the differences in terms of climate impact were negligible.

Lidfeldt et al. 2022) have shown that car can greatly influence the environmental impact of textile recycling.

2.4 Reduced primary fibre production

Recycling yield refers to the proportion of materials collected for recycling that are converted into recycled fibres ready for subsequent production, such as yarn spinning. This corresponds to parameter E in the Monte Carlo analysis (See Section 3.6).

The decarbonisation rate of the Asian energy system by 2035 is represented by parameter B in the Monte Carlo analysis (See Section 3.2). Here we assume that the climate impact of reduced primary fibre production decreases in alignment with the overall decarbonisation rate of the Asian energy system until 2035.

The climate and water-deprivation impacts of reduced primary fibre production correspond to parameters D1 and D2, respectively, in the Monte Carlo analysis.

The replacement rate indicates the extent to which the production of recycled fibres offsets the primary production of primary textile fibres, which is represented by parameter H in the Monte-Carlo analysis (see Section 3.9).

2.5 Compensation for the decrease in recovered energy

The proportion of fibres incinerated versus landfilled (i.e., the proportion of fibres incinerated out of those either landfilled or incinerated) in the EU by 2035, is derived from parameter G in the Monte Carlo analysis (see Section 3.8).

The replacement rate, represented by parameter H in the Monte Carlo analysis, (see Section 3.9), is considered in the equations for the compensation for the loss of recovered energy. The reason for this is that only when primary fibre production is replaced, is the quantity of textile fibres being incinerated decreased with a subsequent need to compensate the loss in energy recovered. Note that this parameter is also considered when estimating the climate and water-deprivation impacts of reduced incineration and landfilling (see Section 2.3).

Table S2 provides the assumptions and references for the quantities of heat and electricity generated from incinerating textile waste, as well as the climate and water-deprivation impact of electricity and heat production in the EU.

Table S2. Assumed quantities of heat and electricity generated from incinerating textile waste, and assumed climate impact (CI) and water deprivation impact (WDI) of electricity and heat production in the EU.

Process	Data	Reference
Incinerating textile waste in combined heat and power plant	Heat generated: 1553 MJ/t textile waste. Electricity generated: 854 MJ/t textile waste.	Database: Sphera's MLC database, version 10.7.1.28. Dataset for incineration of textiles in Europe.
Electricity production	CI: 0.00033 t CO2 eq./kWh WDI: 0.072 m^3 world eq./kWh	Database: Sphera's MLC database, version 10.7.1.28. Dataset for average European electricity.
Heat production	CI: 0.00023 t CO2 eq./kWh WDI: 0.0065 m^3 world eq./kWh	Database: Sphera's MLC database, 10.7.1.28. Dataset for average European district heating.

Here we assume that all losses of recovered energy are fully compensated. However, in practice, a reduced energy supply would likely lead to higher energy prices, thereby lowering demand and resulting in only partial replacement of recovered energy losses. As with our approach for reduced primary fibre production (see Section 2.4), we could have applied a replacement rate below 100%. This adjustment of the model would have a negligible effect on the results, given the relatively small contribution of recovered energy compensation to the overall results (as shown in Section 3 of the main article).

3 Parameters randomized in the Monte Carlo analysis

The Monte-Carlo simulations in this study are done for the equations presented in Section 2.3 of the main article and the uncertain parameters (A to I) represented by probability distributions. The subsections below detail these parameters and their assumed normal distributions and standard deviations.

Consequential LCAs typically use marginal data representing the environmental impact of the technology (of production, transports, end-of-life processes, etc.) influenced by a change in demand. This is often the technology with the highest marginal cost. Marginal data can vary depending on the time horizon and the scale of change considered. For instance, short-term marginal technologies depend on a change in the utilisation of existing production capacity, while long-term marginal technologies depend on a change in the production capacity (Ekvall 2019), which in turn depends on future changes in technology, demand, and policy – factors which are inherently uncertain. Short-term marginal technologies often have relatively high environmental impact, while long-term marginal technologies may have lower environmental impact due to advancements in technology or policy-driven changes.

This study focuses on long-term (up to 2035) and large-scale (a 1.6 million tonnes increase in textile recycling) marginal technologies across collection, sorting, transports, recycling, fibre production, and end-of-life processes. Instead of relying on speculative or uncommon future technologies, we assume the use of common or average technologies employed today. We assume that uncertainties associated with this approach are addressed by the normal distributions assigned to the environmental impacts of these assumed (marginal) technologies. Additionally, for the climate impact assessment, uncertainties regarding the future marginal technologies are further addressed by the decarbonisation parameters and their probability distributions.

If nothing else is stated, the standard deviation of each randomized parameter is assumed to be one fourth of the estimated mean.

3.1 Parameter A: Decarbonisation rate of the EU energy system until 2035

This parameter influences the climate impact of all processes located in the EU, including collection, sorting, and recycling processes. Note that this concerns transports as well as industrial processes using any type of energy. The rate is expressed per work or function delivered by these processes, as detailed in the equations in Section 2.3 of the main article. Simplified, this represents the decarbonisation per energy unit consumed, focusing not on the absolute reduction in the EU energy system's climate impact, but on the relative reduction per unit of energy used.

To estimate the decarbonisation rate per energy unit, our starting point is EU's climate impact reduction targets, which are defined in absolute terms. The EU is legally obliged to reduce the greenhouse gas emissions by at least 55% until 2030, supported by the "Fit for 55" legislative package (European Council 2023). Additionally, the EU aims for climate neutrality by 2050, where any remaining emissions shall be offset by carbon-removal technologies or increased carbon uptake in the land-use sector (European Parliament 2023). Consequently, there is an anticipated decarbonisation of energy use by 2035, which would be a continuation of current trends (Zachmann et al. 2021, Ancygier et al. 2020).

Next, we look at studies with scenarios of future CO_2 intensity of European electricity mixes. Ancygier et al. (2020) review scenarios for 2030, 2040, and 2050, and relate the scenarios to what is required to be compatible with the Paris Agreement, which the EU targets aim to reflect. For 2030, the scenarios project CO_2 intensities between 26 and 170 g CO_2 per kWh (disregarding a scenario that assumed 0 g CO_2 per kWh for all time horizons). For 2040, the scenarios range between 0 and 70 g CO_2 per kWh. Assuming a linear progression between 2030 and 2040, CO_2 intensity in 2035 is estimated to fall between 13 and 120 g CO_2 per kWh.

For comparison, the EU's CO_2 intensity in 2020 was 265 g CO_2 per kWh (Ancygier et al. 2020). The scenarios reviewed by Ancygier and colleagues therefore indicate a decarbonisation of the EU electricity mix of approximately 55-95% from 2020 until 2035.

But electricity constitutes only part of the EU energy mix. The combustion of fuels for heating and transports also plays an important role. Decarbonisation in these sectors is largely expected to be achieved by electrification rather than improvements in the fuels themselves as such. Considering both electrification and fuel improvements, International Transport Forum (2020) presents two scenarios representing different levels of ambition for CO_2 emissions reduction per tonne-kilometre in Europe from 2015 to 2050. The "current ambitions" scenario estimates a decrease in emissions from about 35 g CO_2 per tonne-kilometre in 2020 to about 25 CO_2 per tonne-kilometre in 2035. In contrast, the "high ambitions" scenario estimates a decrease from about 35 g CO_2 per tonne-kilometre in 2020 to about 15 CO_2 per tonne-kilometre in 2035. These scenarios suggest a decarbonisation rate of approximately 30-60% in transport emissions over this period.

Based on the analysis above, we assume a mean decarbonisation rate of 45% for the EU energy system until 2035. This rate lies in the middle of the range outlined of the transport decarbonisation scenarios. Although the rate is higher than the "current ambitions" scenario for transport decarbonisation, it is lower than the decarbonisation rates in all the electricity mix scenarios reviewed by Ancygier et al. (2020). It is important to note that out of energy consumption in the EU in 2020, only 25% consists of electricity (Eurostat 2023b), although this proportion is expected to rise due to increasing electrification. Still, our choice of the mean decarbonisation rate reflects a slightly conservative stance compared to the electricity scenarios reviewed by Ancygier and colleagues, and the EU's political target of a 55% decarbonisation until 2030. However, a conservative stance is justified considering that the current projections of the greenhouse gas emissions of the EU member states are not aligned with the 55% target (European Commission 2024). Nevertheless, with the assumed standard deviation of one fourth of the mean, the probability distribution covers many of the more optimistic scenarios of Ancygier and colleagues as well as the 55% target, as shown in Figure S1.

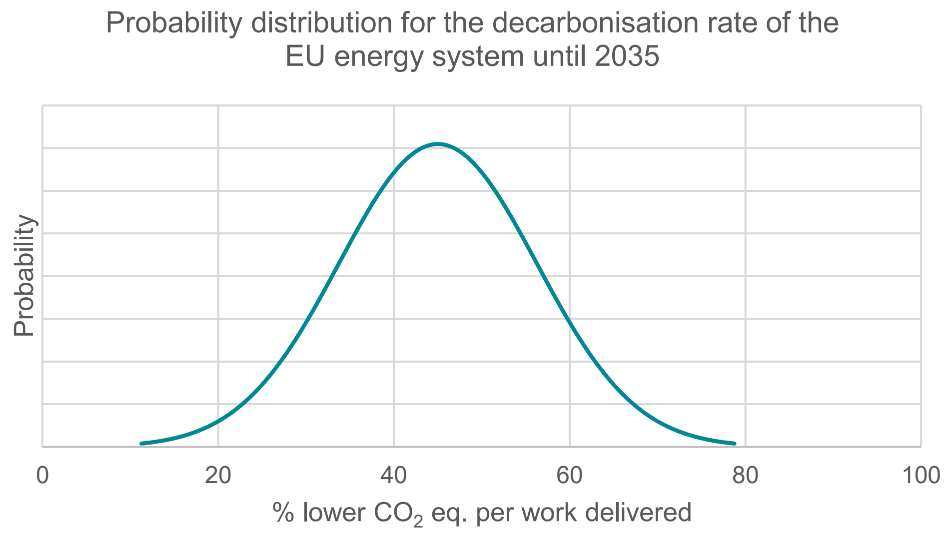

Figure S1. Probability distribution of parameter A, the decarbonisation rate of the EU energy system until 2035.

3.2 Parameter B: Decarbonisation rate of the Asian energy system until 2035

This parameter is assumed to influence the climate impact of all processes assumed to be located outside the EU, primarily in Asia, including recycling and fibre production processes. It concerns the decarbonisation of industrial processes and transports, regardless of the energy source used. Similar to parameter A (decarbonisation rate of the EU energy system until 2035), this can be simplified as the decarbonisation per energy unit. In other words, it focuses not on the absolute reduction in the climate impact of the Asian energy system by 2035, but on the relative reduction in climate impact per energy unit consumed.

As a starting point, we use targets and scenarios for reduced climate impact in Asia as the basis for defining the mean of this parameter. We mainly consider China and India as they are two powerhouses in the Asian textile industry. China has set a target to achieve net-zero climate impact by 2060 (The World Bank 2022), with a goal to reduce its carbon intensity – the greenhouse gas emissions per unit of gross domestic product (GDP) – by 65% by 2030 compared to 2005 levels, continuing recent trends (European Parliament 2022). India, with a target of achieving net-zero climate impact by 2070, aims for 40% non-fossil electricity generation by 2030, up from approximately 25% in 2022, and 50% cumulative electric power installed capacity from non-fossil fuel-based energy resources by 2030 (Climate Action Tracker 2023). With the current policies, India is expected to exceed this target, achieving 60-70% non-fossil electricity by 2030 (Climate Action Tracker 2023).

Although both China and India are projected to increase their absolute climate impact due to growing populations and economies, the carbon intensity per GDP is expected to decrease because of the above-described targets and policies. Here it is important to note that carbon intensity per GDP is not the same as carbon intensity per energy unit.

Considering scenarios of future use of carbon-intensive versus low-carbon energy sources, it is interesting to consider the Announced Pledges Scenario (APS) provided by the International Energy Agency (2021). This scenario aligns with China's net-zero 2060 target. Under the APS, primary energy use of based on coal, oil, and natural gas is projected to rise by about 6% by 2030, compared to 2020, while renewable and nuclear energy use is projected to rise by nearly 80% over the same period (International Energy Agency 2021, Table 2.1). In absolute terms, this results in an increase in the non-fossil energy share from 15 to 23%. However, the greatest shift in the non-fossil energy share is anticipated after 2030, with the APS projecting a rise to 74% by 2060. This suggests that the decarbonisation per energy unit in China up to 2035 will be considerably lower than the above-mentioned reduction in carbon intensity per GDP. In other words, the APS underscores the continued reliance on fossil fuels in the short term and the importance of considering this in decarbonisation assessment.

Based on the analysis above, we assume lower decarbonisation rate in Asia than in the EU by 2035 (parameter A). Generally, net-zero targets in Asia are set further into the future compared to the EU, and policies and technological transitions – such as the electrification of transport and the phasing out of coal – are expected to occur later. This is also reflected in the Paris Agreement, where Asia has lower CO_2 reduction targets than the EU (Climate Watch 2023). Because of this and given the lack of specific scenarios on decarbonisation rates per energy unit – both for specific Asian countries and for the region as a whole – we assume the mean decarbonisation rate to be 22.5% by 2035, which is half of the assumed mean for the EU rate (parameter A). To account for the high uncertainty of this assumption, we assume a higher standard deviation: one third of the mean as opposed to the default one fourth. This results in the probability distribution presented in Figure S2.

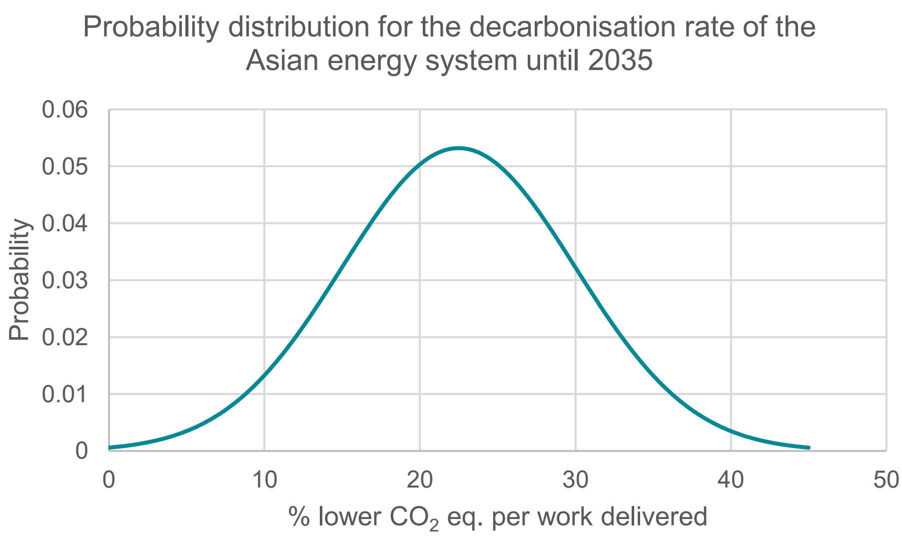

Figure S2. Probability distribution of parameter B, the decarbonisation rate of the Asian energy system until 2035.

3.3 Parameter C: Proportion of recycling occurring outside the EU in 2035

We found no data on the present share, or projections for the future share, of recycling for the EU market occurring within or outside the EU. While Asia dominates global fibre production today, there are significant European initiatives in textile recycling. For instance, Dahlbom et al. (2023) identified 33 fibre-to-fibre recycling actors in Europe, that presently operate at a commercial scale, with an estimated total potential capacity of 1.3 million tonnes of fibres per year by 2025. This can be contrasted with the 585 000 tonnes increase in textile-to-textile recycling being assessed in the present study. Since Dahlbom and colleagues conducted their study, the fibre-to-fibre recycling industry in Europe has had difficulties, for example manifested by the bankruptcy of Renewcell (Innovation in Textiles 2024). As such, the scale of fibre-to-fibre recycling in Europe in 2025 will be nowhere near the potential capacity outlined by Dahlbom and colleagues. However, given the numerous additional European textile recycling initiatives at research and development stage, it is reasonable to expect that most textile recycling for the EU market will occur in the EU by 2035.

Based on the above analysis, we assume the mean of the proportion of recycling occurring outside the EU to be 30%, with a standard deviation of one third of the mean (instead of the default one fourth), reflecting the lack of direct data to support the assumed mean. These assumptions result in the probability distribution shown in Figure S3.

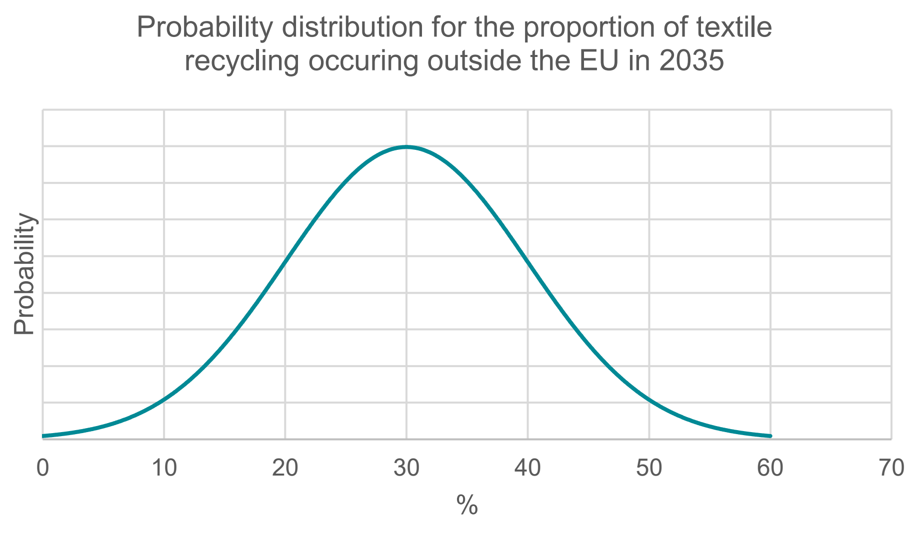

Figure S3. Probability distribution of parameter C, the proportion of textile recycling occurring outside the EU in 2035.

3.4 Parameter D1: Climate impact of reduced primary fibre production

This parameter reflects the climate impact of production of primary fibres that are presumably avoided when textile recycling increases today.

The climate impact of primary fibres varies considerably, depending on fibre type and production conditions. Excluding the biogenic carbon stored in fibres (as it is ultimately released during incineration or landfilling), the typical climate impact of textile fibres ranges from 0.5 to 10 t CO_2 eq. per tonne fibres (Sandin et al. 2019b). The average climate impact of avoided fibres can be assumed to lie approximately in the middle of this range.

Certain outlier fibres, such as wool, cause higher climate impact. However, these outliers represent a relatively small proportion of the global fibre market. For instance, wool accounts for less than 1% of the global fibre market (Textile Exchange 2022). Consequently, we assume that outliers have a negligible effect on the average climate impact of reduced primary fibre production.

Furthermore, the fibres assumed in an LCA model of Swedish clothing consumption (Sandin et al. 2019a) exhibit an average climate impact of 4.25 tonnes CO_2 eq. per tonne fibres. This LCA model considers a mix of common fibres – cotton, polyester, polyamide, viscose, elastane – used in a variety of common garments. The model represents generic and average garment categories, considering the losses of fibres from fibre production to a garment purchased by a customer. We calculated the average based on the fibre production's contribution to the overall climate impact of Swedish clothing consumption, as presented by Sandin et al. (2019a).

The Ecoinvent database provides global market datasets for conventional cotton and polyester fibres. If we consider the fossil greenhouse gas emissions (GWP-fossil) and emissions from land use and land use change (GWP-luluc), and thereby disregard the biogenic carbon temporarily stored in the cotton fibres, version 3.10 of Ecoinvent yields climate impact of cotton and polyester of about 3.8 and 4.5 tonnes CO_2 eq. per tonne fibres, respectively. Compared to Ecoinvent 3.8, the impact for cotton has decreased with almost 10%, indicating that the average impact per fibre in the LCA model of Sandin et al. (2019a) would also decrease if a more updated dataset on cotton was used (they used Ecoinvent 3.5).

Based on above sources and discussion, we assume a mean of 4.0 tonnes CO_2 eq. per tonne reduced primary fibre production, with the corresponding probability curve illustrated in Figure S4.

This mean is considered representative for home textiles as well, as they are commonly made of similar fibres and there are no large, general differences in the climate impact of the common fibre types (the variations tend to be greater between production sites/plantations than between fibre types; Sandin et al. 2019b). Notably, the fibre mix of Swedish clothing consumption in Sandin et al. (2019a) includes a higher proportion of cotton compared to global averages. This is backed up by Statista (2023) which reports that new clothing purchased in the EU in 2015 consisted of 43% cotton and 19% polyester. While fibre mixes influenced by large-scale increases in textile recycling may diverge from both EU and global averages, the climate impacts of cotton and polyester are broadly similar.

Thus, uncertainties regarding the replaced fibre mix are assumed to be adequately captured by the assumed normal distribution.

Note that the probability distribution for parameter D1 is based on estimates of the average climate impact of textile fibres produced in recent years. Anticipated improvements in fibre production until 2035 – such as the adoption of new technologies, improved farming practices, or an increased use of renewable energy – are accounted for through the decarbonisation parameters (A and B). Thus, the probability distribution for parameter D1 primarily reflects the uncertainties in the types of fibres being replaced and the variations in climate impact within specific fibre types, but it does not encompass the uncertainty related to future decarbonisation of fibre production processes.

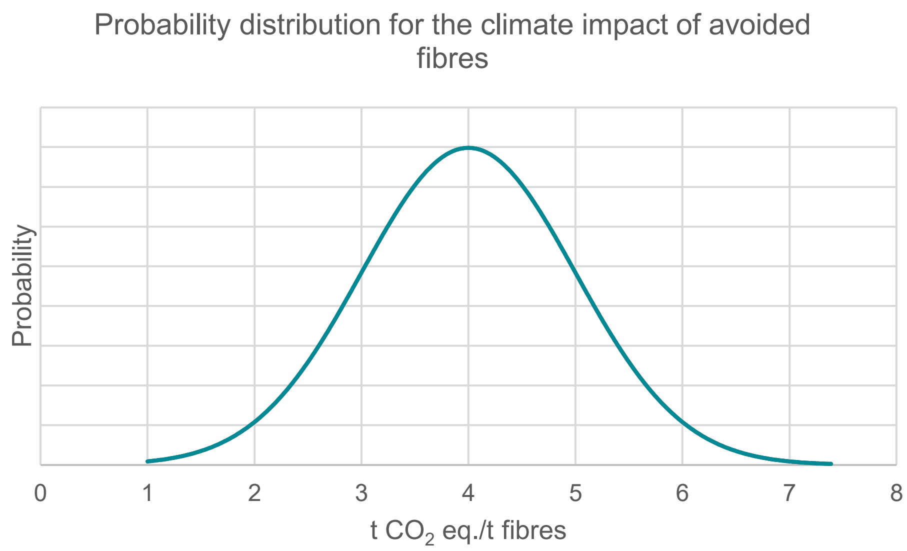

Figure S4. Probability distribution of parameter D1, climate impact of reduced primary fibre production.

3.5 Parameter D2: Water-deprivation impact of reduced primary fibre production

This parameter concerns the water-deprivation impact of production of primary fibres that are presumably avoided by an increase in textile recycling.

The water-deprivation impact varies considerably, depending on fibre type and production conditions. The variations are greater than for climate impact (see Section 3.4). For instance, cotton cultivation exhibits extreme variability, ranging from near zero for rain-fed cotton grown in wet regions (e.g., see Sandin et al. 2019b) to several hundred thousand m^3 world eq. per tonne of fibres in regions with water scarcity.[3]

The fibres used in an LCA model of Swedish clothing consumption (Sandin et al. 2019a) show an average water-deprivation impact of 44000 m^3 world eq. per tonne. This value

[3] As an example, combining the blue water use of cotton grown in Turkmenistan according to Chapagain et al. (2006) with the cotton-specific characterisation factors for Turkmenistan from Boulay et al. (2019), using the AWARE method, yields a water deprivation impact of 374 000 m^3 world eq.

reflects a mix of common fibres – cotton, polyester, polyamide, viscose, elastane – used in a range of common garments modelled to represent generic and average garment categories. We calculated the average based on the fibre production's contribution to the overall water-deprivation impact of Swedish clothing consumption, as presented by Sandin et al. (2019a). Cotton cultivation is the dominant contributor to this value.

We estimate the mean water-deprivation impact of avoided primary fibre production to be 33,000 m^3 world eq. per tonne, representing 75% of the average value of Sandin et al. (2019b). We assume a lower value than Sandin et al. (2019b) as we anticipate the proportion of cotton among avoided fibres in the EU in 2035 to be lower than that observed in the LCA model of Swedish clothing consumption from 2019. The main reasoning for this is that fossil fibres is gaining market shares and as polyester is assumed to constitute a substantial proportion of the future fibre-to-fibre recycling, as is further discussed in Section 3.7. To account for the significant uncertainty, we assign a standard deviation equal to one-third of the mean, resulting in the probability distribution shown in Figure S5.

Note that the probability distribution for parameter D2 is determined based on estimates of the average water-deprivation impact of textile fibres produced in recent years. Unlike the climate-impact assessment, which accounts for future decarbonisation, potential changes until 2035 are neglected. For water deprivation, we assess the future to be more uncertain in terms of whether water-deprivation impact will be higher or lower compared to today. Changes due to technologies or improved farming practices may reduce the water quantities used. However, climate change may cause increased water scarcity which may aggravate water deprivation and thereby increase the characterisation factors. Additionally, there is a lack of politically set targets on a global level to reduce water-deprivation impact, which contrasts with the issue of climate change. The uncertainty of this parameters is further explored in the sensitivity analysis.

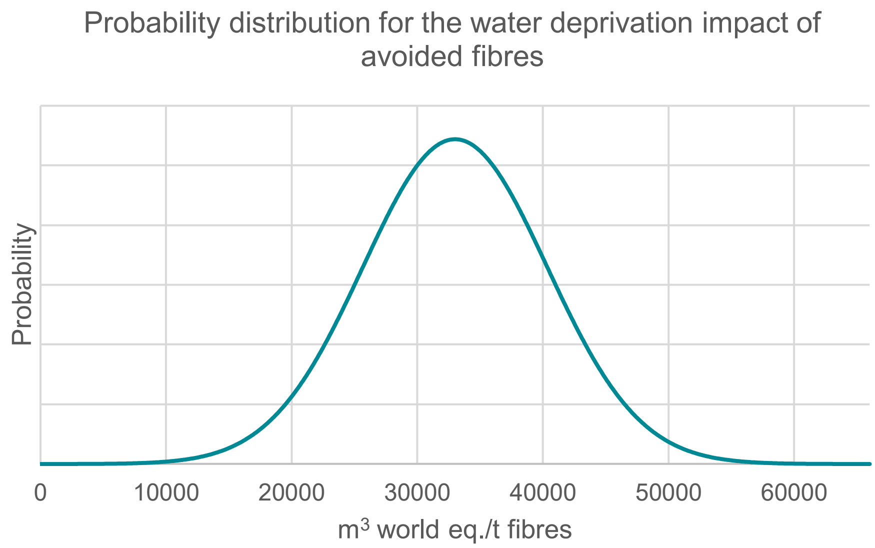

Figure S5. Probability distribution of parameter D2, water-deprivation impact of reduced primary fibre production.

3.6 Parameter E: Recycling yield

This parameter represents the yield from collected discarded textile products to fibres ready for subsequent production processes, such as yarn spinning, and nonwoven production

For chemical recycling[4] of cellulose fibres – resulting in dissolving pulp that can substitute dissolving pulp, derived from sources such as wood or bamboo, in viscose production – parameter E thus accounts for the yield across the entire process until new fibres. This yield definition is used for chemical recycling although the substitution of primary resources – if it occurs, see Section 3.9 – takes place at the dissolving pulp stage rather than the fibre stage.

Lidfeldt et al. (2022) present four case studies on textile recycling, reporting yields of approximately 90% for mechanical recycling of cotton and wool, and 60% and 57% for chemical recycling of polyester and cotton (into viscose), respectively. Peters et al. (2019) studied recycling of polyester/cotton blends, assuming a yield of 65%. Hagoort et al. (2013) studied polyester recycling from post-consumer clothing and assumed a yield of 60%. Östlund et al. (2015) present three scenarios for mechanical recycling, with yields ranging from 60% to 90%. Esteve-Turrillas and de la Guardia (2017) assume a 96% yield for mechanical recycling of cotton.

Above references suggest considerable differences in yield between chemical and mechanical recycling. For textiles discarded in the EU in 2035 that undergo textile-to-textile

[4] The references to "chemical recycling" and "mechanical recycling" in this article is a simplification of reality, as chemical recycling of textiles always include mechanical pre-treatment steps and mechanical recycling sometimes involve chemicals. This is further discussed by Sandin and Peters (2018).

recycling, it can be assumed the majority (perhaps 80-90%) will be chemically recycled, as most new recycling initiatives and investments target chemical recycling (Dahlbom et al. 2023). This aligns with conclusions by Dahlbo et al. (2017), who assume 90% chemical recycling in future scenarios of recycling in Finland.

Therefore, the average recycling yield is expected to align more closely with the reported chemical recycling yields (57-65%) than with the higher yields typically associated with mechanical recycling (80-90%). Based on this, we assume a mean recycling yield of 65%, with a standard deviation of one tenth of this value. This reflects the low likelihood of the average yield exceeding 85% or dropping below 45%. The resulting probability distribution is illustrated in Figure S6.

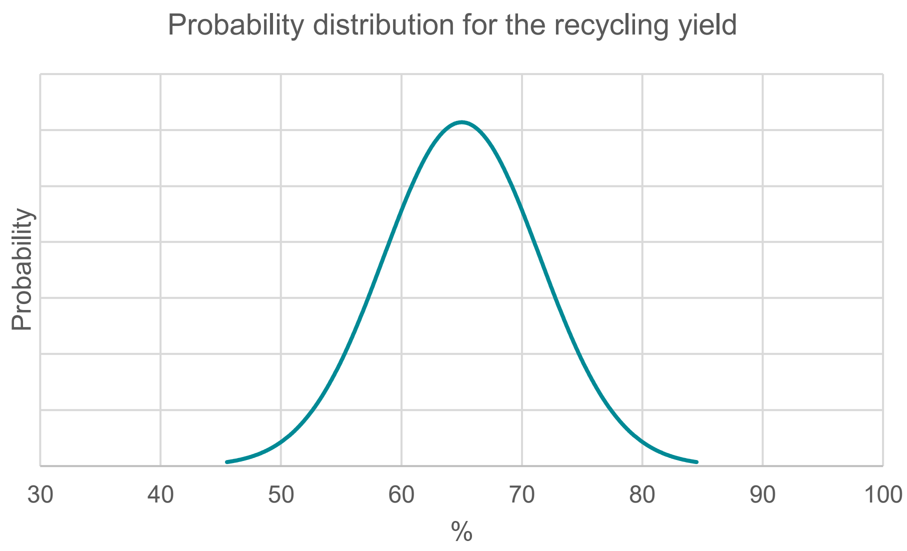

Figure S6. Probability distribution of parameter E, recycling yield.

3.7 Parameter F: Proportion of fossil fibres among recycled fibres

This parameter represents the proportion of fibres originally derived from fossil resources ("fossil fibres") among those being recycled in the EU in 2035. We assume this parameter influences the climate impact of reduced incineration since incineration of fossil fibres, unlike non-fossil fibres, contributes to the climate impact indicator used in this study.

Data from Textile Exchange (2022) imply that the proportion of fossil fibres among recycled fibres was about 70% in 2021. However, 99% of these fossil fibres are derived PET bottles and thus represent recycling that is not within the scope of our study, as we focus on fibre-to-fibre textile recycling. More relevant is to look at the proportion of fossil fibres among clothing purchased in the EU, which in in 2015 was about 27% (Statista 2023). The share in 2023 can be assumed to be considerably higher than this, as the proportion of fossil fibres has been steadily increasing globally, a trend that is projected to continue (Textile Exchange 2022). Furthermore, because of recent advancements in technology and investments in polyester recycling (Syre 2024, Rewin 2024) we assume that polyester recycling will be an increasingly viable option and constitute a substantial proportion of the fibre-to-fibre recycling in the 2035 scenario. Therefore, we assume that the mean for parameter F is 40%, which results in the probability distribution presented in Figure S7.

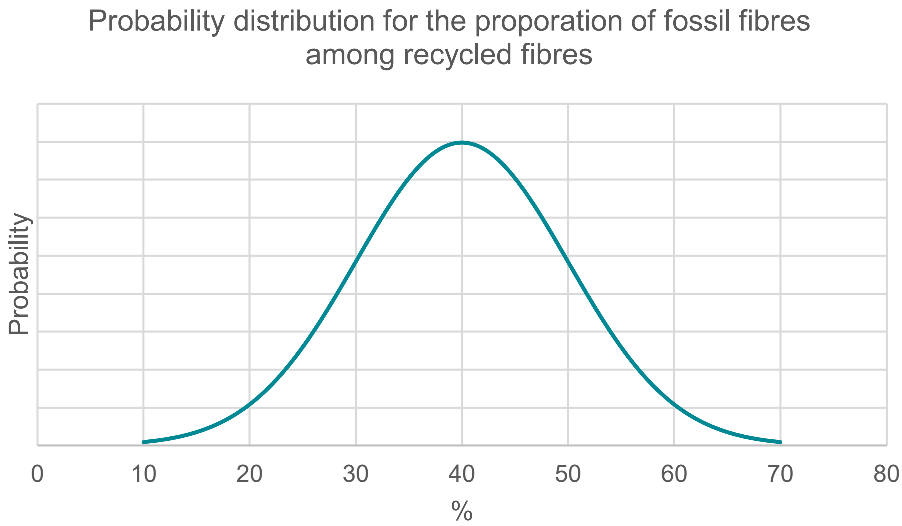

Figure S7. Probability distribution of parameter F, proportion of fossil fibres among the recycled fibres.

3.8 Parameter G: Proportion of landfilled textiles, among landfilled or incinerated textiles

This parameter is used to estimate the climate-impact consequences of reduced incineration and landfill (see Section 2.3) and compensation for the loss of recovered energy (see Section 2.5). It reflects the proportion of textiles discarded for landfilling, among those discarded for landfilling or incineration, in the EU in 2035. We assume that fibres recycled in 2035 would have been landfilled or incinerated if they had not been recycled.

According to Köhler et al. (2021), the proportion of textiles being landfilled, from those sent to landfilling or energy recovery (i.e., incineration), is on average 48% for seven selected EU countries, based on data from one of the years 2013-2019 for each country, with each country weighted equally. In LCAs following the Product Environmental Footprint Category Rules (PEFCR) for T-shirts placed on the EU market (European Commission 2021), the assumption for end-of-life modelling shall be 49% landfilling, 40% incineration, and 11% recycling. This would give a value of parameter G of 55%. This is also the value assumed by Trzepacs et al. (2023) in their study of the environmental impact of reuse in Europe.

However, parameter G can be expected to decrease considerably by 2035, driven by regulatory changes such as the EU Landfill Directive, which restricts landfilling of all waste suitable for recycling or energy recovery by 2030, and limits the proportion of municipal waste (including textiles) being landfilled to 10% by 2035, down from 24% in 2018

(European Commission 2023). Assuming the same rate of decrease in landfilling for textiles, and a starting point of 50% landfilling in 2018, we assume the mean of parameter G to be 21%. This results in the probability distribution presented in Figure S8.

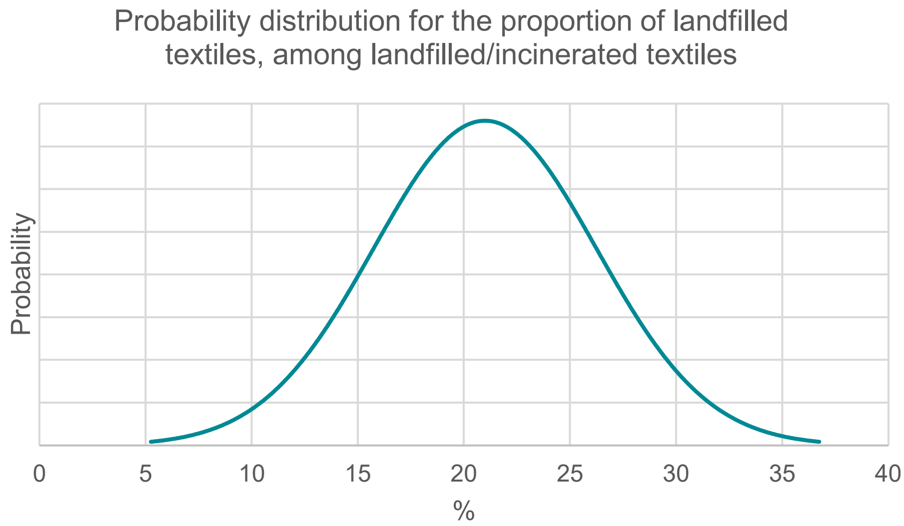

Figure S8. Probability distribution of parameter G, proportion of landfilled textiles, among landfilled or incinerated textiles.

3.9 Parameter H: Replacement rate

The replacement rate refers to the extent to which recycled fibres substitutes primary fibres. This substitution is influenced not only by the functional equivalency of the fibres – which is the meaning of replacement rate adopted by, for example, McGill (2009) and Schmidt et al. (2016) – but also by the price elasticity of demand for the fibres. Both functional equivalency and price elasticity are affected by the type and quality of the recycled fibres and the fibres they are assumed to replace. A fibre is more likely to substitute fibres of similar type and quality. For example, recycled viscose is more likely to replace primary viscose than cotton. But different fibre types also substitute each other, albeit to a lesser extent. For instance, Agbadi (1988) concludes that a 1% increase of per-capita consumption of synthetic fibres leads to a 0.1% reduction in per-capita consumption of cotton.

In the present study, we aimed at estimating the average replacement rate for fibres being recycled in the EU in 2035. To inform this estimate, we consider the assumptions of replacement rates in previous studies of textile recycling.

Sandin and Peters (2018) conclude that LCA studies of recycling or reuse often assume a 100% replacement rate, typically without justification. Other replacement rates are sometimes assumed, particularly in studies of reuse, such as done by Trzepacz et al. (2023), Damgaard et al. (2019), Fortuna and Diyamandoglu (2017), and Castellani et al. (2015). However, there are a couple of studies of textile recycling that assume replacement rates below 100%. McGill (2009) estimates replacement rates between 67% and 80% for textile downcycling. Corsten et al. (2013) assume a 50% replacement rate for both textile recycling and reuse, though this assumption lacks supporting data. Apart from these two studies, we have not found any references on replacement rates of textile recycling.

Based on the range provided by of McGill (2009), and assuming that fibre-to-fibre recycling has a relatively high replacement rate compared to downcycling, we assume the mean of parameter H to be 75%. As a replacement rate above 100% is highly unlikely, we assume a lower standard deviation than for the other parameters: one eight of the mean value. This results in the probability distribution presented in Figure S9. Due to the lack of underlying data this is a particularly uncertain parameter, and therefore we test the influence of assuming substantially lower and higher replacement rates in the sensitivity analysis (see Section 3.2 in the main article).

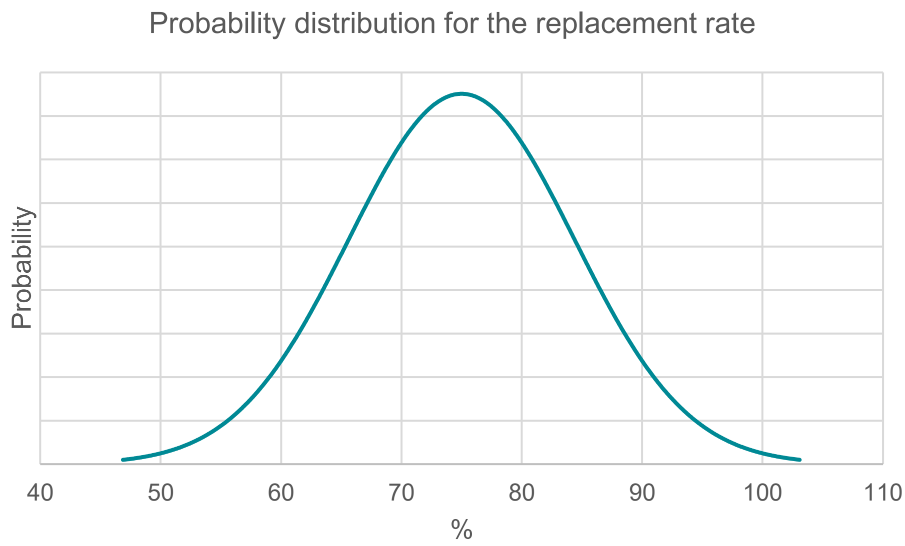

Figure S9. Probability distribution of parameter H, replacement rate.

3.10 Parameters I1 and J1: Climate impact of textile recycling processes today in the EU and in Asia

Parameters I1 and J1 reflect the climate impact of recycling, encompassing all processes from the transport from the sorting facility to the recycling facility and all further processes to produce filament or staple fibres ready for subsequent production processes (e.g., yarn spinning, nonwoven production, etc.). Parameter I1 reflects recycling processes in the EU, and parameter J1 reflects recycling processes in Asia. The parameters are expressed per tonne fibres produced, as this is the level at which primary fibre production can be replaced (which is one of the five consequences considered, see Section 2.4).

By accounting for all processes up to the production of fibres, this approach integrates both mechanical recycling (which preserves fibres) and chemical recycling (which does not). Consequently, a single climate-impact estimate can be derived for each region (EU and Asia) irrespective of the recycling method. Furthermore, this enables a single-climate estimate also for the reduced primary fibre production (parameter D1). In the end, this approach significantly simplifies the calculation of the climate-impact consequences.

Although parameters I1 and J1 reflect the average climate impact of textile recycling processes existing today, the composition of recycling methods must be adjusted to reflect the expected situation in 2035. Given the trends and investments in recycling technologies (Dahlbom et al. 2023), it is reasonable to assume that chemical recycling – which generally has a higher climate impact than mechanical recycling – will dominate in 2035. Specifically, we estimate 80-90% chemically recycling in 2035, which aligns with the 90% chemical recycling in scenarios of future recycling in Finland assumed by Dahlbo et al. (2017).

The choice of allocation method is also an important aspect to consider when assessing the environmental impact of recycled fibres. Allocation determines how the environmental burden of the initial resource extraction is distributed between the primary use of the fibre (as a primary fibre) and its secondary use (as a recycled fibre). Cut-off is the most applied method (Sandin & Peters 2018). Under this method, the textile waste input to the recycling process is considered free of any environmental burden, while, the subsequent user of the recycled material assumed the full environmental burden of the recycling process, as well as the collection and sorting processes in most cases. As we're interested in the climate impact of the recycling processes (up to fibres) and the transport from sorting to recycling, we disregard data that are based on allocation methods other than cut-off, that includes environmental burdens from the original resource extraction. If the data includes upstream processes such as collection and sorting, the contributions from these processes must be accounted for and, if possible, subtracted.

Lidfeldt et al. (2022) provide recent data of textile recycling. Based on the LCA model of Lidfeldt et al. (2022), we calculate the climate impact of mechanically recycled cotton and wool to 0.36 and 1.4 tonnes CO_2 eq. per tonne fibres, respectively, and chemically recycled viscose and polyester to 2.8 and 4.7 tonnes CO_2 eq. per tonne fibres. These calculations exclude collection and sorting. For wool recycling, the data referenced in Lidfeldt et al. (2022) is primarily derived from Wiedemann et al. (2020, 2022) and Martin and Herlaar (2021).

Duhoux et al. (2021) studied mechanical recycling and reported a climate impact of about 0.22 tonnes CO_2 eq. per tonne cotton entering the process. Given typical yields for mechanical recycling (see Section 3.6), this would result in slightly higher impacts per tonne exiting the process. Esteve-Turrillas and de la Guardia (2017) also studied mechanically recycled cotton and found a similar impact of 0.21 tonnes CO2 eq. per tonne cotton. Bianco et al. (2022) estimated the climate impact of mechanically recycled wool to about 0.63 tonnes CO_2 eq. per tonne, with an uncertainty range of 0.1 to 0.9 tonnes CO_2 eq. Notably, the impacts reported for both for cotton and wool recycling are much lower than for producing primary fibres (Sandin et al. 2019b).

There are few recent studies that provide relevant data on chemical textile-to-textile recycling. To address this gap, we consider older studies and also studies on bottle-to-fibre recycling.

Hagoort (2013) studied several polyester recycling routes in Asia, reporting climate impacts of about 3 to 4.5 tonnes CO_2 eq. per tonne yarn, including nearly 0.5 tonnes CO_2 eq. from transports such collection and a small contribution from sorting. Hagoort compared these values to primary polyester fibres with climate impact of almost 6 tonnes CO_2 eq. per tonne yarn.

Shen et al. (2010a) studied bottle-to-fibre recycling in Europe (mechanical, semi-mechanical and chemical recycling routes), finding climate impacts ranging from 1330 to 2820 kg CO_2 eq. per tonne fibres. They compared these with primary PET fibres produced in Europe, which have a climate impact of 5540 kg CO_2 eq. per tonne fibres. The study excluded impacts of sorting, baling, and compacting the bottles, but included the impact of transport to the recycling facility.

As implied by above reference and concluded in a review of LCA studies of textile reuse and recycling (Sandin & Peters 2018), recycled fibres generally have lower climate impact compared to primary fibres. However, there are exceptions where the impacts are comparable. For instance, Lidfeldt et al. (2022) found that viscose produced from chemically recycled cotton has a climate impact similar to its primary fibre benchmark – but this benchmark assumed dissolving pulp produced in Sweden, which leads to viscose with relatively low climate impact. In contrast, Schultz and Suresh (2017) found that viscose from recycled cotton often has much lower climate impact than primary fibres. However, their results are not included here due to their use of an unconventional climate-impact assessment method, making it unsuitable to directly compare with the other mentioned results.

Another key observation from the literature is that studies of recycling in Asia report higher climate impact than studies of recycling in Europe, as is expected considering the higher climate impact of energy mixes in Asia compared to Europe. The same is true for fibre production in general, for example as shown by Shen et al. (2010b) and Schultz and Suresh (2017) for regenerated cellulose fibres.

Before estimating the means of parameters I1 (climate impact of recycling processes in the EU) and J1 (climate impact of recycling processes in Asia), we summarise the three main points from above paragraphs:

- The mean should be closer to the data for chemical recycling than for mechanical recycling, as we assume that chemical recycling will be the more commonly used recycling route in 2035.

- The mean should be notably lower than the mean assumed for the reduced primary fibre production (4 t CO_2 eq./t fibres, see Section 3.4) as (i) almost all LCA studies of recycled fibres show climate-impact benefits compared to primary fibres, and (ii) most recycling facilities in the 2035 scenario remains be built, and they are expected to be relatively modern, large-scale, and energy-efficient compared to the recycling facilities studied in the above mentioned studies.
- The mean for the climate impact of recycling processes should be lower in the EU (parameter I1) than in Asia (parameter J1).

Based on above, we assume parameter I1 to be 2.0 tonnes CO_2 eq. per tonne fibres and parameter J1 to be 3.0 tonnes CO_2 eq. per tonne fibres. Due to the high uncertainty, we assume a standard deviation of one third of the mean (which is a larger standard deviation than the default one assumed in this study). See Figure S10 and Figure S11 for the resulting probability distributions for parameters I1 and J1, respectively.

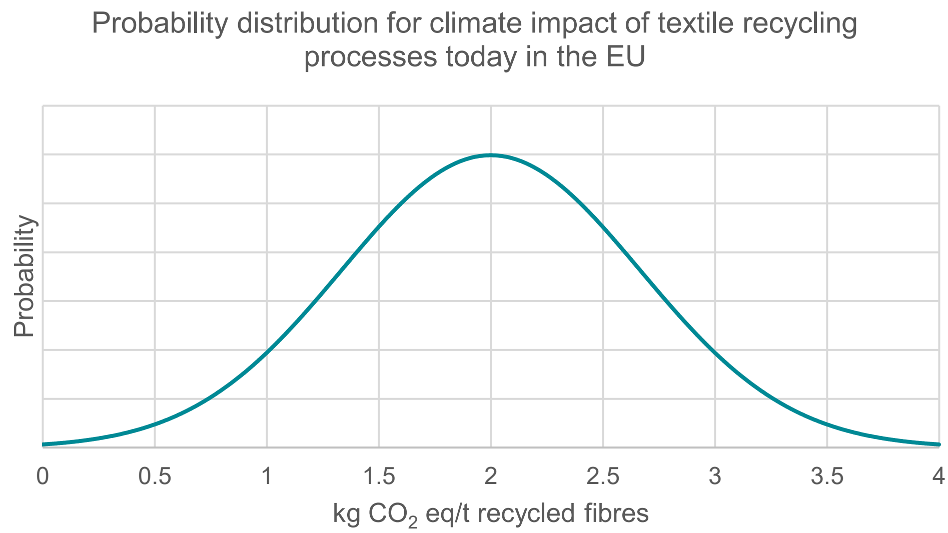

Figure S10. Probability distribution of parameter I1, climate impact of textile recycling processes today in the EU.

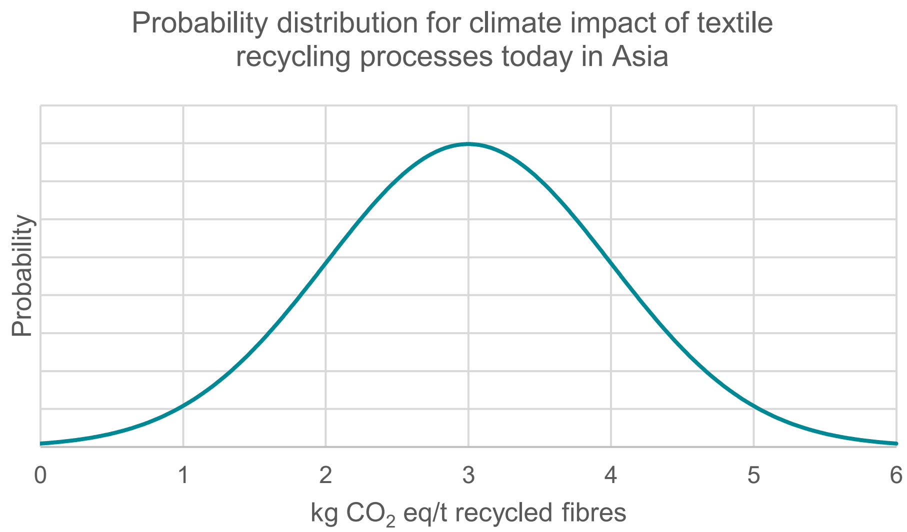

Figure S11. Probability distribution of parameter J1, climate impact of textile recycling processes today in Asia.

Compared to a business-as-usual scenario, the increased-recycling scenario assessed in this report assumes a shift towards fibre production occurring more extensively within the EU. This assumption is reflected in the parameters: while the reduction in primary fibre production is assumed to primarily occur in Asia (see Section 3.4), the recycling processes are assumed to occur largely within the EU. This outcome aligns with expectations, as the raw material – increasingly discarded textiles – is increasingly sourced from within the EU.

It is also important to note that in our model, recycling (up to the stage of fibres ready for subsequent production) is assumed to occur entirely either in Europe or Asia. However, in practice, there will likely be instances initial recycling processes are carried out in the EU, and subsequent fibre production occurs in Asia. This underscores the fact that our model, like all models, is a simplification of reality.

3.11 Parameters I2 and J2: Water-deprivation impact of textile recycling processes in the EU and in Asia

Similar to our assessment of climate impact (see Section 3.10), we calculated the water-deprivation impact of textile recycling processes, excluding collection and sorting, using the LCA model by Lidfeldt et al. (2022). For mechanically recycled cotton and wool, the results are 67.1 and 106 m^3 world eq. per tonne fibres, respectively. For chemically recycled viscose, the results are 1,230 m^3 world eq. per tonne dissolving pulp and 3,320 m^3 world eq. per tonne fibres. For chemically recycled polyester, the result is 388 m^3 world eq. per tonne PET granulates.[5]

[5] The result for chemical recycling of polyester expressed per recycled PET fibres was negative, which we could not explain and therefore chose to ignore.

Duhoux et al. (2021) provide rough estimates of water-deprivation impact for several recycling technologies:

- Mechanical recycling of cotton, polyester, polyamide, and polycotton: 49.3 m^3 world eq. per tonne fibres recycled.
- Chemical recycling of cotton into dissolving pulp: 1,550 m^3 world eq. per t sulphate pulp, and 1,180 m^3 world eq. per tonne sulphite pulp.[6]
- Chemical (enzymatic) recycling of polycotton: 14,800 m3 world eq. per tonne fibres recycled (this is based on data that has not been validated for industrial scale and is therefore highly uncertain).

Schultz and Suresh (2017) provide data on the net water consumption of production of viscose fibres in Germany, where dissolving pulp is derived from recycled textiles. They report a water consumption of 377 m^3 per tonne fibres. Assuming all processes occur in Germany (which is a simplification), and an average characterisation factor for water consumed in Germany of 1.36 m^3 world eq. per m^3 water (obtained from Sphera's LCA software), this results in a water-deprivation impact of 513 m^3 world eq. per tonne fibres produced.

The other references used when determining the climate impact of recycling processes (see Section 3.10) did not include data useful for determining the water-deprivation impact of recycling processes.

Based on the data above, it appears that chemical recycling generally has substantially higher water-deprivation impact compared to mechanical recycling, although there are large variations between different chemical recycling processes. As we assume chemical recycling is 80-90% of recycling in 2035 (see Section 3.6), the estimated means for water deprivation should be in the range indicated by the results for chemical recycling.

We assume 1,500 m^3 world eq. per tonne fibres as the estimated means both for parameters I2 and J2. This is the middle of the range of above numbers for chemically recycled fibres (not considering the very uncertain and remarkably high results of chemical recycling of polycotton). Note that we assume the same parameter for textile recycling in the EU (parameter I2) as in Asia (parameter J2), as we have not found any data that indicate a general difference in the water-deprivation impacts of recycling processes in the two continents. If more data is available in the future, this is an aspect of the model to improve in a potential update of this study. Due to the high uncertainty, we assume a standard deviation of one third of the mean (which is a larger standard deviation than the default

[6] Here the number provided by Duhoux and colleagues is 1.18 m^3 world eq. per tonne pulp, but we conclude there must be an error with a factor 1000 for all sulphite pulp results, based on a comparisons with other results in the same study (other results provided for sulphite pulp, for avoided production of sulphite pulp from primary resources, and results on sulphate pulp).

one assumed in this study) and explore the uncertainty of this parameter further in the sensitivity analysis. See Figure S12 for the resulting probability distribution for parameters I2 and J2.

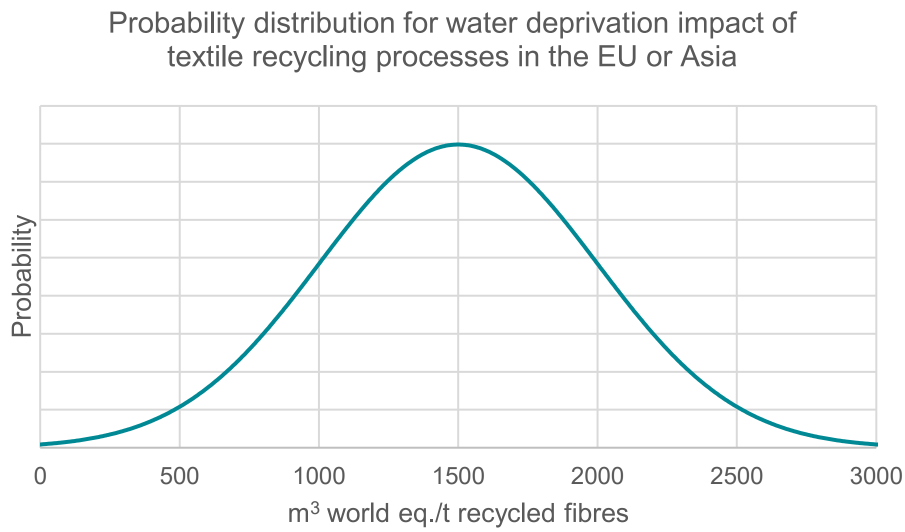

Figure S12. Probability distribution of parameters I2 and J2, water-deprivation impact of textile recycling processes in the EU or Asia, respectively.

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