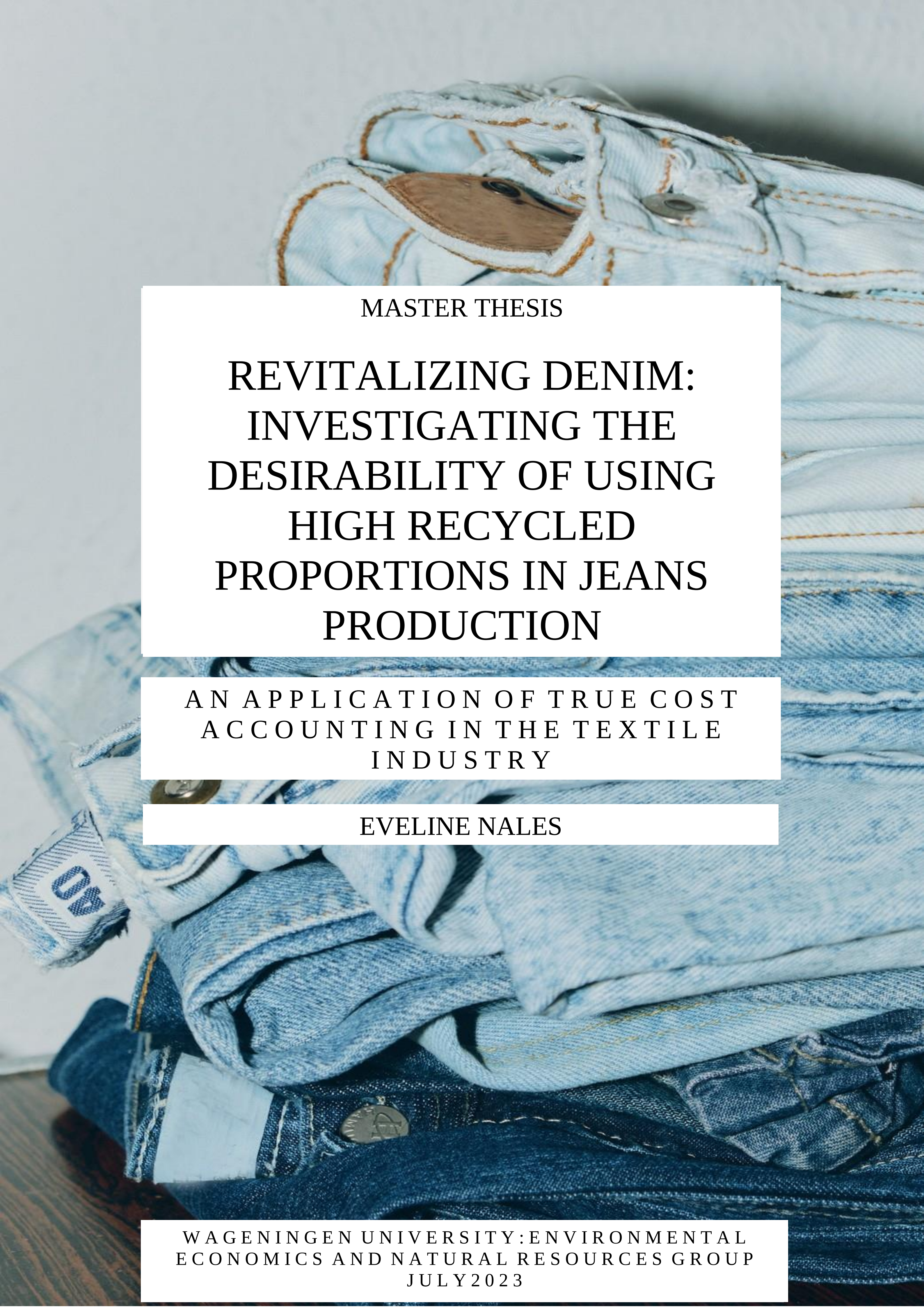

MASTER THESIS

REVITALIZING DENIM: INVESTIGATING THE DESIRABILITY OF USING HIGH RECYCLED PROPORTIONS IN JEANS PRODUCTION

AN APPLICATION OF TRUE COST ACCOUNTING IN THE TEXTILE INDUSTRY

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Revitalizing denim: investigating the desirability of using high recycled proportions in jeans production

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Abstract

As the ongoing textile waste problem in the Netherlands continues to persist, developing a system that considers post-consumer waste is crucial. Using post-consumer recycled textile in new garments provides several advantages over using virgin material. It has the potential to reduce environmental impacts of garment production, reduce textile waste and to be produced locally. Therefore, the Dutch Circular Textile Policy Program states that by 2030, all new textile products must consist of at least 50% sustainable material, of which a minimum of 30% is recycled. However, using post-consumer recycled textile in garment production presents some challenges. The production process of post-consumer recycled material carries inherent environmental impacts. Moreover, its use often decreases the quality and therefore the lifespan of the garment. Also, there are concerns about its cost-effectiveness. This thesis explores whether it is desirable to use as much recycled material as possible when producing a new pair of jeans and what the optimal recycled proportion is. To find this optimum, it applies a mathematical programming model that incorporates the outcomes of the true cost accounting (TCA) assessment of a pair of 100% virgin and 100% recycled jeans. In this way, it investigates the trade-offs of including post-consumer recycled material in jeans production on the jeans' environmental costs, private costs and lifespan. The findings reveal that while using a higher proportion of recycled material reduces the environmental costs of jeans, it leads to higher private production costs and a shorter lifespan of the garment. Under the current circumstances, it is not desirable to strive towards using 100% recycled material in the production of new jeans. Instead, this study finds an optimal recycled proportion of 0.2. However, this study does show the challenges that arise when using the TCA method. Therefore, it can also provide insights for policymakers, companies and researchers in the practical use and reliability of the TCA method.

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1 Introduction

1.1 Recycling as the solution to textile pollution

The global textile, clothing, and fashion industry is among one of the most polluting industries in the world (Fletcher, 2012). The fashion industry is estimated to be responsible for 2-8% of global greenhouse gas emissions, 20% of global wastewater, and 9% of annual microplastics losses to the ocean (UN, 2023). It also consumes more energy than the aviation and shipping industries combined (UNFCCC, 2018).

These impacts of the fashion industry are not being reflected in consumer behaviour. One reason for this may be the fashion paradox, which states that consumers are more likely to consider the environmental effects of their non-fashion purchases, but tend to justify their fashion consumption (Joy et al., 2012). Also, companies are constantly introducing new clothing designs, creating an impulse to buy more. As a result, each garment is worn less and less, reducing its lifespan, and increasing the demand for new items (Pal, 2017). It is estimated that of the 150 billion garments that are produced annually, over 50% are discarded within less than a year (Mazotto et al., 2021). In case of the EU, the destination and future use of this textile waste is highly uncertain, with a significant portion being sent to Asia and Africa. A considerable amount of this waste ends up in open landfills and informal waste streams in these regions. This situation may get worse as a result of all EU countries being required to separately collect their textile waste by 2025 (EEA, 2023).

This ongoing textile waste problem cannot be effectively addressed without providing solutions specifically targeted at post-consumer textile waste (McCauley & Jestratijevic, 2023). Recycling has great potential for the textile sector to reduce its waste, reduce environmental impacts of production and contribute to the circularity of the industry (Baruque-Ramos et al., 2017). By avoiding virgin fibre production in the textiles supply chain and replacing it with recycled fibres, several environmental benefits can be achieved. These include reducing CO_2 emissions, decreasing water consumption, reducing acidification and eutrophication by using less pesticides and dyeing materials, and lowering landfill waste (Johnson et al., 2020; Shen & Patel, 2010). Other benefits include that recycling, especially compared to other circular economy practices, can more easily be coupled with existing business practices (Levänen et al., 2021). Furthermore, the EU Commission's policy on the circular economy (2015) envisions socio-economic and socio-environmental benefits. These include new business opportunities (e.g. local fibre production), innovative production and consumption methods, and social integration (Filho et al., 2019).

In an effort to improve the circularity of the fashion industry, the Dutch government has taken measures to promote textile recycling. In 2020, this was done by setting up the ‘Denim Deal’, in which signatory parties state to use at least 5% post-consumer recycled cotton in their denim garments (Rijksoverheid, 2020d). Moreover, in 2021, the Netherlands sent a joint paper together with ten other member states to the European Commission regarding a more sustainable fashion industry. Among other things, this paper calls for compulsory clothing recycling practices (Rijksoverheid, 2021). Most importantly, as of January 1, 2023, the Dutch government has implemented an extended producer responsibility (EPR) scheme for textiles and garments. This scheme makes producers collectively responsible for the reuse, recycling, and reprocessing of used apparel and fabrics. The goal of this program is to have recycled and reused at least 50% of the textile products that have been introduced to the market by 2025, with 60% of this being recycled and at least 25% of the total recyclable portion being recycled back to fibres (Rijksoverheid, 2022). The goal of this fibre-to-fibre recycling is to increase the proportion of recycled fibres in new textiles by as much as possible. By 2030, it will be mandated that new textiles produced must include a minimum of 30% recycled fibres (RIVM, 2023).

However, the use of recycled material in garments has potential downsides. One issue of concern is the high level of emissions produced during the process of turning used clothes into new materials (Levänen et al., 2021). Moreover, the sorting process for post-consumer textile waste requires significant labour input, causing concerns regarding its cost-effectiveness (Pensupa, 2020). Additionally, there are concerns about the quality of recycled materials, as its use generally has a negative impact on the properties of the garment that is made from it (Bain, 2015, Ütebay et al., 2019a). Given the multiple downsides and varying environmental implications associated with recycling, it may be worth questioning whether striving for the highest possible recycled proportion in new garments is actually desirable from an environmental and economic perspective.

To better understand the environmental impacts of textile recycling, multiple life cycle assessments (LCA) have been conducted. However, these LCAs present their findings in natural units, such as total mass (kg) of emissions or land size (m^2). This presentation can make it challenging to compare and weigh the different environmental impacts (Sandin & Peters, 2018). The decisions resulting from these LCAs then depend on the relative importance placed on each type of environmental impact. To enhance comparability among different scenarios, certain LCA studies have opted to focus on a single environmental impact category. For example, Franco-García et al (2019), La Rosa & Grammatikos (2019) and Levänen et al (2021) exclusively examine the contribution to climate change. However, this approach does carry the risk of burden shifting between different environmental impacts (Hauschild et al., 2018).

To overcome these limitations and facilitate easier comparison and weighting of different environmental impacts, the method of true cost accounting (TCA) has emerged. TCA is increasingly gaining recognition among practitioners and policymakers (Taufik et al., 2023). It provides a framework that allows for the integration of information from various stages of the supply chain (agricultural, manufacturing, distribution, and retail), to quantify impacts and assign a value to them. This approach enables the quantification and comparison of the unintended negative external effects of producing a particular product on the environment and society (Gemmill-Herren et al., 2021).

According to Johnson et al (2020), who performed a literature review on the supply chain of waste cotton recycling, previous studies have only focused on specific aspects of recycled textile. For instance, Wendin (2016) conducted a LCA on recycled cotton at H&M and reports that mechanically recycled cotton has lower environmental impacts than virgin cotton. However, the study did not evaluate the economic feasibility and quality reduction of recycled cotton fibre that may limit the applications of the final product. To promote the sustainable use of waste cotton, it is critical to understand not only the environmental benefits, but also the product quality and economic competitiveness compared to virgin counterparts (Johnson et al., 2020, Sandin & Peters, 2018). To understand if the Dutch government's approach of stimulating the use of the highest possible proportion of recycled material in new garments is actually desirable, there is a need for a comprehensive evaluation that considers various blending ratios of virgin and recycled cotton. Therefore, this thesis aims to explore the optimal recycled proportion in jeans through the use of true cost accounting and considering the trade-offs between recycled content, product lifespan, and environmental impacts.

1.2 Research questions

This research divides the true costs into "private costs" and "environmental costs". Environmental costs include the damage the environment, e.g the cost of greenhouse gas emissions. Private costs include raw material input, e.g wages and electricity costs (Gemmill-Herren et al., 2021). Therefore, the research questions are as follows:

1. What are the private costs associated with producing jeans from virgin cotton?
2. What are the private costs associated with producing jeans from post-consumer recycled cotton?
3. What are the most significant impact categories to consider when evaluating jeans production?
4. What are the environmental costs associated with producing jeans from virgin cotton?
5. What are the environmental costs associated with producing jeans from post-consumer recycled cotton?
6. What is the impact of adding post-consumer recycled material on the lifespan of a pair of jeans?
7. What is the optimal proportion of post-consumer recycled material to use in the production of jeans?

1.3 Methodology

The methodology of this study consists of three main components: a modelling framework, data collection through literature review, and a sensitivity analysis.

This study addressed the first two research questions through a literature review. For questions three to five, the study performed a literature review and a taxonomy of life cycle assessments (LCA) of both virgin and recycled jeans. It obtained monetization factors mostly from the True Price Foundation to calculate the environmental costs. This study addressed question six through a literature review on the quality and lifespan of jeans. It identified an equation that relates the lifespan of jeans to the recycled proportion. This study addressed question seven through the creation of an optimization model that aims to consider the impacts on environmental costs, private costs and lifespan of recycling jeans in order to find the optimal recycled proportion.

2 Background

This section provides an overview of some relevant background information to this study. It begins by outlining the current state of textile waste and recycling in the Netherlands. Here, the existing textile waste streams, various recycling options, current textile recycling policies and drawbacks of recycling are discussed. Then, the method of true cost accounting is introduced by examining its origins, application, and the associated criticisms and limitations.

2.1 Textile recycling in the Netherlands

2.1.1 Textile waste streams

There are two kinds of textile waste streams, pre-consumer waste and post-consumer waste. During the cutting and sewing processes, about 10-20% of pre-consumer textile waste is generated (Lau, 2015). Pre-consumer waste is often well-defined and homogeneous, as it can be collected before the garment is produced. These characteristics increase the possibility of cost-effective and high-quality recycling (Harmsen et al., 2021). Furthermore, pre-consumer waste does not have the hygiene and collection challenges associated with post-consumer waste (Lau, 2015). Post-consumer textile waste refers to clothes discarded by customers, either in recycling bins or through collection services (Chevalier, 2022). Post-consumer textile waste that is not collected, mostly ends up in residual waste. The proportion of textiles in the residual waste in the Netherlands is increasing. In 2000, textiles made up only 3% of residual waste, but by 2018, that percentage had risen to 6% (Rijkswaterstaat, 2019).

As can be seen in figure 1, 45% of all textile waste is collected separately from residual waste. The collected textiles are sorted manually based on usability, quality, type, and material. Still wearable textiles (50%) are either pressed into bales and sold and shipped by type (90%) or sold in the Netherlands through for example thrift stores (10%) (VHT, 2023).

A significant portion of wearable textile waste is sold to other countries, primarily Eastern Europe and Africa. Of the fractions suitable for recycling, 4% remains in the Netherlands. Recycling destinations include Europe, India, and Pakistan, where further sorting, insulation, and processing into yarns occur (VANG, 2023).

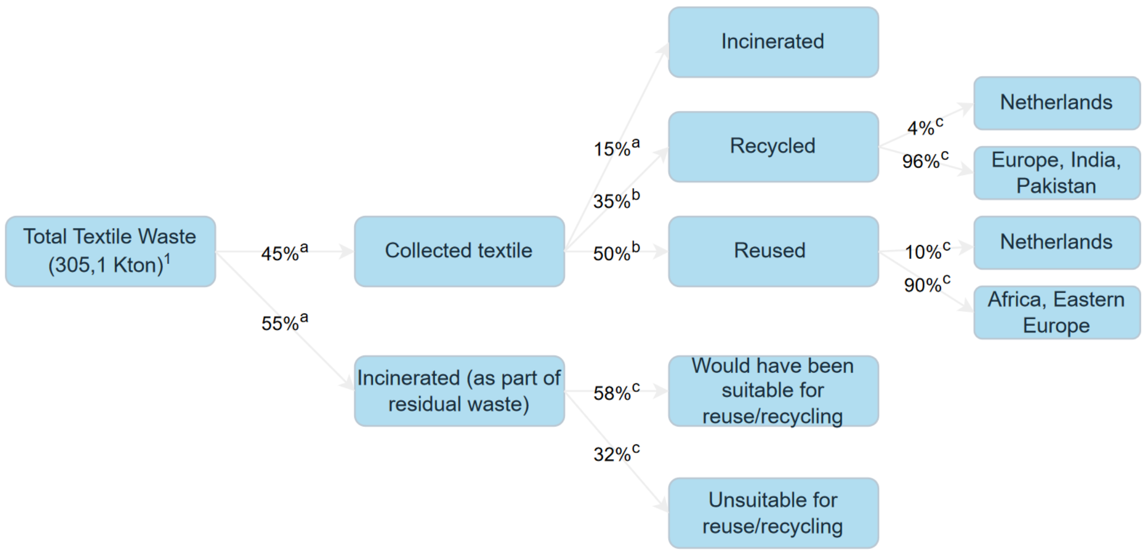

Figure 1: Overview of textile waste streams in the Netherlands. Sources: a.(Rijksoverheid, 2020c), b.(VHT, 2023), c. (VANG, 2023)

2.1.2 Types of recycling

35% of the post-consumer textile waste that is collected is being recycled. This can occur in different ways. First, textiles can be cut and shredded into a textile type for a different use than the original product. If the new product is of lower quality, it is considered downcycling. Examples include clothing being turned into cleaning cloths and insulation material for cars (VANG, 2023).

Second, post-consumer textile can be mechanically recycled. Mechanical recycling is the recovery of textile fibres by physically separating the fibres. The mechanically recycled fibres are then mixed with primary fibres to ensure sufficient quality when used in new products (VANG, 2023). However, this technique is still in its infancy stage (Harmsen et al., 2021; RIVM, 2023). Currently, only 1% of the material from both pre- and post-consumer waste is recycled into new clothing (McArther Foundation, 2018). A big advantage of mechanical recycling is that it does not require monomaterials (a textile product that is 100% made of one type of fibre). However, recyclers often prefer to receive material of one colour with as few other additions as possible, such as elastane. Accessories, coatings, and zippers have to be removed manually before recycling (van der Wal & Verrips, 2019). When it comes to denim, most mechanical fibre-to-fibre recycling is done in small projects and focuses on creating new denim from old denim. Currently, there are a few facilities in the Netherlands that mechanically recycle post-consumer textile, such as Frankenhuis in Haaksbergen and Wolkat in Tilburg, but their capacity is limited. Until now, most post-consumer recycling for the Dutch market takes place in Spain and Turkey (van Raan, 2019).

Finally, there is chemical recycling, which involves breaking down textiles into their basic chemical components. The fibres generated by this process are of higher quality and the colour of the secondary material is not important (van der Wal & Verrips, 2019). However, chemical recycling technologies are still scarce and are currently relatively expensive and energy intensive. There are also technical barriers to overcome in order to meet strict requirements in terms of pollution and material composition. For this technique, the secondary material needs to consist of only one material or needs to be separated intensively (chemically), causing a significant portion of textile to go to waste (van der Wal & Verrips, 2019). Chemical recycling has the potential to create 100% recycled textiles. The technology is not yet ready for commercial use and is more costly than mechanical recycling. In the Netherlands, a Dutch chemical recycling firm, SaXcell, has recently emerged and anticipates the ability to use their technology on a large scale in the near future (van Raan, 2019).

2.1.3 Policy on textile recycling in the Netherlands

To increase the circularity of the textile industry, the Dutch ministry of infrastructure and water management has introduced the Circular Textile Policy Program 2020-2025. This program includes the ambition that the textile chain will be fully circular by 2050, meaning that all textiles collected in the Netherlands must be reused and recycled (Rijksoverheid, 2020b). By 2025, all new textiles brought to market must contain at least 25% (post-consumer) recycled or sustainable material. By 2030, textile products must consist of at least 50% sustainable material, of which a minimum of 30% is recycled. Additionally, the goal is to recycle a total of 30% of textile waste in the Netherlands by 2025 and 50% by 2030. Furthermore, the government wants to encourage the reuse of textiles (Rijksoverheid, 2020b).

To stimulate the recycling of post-consumer material, two new initiatives have been introduced by the Dutch government in recent years. First, the denim industry, which is relatively large in the Netherlands, has the ambition to lead the way in using post-consumer recycled cotton fibres in the production of new denim clothing. As a result, the denim deal was introduced in 2020 in the Netherlands. In this deal, parties commit themselves to the joint ambition of working as quickly as possible towards a new industry standard of at least 5% post-consumer recycled cotton fibres in all denim clothing items (Rijksoverheid, 2020d). Second, the extended producer responsibility (EPR) for textiles was introduced in January 2023. EPR means that the producer (the one who brings the garment to the Dutch market) is made responsible for their product throughout its entire lifecycle, including the end-of-life phase. This can be collectively taken up by having producers pay a fee for each piece of clothing they bring to market. The fees that producers must pay are determined based on factors such as sustainability, repairability, reusability and recyclability, and the presence of hazardous substances. The more sustainably produced, the lower the fee. This fee differentiation provides producers with an incentive to act sustainably (Rijksoverheid, 2020a). The goal of the scheme is to have recycled and reused at least 50% of the textile products that have been introduced to the market by 2025. 60% of this portion will be recycled and at least 25% of the total recyclable portion will be recycled back to fibres and incorporated in new garments (Rijksoverheid, 2022).

2.1.4 Drawbacks of recycling

While textile recycling is seen as a crucial step towards creating a sustainable textile sector, and fashion brands are increasingly integrating textile-to-textile recycling into their production processes, it is important to note that recycling is often not regarded as the preferred strategy by academics in the implementation of a circular economy (CE). Kirchherr et al (2017) conceptualizes the circular economy using the 9R framework. Among the 9 strategies outlined, recycling ranks near the bottom. This perspective is supported by Sandin & Peters (2018), who found that in nearly all studies on implementing CE practices in the textile industry, other strategies, such as reuse, are preferred over recycling.

It is important to acknowledge the presence of significant challenges and drawbacks of recycling (Watson et al., 2017). One of the main challenge in mechanical recycling of post-consumer textile waste is the reduced quality due to the complex blend of fibres and their shortened length. These result in lower quality, strength, and softness compared to virgin cotton fibres (Chevalier, 2022; Radhakrishnan & Kumar, 2018). Additionally, the private costs of shredded textile waste fibres present a challenge, as they often are higher than virgin materials due to the labour-intensive sorting process (McCauley & Jestratijevic, 2023; Pensupa, 2020). Furthermore, there are knowledge and attitude barriers that hinder recycling efforts (Anthouli, 2013). For example, some consumers may have concerns about the susceptibility to bacterial growth and infection. These factors contribute to investors becoming indifferent towards the waste-clothing recycling industry (Xie et al., 2021).

There has also been criticism on the benefits of recycling for the environment. For instance, Milne et al (2006) refers to recycling as an example of the "journey metaphor" of environmental sustainability. Here, retailers often make promises about the possibility of a future fashion industry running on recycled materials and generating no waste. It justifies the lack of clear results in the present by deferring commitments to a distant future. By promoting take-back systems and recycling, retailers are essentially asking stakeholders to accept the current unsustainable state of the industry. Paradoxically, these systems may even contribute to accelerating the consumption cycle of clothing. Rather than leaving clothes unused in wardrobes, they encourage individuals to participate in the take-back systems, making room for new items and allowing them to buy more without guilt (Corvellec & Stål, 2019). There is an ongoing debate regarding whether increased recycling can effectively offset the overall environmental impact of the textile industry if production and consumption continue to grow (Koligkioni et al., 2018). For instance, there is a risk of the rebound effect, which occurs when improvements in technological innovations do not meet their environmental expectations due to (behavioural) economic mechanisms. This is also present in the Dutch textile industry (Siderius & Poldner, 2021). Despite the multiple challenges and limitations of recycling, the focus of this study revolved around the challenges of the environmental impacts of the mechanical recycling process, quality reduction and increased private costs when using recycled materials.

2.2 True Cost Accounting

2.2.1 Negative externalities

The trends towards mass-production and overconsumption, and the disregard for their consequences, has not only become normalized, but has also become very profitable. However, the price that the planet and vulnerable populations must pay for the production of clothing is not cheap (Costa, 2015). These unintended negative by-products of economic activities are called negative externalities. They occur when the actions of economic agents result in uncompensated physical or economic consequences for others. These externalities arise when there are no well-defined property rights, often due to the high costs of establishing clear boundaries (Vatn & Bromley, 1997). Negative externalities are problematic for various reasons. First, externalizing costs leads to overconsumption of natural, health, and social resources by economic agents, incentivizing them to consume more than what is socially efficient. In competitive markets, resource allocation tends to be efficient. However, real-world conditions can cause market failures. In the case of negative environmental externalities, nonrival and nonexcludable public goods like clean air, water, and biodiversity, are used by private entities. Due to the lack of incentives for production and maintenance of these goods, environmental resources are being overconsumed and under-priced (Randall, 1983; Reinhardt, 1999). Second, negative externalities distort market prices, hindering societies from reaching their maximum potential by misrepresenting the true value of products (Gemmill-Herren et al., 2021; Michalke et al., 2022). Consequently, crucial economic indicators fail to account for the environmental and health costs associated with production, resulting in lower average living standards. Third, the erosion of natural capital breaches the rights of marginalized groups and future generations to decent livelihoods, causing social injustice (Hendriks et al., 2021).

Currently, society as a whole bears the costs of externalities, such as the increasing prices of clean water (Barraqué, 2003). Given the embedded assumptions about behaviour and rationality in the traditional market and economic models, negative externalities are likely to increase over time. As competition intensifies, companies will remain alert for emerging opportunities to reduce costs. Many scholars perceive negative environmental externalities as a market failure. Though, some even argue that it represents a failure of the underlying model itself (Vatn & Bromley, 1997).

2.2.2 The use of true cost accounting

To address the issue of negative externalities, various economists, researchers, and sustainability advocates have explored and promoted the concept of ‘true pricing’. One of these advocates, is the Dutch non-profit organization True Price Foundation (now Impact Institute), which was established in 2012 (True Price, 2023).

True pricing, which internalizes some externalities in product prices to reveal 'hidden costs', is considered a potential solution to the overconsumption and under-pricing of natural resources (Baker et al., 2020; Hendriks et al., 2021). It bridges the gap between market prices and the actual production costs of products by placing a value on external costs (Gemmill-Herren et al., 2021). Therefore, it aligns with the Polluter Pays principle of the UN Sustainable Development goals.

The true price is calculated with the true cost accounting (TCA) method (Michalke et al., 2022). In this study, true costs refer to the combined internal (private) and external (environmental) costs. The calculation of additional 'social costs' was beyond the scope of this study. However, it is important to note that the apparel industry has historically been dominated by countries with lower wage rates due to its labour-intensive nature (Yunus & Yamagata, 2012). Reports have revealed widespread violations of human rights in garment factories, including inadequate wages, unsafe working conditions, bonded labour, child labour, harassment and incidents like building collapses and fires in sweatshops (Brooks, 2015; Impact Institute, 2019; Nathan et al., 2016). Despite corporate pledges to improve this, progress in these areas remains limited (LeBaron et al., 2022).

When determining the true cost of negative environmental externalities in monetary terms, two questions arise. First, which externalities should be considered and what is their quantified impact? Second, how to quantify and monetize them? To address the first question, life cycle assessments (LCA) are reviewed to evaluate and identify the relevant externalities. LCA is "a tool to assess the potential environmental impacts and resources used throughout a product's life cycle, i.e. from raw material acquisition, via production and use stages, to waste management" (ISO, 2006, p.1). Using a life cycle perspective prevents burden shifting between stages or processes, which may occur when efforts to reduce environmental impacts in one area inadvertently lead to increased impacts in other areas. This potential burden shift is also prevented by considering a wide range of environmental impacts, such as climate change, freshwater use, eutrophication, and more (Hauschild et al., 2018).

To address the second question, these negative impacts, which are expressed in natural units, can be multiplied by monetization factors. According to the True Price Foundation (2021), these monetization factors are determined by considering four types of costs that, when combined, create the remediation cost for an impact. First, restoration costs involve restoring people's circumstances regarding wealth, health, capabilities, or environmental conditions to the original state. Second, compensation costs entail compensating affected individuals for damages caused by the production or consumption of a product. Third, prevention of re-occurrence costs include expenses incurred to avoid or prevent future negative impacts associated with a product (e.g supply chain audits). Finally, retribution costs refer to fines, sanctions, or penalties imposed by governments for violating laws. The value of monetization factors is not solely determined by their cost perspective, but also by the areas of impact they include (e.g., human health, resource scarcity), the use of equity weighting, the level of discounting applied and the geographical scope. Among these criteria, practitioners should particularly focus on the geographical scope, as it is the most influential criterium (Arendt et al., 2020).

TCA is increasingly gaining recognition among practitioners and policymakers, especially in the food sector (Taufik et al., 2023). Various organizations, including the Natural Capital Coalition, the World Business Council for Sustainable Development, and TEEB, are working to further expand its application (FAO, 2023). There are differing perspectives on the implementation of TCA, with some suggesting that it should involve (voluntarily) pricing harmful products proportionally higher, while others suggest using it purely for informing stakeholders (Michalke et al., 2022). TCA provides insights into the varying impacts of different products, which can be utilized by stakeholders in production and consumption processes (Taufik et al., 2023). This approach has the potential to incentivize the private sector to generate positive externalities, enhance transparency, and support the transition towards a more sustainable system. Furthermore, applying TCA brings attention to issues that are often overlooked by decision-makers, and it can contribute to limiting social injustice and addressing underlying causes of conflict (Baker et al., 2020; Hendriks et al., 2021). In fact, Gemmill-Herren et al (2021) even argues that TCA has the potential to establish a framework for systemic transformation.

2.2.3 Criticisms and limitations

However, TCA also faces criticism and limitations. One of the main criticisms against TCA is that it is a market-based solution, which puts nature under the rule of the free market. Some people oppose this idea, stating that the Earth should be recognized as our common home and managed accordingly (Rundgren, 2017). Also according to Michalke et al (2022), informational campaigns and individual behavioural changes alone are insufficient to overcome existing institutional barriers. Additionally, (Vatn & Bromley, 1997) critiques market-based interventions, as they tend to address symptoms rather than the underlying problem that gives rise to externalities in the first place.

Second, it is questioned if monetary valuation is even appropriate nor possible. Evaluating the economic cost of social damage, such as clean air, water, culture, climate stability, and biodiversity, is complex and subjective and cannot be quantified in a single figure (Baker et al., 2020). Often, it relies on methods like the market price, revealed preference or stated preference approach, which may not accurately reflect the true economic impact (Arendt et al., 2020; Van Grinsven et al., 2013). Current methodological advancements still struggle to capture the full extent of economic damage in a realistic manner (Michalke et al., 2022). Arendt et al (2020) identifies the valuation of biodiversity in particular as one of the major weaknesses of valuation methods. Capturing the full extent of impacts is also an issue when performing life cycle assessments (LCA), which require simplifications and generalisations in modelling. Considering these and the fact that calculated impacts are aggregated over time and space, it is more accurate to say that LCAs and therefore also TCAs calculate potential impacts rather than actual impacts (Hauschild et al., 2018).

Third, the ability to create a standardized approach to TCA is questioned due to the significant variations in calculating costs across different regions and even within the same country. Also, the True Price Foundation (2021) highlights the presence of inconsistencies in assumptions and modelling choices among indicators, primarily due to a lack of standardization. Standardization is crucial for comparability and engagement, as unclear and differing calculations may lead to distrust and improper applications (i.e. greenwashing) (Baker et al., 2020). Though Hendriks et al. (2021) argues that despite imperfections, TCA can still provide valuable information to actors in the supply chain.

Finally, an important ethical concern is raised regarding the potential scenario where true prices are paid to address externalities. Paying true prices can simply become a way for people to compensate for their externalities by appropriation of marginalized people's resources, e.g. by tree planting in Africa. This raises concerns about fairness and equity (Rundgren, 2017).

3 Methodology

This chapter outlines the research methodology of this study. The methodology consists of three parts, first a data collection through literature review, second, a modelling framework and third, a sensitivity analysis.

3.1 Literature research and parametrisation

To run the model described in section 3.2, this study performed a literature to investigate the scope of the research and to identify the values of input factors.

3.1.1 Private costs

This study conducted a literature review to understand the private costs of the production of virgin and recycled jeans, including the entire process from raw material creation to manufacturing and transportation, excluding selling costs. I examined various sources, including scientific articles, reports, websites of textile selling companies and used personal communication through email. Keywords used in databases to find relevant literature articles are jeans production costs, post-consumer recycled cotton yarn and mechanical recycling costs. The investigated databases were Google Scholar, WURlibrary, Sciencedirect and Google. The literature review followed an iterative process, with additional sources continually being added during the course of the study.

3.1.2 Environmental costs

A TCA assessment typically starts by identifying the scope of the assessment, establishing the unit of analysis and the system boundaries (Hendriks et al., 2021). This study focussed on the trade-off between private costs, lifespan, and environmental impacts of a pair of jeans sold on the Dutch market. It performed a short literature review on both the life cycle of recycled and virgin jeans to better understand the system boundaries. To be able to compare life cycle assessments, I standardized their outcomes with the functional unit of one pair of average-sized jeans of 0.67 kg (Hedman, 2018; Şener Fidan et al., 2023). Due to wastages during the production process, the weight of different materials for one pair of jeans differs (figure 2). To produce a single pair of jeans, 0.744 kg of denim fabric is needed as there is an estimated wastage of 10% to 15% due to cutting during the production process (Akı et al., 2020). For one pair of jeans, an estimated 0.882 kg of cotton yarn is required, with an expected wastage of 15.6% during the weaving process (Kazan et al., 2020).

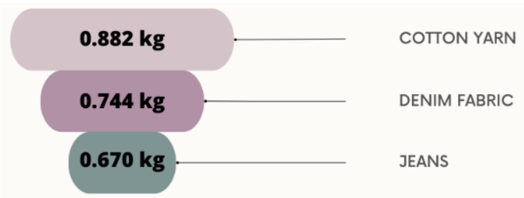

Figure 2: The weight of different types of material needed for one pair of jeans

This study conducted a literature review to make a selection of impact categories to be included when determining the environmental costs. This was done in several steps. First, this study conducted a short literature review to better understand which environmental impact categories are relevant for textile production and what their definition is. I reviewed scientific articles on LCAs, environmental impact categories and post-consumer recycling of cotton. Second, I prepared a taxonomy that lists the assessed environmental impact categories in LCA studies on virgin jeans, denim fabric and cotton, as well as on recycled jeans, denim fabric and cotton. Since the literature on the mechanical recycling of post-consumer textiles is limited, there were no age restriction on the literature findings.

Third, I made a selection of impact categories based on the availability of LCA data on recycled jeans, denim fabric and cotton. As studies on recycled material are less available, I assessed the researched impact categories of these studies first. To determine which studies are suitable for this study, I filtered the studies from the taxonomy according to the following criteria:

1. The study researches the environmental impacts of recycled jeans, denim fabric, cotton fabric or yarn.
2. The study displays its results in actual units.
3. The study includes results on the use of 100% recycled material.

Fourth, I made a selection of impact categories based on the available data on monetization factors. This study primarily derived monetization factors from the handbook of the Dutch True Price Foundation (now Impact Institute) (2021). However, if any of the impact categories from step three were not addressed by the True Price handbook, I consulted the handbook from CE Delft (Schroten et al., 2018).

To determine the environmental impacts of a pair of 100% virgin and 100% recycled jeans, I consulted the LCAs from step three. From these studies, I selected one base study from which I collected most of the LCA data. The selection of the base study depended on how representable the study is for jeans production from post-consumer recycled cotton on the Dutch market. The remaining studies are complementary studies, which I consulted to compare the findings from the base study to. Also, in case the base study lacked information on certain impact categories, I consulted these complementary studies. I standardized the LCA outcomes according to the functional unit described in figure 2. After I selected the most representative environmental impact values from the base and complementary studies for 100% virgin and 100% recycled jeans, I calculated the environmental cost. This means multiplying the monetization factors with the corresponding environmental impact values (Gemmill-Herren et al., 2021). Then, I calculated the relative environmental cost in percentages of the total environmental costs of either 100% virgin or 100% recycled jeans. Finally, I discarded all environmental impact categories with a relative environmental cost of less than 1% and therefore did not include them in the model.

3.1.3 Lifespan

This study conducted a literature review to understand the lifespan of virgin jeans and the impact of adding recycled material on the lifespan of jeans. Various scientific articles were examined. Keywords used in databases to find relevant literature articles are post-consumer recycled cotton quality, jeans lifespan and garment quality indicators. The investigated databases were Google Scholar, WURlibrary and Sciencedirect. The literature review followed an iterative process, with additional sources continually being added during the course of the study. As a result of this literature review, this study found an equation for lifespan (4) as an input for equation (1). Here, the lifespan of a pair of jeans is dependent on the recycled proportion.

3.2 Modelling framework for the true costs of jeans recycling

This study introduced a mathematical programming model to quantify the environmental and private costs of recycling jeans. The model aims to optimize the proportion of recycled material by minimizing the total true costs divided by the lifespan under various constraints.

First, the cost minimization problem can be written as:

$$\min_{r,L}\left(\frac{(1-r)C_1 + rC_2}{L}\right) \tag{1}$$

Where r is the recycled proportion, L is the lifespan of jeans, C_1 is the total true costs of 100% virgin jeans and C_2 is the total true costs of 100% recycled jeans. The minimization problem is based on the assumption that the use of recycled material forgoes the production of virgin material in a 1:1 ratio. However, this is often not the case (Castellani et al., 2015). Also, it is assumed that recycling has no other impacts beyond the environmental, private cost and lifespan impacts described in this study, as well as the impacts from forgoing virgin material production. Further, it is assumed that the total true costs change linearly with the recycled proportion added. Also, it is assumed that the costs associated with producing jeans, both virgin and recycled, do not change based on the proportion of recycled material used or any other factors considered in the model.

Total True Costs for either 100% virgin or recycled jeans can be written as:

$$C_j = E_j + P_j \tag{2}$$

Where E_j are the total environmental costs of either 100% virgin jeans (j=1) or recycled jeans (j=2) and P_j are the total private costs of either 100% virgin jeans (j=1) or recycled jeans (j=2). The total true costs (C_j) are assumed to be the sum of private and environmental costs without any interactions or dependencies between the two.

The total environmental costs of 100% virgin or recycled jeans can be written as:

$$Ej = \sum_i I_{ij} M_i \tag{3}$$

Where I_{ij} are the environmental impacts i of either virgin jeans (j=1) or recycled jeans (j=2) in natural units and M_i is the corresponding monetization factor in €/natural unit. Here, assumptions include that environmental impacts can be categorized, quantified and can be assigned a monetization factor. The methodology of true cost accounting is explained more extensively in 2.2.2.

The lifespan of a pair of jeans in hours of active wear can be written as:

$$L = \alpha + \beta r \tag{4}$$

Where α and β are coefficients and r denotes the recycled proportion.

The cost minimization function (1) is subject to the constraints:

$$r \geq 0 \tag{5}$$
$$r \leq r_{max} \tag{6}$$
$$L \geq 0 \tag{7}$$
$$L \leq L_{max} \tag{8}$$

3.3 Sensitivity analysis

Although this study utilized real data, it is important to acknowledge the potential uncertainties arising from variations in textile factories, geographical locations, and the divergent conclusions from different sources regarding input factors. Therefore I performed a sensitivity analysis to test certain input factors. First, I examined all the parameters and other model input factors found during the literature review described in 3.1. Then, I identified the input factors of interest on the basis of two criteria. First, the model input factor is uncertain. Reasons to believe an input factor is uncertain can be for instance due to contradicting literature, a lack of available literature or the input factor being dependent on many other factors. Second, I reasonably expect the input factor to have a large impact on the outcome of the model. To determine this impact, I assessed the relative total environmental costs described in 3.1.2.

The identified input factors were tested by varying the variables by different percentages, both increasing and decreasing them with a maximum of 100%, while holding all other factors constant (ceteris paribus). I examined one additional scenario in which two input factors were simultaneously increased to evaluate their combined impact.

4 Private costs: Virgin jeans

This chapter outlines the private costs involved in the production of 100% virgin cotton jeans. First, an explanation of the term "private costs” is provided. Subsequently, the cost division and the overall expenses associated with manufacturing jeans made of 100% virgin cotton are discussed.

4.1 Defining private costs

The calculation for the private costs of jeans involves several factors, including raw material, energy, water, labour, and auxiliary materials (Fidan et al., 2021). This differs from the price paid by consumers, which also covers additional expenses, such as advertising, design, transport, packaging, wastage, and profits (Brooks, 2015). Given the competitive nature of the denim industry, cost-saving and product quality are crucial considerations during the product design phase (Annapoorani, 2017; Wang, 2006).

4.2 Division of costs

According to Brooks (2015), the production of denim fabric is the most expensive stage of jeans manufacturing, accounting for US$ 2.80 per pair of jeans. Of these costs, yarn production costs accounts for 72.5% (Radhakrishnan & Kumar, 2018). Within the production of yarn, various factors are influencing the cost, including raw material, energy, labour, and capital (Wanassi et al., 2015). Raw material costs are particularly significant, accounting for 39-67% of yarn manufacturing costs, depending on where it is produced (Wanassi et al., 2016). Additionally, system costs make up 51.4% of the total of yarn production costs, with energy costs accounting for 2.9%, and waste management costs only accounting for 0.6% (Radhakrishnan & Kumar, 2018). Apart from yarn and fabric manufacturing, US$ 1.67 can be accounted to labour costs, US$ 1.17 to trims, US$ 1.09 to tax and duty US$ 0.50 to wash and finish, US$ 0.37 to finances and US$ 0.25 to transport (Brooks, 2015).

4.3 Total private costs

In total, the private costs for a pair of jeans are around US$ 7.85 (Brooks, 2015). Tseng & Hung (2014) supports this as it uses private costs of US$ 7.60 for a pair of jeans in their research. However, Mohibullah et al (2021) suggests that this figure may be lower than the actual costs, as it estimates fabric costs alone to be $US 3.49 and trims and accessories costs to be US$ 2.59. This study assumes private costs for a pair of virgin jeans to be US$ 7.85, which corresponds to € 7.13 (Google Finance, 2023).

However, it is important to note that the production costs are greatly impacted by the production location. Egels-Zandén (2016) reveals that producing jeans in Italy can cost four to five times more than in China. Also, in many cases workers in the textile industry face hazardous working conditions and violations of human rights in an effort to reduce production costs, as mentioned in section 2.1.4. These problems are not limited to East Asian nations alone, as inadequate wages and unfavourable working conditions have been documented in garment factories in Western countries such as the UK (Butler, 2022).

5 Private costs: Recycled jeans

5.1 Division of costs

In the case of recycled yarn, raw material costs make up more than half of the total cost (56.1%), followed by labour costs (24%). Other expenses such as auxiliary costs, waste costs, and spare parts for recycling machines constitute 10% of the total cost. The least significant contributors to the total cost of recycled fibre are energy and capital costs (Wanassi et al., 2018).

When using recycled yarn instead of virgin yarn, different private costs associated with jeans production are affected. For instance, by utilizing low-cost textile waste fabrics as raw materials instead of virgin material, the production costs for recycled material are decreased (Filho et al., 2019). Also, when using recycled cotton yarn instead of virgin cotton yarn, there are cost savings related to dyeing, as the recycled cotton yarn is already blue in colour. The savings include reduced costs for water, energy, dye, and effluent treatment after dyeing. In case of 100% recycled cotton, approximately €0.26 of dyeing costs are saved per pair of jeans (Radhakrishnan & Kumar, 2018).

On the other hand, even with the usage of low-cost textile waste fabrics, a considerable amount of textile waste remains unsuitable for processing. This is due to materials possibly being damaged, soiled or damp, containing mixed fibre compositions, and the including metal or plastic (Filho et al., 2019). The collection and sorting of these post-consumer garments is a labour-intensive and costly operation (Chevalier, 2022; Filho et al., 2019). This can be explained by the fact that the current state of technologies for sorting, separating, and processing individual fibre types (often found in blends) is still in development and has not yet been scaled up to industrial practice. Therefore, the collection of non-reusable textiles results in relatively high economic costs for collection, transport, and sorting that are not offset by their subsequent value through recycling (Filho et al., 2019). Also, according to Chevalier (2022), logistics costs have a significant impact on the overall costs of recycled textiles. The importance of the recycling process set-up is highlighted to minimize these costs. Furthermore, the spinning process is costly due to the conduction of multiple tests to determine the appropriate blends of mixed textiles (Chevalier, 2022). Finally, Fidan et al. (2021) suggests that the increased waste ratio in the spinning process of recycled cotton yarn contributes to higher costs when manufacturing recycled denim fabric.

5.2 Total private costs

From the literature, it was challenging to determine whether the production of recycled jeans is cheaper or more expensive than virgin jeans. The manufacturing costs of recycled cotton yarn depend on many aspects, namely the cost of waste material, collecting and sorting, shredding and fibre extraction, and the energy cost (Arafat & Uddin, 2022; Wanassi et al., 2018).

Some studies find that recycled cotton yarn is cheaper than virgin cotton yarn. According to Wanassi et al (2018), the price of one kg recycled yarn is $US 0.38, whereas one kilogram of virgin cotton fibre costs $US 1.15. However, it should be noted that this study did not involve an extensive collection and sorting process, as it utilized pre-consumer material as its input. Fidan et al. (2021) suggests that using recycled material is slightly cheaper than virgin material, as it values recycled cotton yarn between 1.30 and 1.40 $US/kg and virgin cotton between 1.50 and 1.80 $US/kg. Also Arafat & Uddin (2022) suggests slightly lower private costs for recycled yarn. It states that the private cost of yarn, incorporating 30% recycled material, amounts to 2.88 $US/kg, compared to a price of virgin yarn of 3.15 $US/kg. Though, this study also only includes pre-consumer material.

On the other hand, other studies indicate that recycled yarn from shredded textile waste is more expensive than virgin cotton yarn (Filho et al., 2019; McCauley & Jestratijevic, 2023; Pensupa, 2020). This is primarily due to the additional collection, sorting, shredding, and transportation between facilities (Kazancoglu et al., 2020). Investments in the mechanical recycling process result in a relatively low profit margin for shredded fibres compared to virgin cotton fibres (Hole & Hole, 2020; Jia et al., 2020). According to Chevalier (2022), this is the main reason why textile manufacturers have not fully integrated textile waste fibres into their production processes on an industrial scale. Also, Muthu (2020) highlights how challenging it is to set up a profitable mechanical recycling operation with a textile recycling company using leftover textiles. The competitive environment faced by textile waste collectors and shredders reduces the profit margin of shredded fibres further, making it difficult for textile recycling companies to invest in advanced recycling technologies (Neto et al., 2021). Finally, according to Watson et al. (2017), the sales margin from low-quality products made from mechanically post-consumer recycled fibres is not cost-efficient. The studies above do not provide any numerical information on the actual total private costs of recycled material.

Reblend, a Dutch company, has developed a recycled yarn called ReDenim, which is composed of 70% Dutch post-consumer cotton waste and 30% recycled polyester (rPET) (ReBlend, 2023). The yarn is manufactured by the Spanish company Recover, which collaborates with various clothing companies such as C&A, Mud Jeans, and Primark (Recover, 2023). The approximate cost of one kilogram of ReDenim yarn is about € 20, as estimated by de Wit (Personal communication, 24 April 2023). This range of pricing is supported by Manifutura, a German company that supplies recycled textiles to companies of all sizes. They indicate that 1 kilogram of their product, consisting of 55% post-consumer recycled polyester and 45% pre-consumer recycled cotton, costs € 22.5 (Manifutura, 2023).

The variation in information regarding the private costs of recycled cotton could potentially be attributed to the differing focus on pre-consumer and post-consumer recycled cotton in the reviewed studies. When considering post-consumer recycling, the labour-intensive and costly process of collecting and sorting post-consumer garments is included (Chevalier, 2022; Filho et al., 2019). Also, whether or not the studies include investment costs of recycling in their studies could contribute to the variation in information. Given the specific scope of this study, it is important that the private costs of recycled material reflect the utilization of post-consumer recycling in the Netherlands. Therefore this study selected private costs of 20 €/kg of recycled yarn as an input factor for the model, which corresponds to the price of the company ReBlend. This company uses post-consumer recycled cotton, in contrast to Arafat & Uddin (2022) and Wanassi et al (2018), who researched pre-consumer materials and do not mention investment costs. Based on the assumption of 0.882 kg of yarn per pair of jeans (3.1.2), and keeping all other aspects of jeans manufacturing constant, the total private costs for a pair of recycled jeans were estimated to be € 22.27. However, ReDenim does include 30% rPET, which means it does not completely represent the private costs of 100% recycled cotton completely accurately.

6 Environmental impacts of jeans

In this chapter, the results from deciding which environmental impacts were included in the study are outlined, as described in 3.1.2. First, the relevant environmental impact categories for textile production were identified. Second, a suitable set of impact categories was selected according to the available data on the environmental impacts of recycled jeans and monetization factors. Finally, the main (base) study and complementary studies are presented.

6.1 Relevant environmental impact categories for textile production

In nearly all life cycle assessment (LCA) research, there are three areas of protection or safeguard subjects: resource use, human health, and ecological consequence. The Sustainable Apparel Coalition (SAC) provides guidelines for the textile industry, which assesses the importance of the key impact categories for conducting LCAs of clothing (SAC, 2021). Below, the impact categories included in this study are shortly explained.

6.1.1 Abiotic resource depletion

Abiotic resources are non-living resources that can be either renewable, such as wind energy, or non-renewable, such as iron ore and crude oil (Saric & Nellström, 2019). This research included water use and depletion of fossil fuels, metals, and minerals. Water use refers to water that when it is used, it becomes unavailable in its original watershed for both human use and ecosystems. Similarly, the use of fossil fuels reduces the stock available for current and future generations (Huijbregts et al., 2016).

6.1.2 Contribution to climate change

The emission of greenhouse gases, such as carbon dioxide, methane, and nitrous oxide, is a significant contributor to climate change. These emissions lead to an increase in the atmospheric concentration of greenhouse gases measured in parts per billion (ppb), which in turn raises the radiative forcing capacity and ultimately drives up the global mean temperature. This temperature increase can cause various harmful impacts such as extreme weather patterns, reduced agricultural yields, and more frequent natural disasters. As a result, the economy, human health, and ecosystems can all suffer significant damage, including increased risks of disease and natural disasters. Contribution to climate change is expressed in kg CO_2 eq (Huijbregts et al., 2016).

6.1.3 Acidification

When acids are released into surface soils and waters, acidification occurs as a result. This process can have detrimental effects on ecosystems and biological organisms, leading to issues such as fish mortality and forest decline (Guinée, 2004). The acidification potential of a substance is a measure of the maximum amount of acidification it can cause and is typically expressed in terms of SO_2 eq (Baumann & Tillman, 2006).

6.1.4 Eutrophication

Eutrophication is caused by excessive flows of fertilisers containing nitrogen oxides and phosphates into different types of water bodies. These effects include an increase in algal blooming, which leads to oxygen deficiency and harms biological diversity by making the waters more turbid and hindering sunlight penetration. Eutrophication also alters nutritional levels and can harm local organisms (Saric & Nellström, 2019). LCAs can differ between eutrophication of terrestrial, marine and freshwater bodies of water.

6.1.5 Pollution

Pollution includes the release of toxic substances, ozone-depleting substances, radioactive substances, photooxidants and particulate matter into the environment.

Toxicity refers to the potential harm on human health and the environment due to the exposure to toxic substances. The measurement of toxicity is a complex process due to the intricate interactions of numerous systems. There is an abundance of unique chemicals that can cause various impacts, making the assessment challenging. Toxicity can be broadly categorized into two types: human toxicity and eco-toxicity. Human toxicity refers to the harmful effects that occur within the human body as a result of exposure to and production of toxic substances (Kazan et al., 2020). Eco-toxicity can be further divided into subcategories, including freshwater, marine water, and terrestrial eco-toxicity (Baumann & Tillman, 2006).

When ozone layer depletion takes place, the absorption of destructive UV rays from the sun is by the ozone layer is increased. This results in a higher percentage of UV-B radiation penetrating the Earth's surface, which has adverse effects on the health of humans and animals. Additionally, the depletion may negatively impact terrestrial and aquatic ecosystems, as well as biochemical materials and cycles (Baumann & Tillman, 2006).

Ionizing radiation (IR) encompasses the effects of both private exposure to radiation and the release of radioactive substances. Exposure to ionizing radiation can have harmful effects on both human health and animals (Guinée, 2004)

Also, photochemical oxidant formation has the potential to cause adverse effects on ecosystems, crops, and human health (Guinée, 2004). Ozone is a photooxidant that is produced in the lower atmosphere (troposphere) through the reaction of Volatile Organic Compounds (VOCs) and carbon monoxide (CO) in the presence of nitrogen oxides (NO_x).

Particulate matter (PM) is a significant atmospheric pollutant that poses a threat to human health. This pollutant consists of a complex mixture of solid and liquid particles from organic and inorganic suspended material. The PM fractions that have an aerodynamic diameter of less than 10 (PM10) and 2.5 (PM2.5) micrometres are the most extensively studied since they can easily enter and accumulate in the human respiratory system (Lelieveld et al., 2015).

6.1.6 Ecosystems and landscapes

The impact category of land use refers to the harm inflicted on ecosystems resulting from the occupation and alteration of land. Since various regions have distinct species diversity, not all forms of transformation have an equal impact on biodiversity. Thus, the type of land use must be considered, leading to three distinct categories: 'agricultural land transformation', 'urban land transformation', and 'natural land transformation' (Goedkoop et al., 2009).

6.2 Selection of impact categories

6.2.1 Step 1 Available LCA data on recycled jeans, denim fabric and cotton

Amicarelli et al (2022) compared the impact categories used in LCA studies on fibres and finds that the impact categories relating to the scope of "air" are the most extensively investigated, followed by "water", "soil", and "energy". This aligns with the findings of Akı et al (2020) that the most researched impact categories in textile LCA studies are contribution to climate change, freshwater use, land use, eutrophication potential, and abiotic resource depletion.

Appendix A shows a taxonomy that lists the used environmental impact categories for LCAs on virgin jeans, denim fabric and cotton, as well as recycled jeans, denim fabric and cotton. The table indicates that contribution to climate change, acidification potential, eutrophication potential, water use and land transformation are the most researched impact categories in the selected LCA studies specifically on jeans, denim fabric and cotton.

After filtering the studies according to the criteria in section 3.1.2, I selected the studies Fidan et al (2021), Esteve-Turrillas & de la Guardia (2017), Liu et al (2020) and Kazan et al (2020). Table 2 shows a list of impact categories that these studies assess. As the included impact categories in this research for both recycled and virgin jeans should be the same, some impact categories were excluded from this study due to a lack of available data in the case of recycled material. These impact categories are shown in red in table the table in appendix A and are left out of table 2.

Table 2 Selected impact reviewing the available LCA data on recycled jeans, denim fabric and cotton

Author and year of publication	Origin	Contribution to climate change	Acidification Potential	Terrestrial acidification	Eutrophication potential	Fresh water eutrophication	Marine Eutrophication	Water use	Abiotic depletion potential	Fossil depletion	Metal depletion	Cumulative energy demand	Marine ecotoxicity	Fresh water aquatic ecotoxicity potential	Terrestrial ecotoxicity potential	Human toxicity potential	Human toxicity-cancer	Human toxicity-non cancer	Land transformation	Ozone layer depletion	Photochemical ozone formation potential	Photochemical oxidant formation	Photochemical oxidation	Ionizing radiation	Particulate matter formation	Total
(Fidan et al., 2021)	Turkey	1	1		1			1				1														5
(Esteve-Turrillas & de la Guardia, 2017)	Spain	1	1		1			1	1			1		1	1	1	1	1	1	1	1		1			15
(Liu et al., 2020)	China	1	1	1		1	1			1	1		1	1	1	1			1	1		1		1	1	14
(Kazan et al., 2020)	Turkey	1	1		1				1				1			1				1						7
Total		4	4		3	1		2	2	1	1	2	2	2	2	3	1	1	2	3	1	1	1	1	1	39

6.2.2 Step 2 Available data on monetization factors

Table 3 shows the final selected impact categories depending on the available LCA data and the available data on monetization factors. For the impact category ‘land transformation’, data was used from the True Price report from 2020. The unit for the impact from the report from 2021 is ‘MSA (Mean Species Abundance)*ha’, which includes biodiversity. However, the LCAs only expressed this impact category in hectares. Also, for land transformation, the monetization factor for ‘other forest’ was selected. This is relatively low, but cotton cultivation takes place in many different regions with different biomes. This study rejected the impact categories metal depletion, human toxicity-cancer, human toxicity-non cancer, photochemical ozone formation potential and photochemical oxidation due to lacking available data on monetization factors associated with these impact categories.

Table 3 Monetization factors for the selected impact categories

Source	Impact	Footprint indicator	Unit	Monetisation factor
(True Price, 2021)	Abiotic resource depletion	Scarce water use	€/m^3	1.27
(True Price, 2021)	Abiotic resource depletion	Fossil fuel depletion	€/kg oil eq	0.448
(True Price, 2021)	Climate change	Greenhouse Gas emmissions	€/kgCO2eq	0.157
(True Price, 2021)	Air pollution	Acidification	€/kg SO2 eq	4.7
(True Price, 2021)	Water pollution	Fresh water eutrophication	€/kg P eq to freshwater	203
(True Price, 2021)	Water pollution	Marine Eutrophication	€/kg N eq to marine water	14.1
(True Price, 2021)	Air/water/soil pollution	Terrestrial ecotoxicity	€/kg 1,4-DB emitted to industrial soil eq	0.0003
(True Price, 2021)	Air/water/soil pollution	Marine ecotoxicity	€/kg 1,4-DB emitted to seawater eq	0.0019
(True Price, 2021)	Air/water/soil pollution	Freshwater ecotoxicity	€/kg 1,4-DB emitted to freshwater eq	0.0406
(True Price, 2021)	Air pollution	Ozone layer depleting emissions	€/kg CFC-11 eq	56.4
(True Price, 2021)	Air pollution	Photochemical oxidant formation	€/kg NMVOC	0.83
(True Price, 2020)	Land transformation	Land transformation of 'other forests'	€/ha	2050
(Schroten et al., 2018)	Human	Human toxicity potential	€/kg 1,4 DB-eq	0.0991
(Schroten et al., 2018)	Air pollution	Ionizing radiation	€//kg kBq U235-eq.	0.0461
(Schroten et al., 2018)	Air pollution	Particulate matter formation	€/kg PM10 eq	39.2

6.3 Selection of base study and complementary studies

Out of the four studies on recycled material, this study chose Fidan et al (2021) as the base study, which served as the main source of data extraction for the environmental impacts of jeans used in this study. I chose this study for the following reasons. Firstly, it specifically focuses on the production of denim fabric. Secondly, it covers a wide range of processes involved in fabric production. Thirdly, it uses post-consumer cotton waste to produce recycled cotton instead of pre-consumer waste, making it more relevant for the policy changes in the Netherlands. Finally, the production of recycled denim takes place in Turkey, and recycled cotton fibres from Turkey are already being used in products sold in the Dutch market, such as jeans from the company "Kuyichi" (Project CECE, 2023).

The remaining three studies complemented the base study by providing additional information on impact categories. The first complementary study is Esteve-Turrillas & de la Guardia (2017) and examines the environmental impact of the production of 1 kg of recycled cotton yarn at the Spanish company 'Hilaturas Ferre'. The study only considers three crucial stages in cotton yarn production, namely cotton cultivation/collection, ginning/cutting, and dyeing, and excludes the spinning of the yarn, textile production, selling and usage, and final disposal. Nevertheless, it covers the largest number of impact categories (15) among all LCA studies on recycled material. The second complementary study is Liu et al (2020) on the environmental impact of producing 1000 kg of recycled cotton yarn for fabric weaving in China. It covers the washing, breaking, separating, packaging, and spinning stages, but excludes dyeing in recycled cotton yarn scenarios due to the predetermined colours of the waste cotton fibres. The final complementary study is Kazan et al (2020) and examines the environmental impact of producing 250 kg of recycled shirts in Turkey. The study assesses the environmental impacts from raw materials to final products, distinguishing between yarn, fabric, and garment production.

7 Environmental costs: Virgin Jeans

This chapter describes the life cycle for a pair of 100% virgin jeans, followed by an overview and clarification of the selected LCA results. Finally, the environmental costs were calculated.

7.1 The life cycle

Figure 3 illustrates the complete life cycle of a pair of jeans made from 100% virgin cotton. The process starts with the agricultural phase. The amount of water used in cotton production varies significantly across countries, mainly due to differences in both climate conditions and the specific water requirements for growing cotton (Chen et al., 2021). Most LCAs from the table in Appendix A tend to look at cotton cultivation in either China or Turkey. For this study, cotton cultivation took place in Turkey. Turkey is among the countries that have climatic conditions that are less favourable for cotton cultivation due to high evaporative demand and low effective rainfall, resulting in a significant irrigation requirement that puts a strain on local water resources and increases the environmental burden (Chen et al., 2021).

After cotton cultivation, the cotton is harvested, cleaned, and pressed into bales (baling). Then the cotton is separated from the seeds (ginning) (Fidan et al., 2021). The cotton fibres are combed, drawn, and spooled to create cotton yarn (Esteve-Turrillas & de la Guardia, 2017). After dyeing, the yarn is sized and woven to create fabric. This is cut and sewn to make a pair of jeans. The jeans are then ironed, packaged, and sold to the consumer, who can wash and eventually dispose them. The garment may end up in a landfill where energy can be generated, or it can be collected for recycling or reuse (Esteve-Turrillas & de la Guardia, 2017).

Transportation occurs between some of these steps, and while some LCAs consider this factor, the environmental impacts resulting from transportation are relatively insignificant, representing less than 1% of the total environmental impacts (Y. Zhang et al., 2015).

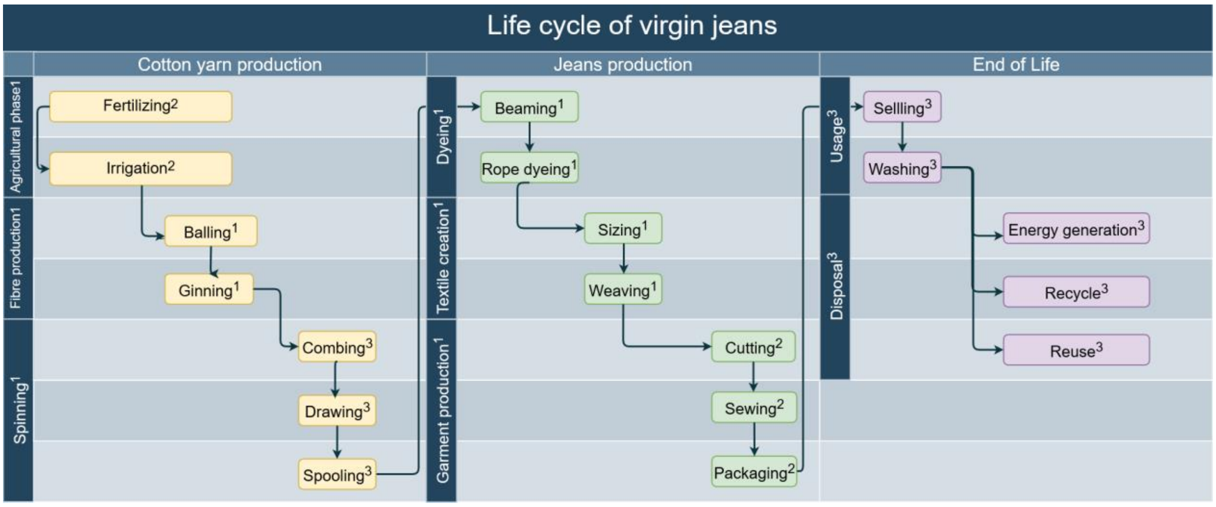

Figure 3 Life Cycle of Virgin Jeans, sources: 1. (Fidan et al., 2021), 2. (Kazan et al., 2020), 3. (Esteve-Turrillas & de la Guardia, 2017)

7.2 The environmental impacts

Table 4 shows the values that were selected for the environmental impacts of virgin jeans, according to the method in section 3.1.2.

Table 4 Virgin jeans: Environmental Impacts and Costs (EC)

Impact category	Unit	Impact	Source	EC (€)	Relative EC (%)
Contribution to climate change	Kg CO2 eq	5.011	(Fidan et al., 2021)	0.79	3.24
Acidification Potential	Kg SO2 eq	0.037	(Fidan et al., 2021)	0.18	0.72
Fresh water eutrophication	kg P eq	0.005	(Fidan et al., 2021)	1.10	4.55
Marine Eutrophication	kg N eq	0.002	(Liu et al., 2020)	0.03	0.11
Water use	m^3	3.099	(Liu et al., 2020)	3.94	16.22
Fossil fuel depletion	kg oil eq	2.538	(Fidan et al., 2021)	1.14	4.69
Marine ecotoxicity	kg DCB eq	9.564	(Kazan et al., 2020)	0.02	0.07
Fresh water ecotoxicity potential	kg 1,4-DB eq	0.0003	(Liu et al., 2020)	0.00	0.00
Terrestrial ecotoxicity potential	kg 1,4-DB eq	0.00003	(Liu et al., 2020)	0.00	0.00
Human toxicity potential	kg DCB eq	0.930	(Kazan et al., 2020)	0.09	0.38
Land transformation	ha/year	0.008	(Arvidsson, 2019)	16.91	69.72
Ozone layer depletion	kg CFC-11 eq	0.00000	(Liu et al., 2020)	0.00	0.00
Photochemical oxidant formation	kg NMVOC	0.005	(Liu et al., 2020)	0.00	0.02
Ionizing radiation	kg U235 eq	0.022	(Liu et al., 2020)	0.00	0.00
Particulate matter formation	kg PM10 eq	0.002	(Liu et al., 2020)	0.07	0.28
Total				**24.26**	**100.00**

7.2.1 Abiotic resource depletion

Kazan et al (2020) states that the majority of fossil fuel depletion (74%) occurs during fabric manufacturing. Though, according to Akı et al (2020) the dyeing process has the greatest impact. Some LCAs express this impact category as non-renewable energy demand in Mj. These results were converted to kg oil eq by multiplying the amount of Megajoules with 0.0238846 (Energy converter, 2023). The selected outcome from Fidan et al (2021) is relatively high compared to other studies.

The agricultural phase is responsible for 88% of water use (Akı et al., 2020). Cotton cultivation requires large amounts of water compared to other fibres (Fletcher, 2014). As there is a large difference between the outcome of the base study from Fidan et al (2021) (1.754 m^3) and of the complimentary study from Liu et al (2020) (3.099 m^3) and because of the importance of water use in true cost accounting, I reviewed some other LCAs to test the accuracy. These are shown in table 5. For this study, I chose the median of the different results, which is 3.099 m^3 from the study by Liu et al (2020).

Table 5 LCA results on water use of virgin jeans

Author and year of publication	water use (m^3)
(Fidan et al., 2021)	1.754
(Liu et al., 2020)	3.099
(Arvidsson, 2019)	2
(Hackett, 2015)	3.781
(Levi Strauss and Co, 2015)	3.772

Sources: (Fidan et al., 2021), (Liu et al., 2020), Arvidsson (2019), (Hackett, 2015), (Levi Strauss and Co, 2015)

7.2.2 Contribution to climate change

The production stages with the highest global warming potential (GWP) values are cotton fibre production, yarn manufacturing, and woven fabric production. The field operations, fertilizer production, and energy supply for the ginning process contribute to GWP in cotton cultivation. Electricity used in the ginning process also contributes to GWP due to fossil fuel-based energy production (Kazan et al., 2020). These findings are supported by Akı et al (2020). The chosen value from Fidan et al (2021) of 5.01 kg CO_2 eq is relatively low compared to the findings of Kazan et al (2020) and Liu et al (2020), who both found 9.7 kg CO_2 eq.

7.2.3 Acidification potential

In the production of jeans, the primary sources of acidification potential are seed cotton cultivation (61%) and fabric manufacturing (28%). The main cause of acidification during cotton cultivation is field operations, particularly the use of fertilizers which results in emissions of ammonia and nitrogen monoxide. In fabric manufacturing, the dyeing stage is the main contributor due to extensive use of chemicals which lead to emissions of NO_x and SO_2 (Kazan et al., 2020). The chosen value from Fidan et al (2021) is very similar to the outcome of Kazan et al (2020).

7.2.4 Eutrophication

The agricultural stage is responsible for the most significant eutrophication impacts, primarily due to field operations and the use of PO_4 fertilizer. As a result, phosphate and nitrate content runoff from fertilized lands, which increases the eutrophication potential (EP) of surface waters (Akı et al., 2020; Kazan et al., 2020). In this study, freshwater eutrophication is expressed in kg P eq to freshwater (True Price, 2021). As some LCAs, express this in kg PO_4 eq, it was necessary to convert this amount to P by dividing the weight by 3 (EPA, 2012). The chosen value from Fidan et al (2021) for freshwater eutrophication is very similar to the outcome of Kazan et al (2020). The chosen value for marine eutrophication is more uncertain, as it is from the study by Liu et al (2020), which only includes the production of cotton yarns.

7.2.5 Pollution

The fabric manufacturing stage is responsible for the largest share of eco-toxicity impacts. In fabric production, the use of high-quality process water and natural gas increases the eco-toxicity potential. Additionally, electricity production in fossil fuel power plants results in high levels of inorganic emissions (Kazan et al., 2020). The emitted emissions flow into aquatic systems and increase eco-toxicity impacts. Also, the use of fertilizers and pesticides during the agricultural stage contribute to eco-toxicity. The chosen value from Liu et al (2020) for terrestrial ecotoxicity could be on the lower side, as this study only researches the production of cotton yarns and does not include fabric and garment manufacturing. Similarly, the chosen value from Liu et al (2020) for freshwater ecotoxicity could be on the lower side. Possibly for the same reason the chosen value from (Kazan et al (2020) is higher than that from Liu et al (2020) for marine ecotoxicity. Also, around 70% of the human toxicity impacts occur during the fabric manufacturing stage. This is primarily due to the emission of volatile organic compounds (VOCs) from the burning of natural gas. The chosen value from Kazan et al. (2020) is again higher than that from Liu et al (2020).

The main source of ozone layer depletion is fabric manufacturing (91%). During fabric production, wet processes make use of high-quality process water and may result in the emission of halogenated organic matter to the air. Additionally, the treatment procedures consume electricity produced from fossil fuels, which can further increase ozone layer depletion. According to Arvidsson (2019) the biggest ionizing radiation impacts take place during the use and end-of-life phase of jeans, mainly during washing and drying of the jeans. As for photochemical oxidant formation, the fabric manufacturing stage contributes most (Saric & Nellström, 2019). According to Saric & Nellström (2019), the agriculture phase contributes most to particulate matter formation. The chosen values for ozone depletion, ionizing radiation, photochemical oxidant formation and particulate matter formation from Liu et al (2020) could be on the lower side, as this study only researches the production of cotton yarns.

7.2.6 Ecosystems and landscapes

According to Akı et al (2020), 99% of land transformation comes from the agricultural phase for cotton fibre production. Because only the study from Liu et al (2020) provides an outcome on land transformation and because of the importance of the impact category in true cost accounting, I reviewed other LCAs that specifically study land transformation. The outcomes are in table 6. For this research, I chose the median, which is 0.0083 ha/year from the study by Arvidsson (2019).

Table 6 LCA results on land transformation of virgin jeans

Author and year of publication	Land transformation (ha/year)
(Liu et al., 2020)	0.00051
(Arvidsson, 2019)	0.0083
(Hackett, 2015)	0.012
(Levi Strauss and Co, 2015)	0.0115
(La Rosa & Grammatikos, 2019)	0.00644

Sources: (Liu et al., 2020), Arvidsson (2019), (ett, 2015), (Levi Strauss and Co, 2015),(La Rosa & Grammatikos, 2019)

7.3 Environmental costs

The environmental costs are displayed in table 4. The total environmental cost for the production of a pair of 100% virgin cotton jeans is € 24.26. The impact categories that have the greatest influence on the total environmental costs of virgin jeans are land transformation, water use, fossil fuel depletion and freshwater eutrophication.

8 Public Environmental costs: Recycled Jeans

This chapter describes the life cycle for a pair of 100% recycled jeans, followed by an overview and clarification of the selected LCA results. Finally, the environmental costs were calculated.

8.1 The Life Cycle

Figure 4 illustrates the complete life cycle of a pair of jeans made from 100% recycled cotton. In contrast to virgin jeans, the process starts with the collection and sorting of post-consumer textile waste. The fabric is then mechanically recycled through washing, shredding, and cutting (Fidan et al., 2021). The recycled cotton fibres are then combed, drawn, and spooled to create cotton yarn (Esteve-Turrillas & de la Guardia, 2017). Some studies suggest that dyeing the recycled material is unnecessary since it is already coloured (Esteve-Turrillas & de la Guardia, 2017; Liu et al., 2020). However, Fidan et al (2021) includes dyeing in the process. Further steps are the same as the life cycle of virgin jeans, which can also be seen in figure 3.

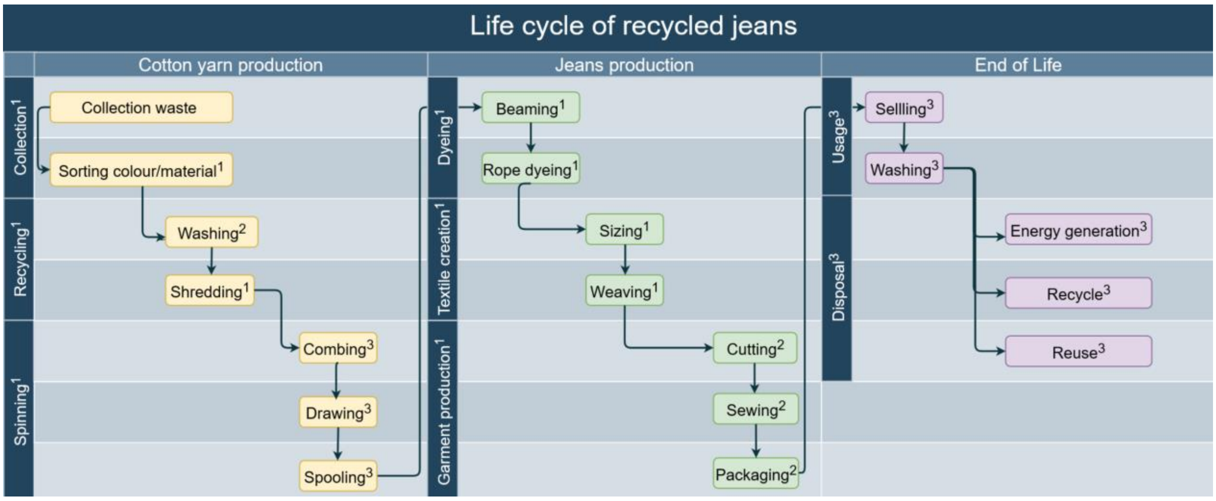

Figure 4 Life cycle of Recycled Jeans, sources: 1. (Fidan et al., 2021), 2. (Kazan et al., 2020), 3. (Esteve-Turrillas & de la Guardia, 2017)

8.2 The environmental impacts

Table 7 shows the values that were selected for the environmental impacts of virgin jeans, according to the method in section 3.2.2.

Table 7 Recycled jeans: Environmental Impacts and Costs

Impact category	Unit	Impact	Source	EC (€)	Relative EC (%)
Contribution to climate change	Kg CO2 eq	2.503	(Fidan et al., 2021)	0.39	13.86
Acidification Potential	Kg SO2 eq	0.014	(Fidan et al., 2021)	0.07	2.32
Fresh water eutrophication	kg P eq	0.002	(Fidan et al., 2021)	0.32	11.13
Marine Eutrophication	kg N eq	0.002	(Liu et al., 2020)	0.03	0.89
Water use	m^3	0.037	(Fidan et al., 2021)	0.05	1.64
Fossil fuel depletion	kg oil eq	0.968	(Fidan et al., 2021)	0.43	15.29
Marine ecotoxicity	kg 1,4-DB eq	2.790	(Kazan et al., 2020)	0.01	0.19
Fresh water ecotoxicity potential	kg 1,4-DB eq	0.000	(Liu et al., 2020)	0.00	0.00
Terrestrial ecotoxicity potential	kg 1,4-DB eq	0.002	(Liu et al., 2020)	0.00	0.00
Human toxicity potential	kg 1,4-DB eq	0.155	(Kazan et al., 2020)	0.02	0.54
Land transformation	ha	0.000	(Esteve-Turrillas et al., 2017)	0.00	0.00
Ozone layer depletion	CFC-11 eq	0.00000	(Liu et al., 2020)	0.00	0.00
Photochemical oxidant formation	kg NMVOC	0.005	(Liu et al., 2020)	0.00	0.14
Ionizing radiation	kg U235 eq	0.067	(Liu et al., 2020)	0.00	0.11
Particulate matter formation	kg PM10 eq	0.039	(Liu et al., 2020)	1.53	53.89
Total				**2.84**	**100.00**

8.2.1 Abiotic resource depletion

Some LCAs express the impact category of fossil fuel depletion as non-renewable energy demand in Megajoules (Mj). These results were converted to kg oil eq by multiplying the amount of Mj with 0.0238846 (Energy converter, 2023). This study selected the value from Fidan et al (2021) of 0.97 kg oil eq was chosen, which is exactly the same as Liu et al (2020) suggests and quite similar to the outcome of Kazan et al (2020)

For water use, this study selected the value from Fidan et al (2021) of 0.037 m^3, which is relatively low compared to the 0.474 m^3 Liu et al (2020) suggests. However, this can possibly be the result of the additional 'washing' step Liu et al (2020) incorporates.

8.2.2 Contribution to climate change

This study chose the value from Fidan et al (2021), which is relatively low compared to the results from Kazan et al (2020) and Liu et al (2020). A possible explanation is that Kazan et al (2020) includes garment production in their study, whereas Fidan et al (2021) only looks at denim fabric.

8.2.3 Acidification potential

This study selected the value from Fidan et al (2021) of 0.014 kg SO_2 eq, which is relatively high compared to the other three studies.

8.2.4 Eutrophication

This study selected the value from Fidan et al (2021) for freshwater eutrophication, which is relatively high compared to the other three studies. I chose the value from Liu et al (2020) for marine eutrophication. The research only focusses on cotton yarn production, but it is the only study that looks specifically at marine eutrophication.

8.2.5 Pollution

This study selected the value from Liu et al (2020) for terrestrial ecotoxicity. The value only incorporates the production of cotton yarn. I chose the value from Kazan et al (2020) of 2.79 kg 1.4-DB eq for marine ecotoxicity. Though the study from Liu et al (2020) found a value of 0 kg 1.4-DB eq, most toxicity impacts occur during fabric and shirt production, which are not included in this study. I did choose the value from Liu et al (2020) of 0.0004 kg 1,4-DB eq for freshwater ecotoxicity, which could also be on the lower side. I chose the value from Kazan et al (2020) of 0.16 kg DCB eq for human toxicity. Liu et al (2020) and Esteve-Turrillas & de la Guardia (2017) provide much lower values. However, most impact occurs during fabric and garment production, which were not included in these studies.

This study selected the values for ozone layer depletion, ionizing radiation, photochemical oxidant formation and particulate matter formation from Liu et al (2020), which could be on the lower side.

8.2.6 Ecosystems and landscapes

This study selected the value from Esteve-Turrillas & de la Guardia (2017) of 0 ha, which is due to the lack of an agricultural phase in the production process. However, Levi Strauss and Co (2015) does dedicate a total of 2.7 m^2/year land transformation to fabric manufacturing, transport, logistics and customer care.

8.3 Environmental costs

The results for the environmental costs of recycled jeans are presented in table 7. The total environmental cost for the production of a pair of 100% recycled cotton jeans is € 2.84. The impact categories that have the greatest influence on the total environmental cost are particulate matter formation, human toxicity potential, fossil fuel depletion and contribution to climate change.

8.4 Comparison virgin and recycled jeans

After removing the impact categories with a relative environmental cost of less than 1%, as described in 3.1.2, seven impact categories were left to be included in the model described in 3.2. Figure 5 illustrates these categories and compares the actual environmental costs of virgin and recycled jeans. When only including the impact categories with a relative environmental costs of more than 1%, the total environmental cost for recycled jeans is € 2.78, whereas for virgin jeans it is € 24.23. The environmental costs are higher for virgin jeans across all impact categories, except for particulate matter formation. The reduction in land transformation and water use through the use of recycled materials is particularly impactful.

Figure 5 Comparison of the environmental costs of 100% recycled and 100% virgin jeans

9 Lifespan of jeans

This chapter quantifies the relation between lifespan and the recycled proportion. First, it describes the process of determining the lifespan of jeans, including defining the term and identifying the factors that influence it. Then it discusses lifespan of virgin jeans is, followed by an explanation of how the lifespan changes when recycled material is added. Finally, this information is combined to formulate the lifespan equation (4) that will be used in the model in section 3.2.

9.1 Determining the lifespan

9.1.1 Defining lifespan

In Benkirane et al (2019a, p.60), lifespan is defined as “the willingness to wear a garment under normal conditions”. The lifespan of a garment can be evaluated and quantified in different ways. These include the number of years it has been used, the frequency of wears, the number of cleaning cycles, and the number of users (Klepp et al., 2020).

According to Laitala & Klepp (2015), assessing the lifespan of a garment solely in terms of years does not provide enough information into its resource efficiency. Someone can own a large number of clothes, which can increase the chance that a garment is rarely used while still increasing its lifespan in years. Instead, taking into account the number of times a garment is worn is recommended as it reveals whether or not the item is being used effectively (Klepp et al., 2020). This approach is interesting because the longer a garment is actively used by a consumer, the more it can potentially offset the production of new textiles, leading to reduced environmental impacts (Piippo et al., 2022). Therefore, this study measured the lifespan of garments in terms of hours of wear.

9.1.2 Factors influencing the lifespan

According to Piippo et al (2022), the quality of a garment is closely related to its lifespan. However, the quality of a garment can be subjective and objective, and can depend on both extrinsic and intrinsic product attributes. Also, context plays a role in how quality is evaluated (Swinker & Hines, 2006). Laitala et al (2015) identifies up to 70 reasons for garment disposal, including defects, size and fit issues, as well as changes in personal taste and style.

Objective quality is measurable and quantifiable and can be assessed prior to use. In contrast, subjective quality is based on factors such as the garment's quality, the user's behaviour, and their experience of using the garment (Connor-Crabb & Rigby., 2019). Disposal of garments is often attributed to subjective factors such as changes in taste and aesthetics, disliking the style, colour, or print (Piippo et al., 2022) . The personal and subjective evaluation of garment quality can influence how the clothing is worn and cared for, which in turn can have an impact on its physical condition over time. How clothing is worn and cared for depends on the wearer’s lifestyle factors. More harmful daily activities, such as cycling and gardening will deteriorate clothing quality faster than less harmful activities. Garment quality is also dependent on choices made during the laundry process, as well as a wearer’s choice to alter, modify and repair clothing (Connor-Crabb & Rigby, 2019).

While the quality and lifespan of clothing depends on many different factors related to its technical and social durability and practicality, technical quality deficiencies remain the primary reason for clothing disposal (Laitala & Klepp, 2015). Deficiencies in quality, such as holes, tears, a worn appearance, faded colour, loss of elasticity, changes in shape, and pilling, account for the largest group of reasons for clothing disposal (Laitala et al., 2015; L. Zhang et al., 2020). Also, high technical quality enables products to remain in use for longer and even to be used by multiple individuals (Piippo et al., 2022). Therefore, this study only focussed on the objective quality of jeans, which consists of measurable and quantifiable indicators of technical quality that can be assessed prior to use.

9.1.3 The relationship between technical quality and lifespan

Benkirane et al (2019b) discusses the use of a CoQ (Consumer-Oriented Quality) score to estimate a garment’s lifespan based on objective aging factors. The results show a promising correspondence between the CoQ score and the evaluated lifespan in the case of T-shirts. The study identifies five categories of technical damage that influence disposal decisions. Ranked in order of relevance, these include holes, loss of shape, opened seams, loss of colour, and pilling. The ability of clothing to withstand these stress factors are found to strongly impact their lifespan.

In a subsequent study, researchers aimed to establish a relation between the quality score (CoQ) and various manufacturing and structural parameters, including fabric thickness (mm), yarn tenacity (Cn/Tex), elongation at break (mm), and twist ratio (Tpm). Based on their findings, the most significant parameters contributing to high consumer-oriented quality, and therefore an increased garment lifespan, are high tenacity, high elongation at break capacity and a dense structure (Benkirane et al., 2022). To estimate the lifespan of a garment, this study focussed on the indicators of tenacity and elongation at break.

9.2 Lifespan of virgin jeans

Cooper et al (2013) suggests the target lifespan of jeans is 3600 hours, equivalent to 300 usages over four years. However, in reality people typically wear their jeans for 233 times over 3.1 years. This finding aligns with the research from the UK's Waste and Resources Action Programme (WRAP), which states that the average lifespan of garments in the UK is 3 years (Xie et al., 2021). However, Gwozdz et al (2017) conducted a survey across four countries and found that consumers estimated wearing jeans only about 42 times on average, totalling approximately to only 504 hours of wear. Also, wardrobes often contain items that are never worn (Klepp et al., 2020). For this study, the target lifespan for a pair of virgin jeans was assumed to be 3600 hours of wear (300 usages) until the garment is worn out.

9.3 Lifespan of recycled jeans

In general, recycled cotton fibres have lower technical properties than virgin cotton (Ütebay et al., 2019). Blending post-consumer waste results in an even higher degradation of yarn properties than when using pre-consumer waste (Arafat & Uddin, 2022). As cotton ages, it undergoes major structural and complex chemical changes, resulting in a reduction of several technical properties (Johnson et al., 2020).

For example, air permeability decreases when using recycled yarns because these are thicker than virgin cotton yarn (Radhakrishnan & Kumar, 2018). Recycled yarn requires high twist during spinning due to low fibre migration, which results in stiffer and less moisture- absorbent yarn (Arafat & Uddin, 2022). Recycled fibres do not take finishes and dyes adequately (Al-Sabaeei et al., 2021). As the percentage of recycled cotton fibre increases, there is a gradual increase in thin, thick places, neps, and hairiness, while also the weight and thickness increase (Wanassi et al., 2015). Finally, multiple studies mention an impact on technical properties related to strength, such as lower tenacity and decrease in breaking elongation of recycled yarns (Arafat & Uddin, 2022). As described in 9.1.3, this study focussed on the impact of adding recycled material on the lifespan of jeans by examining its effect on tenacity and breaking elongation.

Tenacity is the conventional method for evaluating the durability of textile goods, including yarn and fibre ropes. This measurement is determined by dividing the breaking load of the material by its mass per unit length and is typically expressed in N/tex (Cesar et al., 2009). Typically, high-quality cotton fibres have lengths between 25–65 mm, while lower quality fibres have lengths of 10–25 mm (Johnson et al., 2020). Recycled fibres have lengths of approximately 10-15 mm (Esteve-Turrillas & de la Guardia, 2017). When subjected to tensile loading, short fibres within a yarn tend to slip. Compared to yarn made from 100% virgin cotton, blended yarns containing recycled fibres have lower tenacity values that decrease proportionally with an increase in the percentage of recycled fibres (Arafat & Uddin, 2022; Halimi et al., 2008; Wanassi et al., 2018). The addition of 50% recycled material results in a tenacity reduction of 13.70% (Halimi et al., 2008). This is similar to the finding from Contin et al (2022).

Elongation at break refers to the maximum extension of a fibre before it breaks, expressed as a percentage of the original length. It indicates how easily a fibre can be stretched. When a fibre has high breaking elongation, it is known to be easily stretchable under small loads (Bunsell, 2018). The breaking elongation of yarn is affected by fibre length and the arrangement of the fibres within the yarn. Recycled fibres, with their high number of short fibres, tend to slip out easily under load, leading to lower yarn elongation (Arafat & Uddin, 2022). The studies Arafat & Uddin (2022), Halimi et al (2008) and Radhakrishnan & Kumar (2018) investigate the effects of incorporating recycled cotton into yarn on its elongation. The findings of their studies are presented in figure 6. All studies indicate a decreasing trend in elongation at break with an increase in recycled fibre content. However, there are differences in the extent of this decrease. Arafat & Uddin (2022) reveals a more significant decrease in yarn elongation when recycled material is added compared to Radhakrishnan & Kumar (2018). Halimi et al (2008) demonstrates only a minor reduction in yarn elongation, which is also supported by Fidan et al (2021). Due to the strong variations in results, this study only concentrated on the effects on the lifespan of tenacity reduction rather than the effects of elongation at break as a result of adding recycled material.

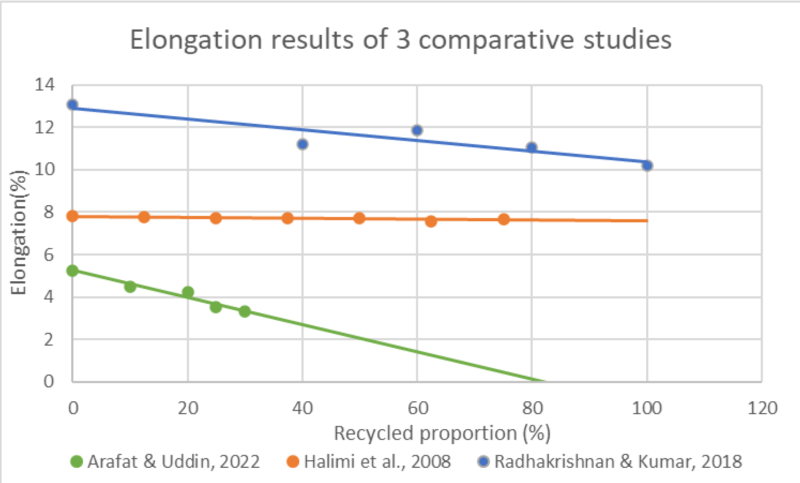

Figure 6 Impact of adding recycled material on the elongation at break of the yarn. Sources:(Arafat & Uddin, 2022; Halimi et al., 2008; Radhakrishnan & Kumar, 2018)

9.4 Quantifying the relation between lifespan and recycling proportion

To be able to quantify the relation between lifespan and recycling proportion, this study made some assumptions.

Firstly, the study assumed that the lifespan of jeans declines linearly in the same way as tenacity when the proportion of recycled material increases, as stated by Arafat & Uddin (2022) and Wanassi et al (2018). The addition of 50% recycled material results in a decline in tenacity of 13.70% (Halimi et al., 2008). Correspondingly, the lifespan also decreases by the same percentage with a 0.5 recycled proportion.

Secondly, the study assumed that the lifespan of 100% virgin jeans corresponds to 3600 hours of wear (Cooper et al.,2013).

Thirdly, the study assumed that the quality and therefore lifespan of jeans remain unchanged with a proportion of 0.2 recycled material. While the tenacity of recycled material tends to decline once it is added, research suggests that a certain amount of recycled material can be incorporated without significantly impacting quality or lifespan. For instance, Halimi et al (2008) finds that under optimal spinning conditions, blending recycled materials in proportions between 15% and 25% results in negligible changes in quality. Additionally, Radhakrishnan & Kumar (2018) recommends adding 20% of recycled material to produce yarn qualities nearly equivalent to those made from virgin cotton. Also, Spathas (2017) notes that fabrics containing mechanically recycled material should not exceed 20-30% recycled fibres before experiencing a significant reduction in quality. On the other hand, Luiken & Bouwhuis (2015) suggests that it is even possible to add 50% of post-consumer denim waste without compromising the mechanical properties. However, this is refuted by Wanassi et al (2016), who states that even in an optimized situation all tensile properties of a 50/50 blended yarn are less (between 73% and 84%) than that of 100% cotton yarn.

Fourthly, the study assumed that jeans can be created using a maximum recycled proportion of 0.5. Even though some studies work with the possibility of using a recycled material proportion of 100%, it is currently technically not possible to produce jeans made of more than 50% mechanically recycled post- consumer material (Fidan et al., 2021; Radhakrishnan & Kumar, 2018).

Based on these assumptions, the parameters from equation (4) are given the following values:

$\alpha = -1643.3$

$\beta = 3928$

The parameters for the constraints (6) and (8) are given the following values:

$r_{max} = 0.5$

$L_{max} = 3600$

Figure 7 illustrates the assumptions this study made to depict the relationship between the proportion of recycled material added and the lifespan.

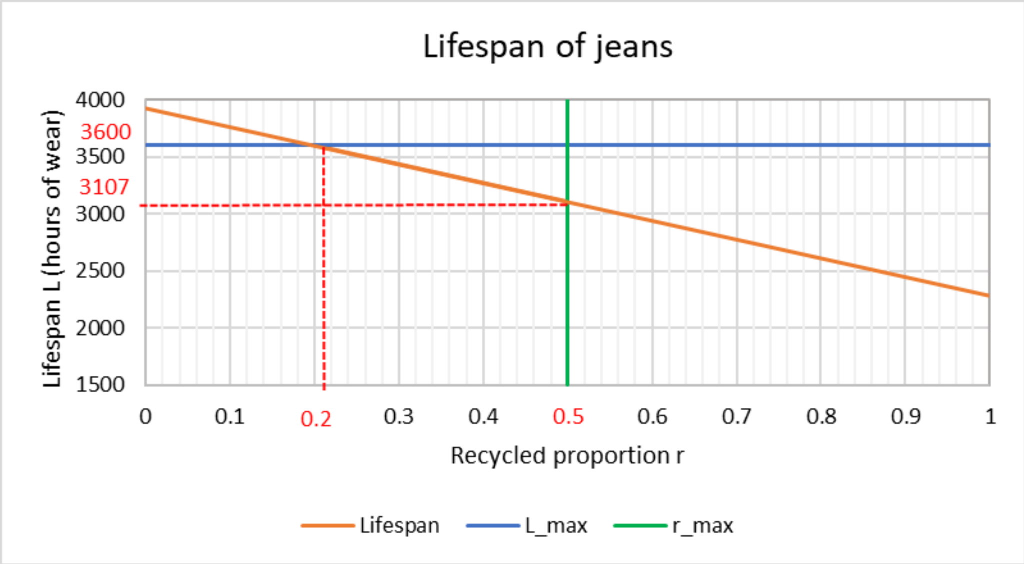

Figure 7 Quantification of the lifespan of jeans

10 Model results

Using the model described in Chapter 3 and the data collected in Chapter 4 to 9, the optimal recycled proportion for a pair of jeans was determined. Appendix B shows an overview of all the parameters used in the model and their sources.

10.1 Optimal recycling proportion

Figure 8 shows the environmental, private and total true costs for each proportion of recycled material. As the recycled proportion of a pair of jeans increases, the private costs increase and the environmental costs and total true costs decrease. At a recycled proportion of 0.47 the private costs exceed the environmental costs. From this figure, it may seem like a recycled proportion of 0.5 is preferred, but this figure does not yet include the effects of adding recycled material on the lifespan of the jeans.

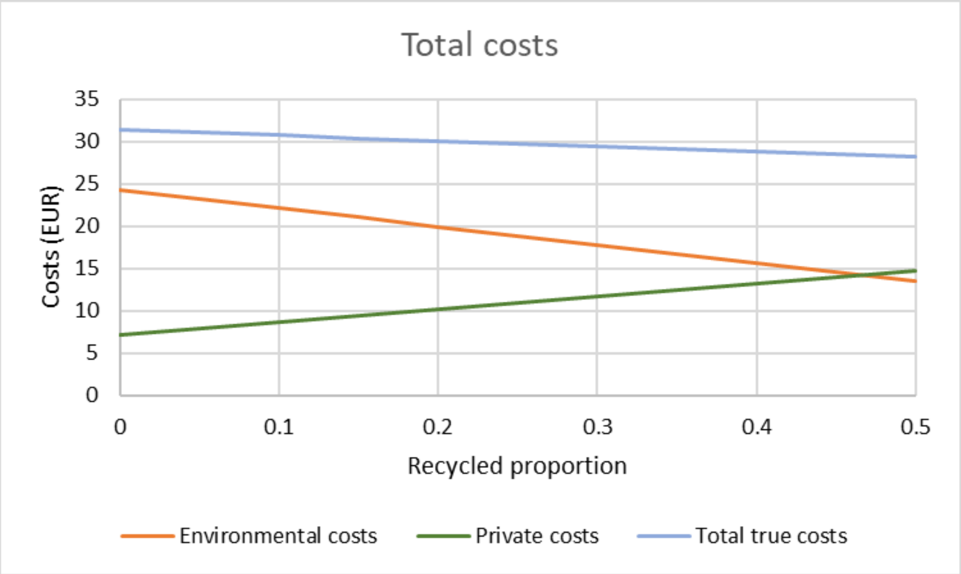

Figure 8 Total costs for each proportion of recycled material added

In Figure 9, equation (4) is added, which represents the lifespan. According to section 3.2 and equation (1), the optimal proportion of recycled material occurs when the total true costs divided by the lifespan is minimized. This occurs at the point where there is the greatest distance between the two graphs in figure 9, which in this case is at a recycled proportion of 0.2.

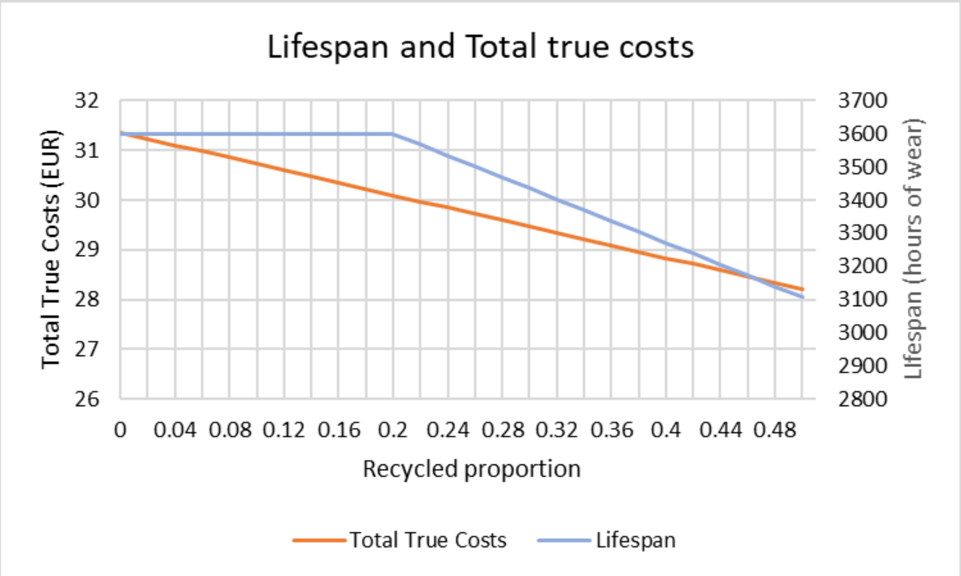

Figure 9 Lifespan and total true costs shown in the same figure

According to the model, the optimal recycled proportion for a pair of jeans is 0.2. In this case, the total true costs per hour of wear are the lowest, namely 0.0084 (figure 10). Figure 8 shows that at a recycled proportion of 0.2 for a pair of jeans, the environmental costs are €19.96 and the private costs are 10.16€, resulting in the total true costs of 30.12€. The lifespan of a pair of jeans with 0.2 recycled proportion is 3600 hours of wear (figure 9).

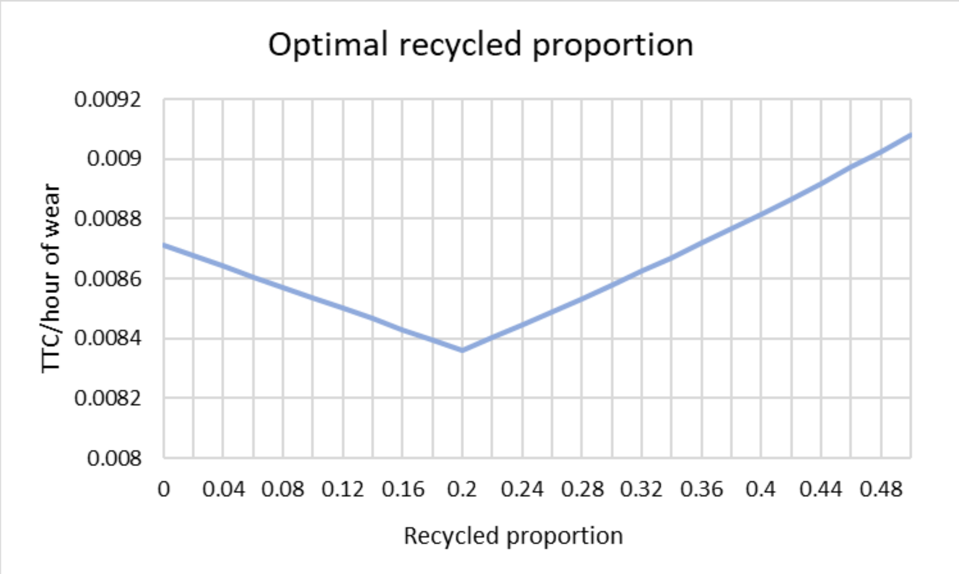

Figure 10 Minimizing the total true costs per hour of wear

10.2 Sensitivity analysis

In the base scenario, figure 10 shows that the optimal recycled proportion is 0.2. Considering the assumption that there is no noticeable reduction in the lifespan of jeans between a recycled proportion of 0 and 0.2, that the environmental impacts for nearly all categories are significantly lower for recycled jeans and the fact that this model included the maximum found private costs for recycling, it is unlikely that the model outcome will be lower than 0.2. Therefore, this sensitivity analysis focusses on determining the level of confidence in stating that the optimal recycled proportion does not exceed 0.2. This means that this sensitivity analysis tests ranges of input factors that would increase the attractiveness of adding more recycled material to jeans than the proportion of 0.2.

This study selected the input factors land transformation monetization factor, land transformation impact, private costs of recycled and private costs of virgin jeans to be tested, because they are reasonable uncertain and impactful (section 3.3). Figure 5 shows the high impact of the land transformation costs when comparing the environmental costs of recycled to virgin jeans. However, the monetization factor is quite uncertain. According to Amadei et al (2021), who reviewed existing approaches to monetary valuation in the context of LCA, the valuation of the land impact category has received considerably less attention and research compared to for instance the valuation of climate change. As described in 6.2.2, this study selected the base monetization factor for land transformation from an older report and represents 'other forests'. However, agriculture can take place in different types of forest, making it uncertain. As for the LCA results on land transformation, table 6 illustrates the variations in outcomes regarding land transformation. In the case of the private costs of recycled jeans, section 5.2 describes the large variations in private costs of recycled jeans. This study selected the value from the source that reported the highest value, as it seemed the most realistic for a Dutch market. However, selecting lower values could impact the final outcome. Finally, section 4.3 describes the limited available data on the private costs of virgin jeans. However, Mohibullah et al (2021) indicates that the chosen value of this study is on the lower side.

Table 8 shows the summarized results from the sensitivity analysis according to the method described in 3.3. Appendix C to E contain the extended results of the sensitivity analysis. When the monetization factor for land transformation varies, the optimal recycled proportion remains at 0.2 until the monetization factor exceeds 3485 €/ha, which is 70% higher than the base value. Similarly, when only the land transformation impact for virgin jeans differs, the optimal recycled proportion stays at 0.2 until the impact exceeds 0.0141 ha, which is also 70% higher than the base value. If either of these two factors change individually up to 70%, the optimal recycled proportion remains unaffected. As for the private costs of recycled jeans, the table indicates that the optimal recycled proportion remains at 0.2 until the private costs of recycled jeans decrease to € 14.48, which is 35%

lower than the base value. In this case, the total private costs for the pair increase by only 6% to € 10.80, while the total environmental costs decrease by 32% to € 13.51 compared to the baseline situation (Appendix D). Finally, in the case of changing private costs of virgin jeans, table 8 demonstrates that the optimal recycled proportion remains constant at 0.2 for all tested values. This holds true even when the private costs experience a significant increase or decrease of up to 100%.

Table 8 Summary of the sensitivity analysis

Input factor	Base value	Tipping point for an increased optimal*	Minimum value from literature	Source Minimum value	Maximum value from literature	Source maximum value
Land transformation monetization factor (€/ha)	2050	>3485	160	(Weidema, 2009)	2960	(True price, 2020)
Land Transformation (ha)	0.0083	>0.014	0.00051	(Liu et al., 2020)	0.012	(Hackett, 2015; Levi Strauss and Co, 2015)
Private costs of recycled jeans (€)	22.27	<14.48	6.49	(Wanassi et al., 2018)	22.7	(de Wit, personal communication, April 25, 2023)
Private costs of virgin jeans (€)	7.13	Not found	2.65	(Jana, 2007)	20	(Wicker, 2016)

**The threshold value of an input factor, under ceteris paribus conditions, at which the optimal proportion of recycling would increase from 0.2 to 0.5. Sources: (Hackett, 2015; Jana, 2007; Levi Strauss and Co, 2015; Liu et al., 2020; Wanassi et al., 2018; Weidema, 2009; Wicker, 2016)*

Given the uncertainty surrounding the impact of land transformation and its monetization factor, it is interesting to analyse the scenario where both factors change simultaneously. When the values for land transformation (ha) and its monetization (€/ha) experience a slight increase, the model's outcome is affected. For instance, if the land transformation impact increases by 30% to 0.0108 ha and the monetization factor increases by 30% to 2665 €/ha, the optimal outcome becomes 0.5 (Appendix C). Considering the potential range of both factors based on the literature assessed (table 8), there is a chance of such a scenario occurring.

11 Discussion

This research aims to explore the desirability of maximizing the recycled proportion in jeans, while considering the trade-offs between recycled content, private costs, lifespan, and environmental impacts. While previous studies have attempted to quantify and compare various characteristics of recycled denim, only one other study has investigated the interplay between these three factors (Fidan et al., 2021). The merit of this study compared to the study from Fidan et al (2021) lies in three things. First, rather than considering recycling as a cost-saving measure, this study acknowledges that recycled material can be more expensive compared to virgin material, which makes it more relevant for the Dutch market. Second, this study encompasses a broader range of impact categories. Of particular importance is the additional inclusion of the land transformation impact category. Third, Fidan et al (2021), as well as studies that research the environmental impacts of recycled denim (Akı et al., 2020; Kazan et al., 2020), encounter a challenge in reaching conclusive decisions due to the lack of comparability among the environmental impacts. Other studies, such as Levänen et al (2021), solve this issue by focusing on a single impact category, often related to climate change. However, such an approach may not provide a comprehensive assessment and can potentially lead to burden shifting issues (Hauschild et al., 2018).

The outcomes of my model generally align with the findings of previous studies. Section 9.4 shows that most studies investigating the qualitative properties of recycled denim find that the optimal recycled proportion falls within the range of 15-30% (Halimi et al., 2008; Radhakrishnan & Kumar, 2018; Wanassi et al., 2018, Spathas 2017). Except for Luiken & Bouwhuis (2015), who suggest incorporating 50% of post-consumer denim waste. Certain studies analysing the quality of recycled material have taken into account private costs (Arafat & Uddin, 2022, Wanassi et al., 2018), although these tends to offer more informative insights rather than directly influencing decisions. When comparing the true cost accounting (TCA) results of a pair of virgin jeans to the results from the Impact Institute (2019), there is a significant disparity. Their research indicates environmental true costs for a pair of 100% virgin jeans of € 10.90, whereas my findings reveal € 22.23. This contrast is peculiar, considering that we likely used similar monetization factors. The notable distinction primarily arises from the land use category, which contributes only € 0.25 to their overall true price, while I attribute € 16.91 to this impact category. One possible explanation is that the LCA data utilized in this study, sourced from Arvidsson (2019), incorporates average data from the United States, India, China, and Australia, while the Impact Institute (2019) assumes cotton cultivation to take place in India. Also, the discrepancy may arise from the utilization of the monetization factor for “land transformation” instead of “land occupation”. The Impact Institute (2019) uses the monetization factor for "land occupation", which is measured in €/MSA (Mean Species Abundance)*ha*yr and includes biodiversity. The LCAs I referred to expressed their findings solely in hectares (ha), so I used the monetization factor for ‘land transformation’. If the environmental cost of land use is indeed lower than stated in this study, the model outcome is unlikely to be affected (10.2).

This study tries to combine and use the existing knowledge on the properties regarding environment, quality and private costs of recycled material. However, there are limitations to this strategy. First, studies on the recycling of cotton fibres for recycled garment manufacturing have remained limited (Contin et al., 2022; Ütebay et al., 2019). With only a limited number of LCAs to choose from (Appendix A), it was challenging to find a study that matches the context of this study and includes as much environmental impacts as possible. To be able to include as many impact categories as possible, LCA outcomes were combined which had different research units, system boundaries and locations. With regard to monetization, especially including impacts from studies with different geographical scope is problematic (Arendt et al., 2020). It cannot be said whether this limitation led to an under- or overestimation of the optimal recycled proportion. Second, many studies on the quality of recycled denim research differing technical quality indicators, making it challenging to compare their outcomes. For this reason, I decided to only consider the quality indicator of tenacity, overlooking many other negative quality impacts of adding recycled material. This may have led to an overestimation of the quality and lifespan of recycled jeans and therefore an overestimation of the optimal recycled proportion. Third, the quality of fibres obtained through waste shredding depends strongly on the quality of the waste material and finishing processes (Ütebay et al., 2019). These variations in quality are the result of different wear and treatment conditions, the unknown fibre blends and varying degrees of aging. This makes it difficult to generalize the quality of recycled denim (Johnson et al., 2020). As a result, the optimal recycled proportion could either be higher or lower, depending on the quality of the textile waste of a specific batch. However, Aronsson & Persson (2020) challenges this notion and finds that heavily worn post-consumer garments retain significant value as raw materials for yarn spinning.

When creating the optimization model, I made several simplifications. The most notable simplification I made, is the assumption that the use of recycled material completely forgoes the production of virgin material and therefore its environmental impacts in a 1:1 ratio. However, Sandin & Peters (2018) highlight the problematic nature of this assumption, as it can lead to an overestimation of the environmental benefits of textile recycling and therefore in this case a possible overestimation of the optimal recycled proportion. Also, this study assumes that recycling has no other impacts beyond the environmental, quality and private cost impacts included in this study. However, the desirability of using a recycled proportion of 0.2 in jeans is influenced by various additional consequences and challenges related to textile-to textile recycling. One such challenges is that the progress of recycling initiatives is threatened by fundamental disagreements concerning ownership and accountability of textile waste (McCauley & Jestratijevic, 2023). Furthermore, challenges with stakeholder collaboration, recycling technologies and infrastructure, for instance in collection and sorting processes, continue to hinder the implementation of recycling practices (Chevalier, 2022). When taking these factors into consideration, the optimal recycled proportion is possible lower. Finally, I made an important simplification in the model regarding the lifespan of a pair of jeans. I assumed that the lifespan decreases linearly with an increase in recycled proportion, similar to the reduction in tenacity. However, translating the information on quality indicators into an accurate lifespan estimation proved to be difficult due to the influence of various factors, as discussed in section 9.1.2. The strategy of enhancing objective garment quality is widely accepted as a means to increase garment sustainability, assuming that higher quality garments will last longer. However, this perspective overlooks the understanding that clothing usage and consumption patterns are driven by various factors, many of which are subjective, intangible, and tied to a product's symbolic value (e.g being outdated) (Connor-Crabb & Rigby, 2019; Fletcher, 2012)). When these subjective aspects are included in the lifespan, Cooper et al (2013) estimates that the lifespan is 2796 hours of wear. If this holds true, the optimal proportion of recycled material could be higher.

It is important to acknowledge the limitations of the TCA method, as these limitations can have an impact on determining the optimal recycled proportion. First, the monetization factors used in this study are uncertain. For consistency reasons, I decided to primarily rely on monetization factors from the True Price Foundation (2021). However, these lack location-specific information, which is very influential (Arendt et al., 2020)**.** Despite the importance of a standardized approach, there is no consensus among researchers regarding the appropriate values for each monetization factor (Baker et al., 2020). The maturity levels of methods and data to measure, value, and attribute externalities vary greatly (Gemmill-Herren et al., 2021). The literature reveals a wide range of outcomes, with certain factors like climate change being extensively studied, while others like water use and land use receiving less attention (Arendt et al., 2020). Even within a more extensively studied impact category like climate change, the range of values varies significantly (Wang et al., 2019). Second, similarly to using a life cycle assessment, the TCA method carries the potential risk of double counting. This arises when certain costs are accounted for multiple times within the analysis. For instance, this can occur when evaluating acidification, eco-toxicity, and eutrophication, which involve partially overlapping substances such as nitrogen compounds. This is particularly difficult when applying the prevention cost approach (True Cost Initiative, 2022). Also within LCAs, double counting of electricity from specific sources can lead to under- or overestimations of environmental impacts (Holzapfel et al., 2023). Third, LCA and therefore also TCA assessments rely on the average performance of processes. Therefore, they do not incorporate the assessment of risks that are rare but are very problematic, such as chemical spills or accidents at factories (Hauschild et al., 2018).

This study has an important limitation in its scope. As outlined in section 2.2.2, it does not include non-environmental social costs. According to Egels-Zandén (2016), studies on the garment industry have tended to focus on integration of environmental sustainability as a strategy where win–win offerings can be readily found through long-term customer cost savings. These often downplay the potential trade-offs in integrating social sustainability. In the case of this study, this limitation arises partly due to time constraints and also due to the challenging task of quantifying these externalities. For example, valuation methods for areas such as health, are quite young and involve uncertainty. The complexity of supply chains and the diverse range of disciplines and data needed result in limited data availability. However, Impact Institute (2019) still tries and allocates € 22 to these non-environmental social costs, which include issues such as bonded labour, harassment, child labour, and discrimination throughout the production process. Particularly, the practice of bonded labour is emphasized, accounting for € 11.95 per pair of jeans. This includes instances like the Sumangali scheme, where young workers (often females) are bound to contracts lasting up to five years and examples of camp labour, where workers are physically confined (Solidaridad, 2012). Given these concerns, a higher recycled proportion may become more desirable in case the workers in the recycling supply chains are treated fairly.

It is important to consider one more significant limitation when using TCA to compare products. While this approach can determine which product is preferable, it cannot determine if being "better" is truly "good enough" from a sustainability perspective. TCA can be viewed as a methodology to identify the most eco-efficient solution among different alternatives (Hauschild et al., 2018). However, the actual eco-efficiency achieved through technological innovations in many cases falls short (Girod et al., 2014). The current status quo regarding certain environmental impacts is inadequate for ensuring sustainable development, as several planetary boundaries have already been exceeded (Moltesen & Bjørn, 2018). Furthermore, the gains in eco-efficiency may be offset by increased demand, leading to a potential rebound effect that diminishes the environmental benefits (Braun et al., 2021). Also, TCA assumes that sustainability can be achieved through a balance between environmental, social, and economic dimensions, where a decrease in one dimension can be compensated by an increase in another. However, this perspective conflicts with the concept of the Earth's carrying capacity, which highlights the importance of maintaining a minimum level of environmental protection to meet human needs, despite other interests (Moltesen & Bjørn, 2018).

Finally, it is important to note that the results from this research might not be applicable in future scenarios. While textile produced with mechanical recycling techniques have worse quality than virgin material, various technological developments for textile are evolving that might increase this quality. In particular, chemical recycling is emerging as a viable solution, enabling the creation of 100% recycled textiles with much better quality compared to mechanical recycling (Aronsson & Persson, 2020; van Raan, 2019). Also, future developments may introduce new technologies capable of recycling textiles composed of different materials, decreasing its production costs (Braun et al., 2021). Furthermore, Xie et al (2021) shows the potential of future policy in promoting waste-clothing recycling, which can reduce collection and processing costs of waste material. For instance, the extended producer responsibility (EPR) scheme, which has also recently been implemented in the Netherlands. This scheme the potential to triple the collection and recovery rate of post-consumer textiles. These advancements could potentially increase the optimal recycled proportion in jeans. However, if the Dutch government's future targets for textile-to-textile recycling rates are met, other challenges might arise. For instance, recycled material might get more than one additional use cycle, possible reducing the fibre lengths and therefore its quality more.

12 Conclusions and recommendations

This thesis investigated the trade-offs of including post-consumer recycled material in jeans production on the jean's environmental true costs, private costs and lifespan. It did so to explore whether it is desirable to use as much recycled material as possible in the production of new garments and what the optimal recycled proportion is. The study reveals that while using a higher proportion of recycled material reduces the environmental costs of jeans, it leads to higher private production costs and a shorter lifespan of the garment. Jeans made from 100% virgin material have lower private production costs (€7.13) but significantly higher environmental costs (€ 24.23). In contrast, jeans made from 100% recycled material have higher private production costs (€22.27), but much lower environmental costs (€ 2.78). The environmental benefits of recycled jeans are particularly evident in terms of land transformation and water use, which have substantially lower environmental costs compared to virgin jeans. However, recycled jeans do exhibit higher environmental costs for particulate matter formation.

Based on the model outcome, the optimal proportion of recycled material in jeans is 0.2. At this proportion, the total true costs amount to € 30.12, with € 19.96 representing the environmental costs and € 10.16 accounting for private costs. The estimated lifespan of jeans at this proportion is 3600 hours of wear. Although the total true costs decrease as the recycled proportion increases, the reduction in lifespan associated with this leads to an optimal proportion of 0.2.

The findings of this study are relevant for the growing interest of governments, companies, consumers and the scientific community in circular economy and true cost accounting (TCA). This study offers several valuable insights that are crucial to consider when formulating policies and making decisions regarding producing, purchasing, and researching recycled textiles. First, it demonstrates the uncertainties and challenges associated with using the TCA method in practical applications. The lack of standardization of the TCA method results in significant uncertainties of the both life cycle assessment (LCA) findings and monetization factors used. Second, in order to actually achieve the environmental benefits from a better quality garment, it is crucial to ensure that garments actually reach their maximum potential lifespan. The lifespan of a garment is influenced by many different factors and often the maximum lifespan of 3600 hours of wear is not achieved. Attention must be given by all actors to other principles of the 9R framework of the circular economy (CE), such as reuse and repair, in order to extend the lifespan of garments.

Specifically for policy makers, this study provides insights for crafting policies and regulations related to the textile industry. For instance, it highlights the benefits that can be obtained by reducing the private costs associated with recycled jeans. If private cost of recycled jeans would be lowered to € 14.48, the optimal recycled proportion would increase to 0.5. In this case, the total private costs for a pair increase by only 6% to € 10.80, while the total environmental costs would decrease by 32% to € 13.51. However, as stated before this maximum benefit can only be obtained when garments actually reach their maximum lifespan. As currently waste management costs account for only 0.6% of the total production costs of virgin jeans, producers seem to lack an economic incentive to increase garments' lifespans and reduce waste. While the Extended Producer Responsibility (EPR) scheme is a positive step towards creating this incentive and a more circular textile industry, its primary focus still lies on recycling. Other additional CE practices are essential to extend the lifespan of garments and minimize environmental impacts. For instance, multiple studies have consistently emphasized the importance of reusing clothing instead of recycling it (Sandin & Peters, 2018).

Furthermore, the findings from the TCA assessment are directly useful for textile companies in understanding the environmental impacts within the supply chains that require attention for improved environmental sustainability. For companies producing 100% virgin jeans, the impact categories with the highest environmental cost are land transformation, water use, and fresh water eutrophication, which mostly occur during the agricultural phase. On the other hand, for companies involved in the production of recycled jeans, the impact categories with the highest environmental cost are particulate matter formation, fossil fuel depletion, and contribution to climate change. The findings also highlight that there are several drawbacks when striving towards 100% recycled garments. First, recycling is not without its impacts, as it generates various types of pollution. Second, the quality of a garment tends to decline when the proportion of recycled material exceeds 0.2, which offsets the environmental advantages. Third, textile recycling is relatively costly. Instead, companies can diversify their approach to reduce environmental costs by also investing resources into other CE practices, rather than solely focusing on recycling. This enables them to reevaluate and reshape their business models.

For the scientific community, this study provides several insights that require attention in future research. First, it gives a glimpse into the possible challenges faced when using TCA. The findings reveal that it can be challenging to seek LCA results that comprehensively cover relevant impact categories, align with the specific context of your research, and are expressed in natural units that align with the units of the chosen monetization factors. When combining different LCA results, inaccuracies can occur, particularly when dealing with studies conducted in diverse locations. Regarding LCA results specifically on recycled cotton, extensive research has been conducted on its contribution to climate change, acidification, eutrophication, and water use, while the impact categories of land use, material depletion, and particulate matter formation require further attention. Additionally, the valuation of impact categories varies across different studies, with some externalities receiving more research attention than others. This study reveals the particular importance and need for further research on the impact and valuation of the land use impact category. Besides, this study shows that it is important to consider the ultimate goal of the TCA assessment. If the aim is to genuinely increase prices for consumers, understanding how accurate calculations are and the effects and destination of the additional funds becomes essential. To address the complexities and uncertainties related to TCA, further research is necessary to identify a more standardized approach to TCA, to identify the risks inherent in TCA and to develop an extensive understanding of its implications. Second, this study highlights the importance of incorporating various dimensions when considering the utilization of recycled material or other CE practices in the textile industry. This makes it possible to identify more potential risks of CE practices for textiles. Third, a limitation to this study is that it does not include non-environmental social externalities, while these are strongly present in the textile industry. Future studies that focus on the garment sector are advised to also consider this dimension when studying the application of CE practices, as this can possibly alter the outcome. Fourth, the findings from this study indicate a need for additional research in ways to enhance the economic feasibility of incorporating recycled material into new garments. Finally, this study suggests that aiming for a 100% recycled proportion may not be desirable given the existing technological limitations. Therefore, it is recommended to conduct research on the economic viability and potential consequences of alternative CE practices, beyond recycling. This broader focus can help policymakers and companies explore other, possibly more efficient, CE practices.

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14 Appendix

14.1.1 Appendix A: Taxonomy on LCA literature

Table 9 Available data on impact categories in LCA studies on jeans, denim and cotton

Author and year of publication	Contribution to climate change	Global warming Potential	Acidification Potential	Terrestrial acidification	Eutrophication potential	Fresh water eutrophication	Aquatic eutrophication potential	Marine Eutrophication	Water use	Abiotic depletion potential	Fossil depletion	Material depletion	Metal depletion	Cumulative energy demand	Non-renewable energy	Ecotoxicity	Marine ecotoxicity	Fresh water aquatic ecotoxicity potential	Terrestrial ecotoxicity potential	Human ecotoxicity	Human toxicity potential	Human toxicity-cancer	Human toxicity-non cancer	Land transformation	Natural land occupation	Urban land occupation	Agricultural land use impact	ozone layer depletion	Photochemical Ozone Creation Potential	photochemical ozone formation potential	photochemical oxidant formation	photochemical oxidation	Ionizing radiation	particulate matter formation
(Fidan et al., 2021)	1		1		1				1					1																				
(Fidan et al., 2021)	1		1		1				1					1																				
(Spathas, 2017)	1		1		1				1																									
(Spathas, 2017)	1		1		1				1																									
(Esteve-Turrillas & de la Guardia, 2017)	1		1		1				1	1				1				1	1		1	1	1	1				1		1		1		
(Esteve-Turrillas & de la Guardia, 2017)	1		1		1				1	1				1				1	1		1	1	1	1				1		1		1		
(Akı et al., 2020)	1				1				1	1														1										
(Akı et al., 2020)	1				1				1	1														1										
(Levänen et al., 2021)	1																																	
(Levänen et al., 2021)	1																																	
(Liu et al., 2020)	1		1	1		1		1	1		1		1				1	1	1		1			1				1			1		1	1
(Liu et al., 2020)	1		1	1		1		1	1		1		1				1	1	1		1			1				1			1		1	1
(Franco-García et al., 2019)	1																																	
(Franco-García et al., 2019)	1																																	
(Moazzem, Crossin, et al., 2021)	1		1		1	1			1															1			1							
(Moazzem, Wang, et al., 2021)	1		1		1	1			1															1			1							
(Moazzem, Wang, et al., 2021)	1		1		1	1			1															1			1							
(Hedman, 2018)	1			1				1	1		1														1	1	1							
(Hedman, 2018)	1			1				1	1		1														1	1	1							
(Hedman, 2018)	1			1				1	1		1														1	1	1							
(B.R. de Haan, 2017)	1		1		1				1	1						1					1							1						
(Hackett, 2015)	1				1				1	1										1				1										
(Saric & Nellström, 2019)	1		1		1							1						1	1		1							1	1		1		1	1
(Baydar et al., 2015)	1		1				1														1									1				
(Kazan et al., 2020)	1		1		1					1							1				1							1						
(Kazan et al., 2020)	1		1		1					1							1				1							1						
(Şener Fidan et al., 2023)	1		1		1													1	1															
(Şener Fidan et al., 2023)	1		1		1													1	1															
(La Rosa & Grammatikos, 2019)	1		1												1																			
(La Rosa & Grammatikos, 2019)	1		1		1				1		1			1			1	1	1		1			1				1				1		
(Morita et al., 2020)	1													1																				
(Zhang et al., 2015)	1				1				1					1		1						1	1							1				
(Levi Strauss and Co, 2015)	1				1				1	1														1										
Total	13	20	20	5	21	5	1	5	21	9	6	1	2	7	1	2	5	8	8	1	10	3	3	12	3	3	6	9	1	4	3	3	3	3

*This study excluded the impact categories in red due to a lack of available data in the case of recycled material.

Sources: Fidan et al., 2021), (Spathas, 2017), (Esteve-Turrillas & de la Guardia, 2017), (Akı et al., 2020), (Levänen et al., 2021), (Liu et al., 2020), (Moazzem, Crossin, et al., 2021), (Moazzem, Wang, et al., 2021), (de Haan, 2017), (Franco-García et al., 2019), (Hackett, 2015), (Saric & Nellström, 2019), (Baydar et al., 2015), (Kazan et al., 2020), (Hedman, 2018), (Şener Fidan et al., 2023), (La Rosa & Grammatikos, 2019), (Morita et al., 2020), (Y. Zhang et al., 2015), (Levi Strauss and Co, 2015)

14.1.2 Appendix B: Model parameters used in the model in Chapter 10

Table 10 Model parameters used in Chapter 10 for reproduction

Model parameters						
	100% virgin jeans	Source	100% recycled jeans	Source	Monetisation factor (€/Impact)	Source
Contribution to climate change (kg CO2 eq)	5.011	(Fidan et al., 2021)	2.503	(Fidan et al., 2021)	0.157	(True Price, 2021)
Acidification potential (kg SO2 eq)	0.037	(Fidan et al., 2021)	0.014	(Fidan et al., 2021)	4.7	(True Price, 2021)
Fresh water eutrophication (kg P eq)	0.005	(Fidan et al., 2021)	0.002	(Fidan et al., 2021)	203	(True Price, 2021)
Water use (m^3)	3.099	(Liu et al., 2020)	0.037	(Fidan et al., 2021)	1.27	(True Price, 2021)
Fossil fuel depletion (kg oil eq)	2.538	(Fidan et al., 2021)	0.968	(Fidan et al., 2021)	0.448	(True Price, 2021)
Land transformation (ha/year)	0.008	(Arvidsson, 2019)	0.000	(Esteve-Turrillas et al., 2017)	2050	(True Price, 2020)
Particulate matter formation (kg PM10 eq)	0.002	(Liu et al., 2020)	0.039	(Liu et al., 2020)	39.2	(Schroten et al., 2018)
Private costs (€)	7.13	(Brooks, 2015)	22.27	7.13-2.50(Brooks, 2015)+0.882(Kazan et al., 2020)*20(de Wit, personal communication, 2023)		
	Lifespan parameters	Source				
α	-1643.3	(Halimi et al., 2008; Contin et al., 2022)				
β	3928	(Halimi et al., 2008; Contin et al., 2022)				
r maximum	0.5	(Radhakrishnan & Kumar, 2018)				
L maximum (hours of wear)	3600	(Cooper et al., 2013)				

14.1.3 Appendix C: Sensitivity analysis for Land transformation impacts and monetization factors

Table 11 Optimal recycled proportion in jeans for differing Land transformation Impacts and Monetization factors

Monetisation factor for Land transformation (EUR) \ Land transformation impact for virgin jeans (ha)	0,0008	0,0017	0,0025	0,0033	0,0042	0,005	0,0058	0,0066	0,0075	0,0083	0,0091	0,01	0,0108	0,0116	0,0125	0,0133	0,0141	0,0149	0,0158	0,0166
205	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20
410	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20
615	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20
820	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20
1025	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20
1230	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20
1435	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20
1640	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20
1845	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50
2050	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50
2255	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50
2460	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50	0,50
2665	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50
2870	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50
3075	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50
3280	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50
3485	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50
3690	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50
3895	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50
4100	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50

*The row and column in green show the base values of either the monetization factor or the impact.

14.1.4 Appendix D: Sensitivity analysis for Private Costs of recycled jeans

Table 12 Optimal recycled proportion in jeans for differing Private Costs of recycled Jeans

	Private Costs Recycled Jeans (EUR)																			
	1,114	2,227	3,341	4,454	5,568	6,681	7,795	8,908	10,02	11,14	12,25	13,36	14,48	15,59	16,7	17,82	18,93	20,04	21,16	22,27
Optimal Recycled Proportion	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,50	0,20	0,20	0,20	0,20	0,20	0,20	0,20
Total Private Costs (EUR)	4,12	4,68	5,24	5,79	6,35	6,91	7,46	8,02	8,58	9,13	9,69	10,25	10,80	8,82	9,04	9,26	9,49	9,71	9,93	10,15
Total Environmental Costs (EUR)	13,51	13,51	13,51	13,51	13,51	13,51	13,51	13,51	13,51	13,51	13,51	13,51	13,51	19,95	19,95	19,95	19,95	19,95	19,95	19,95

*The column in green shows the base values.

14.1.5 Appendix E: Sensitivity analysis for Private Costs of Virgin jeans

Table 13 Optimal recycled proportion in jeans for differing Private Costs of virgin jeans

	Private Costs virgin Jeans																			
	0,71	**1,43**	**2,14**	**2,85**	**3,57**	**4,28**	**4,99**	**5,70**	**6,42**	**7,13**	**7,84**	**8,56**	**9,27**	**9,98**	**10,70**	**11,41**	**12,12**	**12,83**	**13,55**	**14,26**
Optimal recycled Proportion	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20	0,20
Total Private Costs	5,02	5,59	6,16	6,73	7,30	7,87	8,44	9,01	9,58	10,15	10,72	11,29	11,86	12,43	13,01	13,58	14,15	14,72	15,29	15,86
Total Environmental Costs	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95	19,95

*The column in green shows the base values.