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Propelling textile waste to ascend the ladder of sustainability: EOL study on probing environmental parity in technical textiles

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A B S T R A C T

The textile sector is growing, so does the technical aspects of it. This has resulted in more chemical consumption. Recently, technical textiles, with attributional substances received attention due to sustainability factor in terms of their raw material production and manufacturing. Studies are present using life cycle assessment (LCA) results to justify the environmental preference of technical textiles over conventional textiles by environmental parity method. Technical textiles like antibacterial ones are expected for less washing due to low prevalence of odor-causing germs, therefore pose lower environmental impacts than conventional textiles in a long run. At the end-of-life (EOL), waste generated from technical and conventional textiles, are treated as the same - municipal solid waste (MSW), whether they go for landfill or incineration. In reality, environmental impacts of technical and conventional textiles waste cannot be the same regardless of their differences in phases like raw material, production, use and especially EOL phase. LCA "gate-to-grave" approach was employed to study two technical textiles with the same weight but different functionalities, one is flame retardant (FR) treated wool and the other is silver nanoparticles (AgNPs) treated polyester. They are scrutinized in order to have better understanding of environmental parity, especially in their use phase and at the EOL phase. Ten-midpoint categories were used to analyze the environmental impacts during the use phase and EOL phase of the two technical textiles. Results indicate that in use phase, life cycle impact of technical textiles is upfront and alters with the change in number of washes, the types of applied attributional substances and their release rates. At EOL phase, it was found that there is no correlation between the two types of technical textiles in terms of environmental impacts. They are nonreciprocal to MSW or even conventional textile waste.

1. Introduction

Textile sector is considered as one of the most polluting industries (Nørup et al., 2018) and household textiles and clothing are one of the most polluting products in EU alone (Beton et al., 2014). Manufacturing of any textile product requires huge consumption of resources, including energy, water, nutrients and chemicals, which eventually connects to major environmental costs (Yasin et al., 2016c). Textile manufacturing contributes 10% to the world's carbon emissions (Bell et al., 2018). And, textile industry is flourishing by having a remarkable increase during the last decade. According to EURATEX, in 2017 the European textile and clothing industry portraits a €181 billion turnover and €4.9 billion of investments (EURATEX, 2017). Given that, exports of technical textiles outside EU were increased 39% in 2017, with 20% purchases were from the US. Moreover, imports of technical textiles to EU reached 32.6%, or a total of €11.1 billion, following 6% increase in 2017, from which 42.4% was imported from China (EURATEX, 2018). This rise in consumption of textiles is not likely due to an increase in the population, but also rise in prosperity of the population, countries such as China whose affluence commence to approach the levels of American and European (Bartle, 2010; Nørup et al., 2018).

Recently, efforts for the production of sustainable textile products have been realized by the industries and researchers, comprehending the fact that environmental impacts occur at each stage of any textile product system. Fig. 1 shows the environmental interventions of a textile product with the potentials of bringing numerous impacts to the environment Yasin et al., 2016b. Indeed textile industry is one of the most established businesses in the world; yet far from the sustainability, especially the technical

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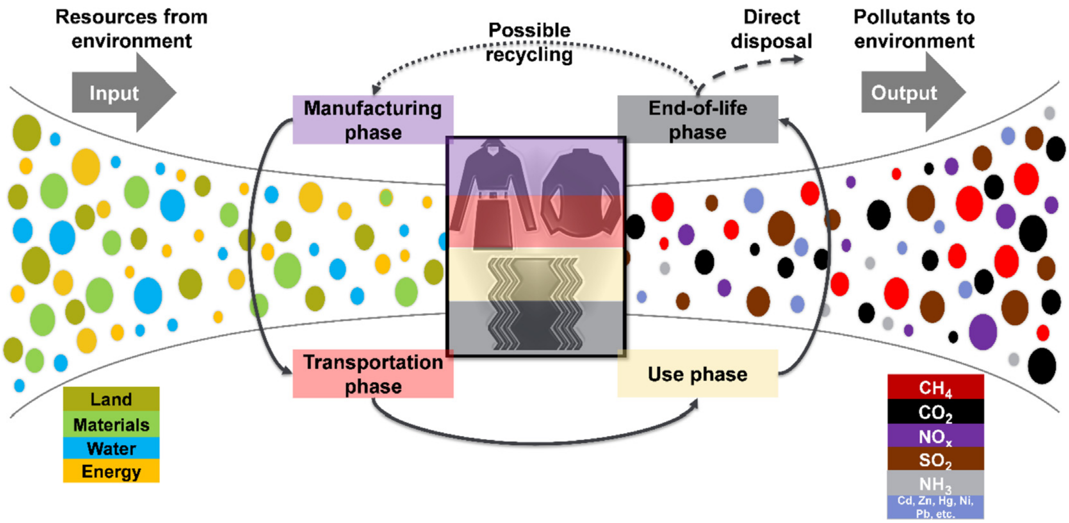

Fig. 1. Life cycle of a textile product system and its environmental interventions at variant phases.

textiles. In respect of waste treatment and disposal of technical textiles, such as flame retardant (FR), water repellent, nano-coated and other functional textiles, have received very little attention. This led to a gap in the literature, for instance, toxicity and associated post disposal environmental issues of technical textile waste (Yasin et al., 2018). Some textile products likely constitute to a smaller fractions of waste, but the potential influence in environmental impacts from production is comparatively high per weight unit (Palm, 2011), and sometimes with minute contribution to environmental emissions could result in the most important route of exposure (Jonkers et al., 2016; Potting and Blok, 1995). Consequently, there is a need for investigating hurdles and possible ways in environmental terms of sorting technical textiles waste at the end-of-life (EOL) and their disposal treatments.

1.1. *Sustainability of textile waste*

A number of European Union legislations are in practice in Europe when it comes to the management, treatment and disposal of waste materials, such as, the Waste Framework Directive (EU, 2008), the Landfill Directive (EU, 2003, 1999) and the Waste Incineration Directive (EU, 2000). These directives are rapidly changing the waste management practices and pushing EU countries to adapt the legislative incentives in more sustainable ways, for instance increase in recovery of valuable resources and reduction of landfill disposal (Wagland et al., 2012). However, a number of socio-economic factors hindered the usage of such directives, for instance, landfill is still practiced with the known fact that it produces leachate and gaseous emissions, including severe pressure of land scarcity (Dong et al., 2018; Muthu, 2015). The only one reason of yet practicing landfills is that it is cheaper than recycling at present (Thompson et al., 2016). In this regard, suggestions have been made to increase landfill taxes to promote recycling and other sustainable ways to reduce or dispose textile wastes (Thompson et al., 2016). Fig. 2 shows general misconceptions about textile wastes at various hierarchal stages and suggested ladder of sustainability.

The total environmental footprints of the production of a virgin material were compared with that of landfills, however, disposal impacts are unfortunately considered as the same for all fiber types. In the study CO_2 equivalent emissions per kilogram virgin fiber production were compared with landfilling and it was found that: 0.4 vs 700 for cotton; 2.8 vs 700 for polyester; and 86 vs 700 for wool (Echeverria et al., 2019). On the other hand, the technologies that used to convert textile waste into energy, that are primarily through incineration, pyrolysis and gasification took an inalienable role in significant reduction of waste mass and volume, energy recovery from unrecyclable materials and complete disinfection (Arena, 2012; Dong et al., 2018). However, incineration approach faces opposition by the public owing to the produced toxic chemicals especially FR MSW, for example polychlorinated dibenzo-p-dioxin and dibenzofurans, therefore the process may cause potential health risk (Duan et al., 2011; Phillips et al., 2014). Incineration of textile wastes being at its intermediate stage, the development and introduction of radically new technologies to be used relatively in a shorter period of time (10–15 years) are not economically feasible (Glushkov et al., 2018). And, the pyrolysis and gasification approaches are yet to be industrialized (Dong et al., 2018). This enforces the textile and fashion companies to not only follow new production patterns and minimize resources usage, such as the use of non-renewable resources, water and chemicals etc. throughout their supply chains, but also to find alternates for disposal phase.

1.2. *Environmental parity! brief overview of textile LCAs*

Since, textiles (and clothing) may last for hundreds of years, they can be used hundreds of times and be inherited for generations (Laitala et al., 2018). In this regard, it is important to evaluate the effects of technical textiles on the impacts of environment and also to realize the potential improvements in a best possible way for their collection, reuse, recycling and disposal during their lifespan (Farrant et al., 2010). The environmental impact of a product is generally assessed via the Life Cycle Assessment (LCA) methodology. LCA has been implemented to a greater extent since the last two decades by many textile and apparel organizations. LCA can supply a detailed guide for the identification of processes, alternates in favor of the environmental impacts and decision makings. Generally, the manufacturing phase of textile products has been reported in the literature, where the environmental impacts are assessed on the fiber and product type and manufacturing processes involved, including mainly the amount of energy and resources used (Muthu, 2015; van der Velden et al., 2014; Yasin et al., 2016c). Similarly, the use phase environmental impacts of a textile product are associated with the number of washes that directly affects the amount of energy being used; fiber type and demographic origin are related to the consumers washing behavior

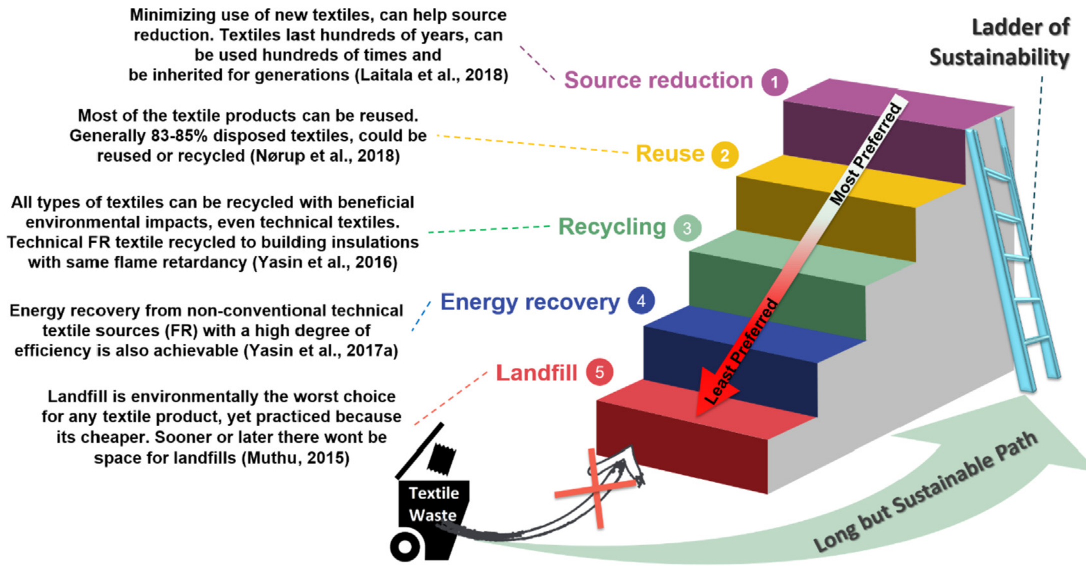

Fig. 2. Schematic waste management hierarchy for technical textiles to ascend the ladder of sustainability.

(Laitala et al., 2018; Yasin et al., 2016c). The manufacturing and use phase of textile products are investigated through both the conventional and non-conventional approaches with various techniques to find the hotspots, including environmental parity.

LCA develops interpretations in accordance with International Standard Organization (ISO), the provisions and the results are usually kept in the boundaries of an organization or a company (ISO-14040, 2006a; ISO-14044, 2006b). Also, with a recent growth in LCA work, most of the results in LCA related to textile products are limited to scientific literature. Accordingly, there are not enough databases of Life Cycle Inventory (LCI) and open source literature to ensemble further research studies. LCA needs to take into account the impact of production delocalization to emerging countries for textile production-consumption chain (Potting and Blok, 1995). Moreover, it becomes essential in a growing environmentally aware consumers and a global demand of green products, to assess all the possible environmental impacts of the textile products, and any related processes and services (Fatarella et al., 2015). In similar context, there is a need of LCA studies focusing on disposal routes to fill in the gap in LCI and the literate, for the betterment of EOL for technical textiles.

Environmental parity is used to compare relative environmental impacts of two products, that at a point, have the same environmental impacts during their life cycle (Hicks and Theis, 2017). However, the EOL phase of textile products is generically limited to landfill and incineration. In addition, nearly all textile products are considered as the same at disposal, regardless of their product or fiber type, functionality or technicality, as there is no specific or standardized way to sort the disposed textiles (Potting and Blok, 1995; Sandin and Peters, 2018; Thompson et al., 2016).

In this study, attention has been paid to one of the growing market products, which is technical textiles and their waste impacts to the environment using LCA “gate-to-grave” methodology. In this context, it is important to understand the environmental parity and disposal impacts of technical textiles along with non-technical textiles at EOL. In this study two type of textile products with different fiber types and technical functionalities were compared, keeping their use phase (number of washes) and EOL scenarios the same. One textile product wool curtain with FR functionality is used in theatres, stadium halls and cinema etc., while, the other polyester curtain with silver nanoparticles (AgNPs) treated for antimicrobial purposes is used in medical centers and hospitals. The concept of environmental impact parity will assist in evaluating the significance of EOL phase of technical textiles, as it has been used to compare the relative environmental impacts of conventional and other unconventional alternative choices (Hicks and Theis, 2017; Jonker et al., 2014).

2. Materials and methods

2.1. Contract to finished textile products

2.1.1. FR treated wool curtain

Unlike domestic textiles, contract textiles are subject to several washing cycles in commercial laundries (and dry cleans). This number of washes of contract textiles are considerable more than domestic household textile. This makes the textile industry to engineer persistent finishes for contract textiles to withstand such laundry demands. Whereas, the use of FRs in contract textiles sector became increasingly important, a corresponding law passed by the UK Parliament stated the end use level of flame retardancy, from high risk areas such as prisons, detention centers, to medium risk sectors like schools, universities, and hospitals, and to low risk areas including hotels and holiday accommodations etc. Indeed, high standard of FR regulations ensures the fire risks, but possesses toxicity risks, as the process systems are not demonstrated properly (Guillaume et al., 2008). Moreover, few studies are available considering the life cycle and EOL of a textile product, and chemical substances along with those technical textiles (Potting and Blok, 1995).

FR technical textiles can be subdivided into two groups. In the first group, textiles are treated with a FR and generally used only at ambient temperatures of as high as 100 °C. The end-use of such textiles are mainly in contract textiles, for instance furnishing textiles and protective clothing (work wear). The second group comprises of textiles that are used for prolonged periods of time at temperatures above 100–150 °C, such as protective clothing for firefighters. These textiles are called high performance, heat and fire resistant textiles, and, mostly comprise different layers of ceramics and glass fibers (Horrocks, 2016).

Wool, a preferable fiber for exhibition halls and indoor usage, due to its comparatively high ignition temperature (570–600 °C), low with burning tendency, low heat of combustion (20.5 kJ/g) and low flame temperature (680 °C), is considered as a FR fiber (Benisek, 1974; Horrocks, 1986). Wool has relatively high limiting oxygen index (LOI) (Martini et al., 2010). Moreover, being non-thermoplastic it makes wool an ideal fiber material providing with good handle and comfort. To fulfil the fire safety regulations, FR textiles made of wool are usually undertaken FR treatment. FR finish for wool requires high levels of heat/flame resistance, and the end uses of FR wool are generally limited to protective clothing, transport and contract textiles (furnishing). For thermal and flame protection, Zirpro finish developed by Benisek in the 1970s (Benisek, 1984) by the International Wool Secretariat (IWS), is well established FR finish for wool fibers (Alongi et al., 2013). The finish based on potassium hexafluorozirconate (K_2ZrF_6) and hexafluorotitanate (K_2TiF_6) chemistry is carried out in dye-bath and produces an intumescent char when exposed to heat. This intumescent char from the zirconium or titanium complex lay char boundary against the penetration of flame and hot gases, and also resistive against mechanical wear (Martini et al., 2010). This makes them most suitable for protective clothing and contract textiles, where molten metal splash hazards are present (Horrocks, 2016). Wool treated with Zirpro FR can last for 50 commercial washes (Martini et al., 2010). Moreover, the FR wool treated with Zirpro has a LOI of 30% (Fung and Hardcastle, 2000).

2.1.2. AgNPs treated polyester curtain

Polyester, a versatile fiber material has been used widely in sports, protective wear and medical textiles. Due to the low surface energy, polyester is prone to microbial growth and odor, therefore it requires functionalization accordingly. Polyester textiles used for medical centers and hospitals as ward screens and bed partition curtain which are used for sanitized atmosphere, are treated with AgNPs. For apparel, AgNPs treated textiles are used not only to reduce odor by microbial inhibition but also for potential laundering reduction (Hicks and Theis, 2017). AgNPs product market value is expected to grow from $682 million in 2013 to $1.8 billion in 2020 (Uskokovic, 2017). In addition, about 30% of 400 tons of produced AgNPs globally are utilized in medical applications (Pourzahedi and Eckelman, 2014; Uskokovic, 2017). There are variety of textile products treated with AgNPs are available in the market for the purpose of reducing infection transmissions. It has been found that textile products such as bedding, table cover, dressing, blind, curtain and clothing, are potential sources of contamination transfer. The production of polyester curtain without raw material acquisition, can consume an amount of 20 MJ/kg energy (electricity) and 25 kg/kg water (Ecoinvent, 2013; Yasin et al., 2016c).

For the synthesis of AgNPs, different chemical and physical methods have been used ranging from chemical reduction to vapor deposition. Among various synthesis processes of AgNPs, biosynthesis (plant extracts) has been reported to gain the most attention, for instance, using leaves of bamboo (Yasin et al., 2013), osmanthus (Ullah et al., 2015), nageia nagi (Liu et al., 2015) and Chinese Holly plant (Ullah et al., 2014). Though, these methods diminish the net life cycle impacts related to toxic reagents, still majority of AgNPs are synthesized using precursor salts in water based nitrate, sodium citrate or borohydride ($NaBH_4$) as the reducing agent, and citrates are used for stabilizing agents (Hicks et al., 2015).

As technical textile, polyester curtain treated with AgNPs is expected for a potential reduction in infection transmission and contamination transfer to serve its purpose in hospitals. In this study, it is used for comparative purpose to be assessed with other technical textiles. Since “gate-to-grave” life cycle analysis model is used, the environmental impacts of use phase and EOL phase, landfill and incineration are evaluated, regardless of their antimicrobial functionality assessment in use phase. This means that the loss of AgNps in treated textile may vary the function (antimicrobial) during the use phase compared to the initial treated textile with higher concentration of AgNPs, this eventually affects the use phase interpretations. In addition, it's difficult to speculate the released AgNPs exposure assessment, because nanoparticles can transform and change properties consequently, moreover, it is still unclear of the specific required amount of Ag for microbial inhibition to occur (Walser et al., 2011). In this study, an initial amount of AgNPs concentration on curtain was considered at 220mg/curtain (Farkas et al., 2011; Geranio et al., 2009; Walser et al., 2011). Correspondingly, the environmental impact of AgNPs treated curtain is taken as a function of the initial AgNPs concentration with consideration of release rate. The post release affects were ignored in this study, for simplicity.

2.2. LCA approach “gate-to-grave” model

LCA is a recognized technique for the assessment of environmental impacts of any products, processes or services (ISO-14040, 2006a). A complete LCA, cradle-to-grave analysis, comprises all stages of a product's lifecycle including extraction of raw material, processing of materials, manufacture, transport, use and maintenance, recycling, and lastly EOL (EU, 2013; ISO-14040, 2006a, p. 140; ISO-14067, 2013, p. 14). Unfortunately, most of the LCA studies assess disposal or EOL stage of textile products assumed equivalent to others (Laitala et al., 2018). For instance landfilling of a technical textile, its end environmental impacts would be different from a non-technical textile even if it is made of the same fiber type, to be considered for the suitability for recycling or biodegradability of the product. Moreover, despite the fact that the existing LCA methodologies and LCI data are incorporated in important environmental factors in today's industrial processes, services and products, information on EOL phase of technical textiles is absent (Potting and Blok, 1995; Yasin et al., 2017). In cases where LCA databases represent a portion of raw materials used in chemical companies, it becomes difficult to obtain the required information. Therefore, estimations by established models are necessary, and gate-to-grave life cycle assessment method has been proved feasible and plausible (Jiménez-González et al., 2000). Such alternates of LCI are taken when factual and literature information are not available, due to manufacturers' legal or intellectual property concerns or delayed responses from the manufactures (Jiménez-González et al., 2000). Similarly, gate-to-grave life cycle approach was used to assess the environmental impacts of the EOL phase of two technical textile products, system boudary used for this study is given in Fig. 3.

In LCA that was based on gate-to-grave model, raw material production and manufacturing of technical textile product were omitted from the model boundaries. Therefore, the above mentioned two end-use products for technical textiles, FR treated wool and AgNPs treated polyester curtain (each 1 kg) were taken as input materials. Transportation from the manufacturing unit to the place of domestic use was provided to the model. The use phase, which includes the number of washes 25 and 100 for wool and polyester respectively, drying, ironing and the amount of detergent needed for each wash was created in the model. The EOL phase was created by separating the textiles into two different disposals, incineration with and without energy recovery and landfill. Transportations to disposals were carried out for the incineration, in order to obtain the heat energy that is produced. Further, study was carried out using EIME software (Bureau Veritas, 2018), where LCI values are calculated using databases such as CODDE, ELCD (JRC-IES, 2012) and Eco-Invent (Ecoinvent, 2013).

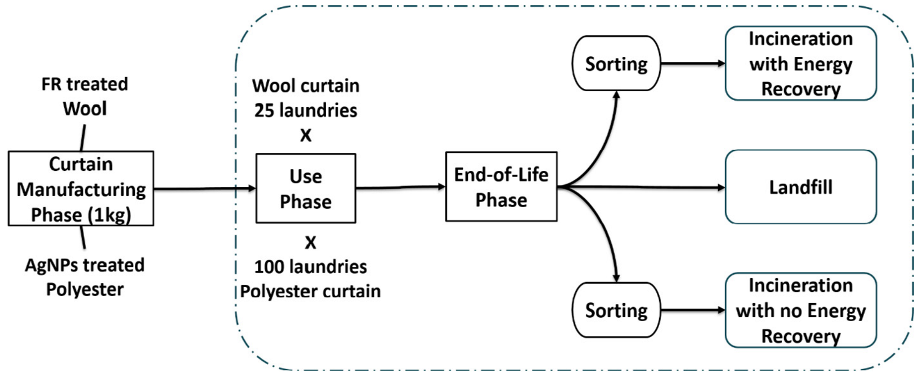

Fig. 3. Schematic illustration of system boundary studied criteria inside the dashed line.

2.2.1. *Laundering consideration for use phase*

The service life of a textile product usually refers to textile lifespan or lifetime, is expressed in number of years, and sometimes by number of washes. However, the effective lifetime is the actual time of textiles in active use, including the number of washes, washing behavior and machine settings. The real data on such entities, is however scarce, as the calculation of the total use period of textile products is difficult as some textile products may inactive, washed less and stored for certain periods of time. The actual service life of a textile product can be estimated by the following method using an example of a textile product that is required to have a functional unit of 10 years of usage in a region: it will need to be manufactured 5 times in case of its service life for two years, or it would need double the time, if the product lasts for a year only (Laitala et al., 2018; Slocinski and Fisher, 2016). Textile products which remain unused do not contribute to functional unit associated to the use phase activities (Laitala et al., 2018). However, such empirical investigation still leads to uncertainties over actual lifetime data, as the retailers and manufacturers may undermine or oversize the items for selling. Moreover, it's difficult to acquire data from consumers for their textile belongings and possessions.

In this study, the laundering impacts of the FR treated wool curtain and AgNPs treated polyester curtain are calculated according to the following settings shown in Table 1.

As aforementioned, nanoparticles for the AnNPs treated polyester tend to change over time and exhibit different properties, in this study their effects including release while drying and ironing are considered as neutral.

2.2.2. *Scenarios building for EOL phase*

EOL technologies for technical textiles particularly textiles with multiple layers of variant technical fibers, such as protective textiles, lack dominating solutions for a wider industrial recycling or are under being developed. Technical textiles, such as protective textiles that ensure the safety of the wearer working in risky conditions have been utilized in a wide range of end-use activities and sectors such as construction and manufacturing, firefighting and law enforcement, oil and gas, health care/medical, mining, military and many more (Fatarella et al., 2015). Recycling of such technical textiles is complex due to the functionality requirements involving various fiber types and chemicals. Currently, landfilling and incineration are practiced for its rapid value recovery and diversification, for instance recovery with or without energy. To cover the EOL scenarios of technical textiles, some assumptions for simplicity are made in order to have value recovery from landfill and incineration. According to the assumptions taken, following EOL scenarios have been defined:

Scenario 1 Landfilling: after finishing their use phase, technical textiles are transported to disposals, to be landfilled within the waste management system.

Scenario 2 Incineration without Recovery: Incineration at disposal of the FR treated wool and AgNPs treated polyester curtain and emissions to be considered.

Scenario 3 Energy Recovery: Heat recovery from incineration to be taken into account for wool and polyester curtains, individually.

2.2.3. *Life cycle impact assessment (LCIA)*

Several potential impacts are brought into the environment from the environmental interventions. The following ten environmental impact categories were assessed at LCIA midpoint levels: Freshwater Ecotoxicity (FWE) [CTUe], Global Warming Potential (GWP) [kg CO_2 eq.], Air Acidification (AA) [AE], Air Toxicity (AT) [m^3], Raw Material Depletion (RMD) [person reserve], Terrestrial Ecotoxicity (TE) [kg 1,4–DB eq.], Photochemical Ozone Creation Potential (POCP) [kg NMVOC eq.], Ozone Depletion Potential (ODP) [kg CFC–11 eq.], Water Depletion (WD) [dm^3], and Water Eutrophication (WE) [kg PO_4^{3} eq.], respectively.

3. Results and discussion

3.1. *LCA values for technical textiles in use phase*

Since the type of textile products are to be made for their specific end uses, so the selection of cleaning methods has to be suitable for the types of products. Machine washing has been taken as the most dominant cleaning method in the world for many years, although people still rely on hand wash in some countries. However, the choice of washing method mainly depends on the textile product type and its functionality, for instance curtains, bedsheets and table covers, and conversely technical textiles are usually

Table 1
Set laundering conditions for technical textiles in use phase.

Types of curtain	No. of washes	Wash temperature, °C	Tumble dry time, min	Ironing condition
FR wool	25	40	40	1300 W, 15 min
AgNPs polyester	100	30	40	1300 W, 15 min

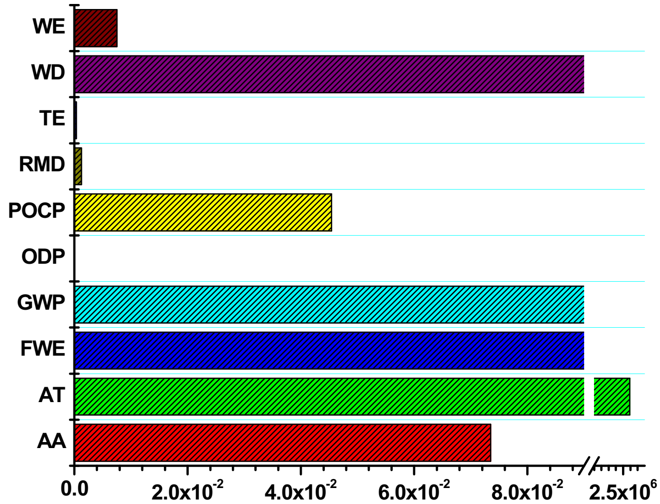

Fig. 4. Use phase LCA values for FR treated wool textile for toxicity impact categories; water eutrophication (WE), water depletion (WD), terrestrial ecotoxicity (TE), raw material depletion (RMD), photochemical ozone creation potential (POCP), ozone depletion potential (ODP), global warming potential (GWP), freshwater ecotoxicity (FWE), air toxicity (AT) and air acidification (AA).

washed by machines. Fig. 4 shows the environmental impacts of FR treated wool curtain as a technical textile product. The impacts in use phase are generally calculated from the energy use while washing, drying and ironing. Wool curtains used in theater halls or elsewhere, are potentially laundered 25 times in its lifespan including washing, drying and ironing. Also, the wash settings are altered depending on the fiber type, wool being resilient fiber is generally washed at different temperature and period of time compared to cotton or polyester. Consequently, the energy consumption and emission values are dependent on these factors. In this study, GWP for the use phase of FR treated wool was taken as 2.16E+01 kg CO_2 eq. The technical textiles as the FR treated wool textiles (curtain, sofa covers) are considered to be washed once a season over a functional unit of ten years usage. It has different energy consumption for other technical textiles, like firefighter uniforms, which are certainly washed more often than curtains.

Fig. 5 shows the environmental impacts of AgNPs treated polyester curtain in terms of consequential energy consumptions in use phase. As a type of technical textiles, AgNPs treated polyester curtains are purposely made to be used for antibacterial activities in hospitals that are opt to be washed more often. Here, it is meant to be washed for 100 times (2.5 times a season) over the period of ten years. However, due to the additional energy consumption while washing the environmental impacts are high. GWP for AgNPs treated polyester curtain was calculated to be 5.61E+01 kg CO_2 eq. It is important to notice that the energy consumption in use phase for any textile product is "case to case" and "country to country" dependent, with an obvious reason of resources consumption and machinery performance including the washing behavior that is different in different countries (Yasin et al., 2016c). Textiles, especially garments made of wool and silk are likely to be washed by dry-cleaning method in contrast of cotton or synthetics garments. Moreover, washing methods for textiles also alter according to gender preferences, for instance men prefer to dry-clean most of their formal or business clothing than women (Laitala et al., 2018, 2017). A textile product formerly discarded quickly if washed less, will relatively have lower environmental impacts in its use phase. Subsequently such textile product will primarily have higher environmental impacts in its next cycle of production phase, due to the increased demand to be placed by a new product (Yasin et al., 2016c).

As aforementioned of the use phase of a textile product, LCA software generally is able to calculate and normalize the environmental impacts based on LCI data for the use of water, and energy consumed by washing, drying and ironing machines and their subsequent emissions. However, for technical textiles data for the release of applied functional chemical materials during the use phase is not available for LCI assessment. For both technical textiles, the release rates of FR species from FR treated wool curtain and AgNPs from AgNPs treated polyester curtain, are identified to be 67% and 2.5%, respectively (Uskokovic, 2017; Walser et al., 2011). The wastewater treatment for these particular functional materials and particulates was not included in the model.

Cumulative comparison of technical textiles for their use phase is rather unclear, and should be considered differently based on their functionalities. For actual comparison, a relative or nomalised use phase environmental impact should be used on the basis of the same weight (e.g. 1 Kg) of the textile products to be assessed, despite their difference in the number of washes undertaken, so does the environmental impact of the use phase. It has already been established that the production phase of technical textiles is different from conventional textiles with same functionality. As a technical textile uses more raw materials and requires additional manufacturing processes to add functionality, hence eventually it will have higher environmental impacts than a conventional textile product. Similarly, in the use phase the use of a technical textile would be higher than a conventional textile due to the leaching of functional substance (e.g. FR or AgNPs) from the technical textile during laundering. Accordingly, the real question is whether the EOL of all technical textiles, which is considered the same at their disposals, actually needs to be distinguished separately.

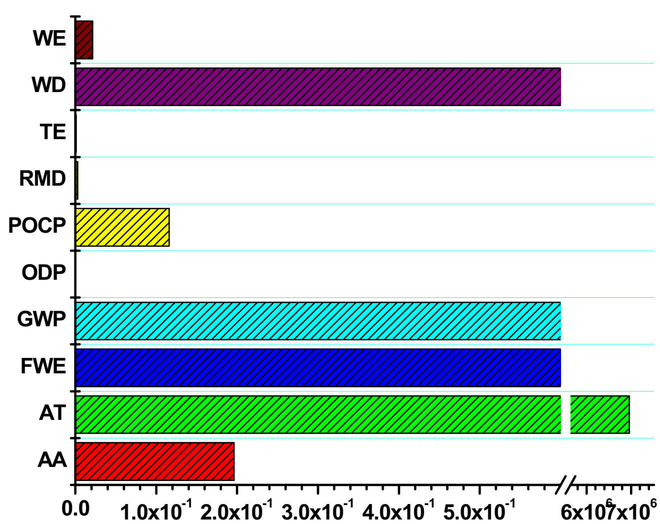

Fig. 5. Use phase LCA values for FR treated wool textile for toxicity impact categories; water eutrophication (WE), water depletion (WD), terrestrial ecotoxicity (TE), raw material depletion (RMD), photochemical ozone creation potential (POCP), ozone depletion potential (ODP), global warming potential (GWP), freshwater ecotoxicity (FWE), air toxicity (AT) and air acidification (AA).

3.2. LCA values for technical textiles in EOL

From Fig. 6, it can be observed that high environmental impacts for landfill scenario 3 are obvious for technical textiles. There is more than 30% of textile waste that is disposed to landfills and about 7% goes for incineration with energy recovery, while about 48% is distributed for reuse within the UK and abroad, which

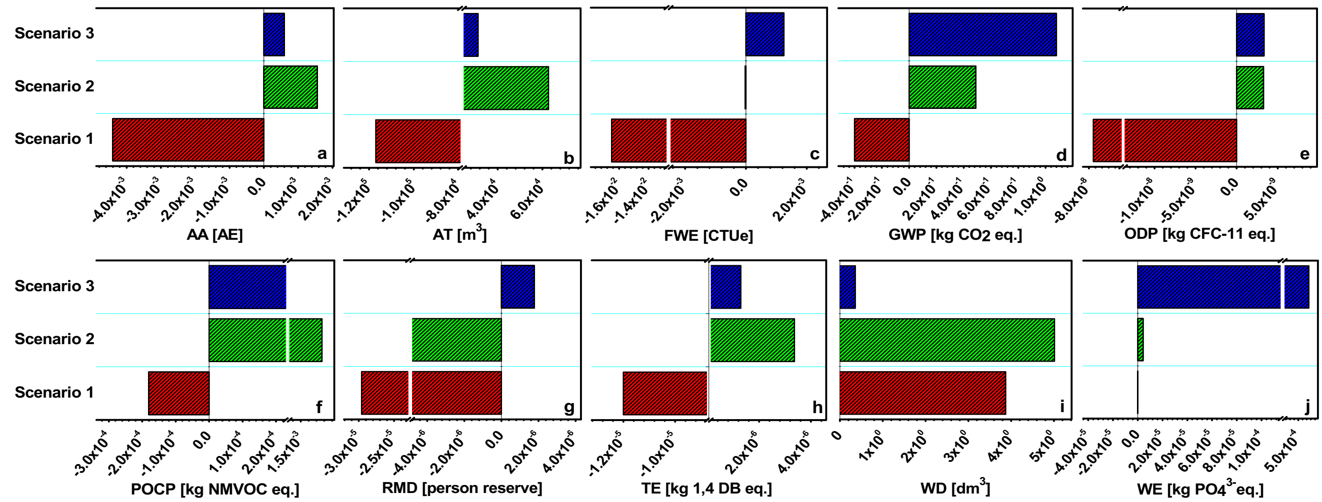

Fig. 6. EOL LCA values for FR treated wool textile with; scenario 1 incineration with energy recovery, scenario 2 incineration without energy recovery and scenario 3 landfill. Calculated for toxicity impact categories; a) air acidification (AA), b) air toxicity (AT), c) freshwater ecotoxicity (FWE), d) global warming potential (GWP), e) ozone depletion potential (ODP), f) photochemical ozone creation potential (POCP), g) raw material depletion (RMD), h) terrestrial ecotoxicity (TE), i) water depletion (WD) and j) water eutrophication (WE).

potentially ends up for landfill and incineration in their second cycle of use (Gracey and Moon, 2012).

In Figs. 6 and 7, it shows that lower environmental impact values for the incineration scenario with or without energy recovery as EOl phase is an environmentally beneficial approach for technical textiles instead of being landfilled. It is clear that incineration scenario with energy recovery is in favor of the indicators responsible for lower impacts at disposal. Having said that, the uncertainties due to the lack of descriptive data from the incineration to the emissions to the air and land and into the water, is rather limited. Therefore incineration should not be considered as an ultimate EOL method for technical textiles. There is a school of thought who does not consider recycling or incineration a sustainable solution of waste management and the two should be combined together as a better solution (Phillips et al., 2014). For instance, recycling of steel, aluminum and glass saves considerable amount of energy, however incineration of plastics is more suitable as of energy inherent in it that could be exploited better through energy recovery (Lea, 1996). Consequently, sometimes recycling is not a better choice than incineration, in case where emissions and energy used for recycling the waste are more than that of generated by incineration (Phillips et al., 2014; Porteous, 2005). It is therefore important to consider lifecycle of different materials within a single product and their residual waste streams along with carbon costs from potential energy recovery parameters.

The incineration of textile waste is ought to be different, as the heat values in terms of percentage from the combustion of waste materials and the carbon content of the combustibles. Here, for instance polyester has high heat value because of its high carbon content and low-to-moderate moisture content (David and Bela,

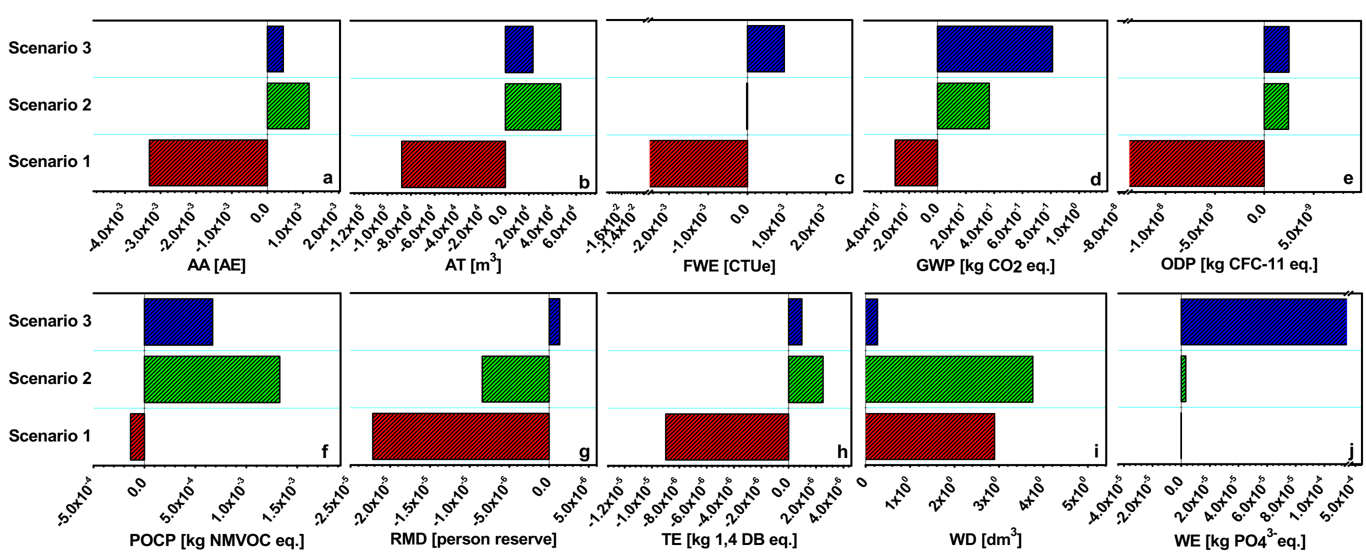

Fig. 7. EOL LCA values for Ag treated polyester textile with; scenario 1 incineration with energy recovery, scenario 2 incineration without energy recovery and scenario 3 landfill. Calculated for toxicity impact categories; a) air acidification (AA), b) air toxicity (AT), c) freshwater ecotoxicity (FWE), d) global warming potential (GWP), e) ozone depletion potential (ODP), f) photochemical ozone creation potential (POCP), g) raw material depletion (RMD), h) terrestrial ecotoxicity (TE), i) water depletion (WD) and j) water eutrophication (WE).

1999). Wool on the other hand, has intermediate-to-low heat values because of its moderate moisture content and intermediate carbon content. The reason of higher CO and CO_2 emissions from FR treated wool is the decomposition of FR components, which starts from an extent where protection of combustible material stops and continuous flaming increases and reacts with the high carbon content of the technical fiber, thus producing higher carbon oxide quantities (Martini et al., 2010).

Air emissions from incineration scenarios with and without energy recovery are generally based on the carbon dioxide and vapors contained in the materials. Carbon dioxide emissions from renewable materials, such as natural fibers are considered to be equivalent to the carbon emissions during their production. Consequently, CO_2 emission from the incineration is generally considered lower or none for renewable materials. In case of incineration of wool fibers, additional emissions like nitrogen NO_x and SO_2 are considered due to nitrogen contents in wool. About 205 g NO_x and 4 g SO_2 are emitted per kg of incinerated wool (Potting and Blok, 1995).

The risk assessment of the use of silver finish or AgNPs on textiles is generally based on cumulative exposure of functional elements (Larsen et al., 2000). Assumption is made that all Ag in the AgNPs treated textile product is to be released within two years, which is shorter compared to the lifespan of the textile product itself. 60% of the Ag from textiles is released after 10 washes with possible discharge from waste water treatment plants that generates risk exposure to sediment organisms. However, 90–95% of the Ag contented in textiles can be removed from waste water treatment plants but still remains activated in sludge system, later to landfills (Hicks et al., 2015).

Considering the EOL fate of nanoparticles applied to textiles, entering the incineration plants has little or no risk of exposure to the environment, as aforementioned (Walser et al., 2011). However, nanomaterials applied to textiles for attributional properties are expected to leach into sediments and groundwater. There is high potential of nanoparticles that are released from different phases of production, manufacturing and usage. There is about 50% of the nanoparticles are reached mostly to the natural and urban soils, 20% and 10% into agricultural soil and the sediments respectively (Caballero-Guzman and Nowack, 2018).

Similarly, the complete combustion of hexafluorozirconate (K_2ZrF_6) and hexafluorotitanate (K_2TiF_6) FR system at high temperatures has not been discovered. The FR tends to generate char from the formation of zirconium or titanium dioxide in the wool fibers from ambient temperature to 350 °C with constant F/Zr ratio. The FR system has been found to create an enhanced intumescent char formation in the condensed phase or flame barrier due to its generated char structure.

The results also indicate that environmental impacts are in the midpoint impact category depending on the life cycle phase of the evaluated textiles. For instance, in use phase, laundering of both the FR treated wool and AgNPs treated polyester curtain contributes mostly to AT, WD, FEW and GWP, in comparison to WD only for their EOL phase. For the impacts related to air environmental category, incineration scenarios with or without energy recovery lead to higher impacts due to the relevant emissions of potentially toxic substances in the air which later causes rise of global temperature, compared to landfill scenarios.

In the landfill scenarios, both Ag treated polyester and FR treated wool have higher GWP, with the increase of the emissions GWP may cause an increased radiation absorption, which is emitted by earth resulting in an increase of natural greenhouse effect (ISO-14067, 2013). Similarly, for WE in landfill scenarios, it covers the impacts of potential macronutrients, such as nitrogen and phosphorus. The enhanced nutrient contents alter with species composition that will effect on the aquatic and terrestrial ecosystems due to imbalanced biomass production, which will further lead to low oxygen levels in aquatic ecosystems (ILCD, 2011).

The processes considered to be important for instance washing, drying and ironing chosen, with 25 and 100 times laundering for FR treated wool and AgNPs treated polyester curtain in the use phase are energy extensive, thus have higher impact values. Depending upon the textile product type and its functionality, washing energy needed may change accordingly. Whereas, it has been established that textile products produce different types of impacts at each lifecycle phase. For instance, the production phases of a textile product use a large agricultural land and are water intensive processes, thus they are associated to TE, and marine and freshwater eutrophication. On the contrary, a considerable environmental impacts are resulted from the use phase, such as human toxicity and freshwater and marine ecotoxicity (EEA, 2014). For the incineration scenarios without energy recovery for both the AgNPs treated polyester and FR treated wool, POCP values are high, which are generally related to emissions attributed to ground level smog, in other words ozone formation. The incineration is linked to air emissions, which cause increased ozone formation from the reaction of CO and volatile organic compounds produced in the presence of NOx in ultraviolet light. Moreover, in Figs. 6 and 7 ODPs which affect the stratospheric ozone layer, are appear negative values for scenario with energy recovery. POCP and ODP have been found risky to human health/ecosystems and plants/crops (Guinée, 2002).

4. General discussion

It is rare to find in literature on FR species migrating from textile products to the surroundings or into the environment (Yasin et al., 2018). However, a large number of research studies shows that FR species can migrate from solid bodies into household dust, such as electronic goods (Schreder, 2017). Adults and children incidentally may ingest the contaminated dust through hand-to-mouth activity (Schreder, 2017). Experiments by researchers have shown that FR species contaminate dust through abrasion of plastic casings and direct migration to dust from the surface of a product (Rauert and Harrad, 2015). In addition, FR species were detected on wipes used for cleaning electronic items at similar high levels presented in the dust (Abbasi et al., 2016).

Wool being natural fiber is biodegradable, means microorganisms can metabolize it the same as other natural/organic wastes, such as wood/paper, yard and food waste. However, some biodegradable wastes are more readily metabolized due to their presence of higher nitrogen and moisture content than others such as grass, food waste, and other green, pulpy yard wastes. Such wastes that have high bioavailability are putrescible. Green wastes, like leaves have intermediate bioavailability, in contrast with cotton, wool and wood that are biodegradable though, but have relatively low bioavailability and are generally considered as non-compostable in solid waste management framework (David and Bela, 1999).

4.1. Incineration of technical textiles and emission calculations

For technical textiles, the incorporation of EOL phase into software for modelling is rather intricate activity and involves many elements to be considered. The module of specific EOL in LCA software for calculating emissions of technical textile fibers (cotton, wool, polyester etc.) is too generic. In LCA, incineration or even landfill is considered the same as municipal solid waste (MSW) for all textile types, regardless of their use and functionality. In addition for incineration with or without energy recovery, the data set presented in LCA is based on the average European waste-to-energy plants for thermal treatment of MSW (Ecoinvent, 2013). According to BREF (BREF, 2006), two-thirds of MSW treated in incineration plant are operated with dry flue gas treatment (FGT) and the other one-third with a wet FGT. Along with that, variant NOx-removal technologies are combined to appliance with different FGT systems in Europe (BREF, 2017, 2006). Also, emissions such as, NOx, HF, CO, NH_3, HCl, VOC, N_2O, SO_2, dioxin, dust and the heavy metals like Cd, Co, As, Cr, Pb and Ni with mean emission values per cubic meter of cleaned flue gas are used for calculation, according to European Commission doctrine Best Available Techniques (BAT) document "Waste Incineration" (BREF, 2006). The components in flue gases like CO_2, H_2O, O_2, and N_2 generated during incineration, contain the majority of the accessible fuel energy as heat (Bosmans et al., 2013). However, emissions of other substances and elements into different residues are generally calculated by transfer coefficient means (BREF, 2017, 2006). Whereas, the treatment of residues such as, bottom ash, that is approximately 220 kg/t of MSW, from which 60% is reused as construction material, after a metal recovery and ageing process. The remaining 40% is disposed into landfill. These generic incineration waste, fly ash and bottom ash require a careful management system to ensure the minimum environmental impacts (Phillips et al., 2014). Moreover, from 42 kg/t of MSW air pollution control residues (boiler ash, slurries and filter cake), 43% is disposed in salt mines and 57% in landfills (BREF, 2018, 2017, 2006). However, the emissions from incineration at first or the last stage (bottom and fly ash) of the process, with or without energy recovery are opt to be different, in terms of distinction between biogenic carbon dioxide (CO_2) emissions and fossil (Yasin et al., 2017), either for technical or non-technical textile waste. Unfortunately the two diverse wastes are considered the same as MSW.

For the incineration of textile waste, it is necessary to consider the energy parameters (Nunes et al., 2018), including the amount of waste percentage being treated for energy recovery as heat or steam, electricity supply in the incineration, and the amount of waste incinerated without energy recovery. The efficiency of energy recovery techniques, delivery of waste convert into grid/network energy is realized as a percentage of the input waste and a lower heating value (LHV) (Beylot et al., 2017). Whereas, the energy production from the incineration of waste is calculated by multiplying it with LHV (in MJ) (Beylot et al., 2017). The LHV does not include latent heat of vaporization of water created during combustion, otherwise its heating value (HHV) will be higher (David and Bela, 1999). Environmental impacts and benefits related to the scenario 1 incineration with energy recovery for different textile fibers are calculated from their relative individual LHV values: 21.2 MJ/kg (34%) for polyester and 23.2 MJ/kg (4%) for wool. By weight, HRR of Zirpro treated wool was taken at 240 kW whereas data such as CO yield (0.065 g/g), CO_2 yield (1.859 g/g) and heat generated from combustion (27.9 MJ/kg) was used in modelling (Martini et al., 2010). HRR of Zirpro treated wool can be taken at 64 kW/m^2 which was found in literate (Fung and Hardcastle, 2000). For an average fiber type, textiles have around 25.1 MJ/kg HHV with a moisture content of 12.7% was used (David and Bela, 1999). And, from the arithmetic mean, for incineration the amount of energy recovered from 1 kg of fiber has been calculated to be 13.2 MJ of thermal energy and 2.5 MJ of electric energy (Schmidt, 2016).

For the scenario 1 incineration with energy recovery of FR wool curtain, it was evaluated from the combustion properties including heat release rate (HRR) and effective heat of combustion (EHC). The credited values were then programmed in LCA for modelization. Here HRR, which is actually a rate at which flame/fire releases energy (Yasin et al., 2016b) and EHC is the measure of the amount of energy (heat) released from the combustion of material that was considered significant. For the incineration scenarios, the AgNPs polyester and FR treated wool with energy recovery is assumed on the heat produced leading to the electricity production, though the emissions in a conventional energy production plant are avoided when renewable materials are incinerated. The energy credit from the incineration of wool is calculated at 15 MJ/kg. For the resultant emission estimations from the incineration of FR treated wool, residual percentage was taken at 25% (Wang et al., 1994). Transfer coefficients express the proportion of each input element that is transferred to the output compartments (gas, liquid, solid) (Astrup, 2015; Beylot et al., 2017; Riber et al., 2008). The mass of any element i transferred to the output compartment o is calculated according to:

$$\boldsymbol{Emission}\ (\boldsymbol{i},\boldsymbol{o}) = \boldsymbol{Mass_{waste}} \times \sum_{\boldsymbol{C}=1}^{\boldsymbol{C_b}} [\boldsymbol{Composition_{waste}}(\boldsymbol{c}) \times \boldsymbol{Compostion}\ (\boldsymbol{c},\boldsymbol{i}) \times \boldsymbol{TC_{combustiables}}(\boldsymbol{i},\boldsymbol{o})]$$

where, Emission (i,o) = mass of element i transferred to compartment o; $\text{Mass}_{\text{waste}}$ = mass of input waste (incinerateables); $\text{Composition}_{\text{waste}}$ (c) = input waste composition in combustible waste category c; Composition (c,i) = composition of waste category c in element i; $\text{TC}_{\text{combustibles}}$ $_{(\text{i,o})}$ = transfer coefficient of element i to compartment o and C_b = number of non-inert combustible waste categories.

It has been reported that there are considerable difficulties in manufacturing and use phase of AgNPs incorporated textiles (Hicks et al., 2015; Hicks and Theis, 2017). In manufacturing phase, silver itself from the manufacturing process of silver nanoparticle textile, was found to be the major contributor of environmental impacts (Pourzahedi and Eckelman, 2015). On the contrary, LCA studies on AgNPs treated textiles are found to neglect the analysis of textile itself while focusing on the impacts of Ag (Hicks et al., 2015). Secondly, in use phase, similar with FR species, it's difficult to assess the Ag release rate from textiles while washing (Walser et al., 2011). Lastly, the EOL of AgNPs treated textiles and its aftereffects at the disposal for instance at landfill is unclear while in incineration (Walser et al., 2011), which is a major research gap. In this study the assumptions were taken that AgNPs were considered being destroyed while combustion, and environmental impacts of AgNPs were neglected.

Although, the focus of this research was the inclusion of chemicals applied to textiles, attributing its technicality evaluated for environmental performance, but FR was rather difficult. For attributional substances, such as FR species for textiles, obtaining LCI data is hindered by the complex manufacturing chains. LCA studies on technical textiles with FRs have mostly neglected analysis of FR species, and instead have focused on the impacts of textile products. The release of FR or any attributional substances for technical textiles in production, use phase or even at EOL, are rarely quantitatively included in the LCI. This makes it difficult to compute the impact assessment and eventually the LCA. Consequently, FR species released during the washing from technical textiles in varying degrees are expected and assumed to enter the wastewater system, and to be treated there.

Apart from the attributional finishing chemicals on technical textiles, another aspect of textile waste needs to take into consideration in LCA studies is the inclusion of textile-related substances. About 10% of known 2400 textile-related substances, such as functional chemicals, dyes and fragrances, are considered potentially risk to human health (Larsen et al., 2000). As aforementioned, EOL environmental impacts of textile waste, even of technical textile wastes, are generally lower compared to other phases of products' lifecycles. Therefore, they are generally excluded from the scope of the screening study. A reader of LCA studies may instinctively assume that emissions of textile-related substances are also included (Roos et al., 2018).

4.2. Disposal labelling of technical textiles

It is also important to understand, that in most of the environmental studies on textiles, collection and/or sorting of textile wastes were fully or partly excluded from the majority of them and some excluded collection and/or sorting as they were assumed to be negligible (Sandin and Peters, 2018). Generally, the textile wastes are directly transported to disposal sites without sorting, such as to landfill and incineration plants, collection and sorting are considered in the case of possible reuse and recycling. The rationale of technical textiles not being sorted from non-technical textiles could be a postulate, that during the use phase, their functionality is decreased with repeated use and cleaning. For instance, FR textiles with a life expectancy of 50 washes as indicated on the label, beyond which the performance may not be certain. To maintain the regular function of that textile product, a new produced one is needed while the older is disposed as non-technical textile. However, this may not be the case in reality. Functionality of a technical textile product sometimes is found to be similar as of new compared to the one completed its life expectancy (after certain number of washes) (Crown and Batcheller, 2016; Yasin et al., 2016a). Restoration of functionality of technical textiles has been reported through additional pretreatment of soiled areas using liquid detergents (Crown and Batcheller, 2016). Additionally the functionality of some technical textiles may deviate at EOL, for instance in cold climates parkas which are cleaned infrequently, may accumulate flammable contaminants with use and it was found difficult or impossible to be removed over time (Crown and Batcheller, 2016; Kerr et al., 2009). Such issues can even hinder general disposals of textile waste, thus highlight the importance of collection and sorting of technical textiles prior disposals. As aforementioned, unfortunately, in most of the environmental studies on textiles, collection and/or sorting of textile waste were fully or partly excluded from the majority of them and some excluded collection and/or sorting were assumed to be negligible (Sandin and Peters, 2018). Considering the EOL of textiles, this inattention has led to a gap in assessing environmental impacts of disposal alternatives.

5. Uncertainty and limitations

5.1. Use phase

It is difficult to use the LCA results with the absolute ability regarding the sustainability of waste incineration, as the results are model dependent and require to consider the localised conditions and implementation (Turconi et al., 2011). The models themselves can have discrepant impacts up to 1400% and thus considerably affects the results that lead to contradictory conclusions (Winkler and Bilitewski, 2007). Moreover, LCA studies without an uncertainty analysis are questionable (Bicer and Dincer, 2018) and may leads to miss interpretation of results. To understand the data variability in a process flow, which is generally in dependent on the established system boundaries, it is essential to adopt uncertainty analysis. Process flow in a phase, for instance laundering in use phase, can highly affect the environmental impacts by choosing LCI data variables that are available in LCA database. Laundering of textiles in LCA use phase with machine wash and hand wash has been considered in many studies (Yasin et al., 2016c) and the results are interpreted according to the respondents in selected geographical locations of the study. Countries where hand washing is dominant over machine washing, like China (Zhang et al., 2015) have shown lower environmental impacts in use phase of t-shirts. Although washing textiles by hands, poses lower environmental impacts compared to machine washing, but hand wash does not always happen in reality. Environmental impacts of use phase launderings are generally calculated on the basis of the number of washes including the quality/quantity of water and detergent used, the amount and sort of energy utilized during washing and the associated set of variable LCI inputs and outputs present in an LCA database.

To understand the uncertainty in use phase of a technical textile, three laundering scenarios were considered for 1 kg textile curtain (without any attributions): with scenario 1 machine washing (washed, tumble dried and ironed), scenario 2 dry-cleaning and scenario 3 hand washing. Figs. 8 and 9 show the use phase LCA values of 1 kg textile washed 25 and 100 times respectively. Scenario 2, which is washed with specific chemicals including tetrachloroethylene, shows higher environmental impacts compared to scenario 1. Scenario 3 in both 25 and 100 washes with hand shows higher environmental impacts in nearly all impact categories, except OPD, RMD and TE. In scenario 3, even ironing is considered, as curtains are generally ironed and hanged, but it shows lower impacts compared to scenario 1 where all laundering processes machine washing, tumble drying and ironing were included. It can be argued that hand washing of 1 kg textile curtain requires no energy for washing machine and drying, but it requires more water and detergent. Also, hand washing of light weight textile product (e.g. shirt) is anticipated to have lower environmental impacts in use phase. It will certainly lead to more usage of resources in the end compared to heavy weight textiles (e.g. curtain) since, curtains are washed much less than shirts. The efficiency of washing machines can be improved for more water efficient or more energy-intensive, considering textile abrasion and hygiene as important indicators (Bao et al., 2017). However, LCI information on waste water from machines are rather difficult to obtain.

Due to the inconsistent methodologies chosen for LCA of chemical products, LCI data is generally not available for most of the chemicals (Van Lieshout et al., 2015). Table 2 shows the LCI input and output demands of resources and emissions of 1 kg textile launderings. The demand of resources such as primary energy of machine washing is certainly higher than hand wash, but hand wash has higher emissions due to high water and washing agent usage, and the treatments. This poses higher environmental impacts in the end. Thus in general textile products (clothing such as shirts, dress, jeans etc.) are not comparable to home textiles (such as curtains, bedsheets, quilt covers etc.) in their associated environmental impacts.

Despite the fact that release of AgNPs in the aquatic environment is a major concern, it's difficult to assess levels of AgNPs released from consumer products into wastewater systems, as some washing machines are programmed to release AgNps with expectation of effluent or sludge treatment (Farkas et al., 2011).

5.2. EOL phase

There are a few LCA studies on wool textiles with an attempt to assess the environmental impacts throughout their lifecycles, based on “cradle-to-grave” or “cradle-to-cradle” life cycles. However, there are limitations in possible comparison to literature due to lack of published data on “gate-to-grave” LCA studies. Majority of the studies were subjected to the manufacturing phase, allowing the readers to have better understanding of primary stages of the wool product system. Interpretations from these cradle-to-grave or cradle-to-cradle life cycle studies, help in adopting clean methods in prior stages of wool processing. A detailed cradle-to-farm gate study in which the authors compared wool production systems

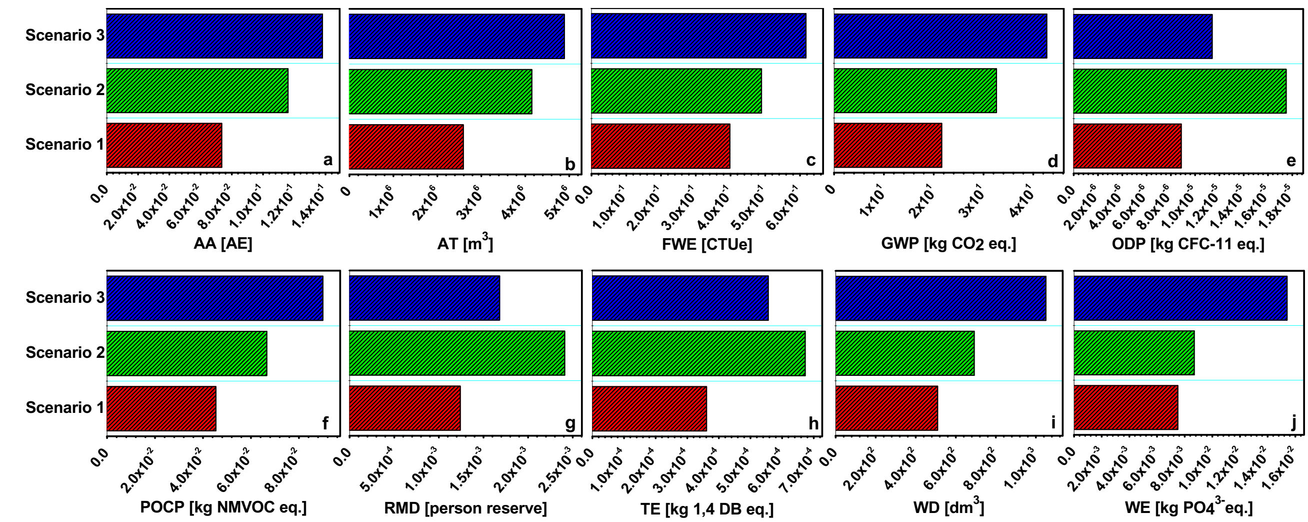

Fig. 8. Use phase LCA values of 1 kg textile washed 25 times under; scenario 1 machine washed (washed, tumble dried and ironed), scenario 2 dry-cleaned and scenario 3 hand washing (hand washed, air dried and ironed). Calculated for toxicity impact categories; a) air acidification (AA), b) air toxicity (AT), c) freshwater ecotoxicity (FWE), d) global warming potential (GWP), e) ozone depletion potential (ODP), f) photochemical ozone creation potential (POCP), g) raw material depletion (RMD), h) terrestrial ecotoxicity (TE), i) water depletion (WD) and j) water eutrophication (WE).

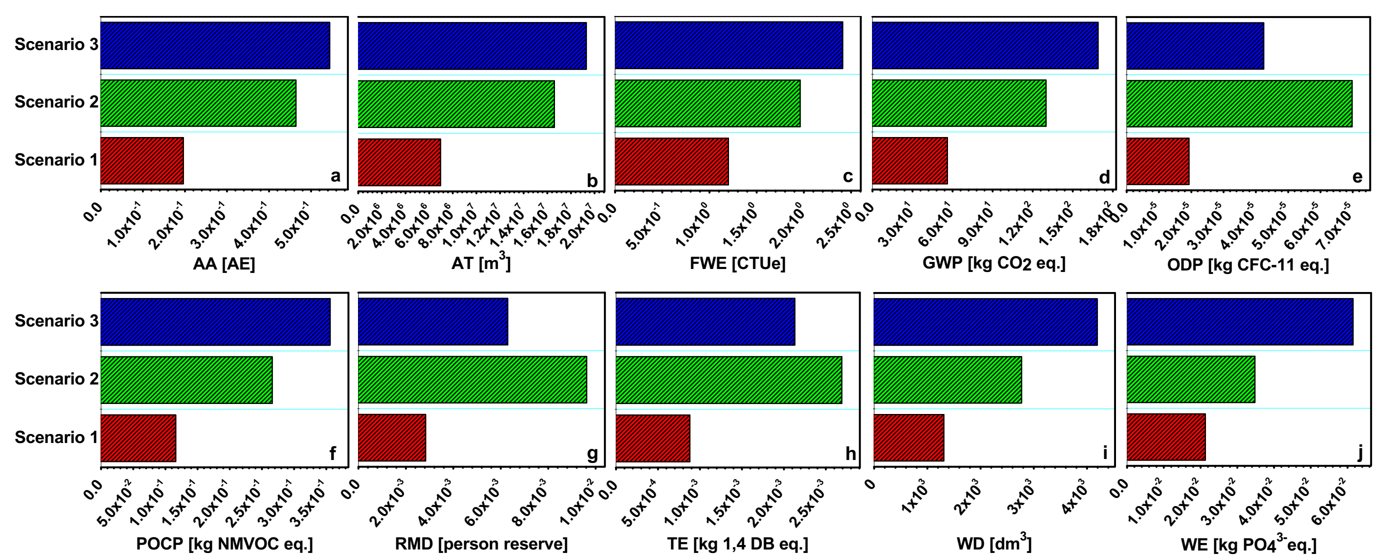

Fig. 9. Use phase LCA values of 1 kg textile washed 100 times under; scenario 1 machine washed (washed, tumble dried and ironed), scenario 2 dry-cleaned and scenario 3 hand washing (hand washed, air dried and ironed). Calculated for toxicity impact categories; a) air acidification (AA), b) air toxicity (AT), c) freshwater ecotoxicity (FWE), d) global warming potential (GWP), e) ozone depletion potential (ODP), f) photochemical ozone creation potential (POCP), g) raw material depletion (RMD), h) terrestrial ecotoxicity (TE), i) water depletion (WD) and j) water eutrophication (WE).

among the three different countries, the United Kingdom, New Zealand and Australia (Wiedemann et al., 2015). In LCA studies, the energy consumption of wool carpet was compared with that of nylon one. In use phase it was found that utilizing 69.19% of the total energy consumption generated 43.61% of the total carbon emissions. While for the production phase nylon carpet was accounted for 68.86% of the total energy consumption and generated 71.49% of the total carbon emission (Sim and Prabhu, 2018). Interestingly, some studies reported that the reduced environmental impacts are allocated to wool, as wool is taken from sheep, and sheep provide other valuable products - milk and meat (Ripoll-Bosch et al., 2013). For the LCA study of textile product system from wool, it is important to consider post production studies of wool as a raw material, as it'll enhance textile products supply chain and fill in the research gap for textile waste management with multi-impact LCA studies.

Moreover, quantification of reuse and recycling of wool products is the key parameter in LCA, in order to prevent or limit the production of new raw material. However, with few data available for such activities and different reuse and recycling practices among different countries, it is difficult to generalize the mode of disposal for wool products (Muthu, 2015). Indeed, benefit of recycling with the lower environmental burdens associated with managing disposal of waste and avoidance of virgin materials production. In previous LCA study, mechanical recycling of technical FR textile products results in reduction of carbon footprint, as manufacturing of virgin materials was avoided (Yasin et al., 2016a). However, it is important to assess more than one impact categories (ISO-14067, 2013), rather than usually assessed single category, which is global warming potential (GWP), reported as greenhouse gas (GHG)

Table 2
Some elementary flows of textile launderings in use phase. Derived from EIME and ELCD Databases (Bureau Veritas, 2018; JRC-IES, 2012).

Environmental Path	Elementary flow name	**Machine washing**	Hand washing
Emission to air	Dinitrogen	3.44e-4 kg/kg	1.16e-3 kg/kg
Emission to air	Nitrogen dioxide	8.91e-4 kg/kg	3.17e-3 kg/kg
Emission to air	Nitrogen monoxide	1.24e-12 kg/kg	8.65e-12 kg/kg
Emission to air	Nitrogen oxides	1.17e-5 kg/kg	1.67e-4 kg/kg
Emission to air	Hydrogen bromide	2.42e-10 kg/kg	1.38e-9 kg/kg
Emission to air	Hydrogen chloride	6.68e-6 kg/kg	1.86e-5 kg/kg
Emission to air	Hydrogen fluoride	3.29e-7 kg/kg	1.02e-6 kg/kg
Emission to air	Hydrogen iodide	2.66e-13 kg/kg	1.52e-12 kg/kg
Emission to air	Hydrogen sulfide	9.06e-7 kg/kg	3.32e-6 kg/kg
Emission to air	Phosphorus pentoxide	3.57e-12 kg/kg	5.10e-12 kg/kg
Emission to air	Waste heat	1.99e+0 MJ/kg	1.18e+1 MJ/kg
Emission in soil	Phosphate (non-agriculture soil)	1.08e-6 kg/kg	3.94e-6 kg/kg
Emission to soil	Sodium (non-agriculture soil)	3.58e-9 kg/kg	2.96e-8 kg/kg
Emission to fresh water	Phosphate	1.55e-7 kg/kg	6.22e-77 kg/kg
Emission to fresh water	Sodium	6.18e-3 kg/kg	2.06e-2 kg/kg
Emission to fresh water	Hydrogen-3	4.61e+0 kBq/kg	4.17e+1 kBq/kg
Emission to fresh water	Hydrogen chloride	1.76e-10 kg/kg	5.99e-10 kg/kg
Emission to fresh water	Hydrogen fluoride	8.59e-12 kg/kg	3.17e-11 kg/kg
Emission to sea water	Sodium	2.20e-7 kg/kg	8.30e-7 kg/kg
Emission to water	Chemically polluted water	1.13e-2 kg/kg	1.62e-2 kg/kg
Emission to water	Hydrogen-3	7.82e-7 kBq/kg	1.12e-6 kBq/kg
Emission to air	Used air	2.55e+0 kg/kg	8.92e+0 kg/kg
Resources from air	Renewable air	1.37e+1 kg/kg	4.62e+1 kg/kg
Resources from air	Primary solar energy	7.72e-3 MJ/kg	5.31e-2 MJ/kg
Resources from air	Primary wind energy	1.67e-2 MJ/kg	6.77e-2 MJ/kg
Resources from ground	Primary geothermal energy	7.74e-4 MJ/kg	3.29e-3 MJ/kg
Resources from ground	Phosphorus	2.10e-7 kg/kg	5.12e-7 kg/kg
Resources from ground	Hydrogen sulfide	1.09e-8 kg/kg	1.55e-8 kg/kg
Resources from ground	Sodium Chloride	3.69e-2 kg/kg	1.02e-1 kg/kg
Resources from water	Primary hydropower	8.80e-2 MJ/kg	7.25e-1 MJ/kg
Resources from water	Ground water	2.16e+0 kg/kg	7.23e+0 kg/kg
Resources from water	Sea water	1.46e-4 kg/kg	3.47e-4 kg/kg
Resources from water	Surface water	9.90e+0 kg/kg	3.37e+1 kg/kg
Resources from water	Water	2.20e-1 kg/kg	7.02e-1 kg/kg
Crude oil	Fuel energy	4.55e+0 MJ/kg	2.30e+1 MJ/kg
Energetic raw material	Feedstock energy	0.00e+0 MJ/kg	0.00e+0 MJ/kg
Mechanical energy	Primary total energy	4.55e+0 MJ/kg	2.30e+1 MJ/kg
Non-renewable fuels	Non-renewable energy	4.43e+0 MJ/kg	2.21e+1 MJ/kg
Renewable fuels	Renewable energy	1.19e-1 MJ/kg	8.57e-1 MJ/kg

emissions in carbon dioxide equivalents (CO_2e) and frequently known as carbon footprint (Muthu, 2015).

Unfortunately, wool technical products were not found to be a LCA topic in literature, with gate-to-grave or EOL perspectives of technical textiles waste management. In addition, EOL scenario building and their interpretations were also found to be absent in most of LCA studies on wool textile products. Recommendations on resource inputs are made for the EOL phase assessment of wool products, for instance water and energy, and emissions of greenhouse gases. Moreover, environmental impacts from chemical leaching to soil and water, such as dyes are opt to be taken into account while conducting environmental assessment. In case of landfill as EOL scenarios, methane is produced from the anaerobic decomposition of the wool, generating greenhouse gas emissions with possibility of direct atmospheric loss or as carbon dioxide following capture and flaring (IWTO, 2016). However, environmental impacts from the attributional finishes to the technical wool textiles are not considered. There is a need to minimize the effluent from the production and use phases of the technical textile products, such as FR treated ones. Presently, the textile finishing industries have overcome such hurdles by selecting cost effective chemicals.

Apart from the fact that FR systems are applied to textiles to avoid any disaster fire event, there is a big debate over the toxicity of FRs, presuming FRs are more toxic than the fire accident itself. On the other hand, majority of deaths are caused by fire effluent toxicity, while the unwanted fires are responsible for majority of injuries (DCLG, 2014; FSM, 2016). Moreover, with well-established chemistry of FR finish for wool textiles with increased applications, such finishes do have some disadvantages, for instance, presence of heavy metals in both the effluent and treatment of Zirpro treated wools (e.g. zirconium). The content of such heavy metals is even present in the most purified effluents, though Zirpro and tetrabromophthalic acid (TBPA) treatments have caught attention by the environmentalists, presently no big pressure to restrict their use (Horrocks, 2016; Mittal and Bahners, 2017).

6. Conclusions and recommendations

This study has tried to elaborate some points for technical textiles waste. Firstly, the LCA method for EOL is practicable if the waste treatment is based on the functionality of the technical textiles rather the common textile waste. In spite of this, development of the method to add information, a part from the fiber content on the labels for technical textiles, to segregate them for their unique functionality and information of wash fastness and release of the functional substance applied.

Secondly, this gate-to-grave or EOL study, demonstrated that the LCA results of any technical textile product at its disposal are also case dependent and should not be considered equivalent to collective textile waste or MSW, regardless of environmental parity being considered or not. With respect to the recycling of technical textile curtains, both employ functionality substances, FR and AgNPs for its flame retardancy and antimicrobial properties. Their life-cycle impact perspectives could be different with loss of their functionality in use phase, for instance, significant loss of AgNPs during laundering compared to well bond FR to fibers. This increases environmental cost of one technical textile (AgNPs treated) in use phase imposing stringent wastewater treatments. Similarly, behavior of other functionality substance on technical textile (FR treated) requires various considerations for either EOLs, landfill or incineration. Which indicates the need of more research to be done in this regard.

Vast environmental impact data exists for the manufacturing and production of technical textiles including their functionality substances, and suggestions associated to reducing the environmental impacts can be found. Data on routes of exposures from functionality substances in use phase and EOL is limited. Thus, existence of numerous research gaps in considerations for technical textiles in all phases of life cycle, enforces the LCA community and textile industries to compute LCI data for technical textiles, especially for EOL phase. Further studies will be carried out to refine the modeled EOL scenarios of technical textiles to more specific disposal alternates.

Conflicts of interest

The authors declare no conflict of interest. The funder has no role in the data collection, in the writing of the manuscript, analyses or interpretation of data, and in the publication of results.

Disclaimer

This work has not been adopted or endorsed to any textile industry/waste management or environment protection agencies in Scotland, United Kingdom. Any views expressed in this research are the mere preliminary views of the authors and should not be considered official statements from abovementioned authorities at any circumstances.

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