



mistra  
future  
fashion



# LCA on recycling of blended fiber fabrics

by  
Greg Peters, Björn Spak  
& Gustav Sandin



**Title:** LCA on recycling of blended fiber fabrics  
**Author:** Greg Peters, Björn Spak & Gustav Sandin  
**Mistra Future Fashion deliverable:** D4.3.1.1  
**Edition:** Only available as PDF for individual printing  
**ISBN:** 978-91-89049-56-7

**Mistra Future Fashion report number:** 2019:14

© Chalmers University of Technology  
412 96 Gothenburg, Sweden  
Phone: +46 (0)31-772 10 00  
[www.chalmers.se](http://www.chalmers.se)

Images: Unsplash  
Layout: Malin Viola Wennberg

#### A Mistra Future Fashion Report

Mistra Future Fashion is a cross-disciplinary research program, initiated and primarily funded by Mistra. It holds a total budget of SEK 110 millions and stretches over 8 years, from 2011 to 2019. It is hosted by RISE in collaboration with 15 research partners and involves more than 50 industry partners.

[www.mistrafuturefashion.com](http://www.mistrafuturefashion.com)



FUNDED BY



The Swedish Foundation for  
Strategic Environmental Research

For additional questions about the research, please contact corresponding author:

Gregory Peters  
Researcher, Chalmers University of Technology  
[petersg@chalmers.se](mailto:petersg@chalmers.se)

Björn Spak  
LCA Researcher, RISE Research Institutes of Sweden  
[bjorn.spak@ri.se](mailto:bjorn.spak@ri.se)

© Division of Environmental Systems Analysis  
Department of Technology Management and Economics  
Chalmers University of Technology  
41296 Gothenburg, Sweden  
[www.chalmers.se](http://www.chalmers.se)

# summary

Recycling garments made from mixed fibers is a challenge on account of the difficulty associated with separating them into their component polymers. For this reason, large quantities of clothing end up in landfill or incinerators. However, progress has been made recently towards the recycling of mixed textiles. Taking laboratory results and scaling them up is a major undertaking, and one which should ideally be subjected to environmental assessment prior to any investment. This report summarizes the environmental assessment work done in the Mistra Future Fashion program which was focused on the potential to recycle fabric made of a blend of fibers.

The process in focus is the Blend Re:wind method developed in the program. The process is evaluated at a small industrial scale, as there are operational questions to examine before large-scale industrial implementation, and the potential to use relatively uncomplicated feedstocks exists at smaller scales. In particular, the main scenario under investigation is the polymeric recycling of textiles from the commercial laundries in Sweden that provide services to the healthcare sector. We estimate a flow of about 850 tonnes of disused fabric is potentially available from this sector in 2023. A scenario based on a single Swedish facility processing this flow was compared with two alternative baselines: the single use of cotton or viscose textiles. The scenarios exclude life cycle stages such as garment production and retailing, as these would not differ between scenarios. A scenario in which only the terephthalic acid was recovered from the polyester, but new ethylene glycol is obtained in the market prior to the synthesis of (partly) recycled polyester, was also examined.

The assessment was performed using environmental life cycle assessment. Key environmental effect categories considered as part of this work include water use, climate change (greenhouse gas emissions), freshwater eutrophication, freshwater ecotoxicity and human toxicity (cancer and non-cancer).

The results indicate that the recycling system is competitive with the single use alternatives for a small majority of indicators. While it underperforms with respect to energy consumption, greenhouse gas emissions and acidification, the outcomes are of the same order of magnitude as the single use alternatives. This is a promising outcome, given the relatively low level of technical development (i.e. laboratory scale) associated with the recycling system versus the long development history and large scale of the single use alternatives. Additionally, this report has identified areas of potential improvement for the recycling system which would make it more competitive.

The scenarios in this report are based on a blend of publicly available data sources and should be considered explorative rather than attempts to model existing businesses. The reader should therefore be cautious when interpreting them. Nevertheless, the results suggest a bright future for the Blend Re:wind process.

# List of abbreviations

AWARE = Available WAter REmaining (an LCIA method)

BRW = Blend Re:wind (recycling process)

CO<sub>2</sub> = carbon dioxide

CTU<sub>e</sub> = comparative toxicity units for ecosystems

CTU<sub>h</sub> = comparative toxicity units for humans

EG = ethylene glycol

H<sub>2</sub>SO<sub>4</sub> = sulfuric acid

LCA = life cycle assessment

LCI = life cycle inventory

LCIA = life cycle impact assessment

ILCD = International Reference Life Cycle Data System

IPCC = Intergovernmental Panel on Climate Change

ISO = International Organization for Standardization

NMMO = 4-Methylmorpholine N-oxide

NMVOC = non-methane volatile organic compound

NaOH = sodium hydroxide

Na<sub>2</sub>TP = sodium terephthalate

P = phosphorus

PEF = product environmental footprint

PES = polyester

rPES = recycled PES

vPES = virgin PES

PET = polyethylene terephthalate

SO<sub>2</sub> = sulphur dioxide

TPA = terephthalic acid

VOC = volatile organic compounds

WWTP = waste water treatment plant

# table of content

1. introduction .....	7
1.1. aim of this report .....	7
1.2. setting the scene .....	7
1.3. scope for recycling of blended material in Sweden .....	8
1.4. what is life cycle assessment? .....	10
2. goal and scope definition .....	13
2.1. goal .....	13
2.2. functional unit .....	13
2.3. system description.....	13
2.4. modelling approaches.....	15
2.5. allocation procedures .....	16
2.6. impact categories .....	16
2.7. software and LCI databases.....	17
2.8. limitations.....	18
3. inventory analysis .....	20
3.1. Blend Re:wind process .....	20
3.1.1. material collection .....	20
3.1.2. cutting and ripping .....	21
3.1.3. hydrolysis .....	23
3.1.4. filtration .....	23
3.1.5. nanofiltration .....	24
3.1.6. acidification .....	25
3.1.7. filtration.....	26
3.1.8. glycol purification .....	26
3.1.9. production of regenerated cellulose fibers .....	27
3.1.10. polyester synthesis .....	28
3.2. single use baseline.....	28
3.3. alternative analysis .....	28
3.3.1. quantitative sensitivity analysis.....	28
3.3.2. rejected alternative scenarios .....	29
4. impact assessment results and discussion .....	31
4.1. hotspots of the BRW scenario.....	31
4.2. Sensitivity analysis and comparison with single use .....	34
5. Conclusions .....	39
references.....	40
appendix - Description of impact categories.....	43

# 1. introduction

## 1.1. aim of this report

The overall aim of this report is to examine the potential to reduce environmental damage by recycling textile blends, i.e. fabrics with several fibers. This report describes a life cycle assessment (LCA) which was focussed on the potential of the Blend Re:wind ("BRW") process for recycling textile blends, and was performed as part of the Mistra Future Fashion research program. We identify the extent to which alternative recycling systems represent an improvement over reference scenarios, and the key parameters which contribute to differences between scenarios. The intended audience for the report is in the first place the Mistra Future Fashion consortium's researchers and corporate partners. This is a curiosity-driven activity which is not directly coupled to a particular decision-maker nor the making of claims for the marketing of a particular commercial product. On the other hand, it is also intended to inform the debate around textile industry impacts and resource management, and has strategic consequences for the design of sustainable textile life cycles.

Previous work on the technical opportunities of Swedish textile recycling has revealed that polymer, oligomer and monomer recycling is hindered by a lack of technologies for sorting and separating textile waste into sufficiently pure fractions (Östlund et al. 2015). Furthermore, LCAs have pointed to the fact that there is a risk that recycling processes could cause increases in e.g. climate impact in case fossil fuels are used to power the processes, especially if the fibers (e.g. viscose) resulting from the recycling process is thought to replace cotton which has relatively low impact on climate change (Östlund et al. 2015).

In general, however, published LCA results indicates that recycling reduces environmental impact compared to incineration and landfilling, mainly from avoidance of manufacturing of virgin materials. Whereas most published studies focus on fiber recycling or polymer/oligomer recycling, monomer recycling (in which, for example, polyester is broken down into monomeric units before reassembly) is less studied (Sandin and Peters, 2018), which is why the current study can provide valuable insights. Previous LCAs have in many cases been limited in choice of environmental impact categories, mainly focusing on climate change while e.g. water depletion and toxicity, which are included in the present study, are of great concern when discussing cotton. Another feature of many previous LCAs is the exclusion of collection and sorting processes, risking underestimation of environmental impacts, the present study tries to come to terms with this.

## 1.2. setting the scene

Along with economic development, the demand for textiles fibers, and consequently the production of fibers, has been growing since the industrial revolution. Since the introduction of synthetic fibers in the 1920s, cotton's share of the global fiber consumption has been trending

downward, although production and consumption has continued to grow in absolute terms. The 1960s marked the real break-through for synthetic fibers, especially polyester, but it was not until the early 1990s that the share of cotton in the global textile fiber consumption was surpassed by polyester. Total demand for textile fibers is expected to increase along with economic development and although production and consumption of cotton is expected to grow in the coming ten years, the growth rate will be substantially lower than it will be for synthetic fibers.

Rising demand for cotton fiber may conflict with for example rising demands for agricultural land for food production, and fluctuations in water availability due to climate change may hamper cotton production. Rising demand for virgin polyester fiber is in conflict with global efforts to combat climate change by consuming less fossil resources. Meanwhile, there are substantial amounts of used textiles being discarded, often destined for waste management systems using incineration or landfilling. Although microbially engineered pathways may be commercially available in the future (for example based on bacterial degradation of polyester – see Austin et al, 2018), at the moment, the systems available for mechanical recycling of used textiles into industrial rags, upholstery or insulation, are typical examples of "down-cycling", where the recycled material is used for a product which cannot replace the original product. If the fibers instead can be utilized to manufacture new textiles (i.e. "closed loop recycling"), production of fibers from virgin sources could be reduced.

Fiber manufacturing contributes a comparatively large part of the total life cycle impact of many textile products. For example, products made from cotton fiber cause major freshwater usage impacts in the fiber production process. Furthermore, the third largest contributor to climate impact for the average Swedish fashion product is fiber production, following fabric production and consumer transport (Roos et al, 2015). The large impact of virgin fiber products indicates a potential environmental benefit to be achieved from extending the life of garments (Zamani et al, 2017) or by recycling their materials, either as fibers, polymers/oligomers or monomers.

## **1.3. scope for recycling of blended material in Sweden**

In the international context Sweden has a well-established system for waste management with high recycling rates, including a well adopted national recovery system for packaging and newspaper. Textile recycling is not as mature although there are charity organizations, and more recently fashion retailers, that collect used textiles for reuse and recycling. The contamination of used textiles with other wastes is one of several obstacles to recycling and therefore large volumes of used textiles are being sent to incineration with energy recovery. As mentioned previously, existing recycling systems are mainly mechanical and result in down-cycling. Pioneering chemical recycling processes, e.g. Re:newcell, where cellulose rich textile waste streams can be turned into recycled regenerated cellulose fiber, are not suited for recycling of fiber blends like polycotton (a mix of polyester and cotton). In contrast, the BRW process is designed to be able to treat polycotton – one of the most common fiber blends. In this process, polycotton materials are separated into solid cotton fibers, which can be used as raw material for regenerated cellulose fibers, and a liquid fraction containing the building blocks of polyester, which potentially can be used to manufacture recycled polyester.



Despite the potential for the BRW process to be applied to mixed waste streams, there remains the key challenge of creating a reliable input flow of appropriate quality recyclable material. For this reason, we have focussed on a particular industry sector with large waste flows. The health and care sector in Sweden is divided into healthcare (i.e. hospitals and primary care facilities), which is the responsibility of the county councils/regions, and care (e.g. elderly care), which is the responsibility of the municipalities. Health and care facilities use textiles for staff as well as for patients and clients, and frequently the textiles are made from polycotton. The facilities' laundry requirements are provided by a few major actors in Sweden. Two private companies, Textilia and Berendsen, have several facilities treating laundry from both municipalities and county councils/regions, as well as others (e.g. hotels). In total, Textilia and Berendsen have eleven facilities around the country specialised in laundry from health and care facilities. Besides the private companies there are five laundries operated by the county councils/regions themselves while some municipalities have their own laundry service or utilise other, local, laundry companies.

Some laundry services apply microchips to textiles for tracking purposes. The major route for care textiles being taken out of service is when they are discarded by the laundries for not meeting quality requirements. Altogether, the health and care sector textile waste is a comparatively homogenous polycotton stream that already is collected and handled by a few major actors, making it something of a "low hanging fruit" for potential recycling interventions. Furthermore, there is a potential for laundries to avoid costs associated with waste handling as well as to claim environmental benefits from sending discarded textiles to recycling.



figure 1 Population and laundry waste projections

Laundry services in Sweden have previously been estimated to generate approximately 763 tonnes unsorted discarded textiles in 2012 (Brismar, 2014). In this estimation there are some other sources outside the care sector, e.g. hotels, restaurants and catering, but these use similar textiles. The estimation relies on figures from 2012 and assuming a linear relationship between population size and amounts of discarded textiles the potential would have been 808 tonnes in 2017 and is projected to be about 854 tonnes in 2023 (see figure 1). Avoiding excess precision, we use a figure of 850 tonnes for the rest of this report.

## 1.4. what is life cycle assessment?

The assessment is based on the life cycle assessment (LCA) method as outlined in ISO 14040 and 14044 (ISO 2006a, ISO 2006b). LCA is an internationally accepted and widely used method capable of assessing a wide range of environmental impacts over the life cycle of products and services. In short, an LCA accounts for all environmentally relevant flows of energy and materials across the system boundaries, from cradle to grave (or cradle to gate, in more limited studies), and uses characterisation methods to “translate” these flows into environmental pressures expressed in impact categories such as climate change, acidification, eutrophication, toxicity and water depletion. In this way, LCA provides an overview of the environmental performance of the studied product and enables the identification of environmental hotspots in the product life cycle. This information can be useful in decision making, such as in prioritising measures for improved environmental performance.

figure 1 Population and laundry waste projections

The LCA procedure consists of four steps, as explained below and illustrated in figure 2

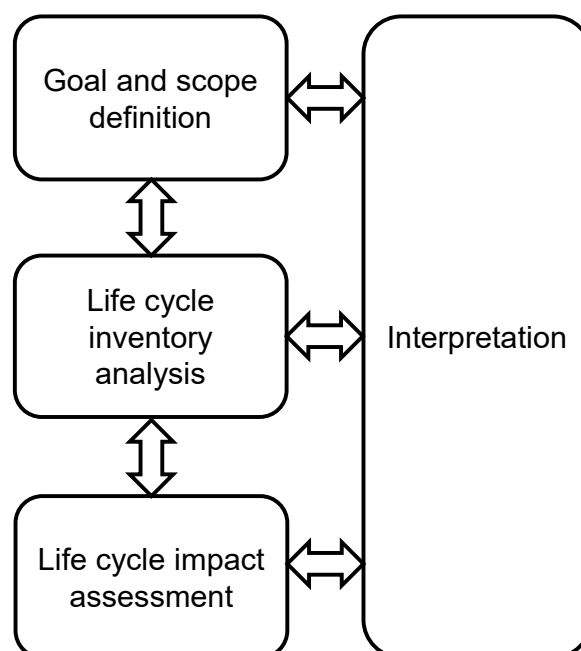
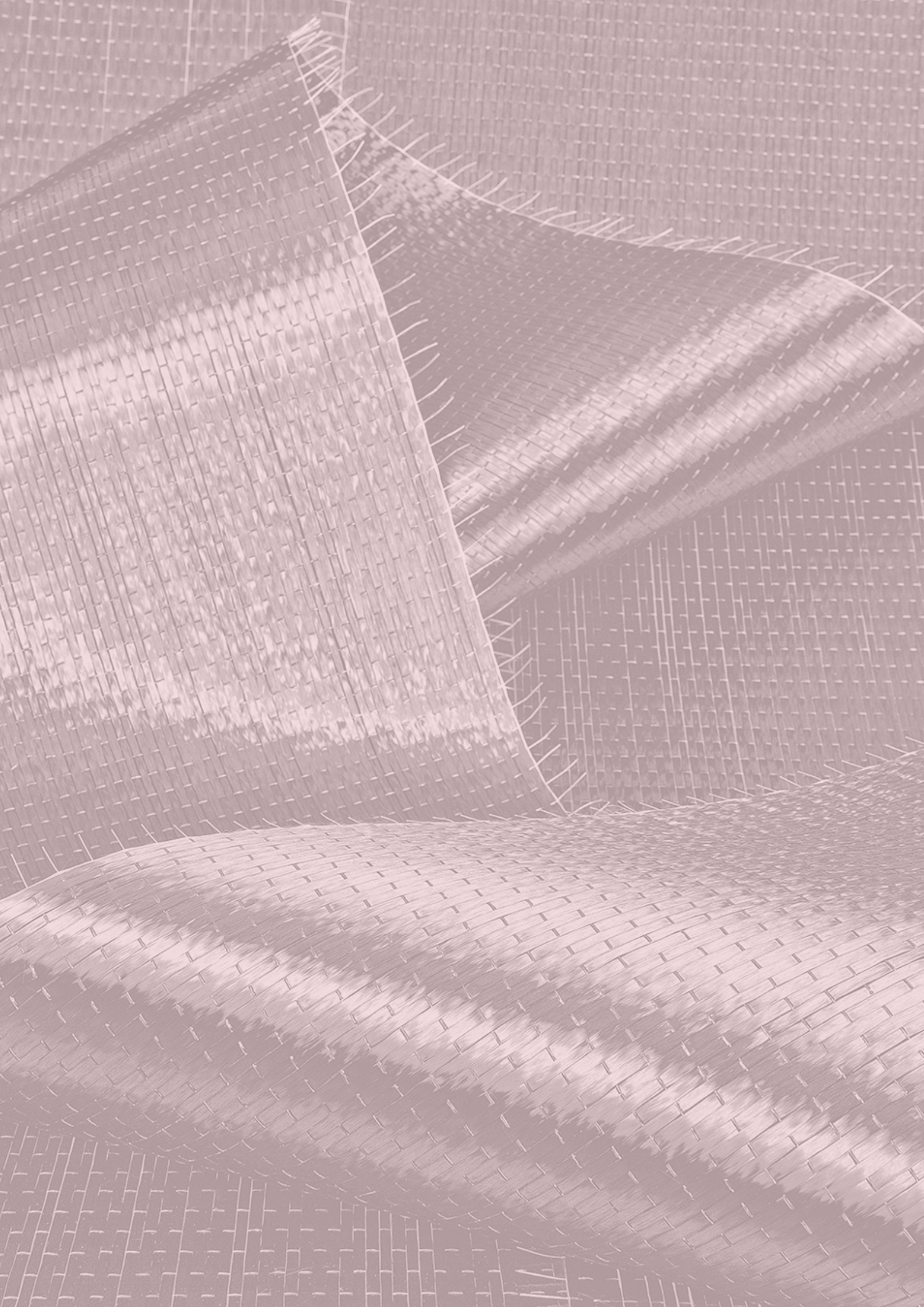


figure 2 Schematic illustration of the four phases of LCA and their interconnectedness.

- I. *Goal and scope definition*: The aim of the assessment, the functional unit and the product life cycle are defined, including boundaries to other product systems and the environment. The functional unit is a quantitative unit reflecting the function of the product, which enables comparisons of different products with identical functions.
- II. *Life cycle inventory analysis (LCI)*: All environmentally relevant material and energy flows between processes within the defined product system, and between the system and the environment or other product systems, are quantified and expressed per functional unit. Flows between the defined system and the environment consist of emissions and the use of natural resources.
- III. *Life cycle impact assessment (LCIA)*: By means of characterization methods, the LCI data is translated into potential environmental interventions, classified into impact categories. The LCIA can also include normalization and weighting, in which results for several impact categories are aggregated on a single yardstick – these steps are not included in the present study.
- IV. *Interpretation*: The result of the LCIA is interpreted, taking into account the goal and scope definition (e.g. the system boundaries) and the LCI (e.g. data gaps and data uncertainties), and recommendations are made to the intended audience.

As illustrated in figure 2, carrying out an LCA is an iterative process, since intermediate results and insights may call for revision of earlier steps.



## 2. goal and scope definition

### 2.1. goal

As previously stated, the overall aim of the study is primarily to inform public debate and business developers regarding the potential to reduce environmental damage by recycling mixed fibers. More specifically – the goal is to find out to what extent the BRW recycling process developed in the Mistra Future Fashion program can provide benefits over business as usual. We aim to make this comparison, identify key process hotspots and parameters of interest for life cycle optimisation of recycling system design and policies towards a circular economy. The report is intended initially for the Mistra Future Fashion consortium researchers and corporate partners, and more generally for policy and business planners within the textile and public sectors.

### 2.2. functional unit

In LCA, a functional unit is an attempt to provide a quantitative definition of the basis for comparing alternative systems. In this study the functional unit is defined as the production of fibers equivalent to the potential production from Sweden's annual commercial laundry waste flow, i.e. 350 tonnes of polyester and 280 tonnes of regenerated cellulose fibers. This functional unit facilitates comparison between processes that create fiber from virgin raw materials, and processes that create fibers from recycled raw materials.

### 2.3. system description

Many different systems can be imagined for delivery of the function described above, so we have identified four key scenarios to explore the environmental performance of recycling systems. The scenarios described in this section of the report are the result of discussions between the manager of the Recycling Theme of Mistra Future Fashion (one of four themes in the research program), Maria Gunnarsson, and the authors. This discussion was guided by the collected experience of many other researchers in the program who have worked on the chemistry of the Blend Re:wind process (including Anna Palme and Stina Björquist at Chalmers University of Technology) and the environmental assessment of textile systems (including Sandra Roos at RISE Research Institutes of Sweden).

Throughout this work, these scenarios are considered to be potential future scenarios. We have attempted to avoid creating scenarios that can obviously only exist in the imagination. Nevertheless, we will assume the existence of certain industrial processes that need not exist today, but may exist in the future. Our task is not to report on what has been demonstrated to work, but to find out whether some ideas about the future are attractive from an environmental perspective.

Diagrams representing the physical flows in the alternative scenarios in this study are shown in figure 3 and figure 4. All the foreground processes numbered in figure 3 are assumed to occur in Sweden. The numbers are consistent with the numbering of the subheadings in [chapter 3.1](#) of

this report. Several variations of the BRW scenario are possible: exactly which cellulosic fiber is produced from the recovered cotton is a question. Viscose, lyocell and cellulose carbamate are three possibilities. Furthermore, compared with the terephthalic acid (TPA), the ethylene glycol (EG) may be less desirable as a recyclate because the distillative purification process is energy intense when the starting solution is as dilute as Palme et al (2017) suggests. Also, ethylene glycol is relatively inexpensive (typically USD 500-1000 per tonne) compared with terephthalic acid and the mass available is relatively small (theoretically 323 kg per tonne of polyester). (The corresponding figures for terephthalic acid are typically USD 800-1200 per tonne, and 865 kg per tonne, which is relatively easily recovered by a pH shift and filtration.) Therefore, the alternative of just recovering the terephthalic acid and using virgin ethylene glycol in polyester synthesis (a “no EG BRW scenario”) was also considered in addition to the basic BRW scenario shown in the figure.

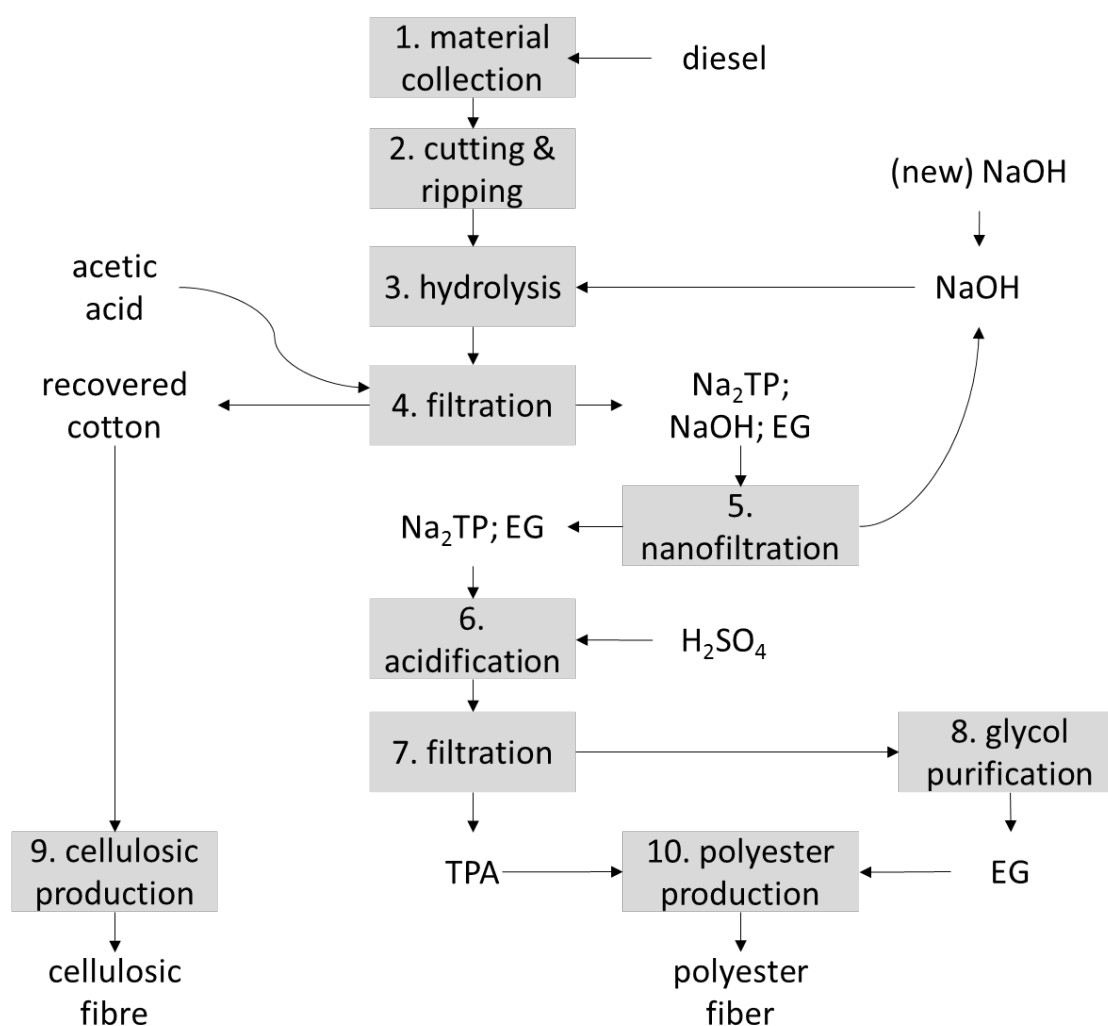


figure 3 Overview of the basic BRW recycling scenario

As shown in figure 4, the normal life cycle of textile products in Sweden ends with combustion with energy recovery for district heating and electricity production. These processes are absent from the recycling scenario.



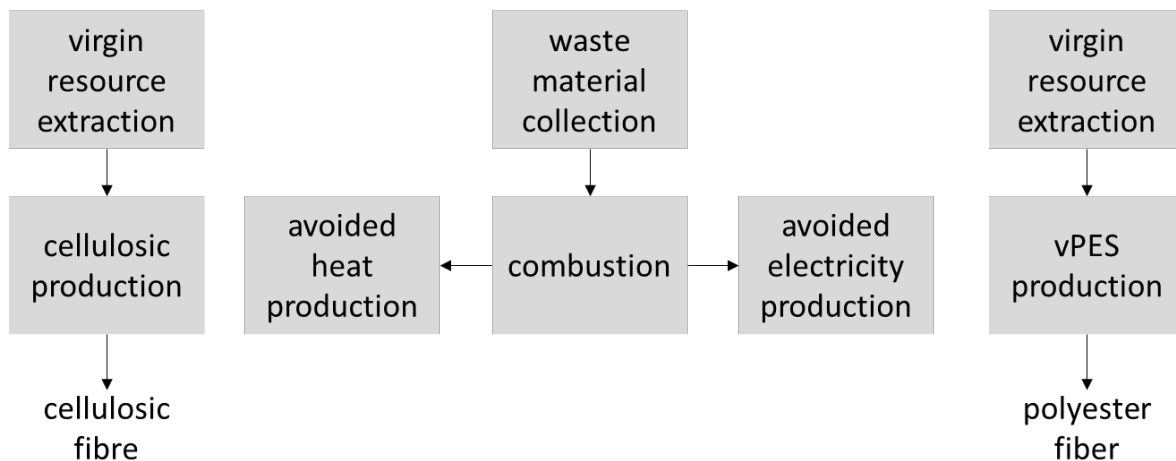


figure 4 Overview of the single-use baseline scenario

## 2.4. modelling approaches

The present study is a process-based LCA, which is “bottom-up” modelling in which the environmental impact of the life cycle is mapped based on its constituting parts – the unit processes – which are modelled separately and in detail. This is in contrast to an input/output (I/O) LCA, in which the life cycle is modelled by assigning a certain share of the flows or impacts of an industrial sector (e.g. Alvarez-Gaitan et al, 2013).

Furthermore, the present study is an attributional LCA. This means that we are attempting to map the product system as it is (or in the case of the present report: as we anticipate it to be), to learn more about the system and its associated environmental hotspots. This is in contrast to a consequential LCA, in which one attempts to map the consequences of a specific change or decision. The choice of an attributional rather than a consequential modelling approach has implications for the definition of system boundaries and the choice of allocation methods. Among others, consequential modelling is more inclined to account for secondary or tertiary affects arising due to market mechanisms.

Also, the study is a prospective LCA, i.e. a study of an emerging, yet non-existing product system (Arvidsson et al. 2017). Such studies are associated with some specific uncertainties, particularly because (i) some processes of the studied system do not yet exist and, in our case, had to be modelled based on pilot or bench scale data combined with some rough estimates on what is possible to achieve in terms of efficiencies; and (ii) background systems (electricity and heat production, production of input chemicals, etc.) change over time, and may therefore be rather different at a time when a commercial scale system has been realized.

## 2.5. allocation procedures

An important choice when conducting an LCA is how to allocate the environmental burden of multi-functional processes between the functions. How to solve such allocation problems is particularly an important choice in studies of products made from recycled feedstock. The key question is whether the incoming recycled (pre- or post-consumer) textile material should be considered to be responsible for any environmental burden of its previous life cycle (primarily, the initial raw material extraction) or whether it should be considered to be free of environmental burden from its previous processes. The first option reflects a view that the recycled material is a co-product of the previous product system, and that, for example, the economic profit of the previous product system, and therefore the demand for it, is influenced by the subsequent recycling of the material. The second option reflects a view that the recycled material is a waste that has no (or negligible) economic influence on the previous production system and should thus be considered to be free of environmental burden. The second option can be described as “cut-off allocation”. If the first option is chosen, the recycled material should be allocated a share of the burden of the initial raw material extraction (then a new allocation problem arises: how this share should be decided).

The second option for allocating the recycled material has been identified as the most common allocation procedure in LCAs of textile recycling in a recently published literature review (Sandin and Peters 2018), and is therefore chosen in the present study. Sandin and Peters (2018) also show it is common practice to apply system expansion and assign credit to the studied product because the presumably replaced production of some product from virgin materials. This approach is therefore also adopted in this study.

## 2.6. impact categories

In LCA, there is a wide range of impact categories to potentially include. For each impact category, there are several characterization methods to choose from. In the present study, the choice of impact categories and characterization methods is based on the choices made in a previous Mistra Future Fashion report (Roos et al. 2015), a selection that reflects important environmental issues facing the textile industry. Some modifications have, however, been made. We examine renewable and non-renewable energy, rather than just non-renewable, since the latter even more strongly correlated with the climate change indicator, while the former is a better (if indirect) indicator of intra-generational equity. We considered including a land use indicator (i.e. LANCA, Beck et al, 2010) but while the original method has been substantially revised in response to perceived shortcomings, the current version has not been implemented in the Gabi software, so we omitted it. We do not present results for ecotoxicity as they are strongly correlated with the results of other indicators. table 1 lists the selected impact categories and characterization methods. The impact categories are further described in the appendix.



table 1 Selected impact categories and characterisation methods

Impact category	Characterisation method	Unit	Reference for characterisation method
Climate change	Global warming potential with a 100 year perspective (GWP <sub>100</sub> ), excluding biogenic CO <sub>2</sub> emissions	kg CO <sub>2</sub> equivalent	IPCC (2013) as implemented in Gabi IPCC (2013) as implemented in GaBi (ILCD PEF recommendation, v1.09)
Acidification	Accumulated exceedence	Mole H <sup>+</sup> equivalents	Seppälä et al. (2006) and Posch et al. (2008) as implemented in Gabi
Freshwater eutrophication	Freshwater eutrophication potential (EUTREND model)	kg P equivalents	Struijs et al. (2009) as implemented in Gabi
Freshwater ecotoxicity	Ecotoxicity potential (USEtox model, recommended + interim)	Comparative toxic units for ecosystems (CTUe)	Rosenbaum et al. (2008) as implemented in Gabi
Human toxicity, carcinogenic	Human toxicity potential (USEtox model, recommended + interim)	Comparative toxic units for human (CTUh)	Rosenbaum et al. (2008) as implemented in Gabi
Human toxicity, non-carcinogenic	Human toxicity potential (USEtox model, recommended + interim)	Comparative toxic units for human (CTUh)	Rosenbaum et al. (2008) as implemented in Gabi
Energy use	Primary energy from renewable and non-renewable resources (net. cal. value)	MJ	Primary energy from renewable and non-renewable resources as implemented in Gabi
Water use impacts	AWARE	Stress-weighted ML	Boulay et al (2017) as implemented in Gabi

## 2.7. software and LCI databases

The Gabi Professional software, developed by ThinkStep, was used for modelling the product system and calculating the LCIA results. The Gabi Professional database (version 8.7, service pack 36) provided LCI data for the background processes. In cases where this was inadequate it was complemented with some data from the current Ecoinvent database (version 3.5) provided by Thinkstep.

## 2.8. limitations

Like many LCAs, the accuracy of this report is limited by life cycle data availability and ongoing methodological debates.

A fundamental issue for life cycle inventory data collection for this study is that it is an attempt to look forward to systems that do not yet exist in the commercial marketplace. There is therefore a need to scale up laboratory scale data to a larger scale. The scale we are considering here would be considered a large pilot plant rather than fully commercial - as long as we limit ourselves to commercial laundry waste (in order to have some certainty about its quality) then we are limited to textile flows at this scale in Sweden. The authors believe that it is necessary for process developers to gain experience operating at this scale before any further expansion is considered, so therefore it is also relevant to perform an environmental assessment at the current scale. Data from such a pilot facility will be necessary in order to refine the quality of this LCA and reduce uncertainty regarding the inventory data. Therefore this is essentially an examination of scenarios developed to understand the potential of particular future options. Some quantitative assumptions had to be made. Building rational assumptions about systems based on today's technologies is a worthwhile approach to grounding an LCA in reality. On the other hand, there will be economies of scale associated with the development of the Blend Re:wind process which cannot be assessed today. Studies like these are dependent on the use of life cycle inventory data collected by other analysts and compiled in the databases named in Section 2.7. This may introduce errors. There may be other issues of which we are unaware.

A key methodological uncertainty in LCA concerns the choice of attributional or consequential approaches. For an introduction to these ideas, see Baumann and Tillman, 2004. Many LCAs include features of both approaches. In this LCA we have attempted to adopt an attributional approach as far as possible, however the norm of considering that byproducts generate benefits through the creation of avoided products is arguably a consequentialist feature of this work.

There are other uncertainties associated with the use of linear impact characterization factors in LCA and the underlying models used to calculate them. This report reflects contemporary LCA practice, but the models are subject to improvement in the future.



## 3. inventory analysis

Life cycle inventory data describes the material and energy flows between processes within the life cycle of a system. In the case of this work, the systems under study are described in section 3.3. By processes we mean the engineered and user activities indicated by the figures in that section. Data representing background systems (energy supply, transportation etc) are drawn from commercial databases as described in Section 2.5 and not elaborated here. Data for other key processes is described in this section of the report.

### 3.1. Blend Re:wind process

An overall mass balance for the process was developed using elementary numerical methods and is presented in table 2 to table 4. The numbering of the processes is consistent between the level three headings in the subsequent text, and those indicated in figure 5.

#### 3.1.1. material collection

The five county laundry facilities operated by local councils and regional governments, and the eleven facilities operated by Textilia or Berendsen are distributed throughout Sweden from Boden in the north to Malmö in the south. The majority of them are located in the southern part of Sweden, reflecting the population density. The distance from them to Gothenburg, the assumed location of a BRW site, varies between 10 and 1270 km with an average distance of 378 km. Whereas these sixteen laundry facilities do not represent all laundries from which collections need to be made in order to achieve the full potential of the recycling system, from their distribution it seems plausible to assume a 400 km average transport distance from the laundries to Gothenburg. Assuming that the annual supply of 850 tonnes of waste had been generated by sixteen laundries, each laundry would generate about 1 tonne/week. Since the value of the goods is low and the used textiles have a long shelf-life when stored correctly, it is reasonable to believe the laundries would store used textiles in a shipping container, accumulating approximately ten tonnes in ten weeks for subsequent transport to the recycling facility. When compacted, this material would occupy most of the volume of a 12 metre shipping container on a 28 tonnes semitrailer truck.

table 2 Annual mass (tonnes) and energy (GWh) balance, collection to hydrolysis

Process	1		2		3			4		
	collection		cutting & ripping		hydrolysis			filtration		
Flow	in	out	in	out	in fresh	in recycle	out	in	out	losses
<i>total fiber</i>	8.5E+02	8.5E+02	8.5E+02	6.9E+02	6.9E+02					
<i>cotton fiber</i>	3.8E+02	3.8E+02	3.8E+02	3.1E+02	3.1E+02		3.1E+02	3.1E+02	2.9E+02	1.9E+01
<i>polyester fiber</i>	4.7E+02	4.7E+02	4.7E+02	3.8E+02	3.8E+02					
<i>Na<sub>2</sub>TP</i>					4.1E+02	9.0E+01	5.0E+02	5.0E+02	4.7E+02	3.0E+01
<i>TPA</i>										
<i>EG</i>					1.2E+02	1.6E+02	2.8E+02	2.8E+02	2.6E+02	1.7E+01
<i>NaOH</i>					5.5E+02	1.1E+03	1.5E+03	1.5E+03	1.4E+03	9.4E+01
<i>acetic acid</i>										
<i>H<sub>2</sub>SO<sub>4</sub></i>										
<i>Na<sub>2</sub>SO<sub>4</sub></i>										
<i>water (fresh)</i>										
<i>water (other)</i>					4.8E+03	1.6E+04	2.1E+04	2.1E+04	1.9E+04	1.2E+03
<i>total mass</i>					6.2E+03	1.7E+04	2.3E+04	2.3E+04	2.2E+04	1.3E+03
<i>electricity</i>			5.1E-01		7.7E-03			4.8E-03		

## 3.1.2. cutting and ripping

Some physical pre-processing of the materials is expected. An emphasis is placed on manual preselection of appropriate polycotton materials in the collection phase, but some garments (e.g. trousers) may have elasticized waistbands which would need manual removal to prevent elastane contamination of subsequent process steps. Zippers and buttons would also be removed. This will result in some material losses, assumed to be 19% of the material inflow (Spathas, 2018). Once these contaminants are removed, ripping and shredding machines can be more reliably operated. These processes are assumed to demand 2.68 MJ/kg inflow in accordance with Spathas (2018).

table 3 Annual mass (tonnes) and energy (GWh) balance, filtration and washing

Process	4b				4c			5		
	cotton acid wash				cotton neutral wash			nanofiltration		
Flow	in ex filter	in ex recycled acid	out	out (aq.)	in	out (prod.)	losses	in	retent- ate	perm- eate
<i>total fiber</i>										
<i>cotton fiber</i>	2.9E+02		2.9E+02		2.9E+02	2.9E+02				
<i>polyester fiber</i>										
<i>Na<sub>2</sub>TP</i>	3.0E+01	5.7E+02		6.0E+02				4.7E+02	3.8E+02	9.0E+01
<i>TPA</i>										
<i>EG</i>	1.7E+01	3.2E+02		3.4E+02				2.6E+02	1.1E+02	1.6E+02
<i>NaOH</i>	9.4E+01							1.4E+03	3.0E+02	1.1E+03
<i>acetic acid</i>	3.6E+02	4.2E+03		4.4E+03	2.3E+01		2.3E+01			
<i>H<sub>2</sub>SO<sub>4</sub></i>										
<i>Na<sub>2</sub>SO<sub>4</sub></i>										
<i>water (fresh)</i>	1.0E+04				2.3E+05					
<i>water (other)</i>	1.2E+03	2.1E+05	2.3E+05	1.1E+04	1.2E+03	1.2E+03	2.3E+05	1.9E+04	3.7E+03	1.6E+04
<i>total mass</i>	1.2E+04	2.2E+05	2.3E+05	1.7E+04	2.3E+05	1.4E+03	2.3E+05	2.2E+04	4.5E+03	1.7E+04
<i>electricity</i>								7.9E-02		

table 4 Annual mass (tonnes) and energy (GWh) balance, nanofiltration to PES synthesis

Process	6		7		8		9		10	
	acidification		filtration		purification		cellulosic production		PES synthesis	
Flow	in	out	in	out	in	out (prod.)	in	out	in	out
<i>total fiber</i>										
<i>cotton fiber</i>							2.9E+0 2	2.8E+0 2		
<i>polyester</i>										3.5E+0 2
<i>Na<sub>2</sub>TP</i>	3.8E+02									
<i>TPA</i>		3.0E+0 2	3.0E+0 2						3.0E+0 2	
<i>EG</i>	1.1E+02	1.1E+02	1.1E+02	1.1E+02	1.1E+02	1.1E+02			1.1E+02	
<i>NaOH</i>	3.0E+02									
<i>acetic acid</i>										
<i>H<sub>2</sub>SO<sub>4</sub></i>	5.1E+02									
<i>Na<sub>2</sub>SO<sub>4</sub></i>		7.4E+0 2	7.4E+0 2	7.4E+0 2	7.4E+02					
<i>water (fresh)</i>										
<i>water (other)</i>	3.7E+03	3.8E+0 3	3.8E+0 3	3.8E+0 3	3.8E+03	5.1E+0 0				6.6E+0 1
<i>total mass</i>	5.0E+03	5.0E+0 3	5.0E+0 3	4.7E+0 3	4.7E+03	1.1E+02	2.9E+0 2	2.8E+0 2	4.2E+0 2	4.2E+0 2
<i>electricity</i>			3.8E- 03		2.6E+00					

### 3.1.3. hydrolysis

The alkaline hydrolysis process included in this LCA is based on the work of Palme et al (2017) and Björquist (2017), particularly the latter with its phase-transfer-catalyst-free variant on the former's process. Principle inputs to the process are listed in Table 4.4 (ranges are from Björquist (2017), the middle value was assumed for the present study).

table 5 Lab-scale hydrolysis conditions

Input or variable	Unit	Value	Range
Polycotton material	g	3	1-5
Sodium hydroxide solution	g	97	
	%	7.5	5-10
Temperature	°C	80	70-90
Stirring	rpm	150	
Time	minutes	390	

To estimate the energy demand associated with the process at large-scale, assumptions about the stirring and heating were necessary. Stirring was assumed to be provided with a 370 W stirrer based on a previous assessment of similarly dilute sludge stirring systems in the wastewater industry (Peters et al, 2009). It was assumed that the reactor system is operated without energy recovery, with the water for the sodium hydroxide solution starting at 15°C and heated using electricity. This initial worst case assumption (i.e. the absence of energy recovery) did not significantly affect the results, and was therefore retained for the final LCA without additional refinement.

### 3.1.4. filtration

The cotton material remaining in the solid phase after dissolution of the polyester through alkaline hydrolysis is not suitable for direct respinning to cotton yarn due to it having too low a degree of polymerization, but can be used as an input to the production of viscose (rayon) or lyocell fibers (Tencel), much like dissolving pulp. Removal of the denatured cotton was assumed to occur using a belt filter press. These devices are relatively easy to use and can handle variations in inflow concentrations associated with startup conditions or other influences. Power consumption for a belt filter accepting 1-2% solids at 90-300 (kg solids)/hr was 1.5 kW, consistent with Peters and Rowley (2009). Three such devices were considered in order to continuously process the cotton after separation from polyester, after acid washing and neutral washing.

In an early iteration of this LCA it was identified that acetic acid consumption could skew the results unless some rational assumptions regarding scale-up were taken. The lab-scale process described in Björquist (2017) suggests the recovered cotton should be initially washed in a 2% acetic acid solution in order to neutralize sodium hydroxide in the liquid absorbed by the recovered cotton. A liter of this solution was used to treat just over a gram of cotton in this way. Based on a mass balance over the sodium terephthalate and considering the scale of losses, we



estimated the filtered cotton has a moisture:solids ratio of 4.4:1, which is within the feasible range of 0:1 to 27:1 (Cotton Inc, 2018). This means that at laboratory scale, over thirty times the stoichiometric acid requirement was employed to ensure complete neutralization. One could make the assumption that after an initial charge, the acid is dosed continuously into the neutralization vessel at the stoichiometric rate, but the need exists for a bleed stream to prevent the buildup of entrained ethylene glycol and sodium terephthalate. Therefore we assumed that the neutralizing acid solution was recycled at a rate of 95% of the inlet acid solution. This value was arbitrarily chosen to prevent the concentration of salt (sodium terephthalate) from rising above 10% of the acid concentration.

After acid washing, a wash with deionized water is used to eliminate the retained acid from the cotton. Losses of cotton during processing are assumed to be 6% (Björquist, 2017). The massive single-use of water for this purpose in laboratory experiments was scaled back on the basis that commercial cotton washing processes use 1% of the laboratory values. In the data provided by Palme et al (2017), the water use is very large compared with the mass of cotton (three orders of magnitude larger). This value would presumably be reduced in a practical larger-scale operation. An indication of the potential for saving water may be apprehended from the ratio of the mass of water to cotton in the laboratory (788:1) versus the range of ratios for Electrolux's current range of washing machines. According to the efficiency data available at [www.electrolux.se/laundry/laundry/washing-machines/](http://www.electrolux.se/laundry/laundry/washing-machines/) at the time of writing, this ranges from 5.7:1 for their best A+++ rated machines to 10:1 for their worst, the A+ rated machines. While other equipment may be more appropriate for industrial conditions, this data indicates that it may be practical to save more than 99% of the water used in the neutral washing step after scaling up from (laboratory) gram scale to kilogram scale batch operations. We therefore scaled the water use down by this factor.

## 3.1.5. nanofiltration

Preliminary assessment of the stoichiometric relationships in the process described by Palme et al (2017) and Björquist (2017) indicated that the Blend Re:wind process will generate large amounts of a salt (sodium sulfate) as a consequence of the dissolution and precipitation reagents (sodium hydroxide and sulfuric acid). This could create scaling problems for downstream processes and ultimately a large mass of waste, in addition to the impacts associated with the requirement for large flows of reagents into the process.

Fortunately there is some literature providing some information on how the sodium hydroxide can be separated for reuse. Choe et al (2005) describe a similar situation in relation to saponification effluent. Saponification of polyester using sodium hydroxide imparts a soft feel to garments, but results in the partial dissolution of the material to disodium terephthalate and ethylene glycol. In their experiments, inlet concentrations of disodium terephthalate, ethylene glycol and sodium hydroxide were in the ranges of 2–4%, 0.5–1.0% and 1–4%, respectively. Using nanofiltration membranes they were able to separate 72–87% and 36–45% of the disodium terephthalate and ethylene glycol, respectively, creating a permeate with 72–84% of the sodium hydroxide depending on the rate at which the influent was recycled in the process. Of course, the more sodium hydroxide was recovered in the permeate, the lower the retention of disodium terephthalate and ethylene glycol. For this LCA we chose the average of the results: 79% of the sodium hydroxide in the permeate, 81% of the disodium terephthalate and of the 40% ethylene



glycol in the retentate. These results can be compared with those of Orecki et al (2006). The latter reported experiments designed to separate ethylene glycol in which, depending on the choice of nanofiltration membrane, between 12 and 70% of the glycol from a 5% solution was kept in the retentate on nanofiltration membranes operating at a pressure drop of 1.5 MPa and a flux of  $50 \pm 2 \text{ dm}^3/\text{m}^2/\text{hr}$ . Unfortunately they did not report the rate of water retention, which hampers the design of a glycol separation processes using their data.

### 3.1.6. acidification

In this step, sulfuric acid is added to the retentate from the nanofiltration step in order to precipitate the disodium terephthalate as terephthalic acid. Disodium sulfate salt is produced both by precipitation of the terephthalic acid (one mole per mole of TPA) and neutralisation of the sodium hydroxide solution (one mole per mole). This creates a very salty solution (16% sodium sulfate) which is nevertheless below the solubility limits for solutions above 15 °C. The salt reaches maximum solubility 49.7 g/100 mL at 32.4 °C (see Figure 4.1). In this LCA model, sufficient heat is assumed to be carried over from the heated hydrolysis reactor to the acidification process to keep the salt in solution.

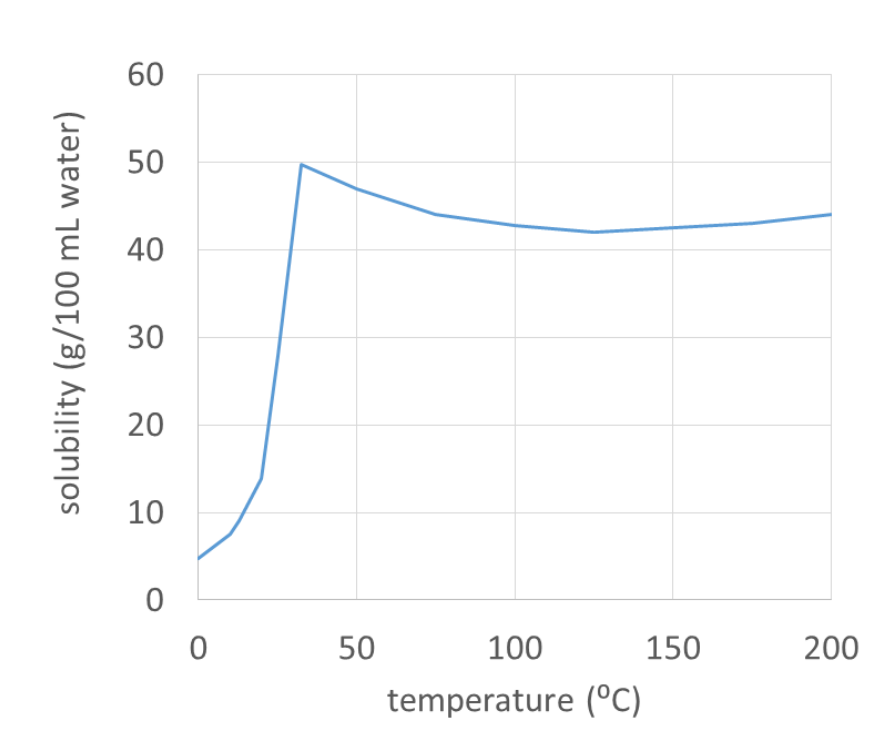


figure 5 Solubility of sodium sulfate

### 3.1.7. filtration

A filtration process is performed after acidification to remove the solid-phase terephthalic acid from the neutralised solution. As with the previous filtration step for the removal of denatured cotton, this is assumed to require a belt filter press drawing 1.49 kW.

### 3.1.8. glycol purification

Compared with the separation from sodium hydroxide, separating ethylene glycol from sodium sulfate appears more challenging. The glycol is only 0.48 nm in size (Dougan, 2011), similar to the ionic radius of sulfate in solution 0.40 nm (Marcus, 1988), about twice the size of water (0.24 nm, Dougan, 2011; 0.28, Marcus, 1988). Moreover, the high solubility of ethylene glycol makes it a challenging chemical to separate from water. Several methods for extracting glycol from the aqueous phase have been suggested. Various authors describe the use of membrane and thermal approaches. For example Jehle et al (1995) described a combined process for concentrating glycol using evaporation, membrane filtration and pervaporation. The evaporation process was central to the design, dividing the 25% glycol mixture into a 0.5% and a 70% stream. To raise the concentration above this to 95%, more efficient separation was obtained using a pervaporation process. The efficiency is higher because in contrast to simpler evaporative processes, the introduction of a water-selective membrane in pervaporation means that only a fraction of the feed change phases (Nik et al, 2006). The pervaporation membranes they preferred had a permeate flux of about 2.6 L/m<sup>2</sup>.hr at 75 degrees Celsius. The overhead product from the evaporation process and the permeate from the pervaporation process were cleaned using a reverse osmosis membrane, which reduced the glycol concentration of the permeate from about 0.5-1% to about a tenth of those figures. Bench-scale tests were performed using a rotary vacuum evaporator (Eurodelta CN 250, Milano, Italy) with a capacity of 10 L/h distillate. This device required 0.1 kWh/L of 25% glycol to separate the liquid into a 0.5% and a 70% glycol stream.

In the case of the Blend:Rewind process, a challenge to evaporative and pervaporative processes like that of Jehle et al (1995) is the high concentration of salt in the solution. Even after implementation of a process like that of Choe et al (2005) to recycle the sodium hydroxide, thus reducing the sulfuric acid demand in the acidification process, the concentration of sodium sulfate in the product solution is an order of magnitude higher than the ethylene glycol concentration. Since ethylene glycol has a higher boiling point than water (i.e. 198°C), the salt would be retained in the ethylene glycol stream in these separation (i.e. dehydration) processes. Additionally, sodium sulfate is not as soluble as sodium chloride and 16% salt, is at fully 32% of its maximum solubility (which is attained at 32°C), so a heightened risk of fouling would be associated with pervaporation processes applied immediately after acidification.

Other technologies for separating the salt and ethylene glycol solution include sorption, ion exchange, distillation and reverse osmosis. For analytical purposes, where ethylene glycol concentrations are in the ug/mL range, Sidisky et al (2010) recommended the use of a microporous carbon molecular sieve material as an absorbent for preconcentration of the glycol.

Methods have also been proposed based on ion-exchange membranes, but only at low salt concentrations (Butyrskaya et al 2012).

At a larger scale, ion exchange resins (typically in bead form) are used to adsorb unwanted ions from solutions. However, this was not considered for this LCA because preliminary assessment of these alternatives suggested large flows of resin regeneration chemicals would be required. The more robust but more energy intensive distillation methods in use in the oil and gas industry work by evaporating both water and glycol, concentrating salt in a brine for crystallization or disposal at sea. Subsequently the water and glycol are separated based on their different boiling points. The process has to occur under vacuum to reduce the operating temperature and reduce the risk of ethylene glycol degradation (Schlumberger, 2016). Dialogue with a manufacturer of glycol distillation equipment indicated an electrical duty of 422 kW for circulation and vacuum pumps and 6.7 MW heating duty. This was assumed to be provided by steam to the distillation column in this LCA, with natural gas as the ultimate energy source. This choice of source was made on the basis that it is an important fuel in the Gothenburg region, and it sits between the possible extremes in terms of the climate impacts assessed in a typical LCA (e.g. between Swedish wind power and marginal German coal power).

### **3.1.9. production of regenerated cellulose fibers**

The most common regenerated cellulose fibers are viscose, lyocell, cellulose acetate and cupro. Regenerated cellulose fibers are conventionally manufactured from virgin pulp, either in integrated mills producing both pulp and fibers or in mills producing fibers from market pulp. Production of regenerated cellulose fibers from recovered sources, e.g. post-consumer paper and cotton textiles, is technically possible and is being researched but there is currently no LCI data available. A notable development in this area is the launch last year of Lenzing's "Refibra" product made from pre-consumer waste textiles and new wood resources.

The environmental impacts from Lenzing's production of viscose and lyocell, the two types of regenerated cellulose fibers that together dominate the market, have been studied by LCA (Shen and Patel, 2010). It has been concluded that the more modern and resource efficient lyocell process has lower impacts than the viscose process but also that integrated production has lower impact than pulp and fiber production in different sites and that the choice of energy source has a large influence on the environmental impact. Altogether, the environmental impacts of regenerated cellulose fibers depend less on the choice of fiber (e.g. viscose vs. lyocell) and more on the kind of production facility (integrated mill or not, electricity supply, heat supply etc). In other words, the range of results for viscose roughly spans the claimed range that the newer cellulosic fibers occupy. Therefore, in this LCA we used data from the Ecoinvent database (i.e.: "Viscose production, global average") that occupies a position in the mid-range of Shen and Patel's estimates of the impacts of viscose production. From this aggregated LCI dataset we subtracted the impacts of the production of bleached sulfate pulp which fully replaced by the recovered cotton fiber in the BRW process.

## 3.1.10. polyester synthesis

Polyester production was modelled based on the data collected by PlasticsEurope and included in the Gabi Professional database. This implies a mix of ester interchange and direct esterification routes. Using the same original data source, the production of ethylene glycol and terephthalic acid was subtracted from the process in proportion to the flows of these substances recycled into polyester synthesis by the Blend Re:wind process. There was a small stoichiometric mismatch caused by higher losses of ethylene glycol than terephthalic acid *en route* to polyester synthesis. This was compensated for by inclusion of virgin ethylene glycol.

## 3.2. single use baseline

If the blended fiber waste is not chemically recycled, its most likely fate in Sweden is either mechanical recycling or (most likely) combustion with energy recovery. The textiles in question are sufficiently damaged that higher-value recycling is unlikely. Opportunities for mechanical recycling into cleaning rags or thermal or acoustic insulation exist but the market is saturated. The route to energy recovery is well-established and unmet demand exists as demonstrated by the importation of municipal solid waste to Sweden for combustion. Therefore combustion with energy recovery was selected for the single use baseline.

Combustion facilities are located throughout Sweden near population centres. Used textiles were assumed to be trucked an average of 40 km to these facilities (i.e. the materials were assumed to be transported only 10% of the distance used in the Blend Re:wind scenario).

The production of materials in the baseline scenario was scaled to be equivalent to the amounts of recycled polyester and regenerated cellulosic materials which are generated in the BRW scenario. Virgin viscose and cotton production data were taken from the Ecoinvent database on the basis described in Section 3.1.9 (but without subtracting the production of bleached sulfate pulp), and virgin polyester production data were taken from the Gabi Professional database in line with Section 3.1.10.

## 3.3. alternative analysis

### 3.3.1. quantitative sensitivity analysis

At this stage of technology development, there is considerable uncertainty about the selection of unit operations for the recycling systems and the appropriate baseline with which to compare them. Therefore, rather than offering false precision via detailed statistical error analysis, the principal approach to sensitivity analysis adopted here is the examination of alternative scenarios.

In order to illustrate the effect of alternative single use systems on the relative performance of the BRW recycling system, a further single use scenario was examined in addition to the one described in Section 3.2. As in the single use baseline scenario, polyester produced from virgin fossil resources was included, but in the alternative baseline scenario, the cellulosic fiber in contention is cotton rather than forest-based viscose fiber. The polyester and cellulosic fiber was assumed to be produced in the same quantity as the polyester and viscose produced by the recycling system. This scenario may be interesting given that current cotton production exceeds forest cellulosic fiber production, so can also be thought to represent business-as-usual.

Preliminary analysis indicated that one of the largest contributors to energy consumption and climate impacts in the BRW recycling system is the purification of ethylene glycol. Therefore a further scenario was considered in the LCA in which the liquid waste stream remaining after separation of terephthalic acid was disposed of to a wastewater treatment facility, and new ethylene glycol made from fossil resources was used in the synthesis of polyester with the recycled terephthalic acid. We speculate that this scenario may be interesting from an environmental and cost perspective.

### **3.3.2. rejected alternative scenarios**

Since the glycol distillation process is a major energy user in this model, some alternative unit operations were considered. Pervaporation was considered for separation of glycol and water. Pervaporation means boiling off part of a mixture using a membrane that only allows that part to pass. In this case, given the fact that glycol has a higher boiling point, it would mean boiling off the water and retaining the glycol. On account of this fact, introduction of pervaporation would be unhelpful since it would mean concentrating the sodium sulfate salt in the product (glycol) stream. Furthermore, Guo et al (2006) report that pervaporation is primarily advantageous for glycol feed concentrations over 70%, a statement supported by a quantitative model presented by Huang et al (2002).

Membranes might be more interesting later in the overall process - having to boil the whole glycol solution to eliminate the salt by distillation could in principle be avoided by preliminarily rejecting the salt via a membrane treatment such as reverse osmosis. We considered this alternative as a potential scenario but rejected it before detailed analysis for two principal reasons. Firstly, the glycol solution after acidification has a very high salt concentration compared with the typical inlet salt concentration of reverse osmosis membranes (i.e. seawater, c. 3.5%). Experimentation with the Dow WAVE model (Dow, 2018) suggested that the glycol solution would need to be diluted several times with fresh water in order for the operating limits of the reverse osmosis elements to be respected. This has the effect of driving up the total liquid volume to be filtered, and with it, the energy consumption of the system. Secondly, the gains to be had from this process are limited since the concentration of glycol in the solution post-acidification is rather low. Consequently, the energy to be saved by only evaporating the water and using membranes to remove salt from the glycol is relatively small in comparison with the likely cost and complexity of adding a reverse osmosis unit operation to the overall process (see Huang et al, 2002).



## 4. impact assessment results and discussion

### 4.1. hotspots of the BRW scenario

figure 6 summarizes the results for the three indicators with the strongest association with energy consumption in this LCA. Note the use of a logarithmic y-axis. There is a common pattern across many of the unit operations shown in the figure, with the processes of hydrolysis, glycol separation, and production of regenerated cellulosic fiber ("cellulosic production" in the below figures) being the outstanding factors for the environment. Perhaps unsurprisingly, production of chemicals for acidification of the sodium terephthalate also plays an important role in contributing to the result for acidification of the environment. The viscose production relies on considerable quantities of chemicals and even though the data used for this LCA reflects the use of internal recycling within the viscose mill, the production and recovery of the chemicals is nevertheless energy intensive, explaining the impacts from cellulosic production.

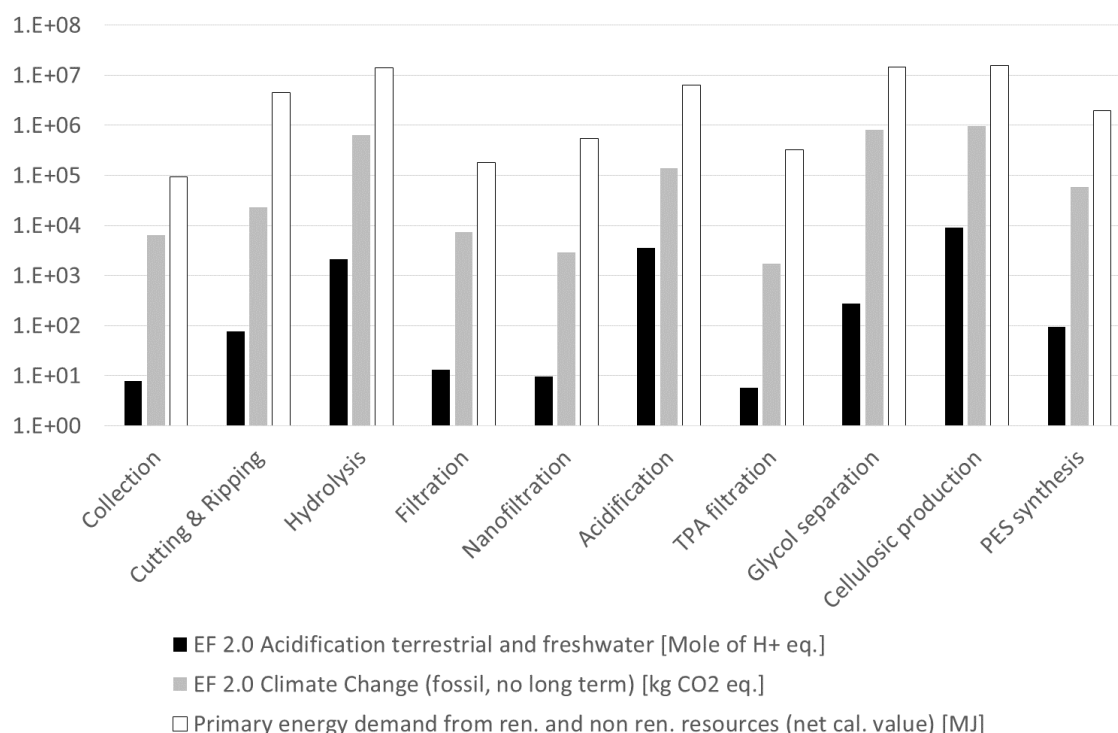


figure 6 Acidification, climate impact and energy consumption for the BRW scenario

The dominant contributor to the impact of the glycol separation process is the production of steam to heat the distillation process. The fundamental energy requirement is unavoidable given the need to separate the liquids from salt contamination using a phase change, but it may be possible to reduce the impacts associated with this process by integrating the recycling facility with other future industrial operations possessing waste heat. On the other hand, the impact of the filtration and hydrolysis processes is dominated by the production of acetic acid and sodium



hydroxide, respectively. Therefore reduction of the impact of these processes within the recycling operation will depend on whether it is feasible to obtain supplies of these commodity chemicals that are environmentally superior to the market average supplies incorporated in the model, or further reducing their use by fine tuning the internal recycling processes built into the model – in particular, the nanofiltration process which recycles the sodium hydroxide, and the return flow of acetic acid in the recovered cotton washing process. The former would in turn also reduce the amount of sulfuric acid required in the acidification process for precipitation of terephthalic acid.

The eutrophication indicator was dominated by the process for turning the recovered cotton into viscose, see figure 7 (note the logarithmic y-axis). This may be a consequence of the production of carbon sulfide and other chemicals for the process. Note that figure 7 excludes a small avoided emission (-2.3 kg P-equivalent) associated with the production of polyester. This is the result of subtracting the production of virgin ethylene glycol and terephthalic acid from an aggregated polyester synthesis process.

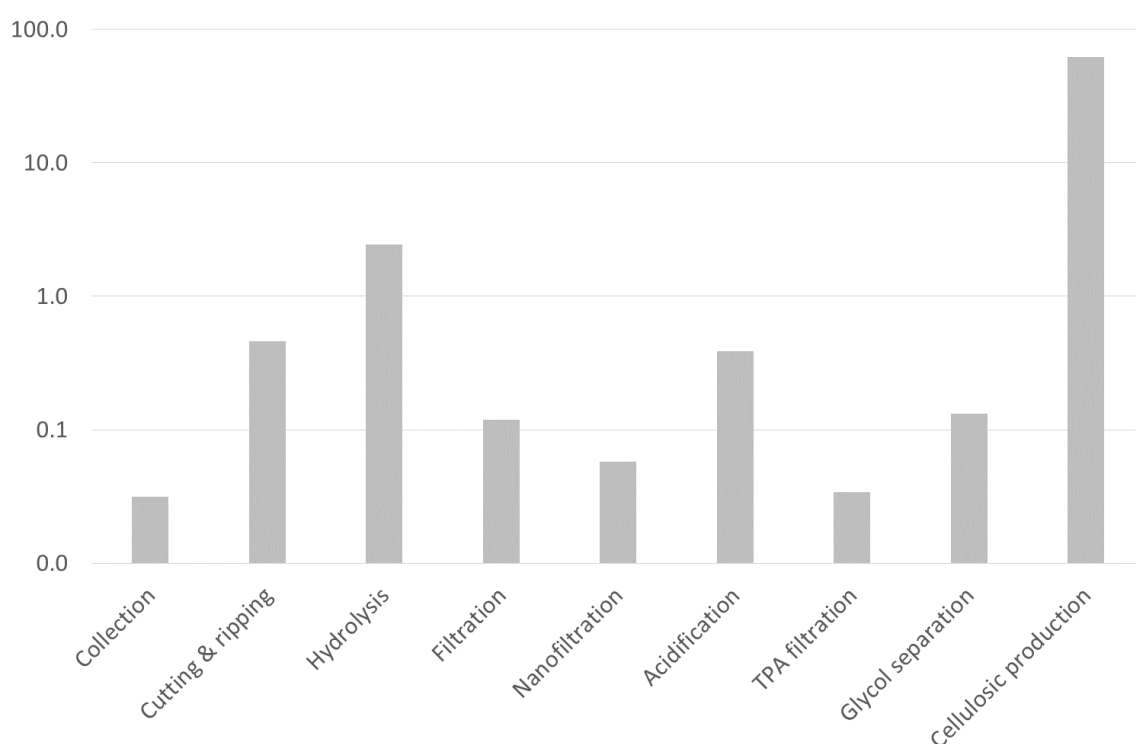


figure 7 Eutrophication in the BRW scenario (kg P equivalent)

The contributions to human toxicity are more evenly spread across unit operations for cancer than for non-cancer effects, as shown in figure 8. (Note the use of a log scale, which enables both indicators to be shown in this figure.) As it is with eutrophication, the single largest contributor to the overall score is the viscose production process. There is also a small reduction in the overall non-cancer score (-0.0009 CTUh) associated with the production of polyester (not shown in the figure). Hydrolysis and acidification appear in the profile on account of their demand for chemicals and water. The freshwater ecotoxicity indicator is not shown here for brevity as the distribution of impacts across processes mimics the human toxicity indicators.



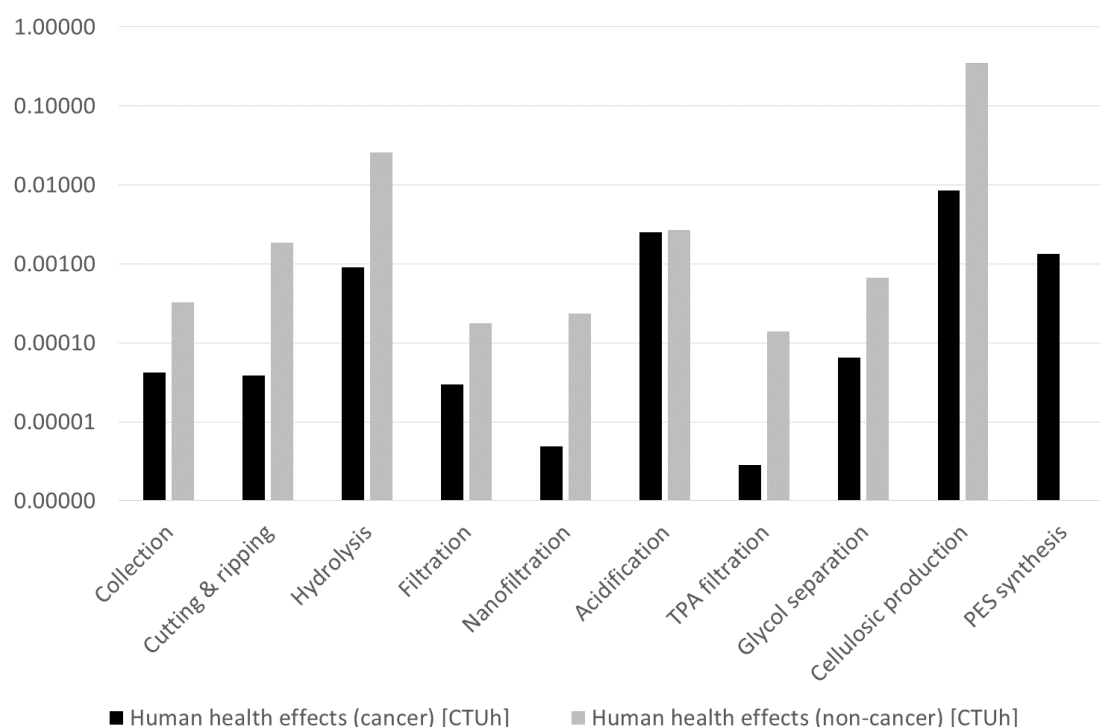


figure 8 Human toxicity in the BRW scenario

The hydrolysis step is the main cause of water use impacts in the textile recycling process (figure 9). This is primarily a consequence of the production of sodium hydroxide, and to a lesser extent, the process water used to dilute it during the BRW process. The viscose production process is the next most important user of water resources. That the ostensibly dry ripping process is a water consumer may surprise some readers but is a consequence of water use in the production of electrical energy.

The data on water consumption are based on a set of characterisation factors implemented in the Gabi Professional software which assume that when data on the location of water use is unavailable, a low characterisation factor is used. This is considered appropriate for this LCA given the location of the processes in mainland Sweden, where generally speaking water shortages do not occur. In particular, if the recycling facility is located in Gothenburg, it is assumed that the water supplied via the Göta River, which drains Sweden's largest lake, is not a limiting factor. This approach and these assumptions may be worth discussing in terms of future climate scenarios or alternative locations for the infrastructure.

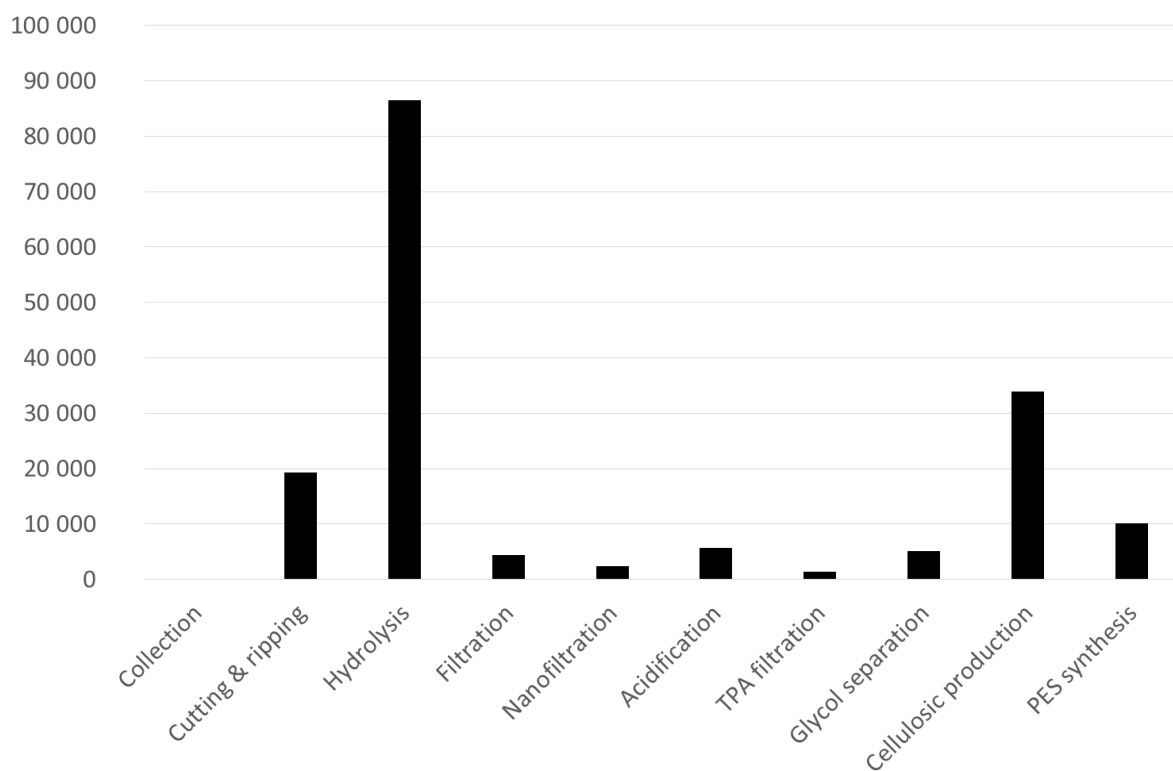


figure 9 Water impacts in the BRW scenario (m³ equivalent)

## 4.2. Sensitivity analysis and comparison with single use

Overall it can be said that the relative performance of the BRW scenarios (with and without ethylene glycol recovery) differs between the calculated indicators. For half of the assessed indicators the basic BRW scenario outperforms at least one of the single use scenarios. However, in absolute terms, the differences in performance are mostly less than an order of magnitude, and indicate that the BRW process does not lead to significantly higher or lower environmental impacts. This is encouraging for the BRW process, since the BRW scenarios are based on scaled-up laboratory data, while the single-use scenarios are based on optimised, commercial-scale, industrial operations.

figure 10 indicates that performance of the BRW process, in terms of its primary energy demand, is of the same order of magnitude as its single-use benchmarks. Eliminating the ethylene glycol recovery step from the process reduces the energy demand to the same level as producing viscose from virgin materials. The single use alternative of using virgin cotton and polyester is the superior alternative to the single use of virgin viscose and polyester. There are many reasons for the overall pattern here. Fundamentally, the recycling scenario is at a bench-scale level of development and has not had the worker-centuries of effort put into improving the energy balance of the system, unlike the level of development of the cotton and virgin viscose alternatives.

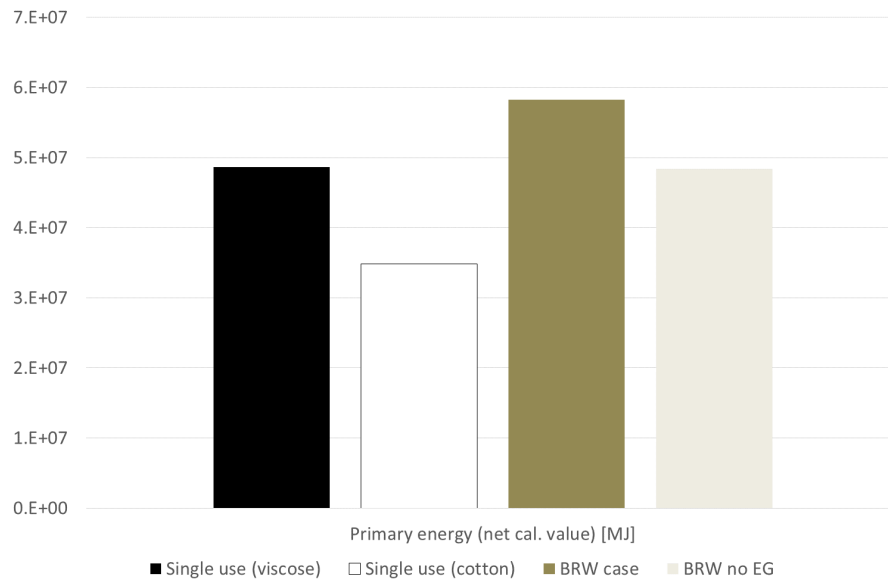


figure 10 Primary energy consumption for alternative scenarios

As figure 11 indicates, the climate change indicator shows a similar profile to the energy consumption indicator across the four alternatives, indicating a strong influence of fossil fuels on the results for both indicators. The results differ from those presented by Zamani et al (2015), in which there appeared to be a significantly better climate impact result for polymer recycling in comparison to energy recovery. The difference may be a result of the use of more recent and detailed data in this LCA, or the decision to avoid the expensive NMMO solvent by selecting the BRW process.

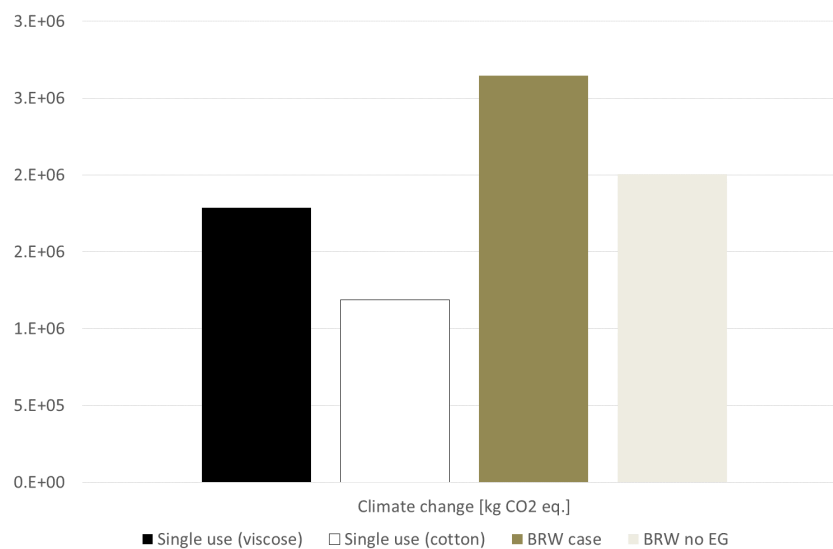


figure 11 Climate impact for alternative scenarios

Despite the role of energy supply and fossil fuel combustion in environmental acidification problems, the acidification indicator (figure 12) exhibits a different profile to the previous two indicators, in particular suggesting the superiority of single-use viscose over single-use cotton. While the process for the production of viscose is superior to cotton production in this respect, both the single-use scenarios benefit from the production of energy from waste material, which prevents other power generation activities. Additionally, as might be expected, the acidification

process and the production of sulfuric acid that it demands, is associated with additional acidifying emissions.

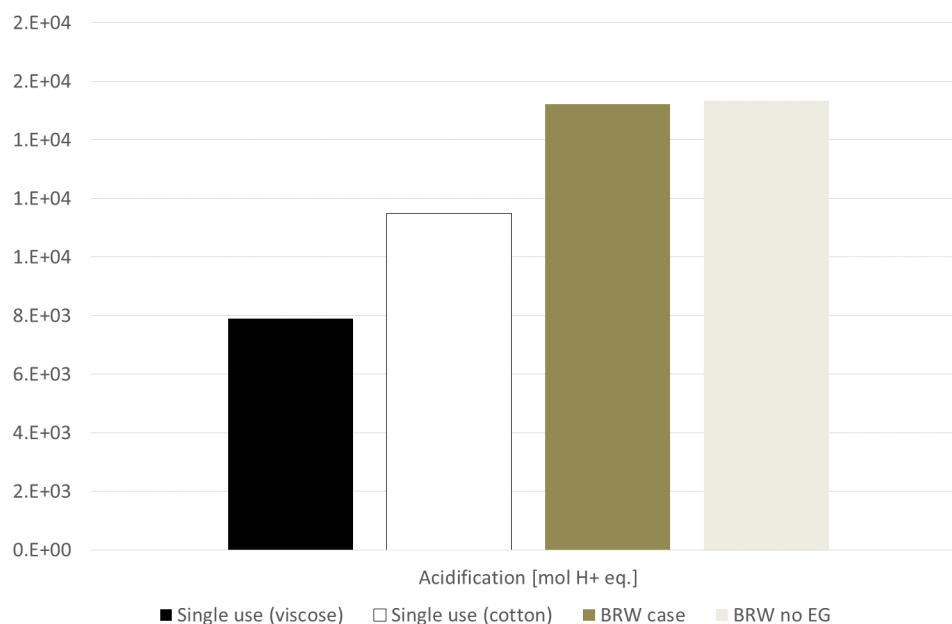


figure 12 Acidification potential for four scenarios

The eutrophication calculations favour the recycling systems over the single use alternatives by a small margin, as shown in figure 13. The relatively poor performance of the cotton alternative is caused by, the use of fertilisers in cotton cultivation.

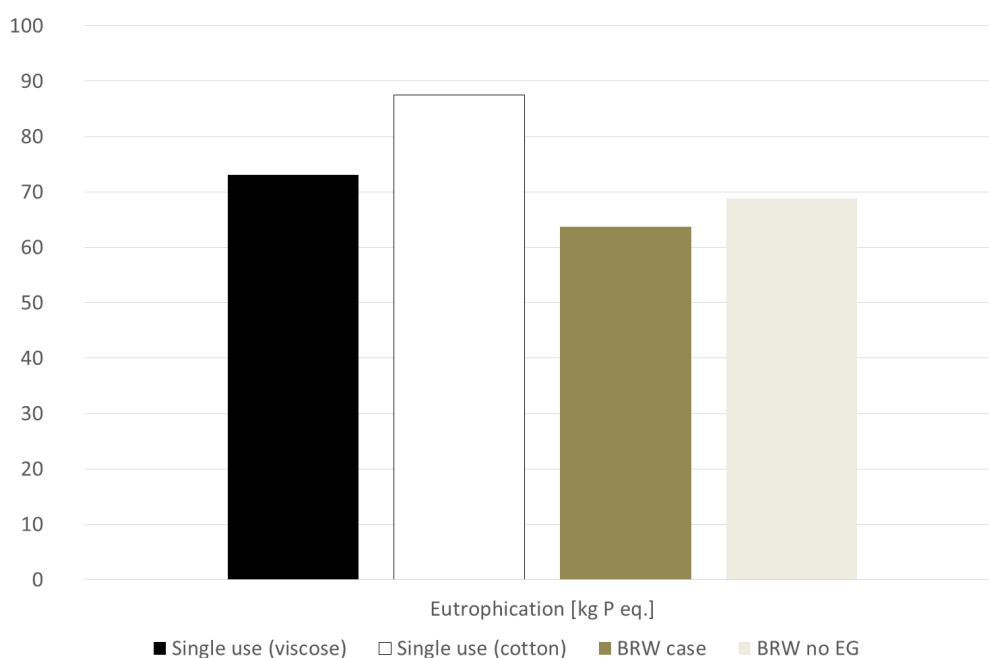


figure 13 Eutrophication potential for alternative scenarios

The toxicity indicators shown in figure 14 present different messages. (Note the use of a log scale, which enables both indicators to be shown in this figure.) The indicator for cancer as a health effect favours recycling using BRW. The largest single contributor to the scores for the recycling

systems is the cellulosic fiber production process, but for cancer effects there are many other contributions connected with production of process chemicals such as acetic acid, sulfuric acid and sodium hydroxide. In the case of this indicator there is a small penalty associated with not recycling the ethylene glycol but on the other hand the results for the BRW scenario and the no EG BRW scenario are very similar for the non-cancer health effects indicator. In the case of this indicator, the single-use cotton scenario seems superior but the single use viscose benchmark generates a very similar result to the BRW scenarios.

(The freshwater ecotoxicity indicator is not shown here for brevity as the distribution of impacts across processes mimics the human toxicity (cancer) and eutrophication indicators.)

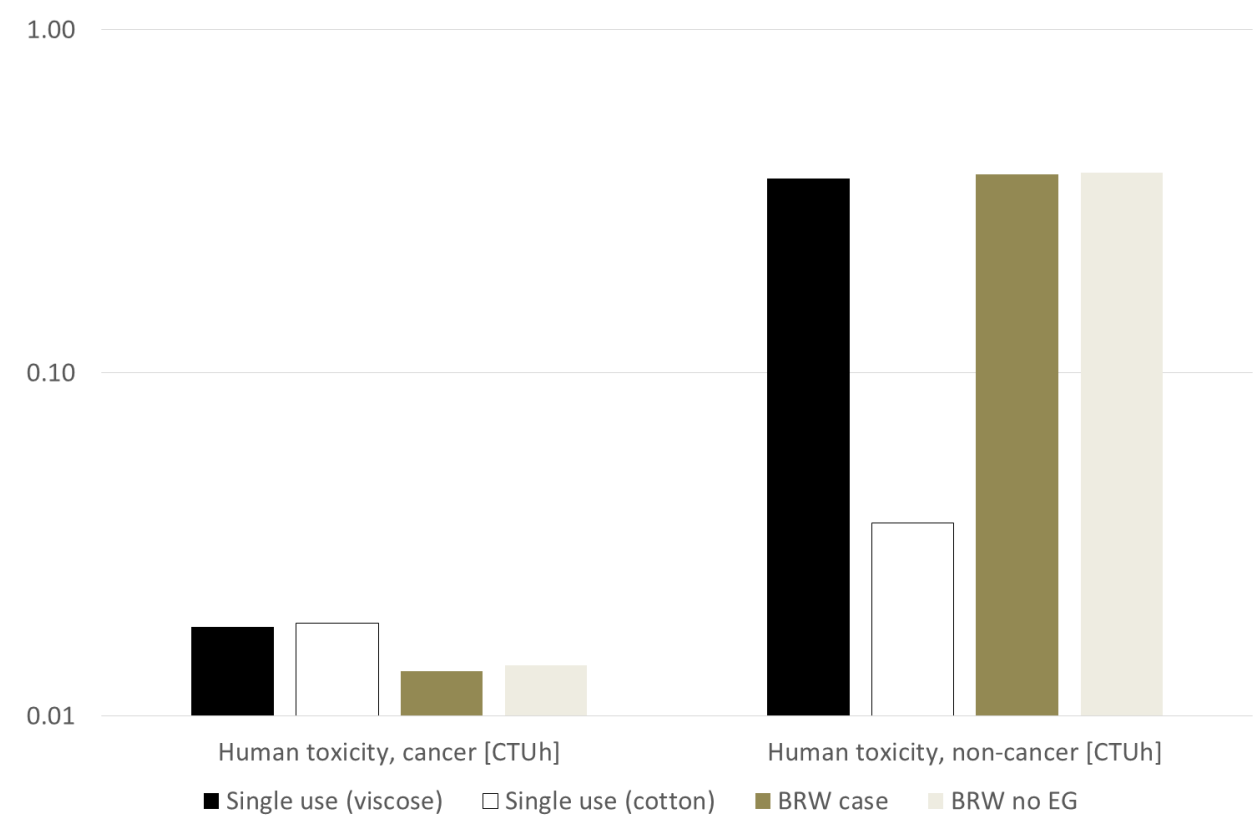


figure 14 Human toxicity for alternative scenarios

An exception to the patterns revealed so far in this report is the water use impact indicator as shown in figure 15. Here the massive consumption of water associated with the irrigation of cotton agriculture, combined with the water scarcity of the country’s in which cotton is grown, is shown in the relative underperformance of the single use of cotton. On the other hand, as it is with the indicators previously shown in this report, recycling mixed fabrics to viscose and polyester is apparently not favoured over the use of viscose from new forest materials.

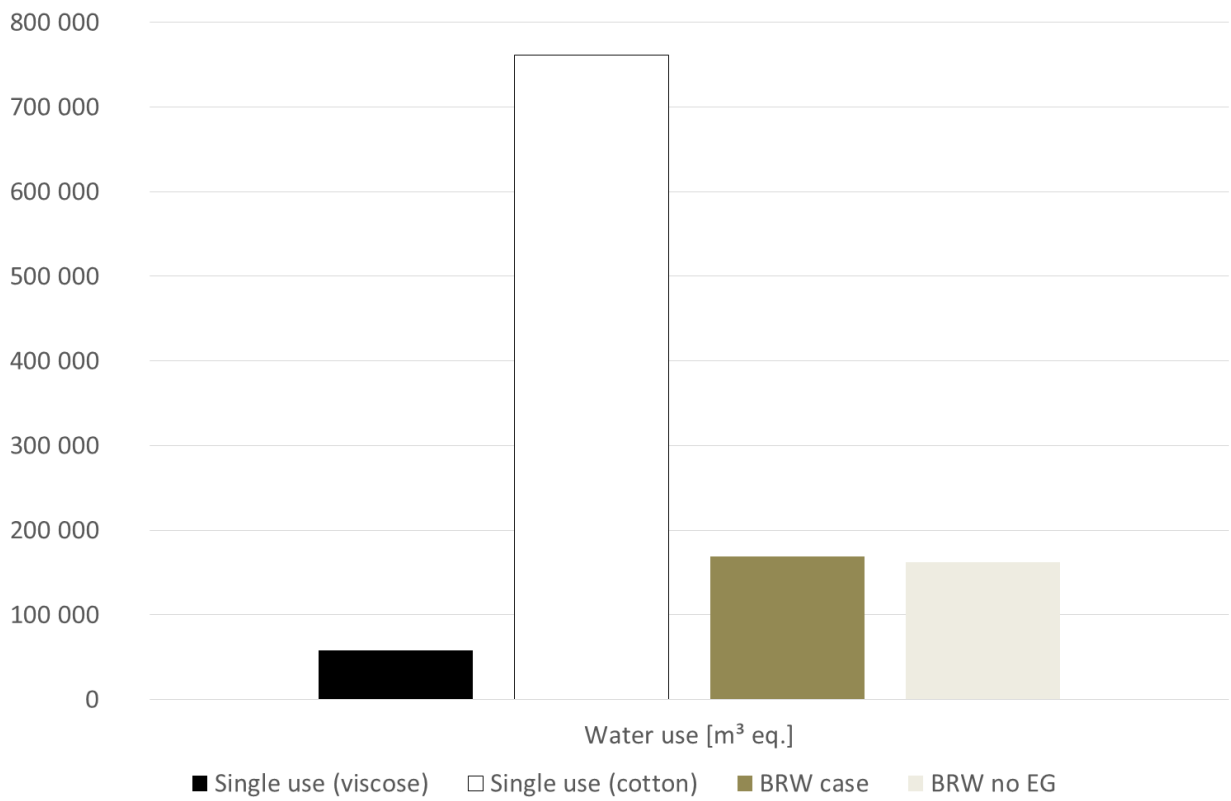


figure 15 Water use impacts for alternative scenarios

## 5. Conclusions

This report is the first public assessment of the Blend Re:wind process using the method of LCA. It compares this recycling process for mixed (polycotton) textile waste (with and without recovery of ethylene glycol) with two single use scenarios. The latter are based on conventional waste handling of textile waste (incineration with energy recovery) and the production of virgin cotton or viscose (the two scenarios) and virgin polyester. The indicators considered included water and energy use, climate changing, acidifying and eutrophying emissions, and toxic emissions with the potential to cause human health and ecological effects.

Half the indicators assessed here favour the recycling scenarios over one or both of the single use scenarios. In most cases the results for the different scenarios are of the same order of magnitude. Since the only publically available data about the Blend Re:wind is based on laboratory scale experiments, it is exciting that this recycling process performs relatively well in comparison to established, industrial-scale production processes for cotton, viscose and polyester. One part of the process which has a considerable influence on many of the indicators is the process for conversion of recovered cotton to viscose fibers. Literature data suggests that the range in the environmental performance of viscose production facilities spans the potential improvement associated with the use of other cellulosic fibers, so selection of the fiber and selection of an integrated production facility are key matters.

Improved results due to economies of scale are to be expected in the future. While some energy consuming processes are constrained by fundamental principles of chemical engineering (e.g. the heat required for distillation of the water/glycol mixture is related to the enthalpy of vaporisation for the chemicals) and therefore not expected to yield significant improvements on scale-up, there are other aspects which are ripe for optimisation. The authors would direct the attention of process developers to the membrane process for separation of sodium hydroxide from the products of polyester hydrolysis as an example of this potential. As stated earlier, developers should also carefully select the cellulosic fiber production facility, which can have a major and positive impact on most of the indicators considered here. We suggest to policy makers that the development of the Blend Re:wind process is worth supporting. We suggest future LCA work should consider additional scenarios, sensitivity analyses and new life cycle inventory data as it becomes available.

# references

- Alvarez-Gaitan JP, Peters GM, Rowley HV, Moore S, Short MD (2013) A hybrid life cycle assessment of water treatment chemicals: an Australian experience. *International Journal of Life Cycle Assessment*, 18:1291-1301.
- Arvidsson R, Tillman A-M, Sandén BA, Janssen M, Nordelöf A, Kushnir D, Molander S. (2017) Environmental assessment of emerging technologies – recommendations for prospective LCA. *Journal of Industrial Ecology*, 22(6)1286-1294.
- Austin HP, Allen MD, Donohoe BS, Rorrer NA, Kearns FL, Silveira RL, Pollard BC, Dominick G, Duman R, El Omari K, Mykhaylyk V, Wagner A, Michener WE, Amore A, Skaf MS, Crowley MF, Thorne AW, Johnson CW, Woodcock HL, McGeehan JE, Beckham GT (2018) Characterization and engineering of a plastic-degrading aromatic polyesterase. *Proceedings of the National Academy of Sciences*, 115 (19) E4350-E4357
- Baumann H, Tillman A-M (2004) The Hitch-hikers Guide to Life Cycle Assessment. ISBN: 978-91-44-02364. Studentlitteratur.
- Beck T, Bos, U, Wittstock B, Baitz M, Fischer M, Sedlbauer K (2010) LANCA Land Use Indicator Value Calculation in Life Cycle Assessment - Method Report. ISBN: 978-3-8396-0170-9. Fraunhofer Verlag.
- Björquist S (2017) Separation for regeneration - Chemical recycling of cotton and polyester textiles. Department of Chemistry and Chemical Engineering. Masters thesis. [publications.lib.chalmers.se/records/fulltext/249870/249870.pdf](https://publications.lib.chalmers.se/records/fulltext/249870/249870.pdf)
- Boulay A-M, Bare J, Benini L, Berger M, Lathuillière MJ, Manzardo A, Margni M, Motoshita M, Núñez M, Pastor AV, Ridoutt B, Oki T, Worbe S, Pfister S (2017) "The WULCA consensus characterization model for water scarcity footprints: assessing impacts of water consumption based on available water remaining (AWARE)." *The International Journal of Life Cycle Assessment*: 23(2)368-378.
- Brismar A (2014) En studie om förutsättningar för pilot, begränsad och fullskalig drift av Re:newcells anläggning i Vänersborg [A study of the prerequisites for pilot, limited and full-scale operation of Re:newcells installation in Vänersborg]. Greenstrategy, Stockholm. (accessed December 2018 at [www.greenstrategy.se](http://www.greenstrategy.se))
- Butyrskaya EV, Belyakova NV, Rozhkova MV, Nechaeva LS (2012) Extraction of ethylene glycol from aqueous salt solutions. *Russian Journal of Physical Chemistry*, 86(11)1721-1725
- Choe EE, Son EJ, Lee BS, Jeong HS, Shin HC, Choi JS (2005) NF process for the recovery of caustic soda and concentration of disodium terephthalate from alkaline wastewater from polyester fabrics. *Desalination* 186:29-37
- Cotton Inc (2018) Interesting and educational facts about cotton. <https://www.cottoninc.com/about-cotton/pressroom/did-you-know/> accessed November 2018.
- Dougan L, Genchev GZ, Lu H, Fernandez JM (2011) Probing osmolyte participation in the unfolding transition state of a protein. *Proceedings of the National Academy of Sciences*, 108:51
- Dow (2018) Water Application Value Engine (WAVE). [water.custhelp.com/app/software/wave](http://water.custhelp.com/app/software/wave). Accessed October 2018.
- European Commission, 2012. International Reference Life Cycle Data System (ILCD) Handbook. Available at: <http://eplca.jrc.ec.europa.eu/uploads/JRC-Reference-Report-ILCD-Handbook-Towards-more-sustainable-production-and-consumption-for-a-resource-efficient-Europe.pdf>. Accessed December 2017.
- Frischknecht R, Büsser Knöpfel S, 2013. Swiss Eco-Factors 2013 according to the Ecological Scarcity Method. Methodological fundamentals and their application in Switzerland. Environmental studies no. 1330., Bern.



- Guinée JB, Gorée M, Heijungs R, Huppes G, Kleijn R, Koning A, et al., 2002. Handbook on life cycle assessment. Operational guide to the ISO standards. Part III: Scientific background. Kluwer Academic Publishers, ISBN 1-4020-0228-9, Dordrecht, 692 pp.
- Guo R, Hu C, Li B, Jiang Z (2006) Pervaporation separation of ethylene glycol/water mixtures through surface crosslinked PVA membranes: Coupling effect and separation performance analysis. *Journal of Membrane Science* 289:191–198
- Huang RYM, Shao P, Feng X, Anderson WA (2002) Separation of Ethylene Glycol-Water Mixtures Using Sulfonated Poly(ether ether ketone) Pervaporation Membranes: Membrane Relaxation and Separation Performance Analysis. *Ind. Eng. Chem. Res.* 41:2957–2965
- IPCC, 2013. Climate change 2013: the physical science basis. Working group I contribution to the fifth assessment report of the Intergovernmental Panel on Climate Change [Stocker TF, Qin D, Plattner G-K, Tignor M, Allen SK, Boschung J, et al. (eds)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. Available at: <http://www.ipcc.ch/report/ar5/wg1/> (Accessed November 2015)
- ISO, 2006a. 14040: Environmental management – life cycle assessment – requirements and guidelines. International Organization for Standardization.
- ISO, 2006b. 14044: Environmental management – life cycle assessment – principles and framework. International Organization for Standardization.
- Jehle W, Staneff Th, Wagner B, Steinwandel J (1995) Separation of glycol and water from coolant liquids by evaporation, reverse osmosis and pervaporation. *Journal of Membrane Science*, 102:9–19
- Marcus Y (1988) Ionic radii in aqueous solutions. *Chemical Reviews*, 88:1475–1498
- Nik OG, Moheb A, Mohammadi T (2006) Separation of Ethylene Glycol/Water Mixtures using NaA Zeolite Membranes. *Chemical Engineering Technology*, 29(11)1340–1346
- Orecki A, Tomaszewska M, Karakulski K, Morawski AW (2006) Separation of ethylene glycol from model wastewater by nanofiltration. *Desalination*, 200:358–360
- Östlund Å, Wedin H, Bolin L, Berlin J, Jönsson C, et al., 2015. Textilåtervinning: Tekniska möjligheter och utmaningar. Naturvårdsverket rapport 6685 (Accessed November 2018).
- Palme A, Peterson A, de la Motte H, Theliander H, Brelid H (2017) Development of an efficient route for combined recycling of PET and cotton from mixed fabrics. *Textiles and Clothing Sustainability*, 3:4
- Peters GM, Rowley HV (2009) Environmental comparison of biosolids management systems using life cycle assessment. *Environmental Science and Technology*, 43 (8), 2674–2679
- Pfister S, Bayer P, Koehler A, Hellweg S, 2011. Environmental impacts of water use in global crop production: hotspots and trade-offs with land use. *Environmental Science and Technology* 45, 5761–5768.
- Roos S, 2012. Livscykelanalys av Tencelfiber (LCA for Tencel fibers). A Mistra Future Fashion report. <http://mistrafuturefashion.com/wp-content/uploads/2015/12/D2.2a-Rapport-Livscykelanalys-Tencel-Swerea-IVF.pdf>. Accessed December 2018
- Roos S, Sandin G, Zamani B, Peters G, 2015. Environmental assessment of Swedish fashion consumption: Five garments – sustainable futures. A Mistra Future Fashion report. <http://mistrafuturefashion.com/life-cycle-assessment-gives-new-understanding-of-fashions-environmental-impact/>. Accessed October 2017.
- Rosenbaum RK, Bachmann TM, Gold LS, Huijbregts MAJ, Jolliet O, Juraske R, et al., 2008. USEtox – the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. *International Journal of Life Cycle Assessment*, 13(7)532–546.
- Sandin G., Roos S., and Johansson M. (2018) Environmental impact of textile fibers – what we know and what we don't know. "Fiber Bible" part 2. D2.1.2.1 Draft report for Mistra Future Fashion. Stockholm

Sandin G, Peters GM, 2018. Environmental impact of textile reuse and recycling – a review. *Journal of Cleaner Production* 184:353-365.

SCB (2018) The future population of Sweden 2018-2070, Demographic reports 2018:1. Statistiska Centralbyrån. [Statistics Sweden].

Schlumberger (2016) *PUREMEG Monoethylene glycol reclamation and regeneration unit*. 16-PRS-180186. [www.slb.com/services/processing-separation/gas-treatment/puremeg-meg-reclamation-regeneration.aspx](http://www.slb.com/services/processing-separation/gas-treatment/puremeg-meg-reclamation-regeneration.aspx) accessed November 2018.

Seppälä J, Posch M, Johansson M, Hettelingh J-P, 2006. Country-dependent characterisation factors for acidification and terrestrial eutrophication based on accumulated exceedance as an impact category indicator. *The International Journal of Life Cycle Assessment* 11(6), 403-416.

Shen L., Patel MK (2010) Life cycle assessment of man-made cellulose fibers. *Lenzinger Berichte* 88, 1-59

Sidisky LM, Stenerson KK, Betz WR, Keeler MJ, McCoy M, Brown J (2010) *The Extraction of Glycols From Water Using ENVI-Carb™ Plus Solid Phase Extraction Cartridges*. Sigma Aldrich document T410135. <https://www.sigmaaldrich.com/content/dam/sigma-aldrich/docs/Supelco/Posters/1/t410135h.pdf> accessed September 2018.

Spathas T (2018) The Environmental Performance of High Value Recycling for the Fashion Industry. Masters Thesis. Chalmers University of Technology.

Struijs J, Beusen A, van Jaarsveld H, Huijbregts MAJ, 2009. Aquatic eutrophication. In: Goedkoop M, Heijungs R, Huijbregts MAJ, De Schryver A, Struijs J, van Zelm R (eds.) *ReCiPe 2008 A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level*. Report I: Characterisation factors, first edition.

Zamani B, Svanström M, Peters G, Rydberg T (2015) A carbon footprint of textile recycling: a case study in Sweden. *Journal of Industrial Ecology*, 19(4) 676-687

Zamani B, Sandin G, Peters G. (2017) Life cycle assessment of clothing libraries: can collaborative consumption reduce the environmental impact of fast fashion? *Journal of Cleaner Production*, 162: 1368-1375.

# appendix - Description of impact categories

Descriptions about the included impact categories shown below are taken from Roos et al. (2015), with some minor modifications.

## **Water use impacts**

Freshwater resources, particularly surface water flows in lakes and rivers, and groundwater flows and stocks (which include deep aquifers with slow recharge rates) are increasingly under stress due to human industrial and agricultural interventions. Excessive consumption of water by for example cotton production can have consequences, not only for other human users of water in the local area, who might have put it to use in food production, but also downstream users and the environment. Environmental impacts of excess water use include the destruction of wetland ecosystems, riparian forests and the extinction of associated animals. Additionally, the reduction in river flows reduces the resilience of river systems to nutrient and other pollutant discharges.

Water use is considered in this LCA using the AWARE method (Boulay et al, 2018). This is essentially a midpoint indicator system, in which the use of water in a catchment is adjusted by a factor between 0.1 and 100 that reflects the scarcity of water in the location of use. We used the method as implemented in the Gabi software by Thinkstep. Thinkstep has implemented the method with three alternative assumptions about water use in uncertain locations: a high characterisation factor (56.9 m<sup>3</sup>-equivalent), a low factor (1.73 m<sup>3</sup>-equivalent) and a midrange factor for OECD and "BRIC" (Brazil, Russia, India, China) situations (34.4 m<sup>3</sup>-equivalent). The location of the BRW facility is not known but given the Scandinavian focus of this work, we selected the low factor implementation of the AWARE method. This was motivated by the fact that the national factor for Sweden is 4.0, which is biased by population distribution, while most of the Swedish countryside has a factor between 0.5 and 1.0. So the value implemented is somewhere between urban and rural expectations in Sweden.

## **Climate change**

Climate change refers to the consequences of increased average temperatures of the earth's atmosphere and oceans. This increase is mainly because of emissions of greenhouse gases such as carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>), nitrous oxide (N<sub>2</sub>O) and chlorofluorocarbons (CFCs) from anthropogenic sources such as the combustion of fossil fuels and deforestation (IPCC 2013). For characterising climate impact, in this report we used the Global Warming Potential (GWP) with a 100 year perspective (GWP100) expressed in kg CO<sub>2</sub> equivalents (IPCC 2013), and assumed that biogenic CO<sub>2</sub> emissions are climate neutral. The latter assumption presumes that within relevant spatial system boundaries (e.g. at a landscape or national level) or within a reasonable time horizon (e.g. within one rotation period: the time period from harvest to harvest), the forestry or agriculture that generates the extracted biomass is carbon neutral. This means that the land management practices ensure that as much carbon is sequestered (above and below ground) as is harvested. In other words, the land is sustainably used with regard to carbon extraction.

## **Acidification**

Precipitation (rain, snow, fog, etc.) deposit acidifying substances from anthropogenic sources (e.g. sulphur dioxide (SO<sub>2</sub>) and nitrogen oxides (NO<sub>x</sub>) released in combustion) to terrestrial and

aquatic ecosystems which may increase pH levels (the concentration of hydrogen ions, H<sup>+</sup>). This may damage freshwater and coastal ecosystems and soils, with consequences such as forest decline, increased fish mortality and damages to buildings (Guinée et al., 2002). Also, heavy metals released due to increased pH levels can damage freshwater resources. For characterising acidification impact, we used the accumulated exceedance method developed by Seppälä et al. (2006), with characterisation factors expressed as mole H<sup>+</sup> equivalents.

### **Freshwater eutrophication**

Nutrients like phosphorus (P) or nitrogen (N) released to freshwater systems may cause increased biological productivity, such as production of planktonic algae. The algae sink to the bottom and are broken down with consumption of oxygen in the bottom layers, causing a dead environment and (among others) increased fish mortality. The most significant sources of nutrient enrichment are the agricultural use of fertilizers, the emissions of nitrogen oxides from combustion and wastewater from households and industry. For characterising freshwater eutrophication impact, we used the EUTREND model method developed by Struijs et al. (2009), with characterisation factors expressed as kg P equivalents.

### **Toxicity**

The toxicity has been evaluated with the LCA method USEtox (Rosenbaum et al. 2008), which is the recommended method by European Commission (2012). USEtox calculates characterization factors for human toxicity and freshwater ecotoxicity at midpoint level. USEtox uses the unit CTU (Comparative Toxic Unit) which is an indirect measure of the number of cases per year caused by toxic effects.

The ILCD handbook (European Commission 2012) recommends that the LCA practitioner should complement the methods with missing characterisation factors if they can have impact on the results. This can be done for processes that are modelled within a project but it is practically impossible to compensate for missing data in database data.

The characterization factor for human toxicity impacts (human toxicity potential) is expressed in comparative toxic units (CTUh), and is the estimated increase in morbidity in the total human population, per unit mass of a chemical emitted. The result is calculated as [CTUh per kg emitted] = [disease cases per kg emitted]. All cases of non-mortal human toxicity impacts, which do not lead to death but to disability and illness, are weighted against their relative severity compared to death.

The characterization factor for freshwater ecotoxicity impacts (ecotoxicity potential) is expressed in comparative toxic units (CTUe), and is an estimate of the potentially affected fraction of species (PAF) integrated over time and volume, per unit mass of a chemical emitted. The result is calculated as [CTUe per kg emitted] = [PAF × m<sup>3</sup> × day per kg emitted].

One CTUe thus equals one cubic meter of freshwater where the species in the ecosystem are exposed daily to a concentration above their no-observed effect concentration (NOEC). An environmental concentration is considered to present an acceptable risk if not more than 5% of all species is exposed above their NOEC.



Mistra Future Fashion is a research program that focuses on how to turn today's fashion industry and consumer habits toward sustainable fashion and behavior. Guided by the principles of the circular economy model, the program operates cross disciplinary and involves 60+ partners from the fashion ecosystem. Its unique system perspective combines new methods for design, production, use and recycling with relevant aspects such as new business models, policies, consumer science, life-cycle-assessments, system analysis, chemistry, engineering etc.

MISTRA is the initiator and primary funder covering the years 2011-2019. It is hosted by RISE Research Institutes of Sweden in collaboration with 15 research partners.

FUNDED BY



The Swedish Foundation for  
Strategic Environmental Research

