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RESEARCH AND ANALYSIS

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Turning trash into treasure

An approach to the environmental assessment of waste prevention and its application to clothing and furniture in Switzerland

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Abstract

According to the waste hierarchy, waste prevention is environmentally superior to recycling or recovery, hence its inclusion in government policy. The assessment and prioritization of waste prevention strategies are impeded, inter alia, by ambiguous definitions and the lack of a sound environmental assessment method. In this study, a systematic approach to the environmental assessment of waste prevention activities (WPAs), covering the whole life cycle of products, was developed. The approach combines material flow analysis and life cycle assessment with a sustainable circular system design framework whilst giving special consideration to pivotal factors such as diffusion factor (share of population engaging in WPA), substitutability (degree to which a new product is replaced), effects on use-phase impacts, and rebound effects. The application of the approach to the case studies of clothing and household furniture in Switzerland revealed lower impact saving potential than assumed initially, due to lack of participation, low substitutability, or high rebounds. For example, reusing clothing locally, instead of exporting it to low-income countries, as currently done, displayed no or even negative impact savings since secondhand clothing in high-income countries is often consumed in addition to new clothing. Drastic scenarios for clothes led to only moderate impact reductions of less than 15%, whereas a take-back scheme for furniture reduced impacts by 70%. Concluding, the four factors (diffusion rate, substitutability, effects on use-phase impacts, and rebounds) proved crucial in the assessment of waste prevention strategies and the approach presented was able to pinpoint improvement potentials of the waste prevention scenarios investigated.

KEYWORDS
circular economy, industrial ecology, life cycle assessment, material flow analysis, sustainable production and consumption, waste prevention

1 | INTRODUCTION

Material production causes more than 50% of current global greenhouse gas (GHG) emissions (IRP et al., 2019). To reduce the environmental impacts attributed to the production, use, and creation of waste, circular economy (CE) and the waste hierarchy promote waste prevention (WP) as a priority (European Commission, 2011). Obstacles to the implementation of WP programs include missing or ambiguous WP definitions, lack of incentives for waste prevention activities (WPAs), the absence of business models, reluctance to change consumer behavior, and the complexity of measuring something that is not there (Corvellec, 2016; Lasaridi et al., 2016; Pongrácz & Pohjola, 2004; Sakai et al., 2017; Wiprächtiger et al., 2021; Zorpas & Lasaridi, 2013).

In this study, we use the circularity strategies (R-strategies) by Potting et al. (2017) to distinguish waste management and WP activities (Wiprächtiger et al., 2021). The R-strategies R0 (refuse), R1 (rethink), and R2 (reduce) include smarter use and manufacturing of products, the strategies R3 (reuse), R4 (repair), R5 (refurbish), R6 (remanufacture), and R7 (repurpose) aim at prolonging the lifespan of products, and the strategies R8 (recycling) and R9 (recover) target the useful application of materials. The strategies refuse (R0) to repurpose (R7) are considered WPAs, while recycle (R8) and recover (R9) are considered waste management activities (see supporting information SI1, Figure S1, for an overview of the R-strategies). R-strategies are often considered more circular and, similar to the underlying assumption of the waste hierarchy, more environmentally friendly, the lower the number (Potting et al., 2017).

Even though WP is generally assumed to be environmentally favorable compared to, for example, recycling, increased circularity does not necessarily lead to reduced environmental impacts (Blum et al., 2020; Zink & Geyer, 2017). In particular, rebound effects, caused by increased productivity or induced by financial savings, can be detrimental for WP measures (Zink & Geyer, 2017). Therefore, the assumption that reduced amounts of waste lead to reduced environmental impacts does not categorically hold true (Leslie et al., 2016; Martinez-Sanchez et al., 2016; Nessi et al., 2015; Wiprächtiger et al., 2020). To demonstrate and compare the environmental benefits of implementing WP measures, environmental assessments are needed (Böckin et al., 2020; Haupt & Hellweg, 2019; Haupt et al., 2018; Leslie et al., 2016; Pivnenko et al., 2016; van Loon et al., 2021). Being able to assess the environmental consequences of WPAs will allow for target and incentive creation as well as quantitative comparison of different WPAs with one another or with waste management activities.

Available WP assessments include exemplary case studies (see, e.g., Beretta & Hellweg, 2019; Laner & Rechberger, 2009; Martinez-Sanchez et al., 2016; Priefer et al., 2016; Privett, 2018). Suggestions for the assessment of WPAs using material flow analysis (MFA) (Brunner & Rechberger, 2016) and/or life cycle assessment (LCA) (Hellweg & Milà i Canals, 2014) have been made, for example, by Cleary (2010), Ekvall et al. (2007), and Nessi et al. (2013). The waste management and prevention LCA by Cleary (2010) enables a comparison of municipal solid waste management scenarios. He proposes a primary functional unit to assess amounts of waste within the waste management system and a secondary functional unit to evaluate the implications of WP on upstream processes by evaluating the substitution of products/services. Nessi et al. (2013) present two approaches for the environmental and energetic assessment of WPAs for municipal solid waste. The first approach includes the treatment of waste and supplementary goods, whereas the second approach is used to compare different types of waste. The functional units of both approaches focus on actually or potentially generated waste. We conclude that there is only a limited number of studies that adopt a shift of perspective from a waste-focused to the whole life cycle (see, e.g., Arushanyan et al., 2017; Söderman et al., 2016). A holistic assessment of the consequences of WPAs on consumption, including effects such as rebounds, is needed to assess WPAs thoroughly. Therefore, the objectives of this paper are (i) to develop an approach to the environmental assessment of WP measures and (ii) to evaluate the suitability of the approach using the case studies of textile clothing and household furniture in Switzerland.

2 | APPROACH TO THE ASSESSMENT OF WASTE PREVENTION ACTIVITIES

The proposed approach aims to environmentally assess WP by providing a structured methodology and highlighting relevant parameters. For this purpose, the WP environmental assessment approach is integrated into an existing framework for sustainable and circular design (SCSD framework) (Wiprächtiger et al., 2020). The SCSD framework includes three phases. In phase 1, the material, product, or system of interest (further explanations mention the system; the method is, however, also valid for products and materials) is identified, the scope of the study set, and the status quo of the target system analyzed using a coupled MFA/LCA approach (Haupt et al., 2018). In the second phase, scenarios using circular strategies for reducing environmental impacts are developed and evaluated. For this phase, we elaborate how scenarios for WP can be defined (Figure 1). Finally, the different scenarios are compared and measures for environmental impact reduction are evaluated in phase 3.

The necessary assumptions for the development and assessment of WP scenarios are elaborated in detail in the following.

2.1 | Phase 1: Status quo assessment

In phase 1, the scope and functional unit are defined, and the status quo is assessed. The latter is carried out using the coupled MFA/LCA approach by Haupt et al. (2018), which links material flows with related environmental impacts. WPAs already in place are identified and assessed. This helps

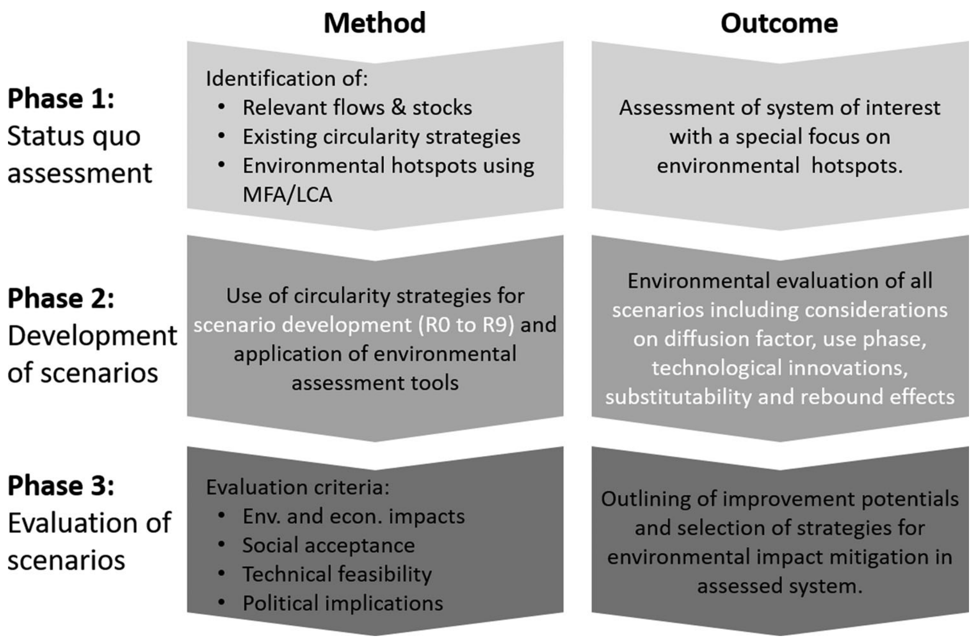

FIGURE 1 Schematic representation of SCSD framework (Wiprächtiger et al., 2020) and, highlighted in white, how the WPA assessment is integrated into the SCSD approach

defining and evaluating *potential* WPAs in the second phase. To assess the circularity and environmental sustainability, indicators such as the LCA-based circularity indicator *retained environmental value* (REV) (see Equation 1) can be applied (Haupt & Hellweg, 2019). The REV indicator compares the impacts of the displaced product (EI_{disp}) and the value retention process (EI_{vrp}) with the environmental impacts of the original good or system ($EI_{original}$). $EI_{surplus}$ accounts for differences during the use-phase, for example, due to different energy efficiencies.

$$\mathrm{REV} = \frac{\sum_{j=1}^{n}\left(\mathrm{EI}_{\mathrm{disp},j} - \mathrm{EI}_{\mathrm{vrp},j}\right) - \mathrm{EI}_{\mathrm{surplus}}}{\sum_{i=1}^{n}\left(\mathrm{EI}_{\mathrm{original},i}\right)} \tag{1}$$

A REV of 1 indicates complete retention of the original environmental value, while a REV of 0 indicates complete loss.

In addition to relative circulatory indicators, such as the REV, the absolute impact of the status quo system is assessed.

2.2 | Phase 2: Development of waste prevention scenarios

Based on the status quo analysis, “what if” WP scenarios are developed and estimated (Höjer et al., 2008). To determine relevant scenarios, the R-strategies by Potting et al. (2017) are first evaluated for feasibility in the given case study. For example, repairing beverage packaging seems infeasible and can therefore be omitted as a WPA in the scenario development phase. This pre-evaluation of relevant R-strategies should be conducted with care to prevent premature elimination of WPAs with high environmental impact mitigation potential.

To compare WP scenarios to a baseline, a business as usual (BAU) scenario, which foresees the continuation of current and planned practice over the chosen time horizons, is developed. For the development of the WP scenarios, aspects such as energy efficiency, durability, and consumer behavior need to be included. Based on existing studies assessing WPAs (see, e.g., Castellani et al., 2015; Haupt & Hellweg, 2019; Manfredi et al., 2011; Martinez-Sanchez et al., 2015; IRP, 2018; Nørup et al., 2019; van Nes & Cramer, 2006), we suggest including the following four factors in the characterization of WP scenarios. A more detailed overview of the above-mentioned references and their importance for parameter selection can be found in supporting information SI1, section 2.

2.2.1 | Diffusion factor

The diffusion factor describes the share of the population or industry willing to engage in a WPA (Manfredi et al., 2011). An example would be the willingness to buy secondhand goods, which can be estimated, for example, through consumer surveys or observed behavior (see, e.g., Edbring et al., 2016).

2.2.2 | Substitutability

Substitutability considers the relative performance of the original compared to the substituting product (Vadenbo et al., 2017). It considers technical, institutionally prescribed, and user-perceived functionality. The first describes, for example, differences in lifespan between a new and a secondhand product. Institutionally prescribed functionality accounts for restrictions or regulations dictated by authorities (Vadenbo et al., 2017). In contrast, user-perceived functionality reflects consumer choices due to available information, personal preferences, etc. (Vadenbo et al., 2017).

2.2.3 | Effects on use-phase impacts

WPAs might require additional material and energy or create waste, which needs to be considered in the assessment. For products with high energy consumption during the use-phase, it is essential to consider potential differences in efficiency compared to newer products (see, e.g., Bakker et al., 2014; Haupt & Hellweg, 2019; van Nes & Cramer, 2006).

2.2.4 | Rebound effects

Rebound effects represent a reduction in expected benefits of the WPA strategy, for example, due to behavioral feedback (see, e.g., Greening et al., 2000; Vivanco et al., 2014; Zink & Geyer, 2017). Income-related (environmental) rebounds may occur as a result of cost reduction, leading to (i) increased consumption of the discounted product (direct rebound), (ii) purchase of other products due to financial savings from the WPAs (indirect rebounds), or (iii) transformational effects (changes in consumer behavior and preferences or altered production) (Colmenares et al., 2019; Greening et al., 2000). Time-related rebounds might occur due to, for example, time saving innovations which allow people to make use of the gained time with other activities (Spielmann et al., 2008). Other rebounds (psychological rebounds) might arise, for example, if needs of the consumer are not satisfied (Hofstetter et al., 2006). In the context of WP, all rebound types can be of concern (see, e.g., Hagedorn & Wilts, 2019; Martinez-Sanchez et al., 2016).Which type of rebound is modeled with which model should be decided on a case-by-case basis.

2.3 | Evaluation of waste prevention scenarios

During the evaluation phase, the WP scenarios are compared to the BAU scenario. An additional analysis of the different WPA on a per-tonne basis can help determine the potential impacts of each R-strategy per tonne of material, independent of the diffusion rate of WPAs. This information may be used for direct process comparisons and for a potential later definition of optimized mixed scenarios (scenarios composed of several R-strategies).

To assess the consequences of measures taken today on the future, the use of a prospective analysis is needed since some WP measures taken today will only show an effect at the end of the lifespan of a product. A dynamic MFA helps to estimate future material flows (Thiébaud et al., 2017). To account for changes regarding the environmental impacts, a consequential approach should ideally be used (Zamagni et al., 2012). Incorporating the dynamic behavior of the system in the analysis allows comparing the different scenarios including all above-mentioned key factors.

For the scenario evaluation, the same indicator(s) as employed in the status quo assessment should be used. For the R-strategies R0–R2, an adjusted version of the REV needs to be used as neither of these strategies *retain* value since they prevent consumption from the start. The adapted indicator for R0–R2 is called comparative environmental value (CEV) and indicates whether switching from a baseline to another scenario is environmentally beneficial. The calculation for the CEV is similar to the REV calculation (see section 3.2.3 in supporting information S1).

The chosen environmental impact assessment methods should address all relevant impacts of the case study (International Organization for Standardization [ISO], 2006). In many cases, the methods recommended by UNEP-SETAC (UNEP-SETAC Life Cycle Initiative, 2016) are a suitable choice (including GWP 100a (IPCC, 2014) and the USEtox methodology (Bijster et al., 2017; Rosenbaum et al., 2008) in addition to comprehensive methods like, for example, ReCiPe (Huijbregts et al., 2017)).

In the following, two case studies are presented to which the assessment of WPAs within the SCSD framework is applied.

3 | CASE STUDIES

Both textile clothing and household furniture contribute a considerable part to the environmental footprint of households (Froemelt et al., 2018). In both cases several WPAs are already applied. While furniture demand is satiable since there is a limit to the number of pieces of furniture that

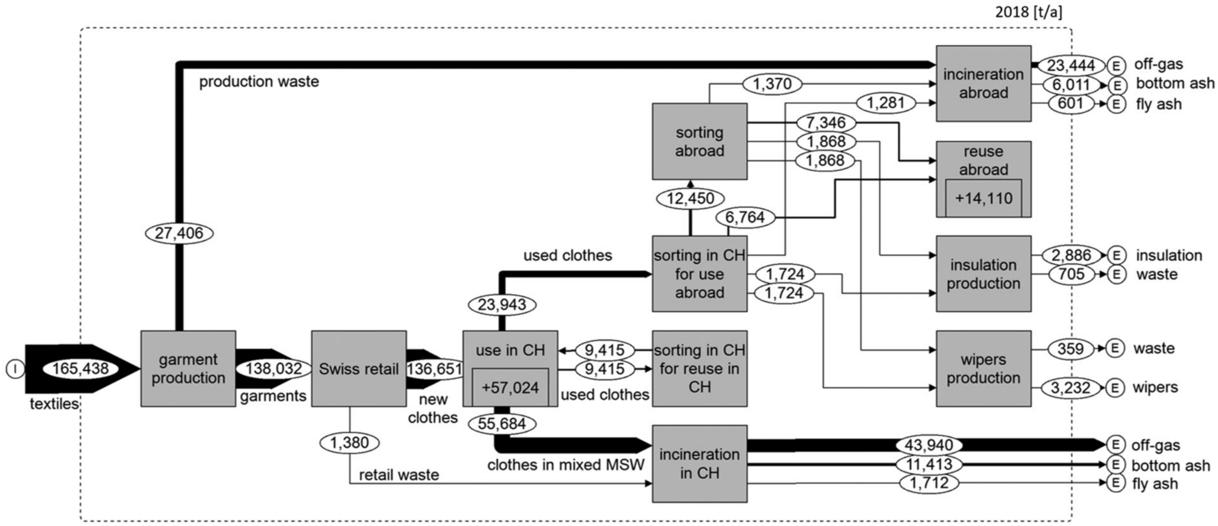

FIGURE 2 Status quo MFA of textile clothing (reference year 2018) (based on Rapp, 2020). MSW, municipal solid waste; MSWI, municipal solid waste incineration. The numbers in the use processes indicate the stock growth

fits into a dwelling, clothing demand is potentially insatiable as garments can easily be stored in great numbers (EllenMacArthurFoundation, 2017; Greenpeace, 2015). Satiable and insatiable demand are of key importance for WP scenarios, that is, when assessing rebounds or substitutability (Zink & Geyer, 2017). Textile clothing and household furniture were chosen as case studies because WPAs exist for both products, they are important factors in shaping household environmental impacts, and they differ with regard to satiable and insatiable demands.

3.1 | Clothing

3.1.1 | Status quo assessment

Scope of the MFA

A dynamic MFA was conducted covering the national and international supply chain of clothes consumed in Switzerland as well as post-consumer clothes, exported and treated inside or outside Switzerland in 2018. Input flow and stock data were collected, while output flows were calculated assuming a Weibull distribution function and an average lifespan of 5 years (Manshoven et al., 2019). The clothing amounts consumed and discarded in Switzerland were estimated using annual data on average household expenditures and clothing prices (for more details see section 3.1.1 in supporting information SI1) (BAFU, 2014; Bundesamt für Statistik, 2015, 2020; Caritas, 2020; Swiss Federal Customs Administration FCA, 2020; Tell-Tex, 2019; TEXAID, 2019).

Goal and scope of LCA

The functional unit *covering the clothing need of the Swiss population in 1 year* was used, which includes the production, transport, and use-phase impacts of purchased clothing and the impacts of WP and waste management activities from post-consumer clothes in 2018. The production impacts were calculated combining the processes described by Eryuruk (2015) (see Figure S2 in supporting information SI1) for the production of garments and end of life options, with corresponding processes in ecoinvent, cut-off, v.3.6 (Eryuruk, 2015; Wernet et al., 2016). Use-phase impacts included energy for washing and drying and detergent production impacts (Sandin et al., 2019). The LCIA calculations were conducted with the software brightway2 and the database ecoinvent, cut-off, version 3.6 (Mutel, 2017; Wernet et al., 2016). In the LCIA, we considered the impact categories climate change (IPCC, 2014), USEtox, human, and ecotoxicity (Bijster et al., 2017; Rosenbaum et al., 2008) as well as all three ReCiPe 2016 endpoint categories (Huijbregts et al., 2017). For further technical details, see sections 3.1.2 and 3.1.3 in supporting information SI1.

Results of the status quo assessment

The analysis of the clothing flows revealed an annual consumption of 16 kg of clothes and disposal of 11 kg per person in Switzerland (see Figure 2), of which 62% end up in mixed municipal waste. Twenty-seven percent of discarded clothes are collected and resold as secondhand clothes abroad, recycled into wipes or insulation, or incinerated. Around 10% are collected for reuse in Switzerland (Caritas, 2020; Tell-Tex, 2019; TEXAID, 2019, 2020). The production of garments and washing and drying were identified as environmental hotspots, confirming the findings of other studies (see, e.g., Schmutz et al., 2021; Sandin et al., 2019). Conceivable WPAs include reuse, repair, sharing of clothes, and reduced clothes consumption (see, e.g., Allianz der Konsumentenschutz-Organisationen, 2020; Kleihd, 2020; RAGFAiR, 2021).

3.1.2 | Development of waste prevention scenarios

WP scenarios were developed based on improvement potentials identified in the status quo analysis and results from existing literature on WP of clothing (see, e.g., Allianz der Konsumentenschutz-Organisationen, 2020; Edbring et al., 2016; Greenpeace, 2015; Kleinhückelkotten & Neitzke, 2019). The improvement potentials include preventing clothes from ending up in the mixed municipal solid waste, and increasing the share of clothes reused in Switzerland. Hence, scenarios of reduced clothing consumption, increased reuse of clothes locally, and repair of damaged clothes otherwise discarded to mixed municipal solid waste were created.

The repurposing of clothes into wipes and insulation, as currently done, was assumed to be continued within all WP scenarios to the same extent as for the BAU scenario, but no scenario was created to specifically address increased repurposing.

The following scenarios were developed:

- BAU: This scenario is a continuation of current practices, assuming the same per capita consumption of clothes and disposal pathways (sorting, reuse domestically and abroad, incineration, and repurposing into wipes and insulation) as in the status quo (Figure 2). Clothing demand was scaled up, taking into account forecasted population growth (Kohli et al., 2020).
- SHARE: In this scenario, three different people are assumed to share (e.g., via a clothing library that rents garments for a fee) the same piece of clothing. No change in garment quality is needed as most clothes are currently disposed of before reaching their end of life (Manshoven et al., 2019; Sandin et al., 2019). Due to a lack of more precise information, it was assumed that 25% of the Swiss population would be willing to share their clothes (diffusion factor), taking into account that some types of clothing will not be shared (e.g., underwear). Increased transport (to and from the consumer) and a higher frequency of washing were considered in this scenario, as suggested, for example, by Zamani et al. (2017).
- REPAIR: In this scenario, all damaged but repairable clothes (around 30% of clothes found in mixed municipal waste (BAFU, 2014)) were assumed to be collected, repaired, and subsequently reused, assuming a substitutability of 80% based on Privett (2018).
- REUSE: All clothes currently exported for reuse abroad are assumed to be reused in Switzerland. For Switzerland, a substitutability of 34% was assumed (Nørup, 2019). This means that only every third secondhand clothing piece replaces a new piece. For low-income countries, a substitutability of 45% was assumed, as determined by Nørup et al. (2019). The lifespans of secondhand versus new clothes was assumed to be equal, as clothes are rarely disposed of because they are damaged but rather because they are not wanted anymore (Greenpeace, 2015).

The total demand of clothing remains unchanged in the four scenarios presented above; consumers are not restricted in their purchases. To assess the influence of reduced consumption of clothing, the following two scenarios were modeled:

- REFUSE: Based on existing consumer studies, it was assumed that due to the share of hardly worn clothes and increased environmental awareness, 25% of the Swiss population (diffusion factor) would be willing to reduce clothes consumption by 15% (Greenpeace, 2015; Kleinhückelkotten & Neitzke, 2019; Laitala & Klepp, 2015).
- SUFFICIENCY: This scenario is a more pronounced version of the refuse scenario. Based on the findings from the German consumer study by Greenpeace (2015), which found that 50% of clothes are worn two times or even less, an overall decrease in clothes consumption in Switzerland by 50% was assumed.

An overview of the different scenarios and corresponding factors are given in Table 1; for more details, see supporting information SI1, section 3.2.

No scenario, which evaluates the shifting from one type of fiber to another (e.g., increased use of cotton), was created as this would require a detailed analysis of the trade-offs between land and water use for cotton-based versus microfiber release from oil-based fibers, which was deemed outside the scope of this study.

Indirect, income-related rebounds were calculated using the model by Shinde et al. (2021). For example, if a sufficiency measure leads to reduced expenses, the money saved may be spent on other consumption. Additional consumption due to the expenditure savings (see last column of Table 1) were computed for Swiss households with an income of 6000–7000 CHF, which corresponds to the median Swiss household income (MyScience, 2021). Time-related rebounds (see, e.g., Hofstetter et al., 2006; Spielmann et al., 2008) were not considered as none of the scenarios were projected to demand significantly more or less time from consumers. The model assumes that some of the money is not immediately spent on consumption but saved, leading to a delayed impact. Those delayed impacts are not considered by Shinde et al. (2021) but could theoretically be included in an extended scenario analysis. For more details on the indirect rebound calculations, see supporting information SI1, section 3.2.2. Direct rebounds, for example, increased overall consumption due to the low substitutability of secondhand clothes, were directly accounted for based on information

TABLE 1 Overview of clothing scenarios and relevant factors

Scenario	Change in new clothing consumption compared to BAU	Diffusion factor	Substitutability	Effects on use-phase impacts	Financial savings (as input to the rebound-effect model)
BAU: continuation of current practice	0%	—		—	—
SHARE: clothing demand covered through sharing, three people share one piece of clothing	−17%	25%[3]	200%[2]	Potentially increased washing	Average savings of **172 CHF/year/household for all households in Switzerland**
REPAIR: repairable clothes (currently ending up in mixed municipal waste) are repaired (lifetime extension) instead of disposed of	−15%	100%[3]	80%[4]	None	None
REUSE: all clothes intended for reuse abroad are reused in Switzerland	−5%	100% of secondhand clothing assumed to be sold	34%[5]	None	Average savings of **82 CHF/year/household** for all households in Switzerland
REFUSE: reduction (15%) of clothing consumption	−4%	25%[1]	—	None	15% reduced expenditures in 25% of all households lead to average savings of **69 CHF/year/household** for all households in Switzerland
SUFFICIENCY: clothing consumption is reduced (−50%)	−50%	100%[3]	—	None	50% of expenditures saved lead to average savings of **913 CHF/year/household** for all households in Switzerland

[1]Based on Greenpeace (2015).
[2]Based on Kleinhückelkotten and Neitzke (2019b).
[3]Own assumptions.
[4]Based on Privett (2018).
[5]Based on Nørup et al. (2019).

about the substitutability of secondhand clothes (Nørup, 2019; Nørup et al., 2019). The scenarios were modeled with a dynamic and prospective MFA/LCA for the years 2025 and 2030 (once and twice the average lifespan of clothing of 5 years) (Manshoven et al., 2019).

It was assumed that a change in the electricity market of Switzerland might occur when transforming toward a more sustainable society, using more renewable energy from biomass and solar panel installations. A sensitivity analysis using a natural gas energy mix instead of a renewable mix was conducted. For more details, see supporting information SI1, chapter 3.1.3.

3.1.3 | Evaluation of waste prevention scenarios

In this section, the climate change results are presented. Analyses using USEtox and ReCiPe 2016 provided results similar to those for climate change and for per-tonne assessment (see supporting information SI1, section 3.3). The renewable energy mix results (presented here) revealed no remarkable differences in the scenarios' environmental impact mitigation potential compared to using a natural gas energy mix (see SI, 2.3.1 and 3.3.2).

The same functional unit as for the status quo assessment (Section 3.1.1) was used. The calculations for the year 2025 include impacts of clothing purchased and disposed of in 2025 (including all up- and downstream processes and related substitution credits for reuse, repair, repurpose into wipes, and for energy recovered from incineration). For the REV calculations, the impact of the annual new clothing demand served as original environmental impact. For the REFUSE and the SUFFICIENCY scenario the CEV was calculated, as neither of the two retains value. A detailed explanation of the REV and CEV calculations can be found in supporting information SI1, chapter 3.2.3.

In Figure 3, the LCA, REV, and CEV results are presented for the year 2025; differences between those results and those of the year 2030 were minimal (see supporting information SI1, section 3.3).

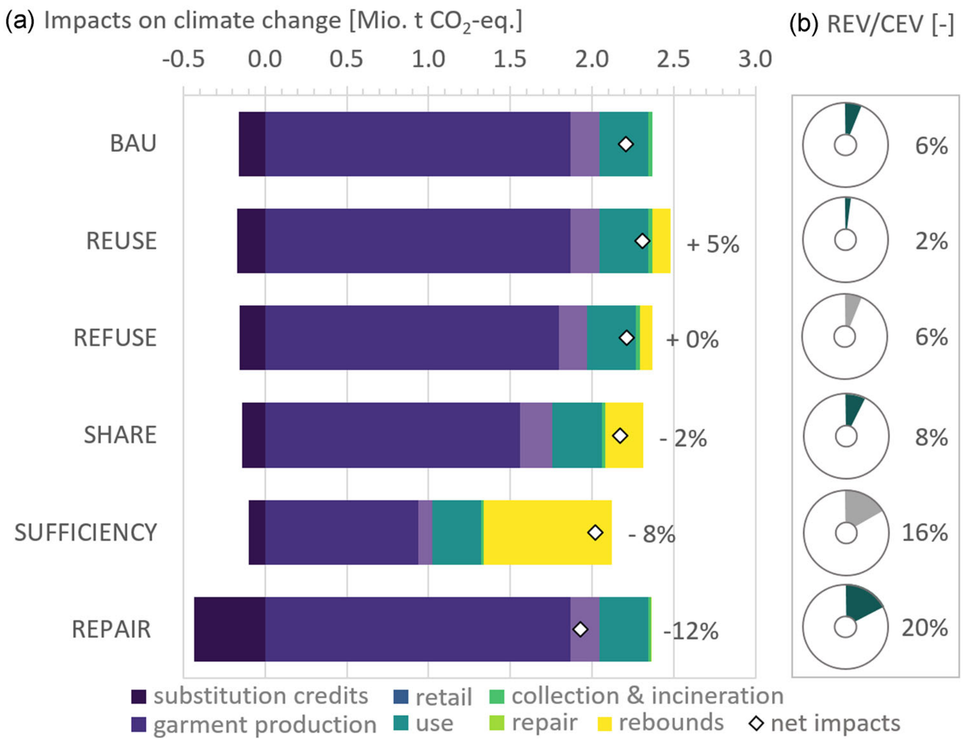

FIGURE 3 Comparison of the different scenarios for clothing for (a) impacts on climate change—relative changes of the impacts of the different scenarios in comparison to BAU are given in %. (b) REV (green) and CEV (grey) for impacts on climate change for the year 2025. The underlying data for this figure can be found in supporting information SI2

REPAIR displays the largest impact savings and the highest REV. Large substitution benefits result by the repair of clothes, assuming a substitutability of 80%. However, the REV and overall saving potential is limited, due to the low number of damaged clothes available for repair. Drastically reduced clothing demand reduces the impacts of clothing production, transport, and waste treatment substantially, as seen for the SUFFICIENCY scenario. These benefits are largely offset by rebound effects. These results highlight that a SUFFICIENCY strategy only leads to significant environmental gains if applied to the total of consumption (thus avoiding rebounds). REFUSE shows only a negligible impact reduction due to the low diffusion factor (25%) and a small decrease (15%) in clothing consumption. Moreover, the CEV for REFUSE is similar to the REV of BAU, as are the total impacts on climate change. For SUFFICIENCY, the CEV is higher than for BAU, illustrating that environmental impacts can be saved and circularity can be increased. The lower impacts for the SHARE scenario are caused by reduced consumption of clothes since three people share one piece of clothing, albeit assuming a rather low diffusion factor (25%), leading to a lower scenario-based REV (8%).

The substitutability for exported clothes (assumed export to developing countries) was assumed higher than for domestically reused clothes (Nørup et al., 2019). This difference in substitutability offsets the benefits from not having to transport the clothes to, for example, Africa, leading to the REUSE scenario having slightly increased impacts compared to the BAU scenario.

3.2 | Furniture

Furniture is responsible for 6% of a Swiss household's environmental footprint (Froemelt et al., 2018). Since the highest environmental impacts attributed to furniture stem from furniture production, preventing furniture from becoming waste and hence preventing the production of primary furniture suggests the need for a suitable measure to reduce such environmental impacts (Iritani et al., 2015).

3.2.1 | Status quo assessment

Scope of the MFA

For the status quo assessment, a dynamic MFA was conducted. It was assumed that most furniture is either wooden (70%) or metal (13%) and that around 80% of furniture was imported (Swiss Federal Customs Administration FCA, 2020). The amounts of wooden household furniture were obtained from the report on the end-use of wooden furniture in Switzerland (Hofer et al., 2019). Metal furniture amounts were estimated using the

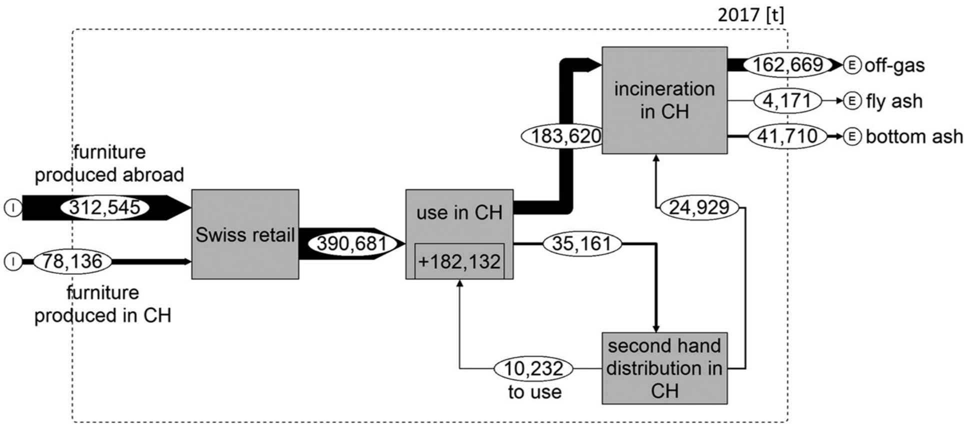

FIGURE 4 Status quo MFA of household furniture (reference year 2017). MSW, municipal solid waste; MSWI, municipal solid waste incineration. The numbers in the use processes indicate the stock growth

end-use of wood report and national import/export statistics (Hofer et al., 2019; Swiss Federal Customs Administration FCA, 2020). The dynamic MFA was conducted assuming a Weibull distribution for furniture lifetime. For more details on, for example, parameters used, see supporting information SI1, chapter 4.1.1.

Goal and scope of the LCA

The functional unit *covering the furniture need of the Swiss population in 1 year* was used. To compile the LCIs for household furniture, wooden furniture was divided into eight categories (living room, bedroom, upholstered, dining room, kid/teenager room, home office, cloakroom, and basement furniture) according to Hofer et al. (2019) (see Table S2 in supporting information SI1). For each category, furniture products and amounts were determined (see Table S3 in supporting information SI1). Where possible, product-specific LCIs were used (e.g., for a table, chair, shelf). If such LCIs were not available, more general LCIs were used, such as storage furniture, enclosed space, or surface area (Wenker et al., 2018). LCI data for metal furniture were mostly obtained from Dietz (2005). For background data, we used the database ecoinvent, cut-off, version 3.6 (Wernet et al., 2016). The life cycle impact assessment (LCIA) calculations for wooden and metal furniture were conducted with the software brightway2 (Mutel, 2017). A detailed compilation of the LCI for wooden and metal furniture can be found in supporting information SI2.

In the LCIA, we considered the impact categories climate change (IPCC, 2013), accounting for the effect of biogenic carbon dioxide (CO_2) emissions and storage effects in biomass, where relevant (Guest et al., 2013), USEtox (Bijster et al., 2017; Rosenbaum et al., 2008), and the aggregating method ReCiPe 2016 (Huijbregts et al., 2017) (see supporting information SI1, section 4.1. for more details).

Results of status quo assessment

Most of the furniture disposed of in Switzerland is incinerated. Metal parts are manually separated from combustible parts and subsequently recycled. Seventeen percent of post-consumer furniture is collected for reuse (via secondhand shops or online), but only around 5% of discarded furniture goes back into reuse; the rest is disposed of in separate collection as bulk solid waste or directly disposed of at the incineration plant (see Figure 4) (Homegate, 2021; Ricardo, 2020).

Initiatives such as furniture-as-a-service or take-back schemes are still in their infancy (INGKA, 2020), while secondhand furniture purchase is more common (Homegate, 2021; Ricardo, 2020).

3.2.2 | Development of waste prevention scenarios

WP scenarios were developed based on improvement potentials identified in the status quo analysis that showed (i) that the largest share of post-consumer furniture is being incinerated and only very few furniture pieces are reused and (ii) that the largest share of impacts is caused by the production of furniture. Hence, scenarios covering reduced production impacts through altered consumption (keeping furniture longer in use) and through targeting the reuse and repair of furniture were created:

- BAU: This scenario is a continuation of current practice, assuming the same per capita consumption of furniture and the same disposal pathways (sorting, reuse, and incineration) as in the status quo (Figure 4). Furniture demand was scaled up, taking into account population growth (Kohli et al., 2020).

TABLE 2 Overview of furniture scenarios and relevant factors

Scenario	Change in new furniture consumption compared to BAU	Diffusion factor	Substitut-ability	Effects on use-phase impacts	Financial savings (as input to the rebound-effect model)
BAU: continuation of current practice	0%	–	–	–	–
REDUCE: increased lifespan of furniture	−27%	100%[1]	–	None	On average, a Swiss household would save **204 CHF/year (2035) and 209 CHF/year (for 2050)**
REUSE: increased reuse of furniture	−32%	63[2]	100%[1]	None	On average, a Swiss household would save **247 CHF/year (2035) and 219 CHF/year (for 2050)**
REFURBISH: furniture is brought back, overhauled, and resold	−83%	83%[3]	100%[1]	None	None
RECYCLING: furniture is made from recycled wood instead of virgin wood	0%[4]	–	–	None	None

[1]Own assumption.
[2]Based on Edbring et al. (2016).
[3]Based on Lidenhammer (2015).
[4]Since not the furniture but the wood was assumed to be recycled (from furniture or other products), no effect on total furniture consumption was observed.

- REDUCE: This scenario looks at the environmental consequences of extending average furniture lifespans from currently 15 to 25 years. To display the best case, a diffusion factor of 100% (concerning the whole population and all furniture in Switzerland) was used. Effects on the use-phase due to prolonged lifespan are zero since use-phase impacts are small, and there is no reason to assume that they will change substantially with lifespan.
- REUSE: 53% of discarded furniture was assumed to be fit for reuse (Curran, 2010) and assumed to be the reuse rate in this scenario. A diffusion factor of 63% was assumed based on the survey of Edbring et al. (2016). The substitutability of secondhand furniture for new furniture was assumed to be 100% (due to satiable demand).
- REFURBISH: For this scenario, it was assumed that furniture would be brought back to the retailer or producer, refurbished, and resold. Eighty-three percent of consumers are willing, with a financial incentive (e.g., coupon), to return their furniture (diffusion factor) (Lidenhammer, 2015). Fifty-nine percent of disposed of furniture can be overhauled and resold (Curran, 2010). It was assumed that each piece of furniture could undergo two refurbishing processes (Krystofik et al., 2018). A substitutability of 100% was assumed for refurbished furniture.

In addition to the assessment of WPAs, a scenario for increased recycling was assessed to test whether WPAs perform better than recycling with regard to furniture.

- RECYCLE: Wooden parts in furniture were assumed to be made from recycled wood (Suter et al., 2017). After the use-phase, furniture was assumed to be incinerated. The lifespan of furniture with recycled wood was assumed to be similar to furniture with virgin wood.

An overview of the different scenarios and corresponding factors are given in Table 2; for more details see supporting information SI1 chapter 4.2.1.

Rebound effects of the different scenarios for furniture were modeled in the same way as for clothing, using the model by Shinde et al. (2021). See section 4.2.1 in supporting information SI1 for a more detailed explanation of the rebound modeling. The scenarios were modeled with a dynamic and prospective MFA/LCA for the years 2035 and 2050 (once and twice the average lifespan of furniture) (Hofer et al., 2019).Analogous assumptions about the electricity mix was made for the furniture case study as for the clothing case study (see Section 3.1.2).

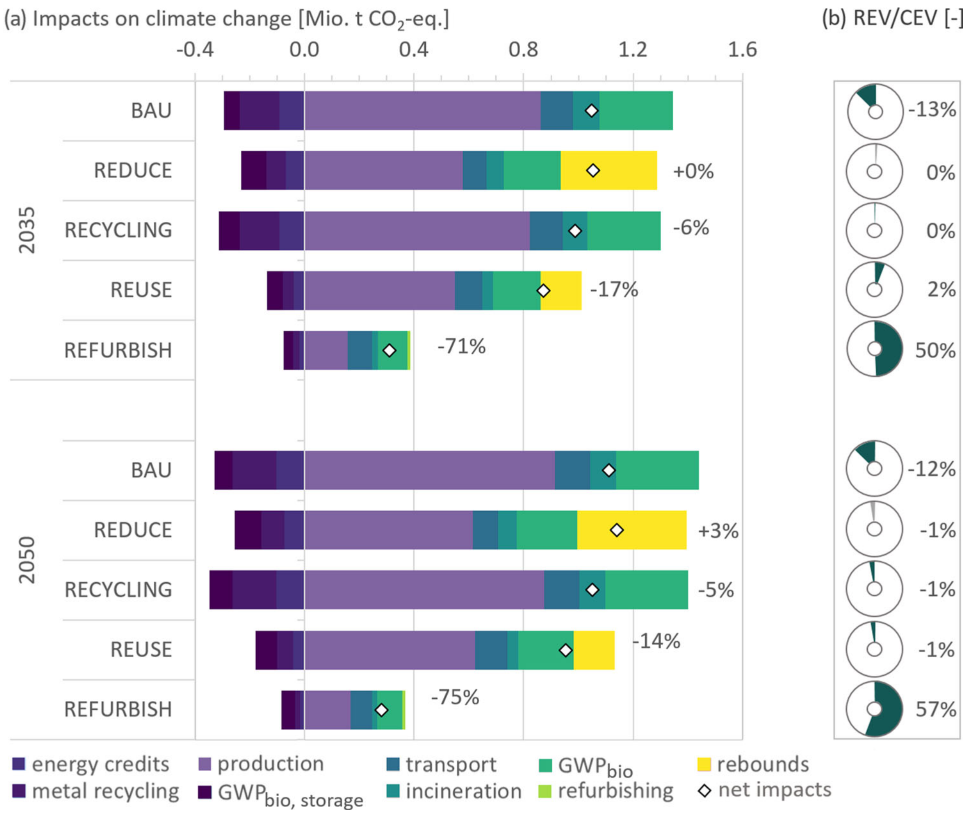

FIGURE 5 Comparison of the different scenarios for furniture on a system-wide scale for (a) impacts on climate change—relative changes of the impacts of the different scenarios compared to business as usual are given in %. (b) REV (green) and CEV (grey) for impacts on climate change for both time horizons assessed (renewable energy mix). The underlying data for this figure can be found in supporting information SI2

3.2.3 | Evaluation of waste prevention scenarios

In this section, climate change results are presented. USEtox and ReCiPe 2016 results can be found in supporting information SI1, section 4.3. The per-tonne assessment results for the furniture case study-specific WPAs are presented in supporting information SI1, section 4.3. The renewable energy mix results (presented here) revealed no noteworthy differences in the scenarios' environmental impact mitigation potential compared to using a natural gas energy mix (see supporting information SI1, 4.3.2).

The same functional unit as for the status quo assessment (see Section 3.2.1) was used, where the presented results always show a snapshot of the respective year. The calculations for the year 2035 include impacts of furniture purchased and disposed of in 2035 (including all up- and downstream processes and resulting substitution credits for reuse, refurbish, recycling, and for energy recovered from incineration). For the system-wide REV calculations, the impacts of new furniture demand served as original environmental impacts. Differences in biogenic CO_2 emissions due to longer use of wood (storage effects) were accounted for with the $EI_{surplus}$ term (see Equation 1).

Since the REDUCE scenario does not retain value, the CEV was calculated instead. A detailed explanation of the REV and CEV calculations can be found in supporting information SI1, chapter 4.1.5.

In Figure 5 the absolute LCA impacts, the REV, and CEV results for the different scenarios and time horizons are shown. LCA results for other impact categories were similar and can be found in supporting information SI1.

The BAU scenario shows a negative REV, caused by the small amount of furniture kept in the system and the large amount of furniture incinerated.

The highest impact reduction was achieved with the REFURBISH scenario. Transport impacts of furniture going to and coming from refurbishing facilities strongly influence the overall impacts. Assuming a doubled transport distance to and from refurbishing facilities leads to an impact increase of 40% for the REFURBISH scenario, whilst still showing the highest impact reduction, with a 60% reduction in CO_2-eq. More detailed results for altered transport can be found in LCA results for the other impact categories were similar and can be found in supporting information SI1, chapter 4.3.1. No rebounds were calculated for refurbished furniture as it was assumed that overhauled furniture would be resold for the same price as new furniture. The REUSE scenario shows higher impacts from indirect rebounds as secondhand furniture was assumed to be cheaper than new furniture.

For the REDUCE scenario, a lifespan extension from 15 to 25 years was assumed, resulting in a substitutability rate of 67%. After use, furniture is assumed to be disposed of in a manner similar to that of the BAU scenario. Therefore, in the REDUCE scenario, more furniture is incinerated and more metal is recycled, generating larger credits for metal and energy substitution than in the REUSE or REFURBISH scenario. Since less money is spent when furniture is kept longer, the REDUCE scenario shows detrimental rebound effects, which entirely offset the benefits from reduced furniture production. Growing and harvesting wood is often not a substantial contributor to climate change impacts. Therefore, the replacement of virgin wood by secondary wood assumed in the RECYCLING scenario did not lead to a significant impact reduction.

4 | DISCUSSION

WP is promoted as the preferable option to reduce the environmental impacts arising from the production, consumption, and disposal of products and materials. In this study, we present a systematic approach for the environmental assessment of WPAs to foster informed decision-making for impact mitigation strategies. The parameters diffusion factor, substitutability, effects on use-phase impacts, and rebound effects were chosen as vital parameters for the development and assessment of WPAs. The approach was then tested on the case studies of textile clothing and household furniture in Switzerland.

4.1 | Waste prevention assessment approach

The application of the approach presented to the case studies demonstrated that the approach is operational and delivers holistic results. For example, although often assumed, the reuse of products is not necessarily an environmentally favorable strategy. In the case of clothing, consumer behavior leads to increased consumption of clothes overall, which does not ultimately lead to environmental impact saving. For furniture, with a higher substitutability of secondhand products, rebound effects are the major negative drawback of the scenario and can reverse the results, as demonstrated in this paper: that is, the environmental benefits of furniture reduction were more than offset by rebound consumption. The model's holistic approach is hence capable of capturing such counterintuitive outcomes. These observations demonstrate the approach's ability to provide a solid structure to evaluate WPAs and identify the most environmentally beneficial one for the case investigated.

The WP scenarios developed and assessed in the case studies presented here were highly influenced by the assumptions about the diffusion factor, substitutability, and the rebound impacts. Obtaining appropriate values for the suggested factors was challenging. For the case studies, we had to refer to different sources, with varying underlining assumptions, to obtain the values for the four parameters. For clothing, mostly studies from Scandinavian countries were used (see Edbring et al., 2016; Nørup, 2019; Sandin et al., 2019), except for the German consumer study by Greenpeace. Thus the consistency for the clothing case study can be assumed to be quite high as the factors were sourced from the same geographic region with the same demographic characteristics. For furniture, data was sourced from the United States and Europe (see Curran, 2010; Krystofik et al., 2018; Lidenhammer, 2015; Privett, 2018), which leads to a higher inconsistency as consumer behavior and also household characteristics from the studies assessed differ more. To reduce the inconsistency and uncertainties, region-specific consumer studies should be used.

Rebound effects proved decisive for some of the scenarios investigated. The model used for the calculation of the rebound effects was developed based on Swiss datasets. In this paper, rebounds were calculated for households with median income. As shown in Shinde et al. (2022), rebounds can vary depending on income class and other properties. Further insights could be obtained by assessing the suitability of the different scenarios to different income groups, since these are differently impacted by the rebound effect.

In other cases, time-related or other rebound effects may be more relevant than income rebounds, and they would have to be considered with other approaches (e.g., Girod & de Haan, 2009; Spielmann et al., 2008). Rebound impacts had a substantial influence in both case studies and should therefore be reduced as much as possible.

To avoid or minimize rebound effects from expenditure savings, consumers could be incentivized to spend their saved money on sustainable consumption options. One means could be a CO_2 tax, which would, if high enough, provide a deterrent to the consumption of high-impact goods (see, e.g., González, 2010; Andreyeva et al., 2011). In addition, changing consumers' attitudes to encourage them to buy half the amount of expensive high-quality clothes rather than double the amount of low-quality garments could reduce rebound impacts and thereby the total impacts of the SUFFICIENCY scenario.

Although not considered relevant in our case studies, WPAs can have impacts on the use-phase. In particular, increasing the lifespan of goods with high use-phase energy demand might be disadvantageous if the technology is not mature and new products display improved energy efficiency and low energy consumption. Thus, the modeling of effects of WPAs on use-phase impacts is, depending on the case study, crucial (Haupt & Hellweg, 2019; IRP, 2018). To avoid increased impacts from, for example, reduced energy efficiency through lifespan prolongation, WP scenarios should take into account product characteristics and the maturity of technology. Böckin et al. (2020) determine, based on product characteristics, the appropriate strategy for resource efficiency, which could be a helpful addition in the WP scenario design process.

4.2 | Waste prevention with regard to textile clothing and household furniture in Switzerland

For the two case studies, textile clothing and household furniture, several what-if scenarios including various circularity strategies were developed (see, e.g., Höjer et al., 2008). The aim of these scenarios was to assess their environmental mitigation potential and to identify necessary adjustments regarding consumer behavior, legal framework, and infrastructure to implement the suggested strategies.

For the two case studies, the scenarios with the highest impact mitigation potential were quite different. For clothing, the SUFFICIENCY scenario demonstrated that a drastic change in consumption patterns could lead to considerable environmental impact savings, if rebound effects were prevented. This scenario illustrates that WPAs may lead to substantial resource and impact savings but need to adopt a holistic view and address consumer behavior as a whole. In the case of clothing, a paradigm shift away from, for example, fast-fashion and a change of attitude toward clothes (less fast-fashion and more durable high-quality clothes) would be necessary to implement the SUFFICIENCY scenario. Since a drastic reduction in consumption may be unrealistic, alternative models, such as sharing of clothes, which reduce the overall amount of clothing produced but not the individual amount and variety of clothing worn, could help to reduce environmental impacts from clothes without renunciation of some of the key benefits (like changing personal appearance) that clothes provide.

Despite being recurrently advertised as a sustainable option, the reuse of clothes is not per se sustainable. In the study presented, the local reuse of clothing did not result in lower environmental impacts compared to the export and reuse of clothing abroad due to the limited substitutability of 34% of secondhand clothing in high-income nations, based on the results of Nørup (2019). However, substitutability is an uncertain and sensitive parameter. In the literature, a range of 25–75% has been reported for high-income nations (Castellani et al., 2015; Farrant et al., 2010). With a substitutability of around 60%, the REUSE scenario would perform like the BAU scenario. Even with 100% substitution, the REUSE scenario would only perform 9% better than BAU due to the amount of clothes available for reuse. Thus, to increase the potential of local reuse of clothes, increased separate collection of garments would be needed.

In all scenarios, except for the SHARE scenario, use-phase impacts were assumed to remain constant (for textiles) or assumed to be negligible (furniture). For clothing, the use-phase impacts were marginally higher (see supporting information SI2 for detailed numerical values) for the SHARE scenario than the other scenarios, as it assumed a higher number of washing and drying cycles and increased transport. The use-phase impacts differed little across the scenarios, as only a small number of clothes was assumed to be shared. Less energy-intensive washing practices as well as clothes that require less frequent washing or less detergent could all contribute to a reduction in the environmental use-phase impacts of clothing (Sandin et al., 2019; Schmutz et al., 2021).

In this paper, the WPAs were modeled separately, although they are not mutually exclusive. For example, scenarios involving the reduced consumption of clothes could be combined with those involving increased repair of damaged clothes to obtain even higher saving potentials than each scenario individually.

For furniture, the scenario with the highest improvement potential was REFURBISH. How often furniture can be overhauled and resold was deduced from a case study on metal office furniture. For household furniture, this might be different since it is more subject to trends and during the refurbishing process no or only small adjustments can be made to the design. While this might matter less for office furniture, consumers may be reluctant to buy overhauled furniture. The highest impacts for the REFURBISH scenario were caused by transport to and from the refurbishing plant. The electrification of transport along with a renewable electricity mix and/or switching to renewable fuels (including hydrogen produced by renewable energy) could further reduce the impacts of the REFURBISH scenario. In this study, no rebound effects from the financial incentive of returning furniture were included. Depending on the extent of this incentive the rebound effects might become more relevant. The REUSE of furniture also poses a good option to prolong the lifespan of furniture with minimal efforts. However, the improvement potential is smaller than for REFURBISH as less furniture can be sold without any touch-up and the rebound impacts can be quite high. Similar to the REFURBISH scenario, changing trends in style, color, and materials used might hinder the reuse of furniture.

4.3 | Future research

The approach presented here combines the known methodologies of MFA and LCA with the SCSD framework for a comprehensive assessment of WP strategies. Consumer surveys specifically tailored to the cases addressed could help to reduce uncertainties and strengthen the underlying assumptions of the four factors utilized (diffusion factor, substitutability, effects on use-phase impacts, rebound effects). Despite being based on different assumptions or not being tailored to the case studies investigated here, the inclusion of the factors in the assessment proved to be crucial to identify shortcomings of some WPAs as well as levers for improvement. In the two case studies investigated, the most promising scenarios were those that required the least fundamental changes in consumer behavior: repairing of clothes or returning furniture to the retailer or producer for refurbishment appears to be more feasible than reducing consumption by a factor of 2. One of the big questions is, however, which policy actions are needed to foster such a scenario and make consumers change their purchasing behavior accordingly.

5 | CONCLUSION

To conclude, WP strategies need to be carefully designed and evaluated. It is not only important to confirm the superiority of WP strategies over recycling and waste treatment, but also to assign priority to the best-performing WPAs. The key parameters of diffusion factor, substitutability, rebounds, and effects on use-phase impacts need to be considered to optimally design, assess, and implement successful WPAs. Adequate policy instruments are required to foster waste-preventing business models. It is equally important to raise awareness among the population in order to achieve high engagement among the population (diffusion factor), high substitutability, and small rebounds. The findings of this study highlight that even with extreme measures, only a limited impact on saving was achieved for the clothing case study, emphasizing the need to implement and combine measures along the whole value chain, for example, by combining incentives for reduced consumption with convenient access to repair services.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that supports the findings of this study are available in the supporting information of this article.

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