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Does increased circularity lead to environmental sustainability?

The case of washing machine reuse in Germany

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online publication: the figure 4 was replaced
including SI files.]

Abstract

This study investigates under which circumstances increases in circularity through the reuse of use-phase-intensive electrical and electronic equipment lead to environmental benefits. We combine dynamic material flow analysis (dMFA) and life cycle assessment (LCA) to assess a Circular Economy strategy toward its environmental sustainability on midpoint and endpoint levels. The hybrid approach measures long-term implications of policy decisions in multiple impact categories and shows the need to comprehensively evaluate Circular Economy activities. We apply the approach to the strategy of setting reuse targets in a case study on washing machines in Germany. As a consequence of a reuse target, the product portfolio changes over time. The resulting stocks and flows are calculated in a dMFA, and attributed with the respective LCA-based environmental impacts. We present cumulated impacts between 2015 and 2050 for scenarios with different reuse targets for 18 midpoints and three endpoints of the impact assessment method ReCiPe 2016, and the cumulative energy demand. The latest proposal of a 5% reuse target results in average impact reductions of 1% compared to “business as usual.” An increase of reuse up to 87% results in an average impact reduction of 9%, ranging from an increase of 1% (*water consumption*) to a decrease up to 26% (*land use*). This shows that even high reuse rates only have a limited leverage on reducing environmental impacts and that it is therefore necessary to include detailed environmental assessments in a holistic evaluation of Circular Economy activities. This article met the requirements for a gold-gold *JIE* data openness badge described at <http://jie.click/badges>.



KEYWORDS

dynamic material flow analysis (dMFA), environmental policy, industrial ecology, life cycle assessment (LCA), reuse, waste electrical and electronic equipment management

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1 | INTRODUCTION

In 2015, the European Commission adopted an action plan for the Circular Economy, which includes measures to stimulate Europe's transition toward a Circular Economy (CE). The realization of these actions will be instrumental in reducing environmental pressures and reaching the Sustainable Development Goals by 2030 (European Commission, 2015). A CE's aim of maximizing resource efficiency by keeping materials at their highest value at all times (Kalmykova, Sadagopan, & Rosado, 2018) can be achieved through reuse, remanufacturing, and refurbishment of products, as well as through recycling of raw materials. The highest resource efficiency is achieved when the original function of a product is maintained, which implies a prioritization of reuse over other CE strategies (Korhonen, Honkasalo, & Seppälä, 2018). However, a sustainable strategy does not always deliver sustainable results (Helander, Petit-Boix, Leipold, & Bringezu, 2019). Possible side effects are rebound effects or burden shifting. A rebound effect occurs when a CE strategy leads to a decrease of product prices that motivates increased consumption and therefore offsets the initial resource savings (Korhonen et al., 2018; Zink & Geyer, 2017). The reuse of resource consuming products poses the additional risk of burden shifting as it avoids the burden of manufacturing, but at the same time leads to a lifetime extension of older, less efficient products, which could generate additional impacts during use. This trade-off demonstrates the need for a comprehensive evaluation of CE activities. Besides the monitoring framework proposed by the EU (European Commission, 2018), several approaches in the scientific community aim to effectively measure CE incentives (Haupt & Hellweg, 2019; Mayer et al., 2018). Helander et al. (2019) assess current approaches for the evaluation of CE activities toward their capability of capturing environmental sustainability. They find that none of the indicators holistically evaluates net environmental pressure and suggests to complement present CE management indicators with environmental indicators related to the respective CE activity.

A CE activity aims to initiate changes in consumer behavior, production patterns, or both. Over the years, these changes cause a shift in the product portfolio in stock. As previously described, it is insufficient to assess the effectiveness of a CE measure solely by the quantity of products in stock, or quantity of products displaced. It is rather important to evaluate the environmental impacts related to these changes in the product portfolio over multiple years. Elia, Gnoni, and Tornese (2017) classified different environmental assessment methodologies according to their potential to measure CE requirements and highlight material flow analysis (MFA) as well as life cycle assessment (LCA) as the two most promising assessment methodologies. We therefore combine a material flow and an environmental perspective to enable a comprehensive evaluation of CE activities that also accounts for possible burden shifting.

In the present study we introduce a dynamic stock model to quantify future product stocks and flows after the implementation of a CE strategy. In our dynamic stock model, which is a particular case of dynamic MFA (dMFA), the estimation of future waste streams is based on past production and product stock characteristics (Elshkaki, van der Voet, Timmermans, & Van Holderbeke, 2005). In a second step we integrate a modular LCA to assess the environmental implications of the resulting product portfolio during all life cycle phases. We apply the approach to the controversial example of reuse of waste electrical and electronic equipment (WEEE) to investigate under which circumstances increases in circularity through reuse lead to environmental benefits.

The combination of MFA and LCA, also known as "hybrid MFA and LCA," appears frequently in recent publications and is utilized to support decision-making in multiple fields of application such as waste management (De Meester, Nachtergaele, Debaveye, Vos, & Dewulf, 2019; Fiore, Ibanescu, Teodosiu, & Ronco, 2019; Haupt, Kägi, & Hellweg, 2018; Hischer, Wäger, & Gaughhofer, 2005; Padeyanda, Jang, Ko, & Yi, 2016; Rochat, Binder, Diaz, & Jolliet, 2013; Seigné-Itoiz, Gasol, Rieradevall, & Gabarrell, 2014; Turner, Williams, & Kemp, 2016; Van Eygen, De Meester, Tran, & Dewulf, 2016; Wäger, Hischer, & Eugster, 2011), the construction sector (Rincón et al., 2013; Ulhasanah & Goto, 2012; Venkatesh, Hammervold, & Brattembø, 2009; Vitale, Arena, Di Gregorio, & Arena, 2017), product consumption, and product population (Kayo et al., 2018; Lavers Westin et al., 2019; Yokota, Matsuno, Yamashita, & Adachi, 2003). If the aim of a study is to evaluate the impacts of policy decisions, it is essential to predict how these decisions will impact future material flows and resulting future environmental impacts. Most of the hybrid approaches reflect the past or current situation with a static MFA, and only some include future scenarios (Kayo et al., 2018; Seigné-Itoiz et al., 2014; Venkatesh et al., 2009; Yokota et al., 2003). One option to incorporate the prospective development is shown by Kayo et al. (2018), who analyze the environmental impacts of wood use by scenario analysis. Seigné-Itoiz et al. (2014) and Venkatesh et al. (2009) use a hybrid dMFA to determine environmental consequences from aluminum recycling and future environmental impacts of Oslo's wastewater pipeline network, respectively. These approaches allow for a description of time-dependent aspects such as the development of the in-use stock and the associated postconsumer flows (Buchner, Laner, Rechberger, & Fellner, 2015). The integration of LCA and dynamic stock modeling can be traced back to Yokota et al. (2003), who assess the environmental impacts induced by a product population over time, taking into account impacts during the use phase and effects of changes in the product performance. In this approach, postconsumer flows are calculated using historical data of input flows and estimations of lifetime distributions (Müller, 2006; Yokota et al., 2003). For the present study changes of in-use stocks are an important determinant, since energy and resource consumption during the use phase are main contributors to the overall impacts of electronic products. So far, impacts during the use phase have rarely been included in a hybrid approach, but findings are of value for researchers and decision-makers in the field of WEEE management (De Meester et al., 2019; Islam & Huda, 2019). We therefore built on the approach by Yokota et al. (2003) and combine dynamic stock modeling and LCA to allow for a country-wide evaluation of a range of environmental impacts of reuse of a certain product group while considering future increases in products' efficiency as well as changes in energy mix and in-use stock.

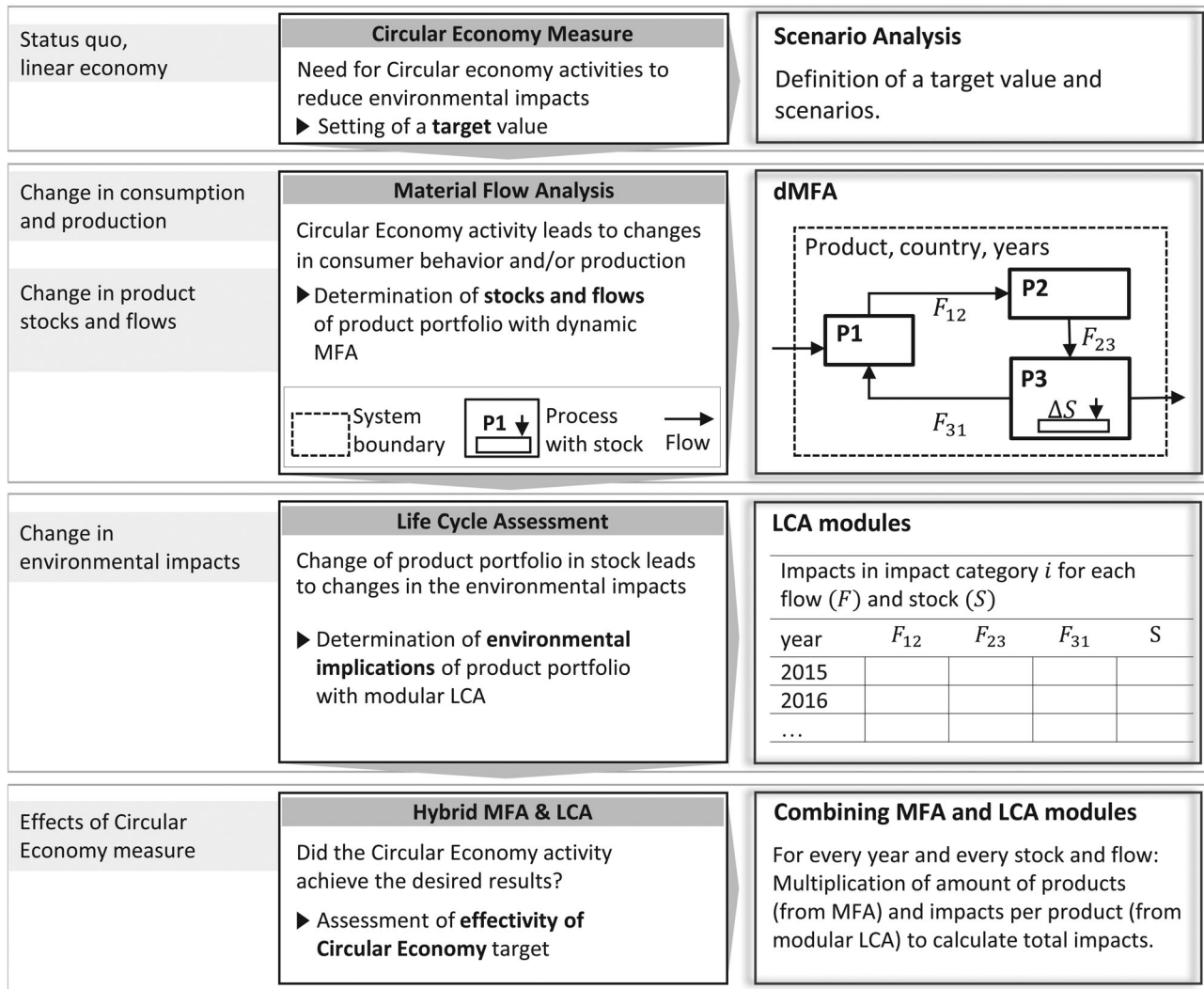


FIGURE 1 Methodological pathway for the assessment of a Circular Economy measure toward its environmental sustainability

By the application of the hybrid LCA dMFA approach to the case of WEEE reuse as an example of a CE activity, this paper contributes to the research on monitoring environmental pressures of CE activities over time. Therefore, the results do not only picture a current situation to inform policy decisions, but also assess mid- to long-term consequences of a CE incentive.

2 | METHODS AND DATA

Figure 1 illustrates the path we follow to comprehensively evaluate a CE measure. First, a target value needs to be defined, as well as different scenarios on the implementation of this target in practice in comparison to “business as usual” (BAU). As a consequence of the CE measure, the product portfolio will change over time. The resulting stocks and flows are assessed by dMFA for each year of the research period. The aim of the LCA is to obtain the environmental impacts of all stocks and flows per product and year, which are then multiplied with the respective number of products to obtain the total impacts per year. Coupling of MFA and LCA requires an inventory for all processes identified within the system boundaries (Haupt et al., 2018). A modular LCA approach is chosen in which the life cycle of a product is represented as interconnected modules (Steubing, Mutel, Suter, & Hellweg, 2016), this allows for the separate calculation of annual impacts for all stocks and flows (representing the life cycle phases) on a single product level (see Appendix A3 in Supporting Information S1). Finally, the environmental impacts can be cumulated over the entire period under consideration. A comparison with the impacts of the BAU scenario allows for an assessment of the effectiveness of the CE measure.

TABLE 1 Reuse scenarios

Scenario		Reuse quotas per year				
		2015 (%)	2020 (%)	2030 (%)	2040 (%)	2050 (%)
1	Business as usual (BAU)	0.5	0.5	0.5	0.5	0.5
2	Moderate increase of PfR target	0.5	5.0	5.0	5.0	5.0
3	Strong increase of PfR target ^a	0.5	14.5	44.1	86.6	86.6

^aBased on Messmann et al. (2019), 14.5% of WEEE could directly be reused without taking any further action. To enable the reuse of an additional 29.6% of the products, changes in collection of WEEE, or storage at collection points are required. To claim the entire reuse potential of 86.6% substantial changes in second-hand markets incentives for improved value conservation by users, and other long-term efforts are necessary.

2.1 | Case study

WEEE is the fastest growing waste stream globally (Mazahir, Verter, Boyaci, & Van Wassenhove, 2019). The annual generation of WEEE is expected to increase to 52.2 million metric tons globally (Balde, Wang, Huisman, & Kuehr, 2017). WEEE is composed of a mixture of materials that demand appropriate end-of-life (EoL) treatment. Therefore, the management of this waste stream is of special interest for a CE and regulated by the Directive 2012/19/EU. Besides preventing the creation of WEEE, it promotes reuse, recycling, and other ways of recovering. Minimum targets for recovery and a combined target for preparation for reuse (PfR) and recycling for different product groups of WEEE are set. These targets support the implementation of the European waste hierarchy (prevention, PfR, recycling, other recovery, and disposal—in that order of priority) in general but do not provide incentives for an increase of PfR in comparison to recycling. This lack of binding requirements for PfR is criticized by different stakeholders (Johnson, McMahon, & Fitzpatrick, 2015; Queiruga & Queiruga-Dios, 2015; RREUSE, 2011). Several organizations such as ComputerAid, RREUSE, ACR+, and the European Environmental Bureau support a PfR target of 5% as suggested by the European Parliament (Esenduran, Kemahlioğlu-Ziya, & Swaminathan, 2016). In any case, potential impacts of such changes in legislation need to be evaluated toward their sustainability prior to implementation (Esenduran et al., 2016; Helander et al., 2019).

This case study explores changes in stocks and flows and the resulting environmental impacts due to different reuse targets for the example of washing machines in Germany from 2015 to 2050. Washing machines are selected, since they present one of the most relevant products by weight out of all collected WEEE in Germany (Boldoczki, Thorenz, & Tuma, 2020). In analogy to prior studies, we assume municipal collection points as the main EoL pathway (Messmann, Boldoczki, Thorenz, & Tuma, 2019). The recovery operations required for the recirculation of waste into the use phase comprise examination, cleaning and if necessary, repair (KrWG §3). We differentiate products by their energy efficiency class. Energy efficiency labels (currently ranging from A+++ to G) are assigned to white goods as defined by the European Commission Regulation 2017/1369 and provide information on the energy consumption during the use phase. In the following we describe assumptions for the parameters, and the data sources used.

2.2 | Scenario analysis of a Circular Economy measure

We model three scenarios to assess the impacts of different reuse targets (Table 1). The current share of reuse in Germany is 0.5% (Messmann et al., 2019). For BAU this status quo is modeled as remaining constant. Scenario 2 represents the implementation of a PfR target of 5% as suggested by the European Parliament. Scenario 3 shows the impacts of WEEE reuse, if the entire reuse potential is exploited and therefore provides an upper bound. The reuse targets for this scenario are derived from previous work on the potentials of PfR of WEEE in Germany (Messmann et al., 2019). In between the given target values for scenario 2 and 3 (Table 1), a linear increase is modeled. The aim of the scenario analysis is not to necessarily predict future reuse policy as accurately as possible, but rather to investigate impacts of different policy decisions. The analysis enables a comparison between scenarios in which no action is taken (scenario 1), the most likely actions are taken (scenario 2) and strong actions are taken (scenario 3). The maximum reuse target in each scenario is reached by 2040. This allows for an investigation of the impacts of this target for the last 10 years of the research period.

2.3 | Dynamic material flow analysis

The system definition in Figure 2 shows all processes, stocks, and flows relevant to (preparation for) reuse of washing machines and specifies the second step (dmFA) in Figure 1. The system boundaries comprise Germany from 2015 to 2050. The flows between the processes P1 (Production), P2 (Use), P3 (End-of-Life), P4 (PfR), and P5 (Recycling) are described in Appendix A1 in Supporting Information S1. The inflow in the system are

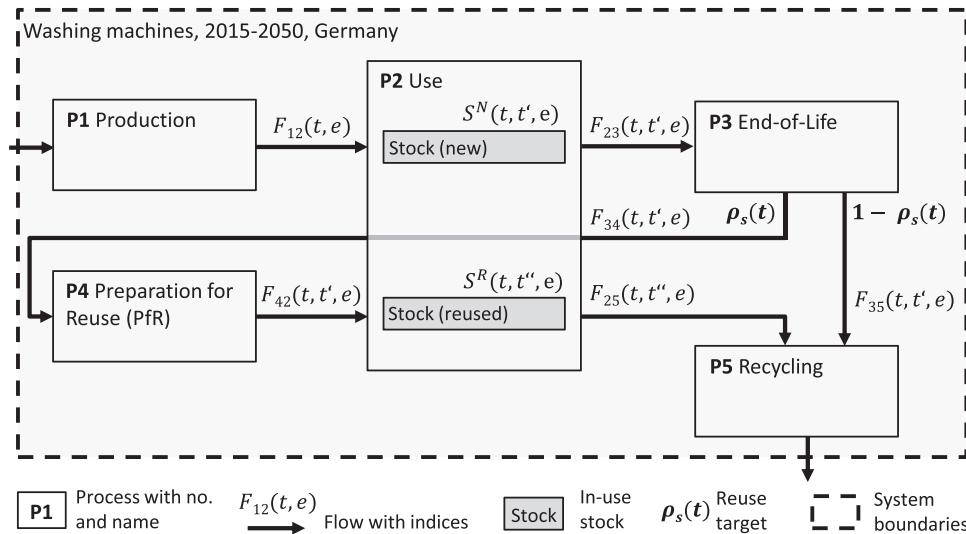


FIGURE 2 Stocks and flows along products' life cycle phases within Germany. *Note:* The total stock is differentiated between the in-use stock of new products $S^N(t, t', e)$ and reused products $S^R(t, t'', e)$. Products are classified by age cohorts, all new products of one cohort enter $S^N(t, t', e)$ in the same year t' , all reused products of one cohort enter $S^R(t, t'', e)$ in the same year t'' . Products are differentiated by their efficiency e . Stocks and flows further depend on model time t . The reuse quota $\rho_s(t)$ describes the share of discarded products that are prepared for reuse in scenario s and model time t . As a consequence, $1 - \rho_s(t)$ is the share of products entering recycling

resources for the production of washing machines and the outflow is recycled material and waste for further treatment. These flows are relevant for the LCA, but are not considered in the MFA model.

The modeling of product flows follows a bottom-up approach, which implies that the model is stock driven, rather than inflow driven (Müller, Hilty, Widmer, Schluep, & Faulstich, 2014). The outflow of products results as a consequence of previous consumption and the respective lifetime distribution. We differentiate between the lifetime distribution for new and reused products to account for a shorter lifetime of PfR products. The outflow of new products $F_{23}(t, t', e)$ is calculated by a multiplication of the original inflow with the respective lifetime distribution. A product that is discarded in year t enters the EoL phase in the same year t . Possible time-shifts due to the storage of products is neglected. The share of these products that enter PfR $F_{34}(t, t', e)$ is determined by the reuse quota $\rho_s(t)$. Thus, the share of products that flow into recycling $F_{35}(t, t', e)$ is defined by $1 - \rho_s(t)$. The number of products that are added to $F_{34}(t, t', e)$ is restricted by the difference between the stock that needs to be met in each year and the stock of remaining products from previous years $\Delta S(t)$. $\Delta S(t)$ is filled with used products that left the use phase in the previous year and are then redistributed in the use phase, as well as newly manufactured products, $F_{42}(t - 1, t', e)$ and $F_{12}(t, e)$, respectively. To fill $\Delta S(t)$ products with the best energy efficiency class e are reused first, since we assume the selection of the most efficient products for PfR. The production of new products is only allowed if $\Delta S(t)$ cannot be met with reused products. The inflow of used products in the in-use stock $F_{42}(t, t', e)$ equals $F_{34}(t, t', e)$. For reasons of practicality, we allow only one reuse cycle. Discarded reused products $F_{25}(t, t'', e)$ are therefore always assigned to recycling.

To determine $F_{12}(t, e)$, the inflow of used products is subtracted from $\Delta S(t)$ and the remaining demand is multiplied with the relative demand per year and efficiency class.

$$F_{12}(t, e) = \left(\Delta S(t) - \sum_{t'=0}^t \sum_e F_{42}(t-1, t', e) \right) \cdot d(t, e). \quad (1)$$

The total in-use stock $S(t, e)$ is composed of new $S^N(t, t', e)$ and reused products $S^R(t, t'', e)$. The respective stock per year and cohort is determined by its original inflow and by the outflows of this cohort until the current year t .

$$S^N(t, t', e) = \begin{cases} F_{12}(t', e) - \sum_{\tau=t'}^t F_{23}(\tau, t', e), & \text{if } t \geq t' \\ 0 & \text{else} \end{cases}, \quad (2)$$

$$S^R(t, t'', e) = \begin{cases} \sum_{\tau=0}^t F_{42}(t''-1, \tau, e) - \sum_{\tau=t''}^t F_{25}(\tau, t'', e), & \text{if } t \geq t'' \\ 0 & \text{else} \end{cases}. \quad (3)$$

When a product enters the recycling process, it leaves the system boundaries. We do not account for any losses or import or export of products. In reality, the manufacturing may take place outside of the considered country. This does not influence the calculation of product stocks, but is

important for the environmental impacts from production. Therefore, the system boundaries do not apply for the calculation of the environmental impacts of production. The model equations in detail as well as further information on the corresponding Python script are presented in Appendix A1 in Supporting Information S1.

The yearly in-use stock that needs to be provided results from the number of households in Germany (increasing from 35.3 million in 1991 to 44.3 million in 2050) and household coverage with washing machines (starting with 87% in 1991 and increasing to 96% in 2016, which is assumed to remain constant until 2050). The in-use stock increases from 30.5 million to 42.6 million in 2050. Detailed data and calculations are provided in Appendix A7 in Supporting Information S2. The market share per efficiency class is given from 2004 until 2017. We derive future trends based on the historical data (see Appendix A8 in Supporting Information S2) and model the introduction of new energy efficiency classes in 2021, 2031, and 2041 (this assumption is based on the currently planned rescaling of efficiency labels in 2021 by German law) to account for future improvements in efficiency. For each efficiency class, an interval of annual electricity consumption (kWh/a) is assigned (from most to least efficient: A2041, A2031, A2021, A+++ , A++ , A+ , A , B , C , D). The yearly electricity demand ranges from 121 kWh/a (A2041) to 369 kWh/a (D) (for details see Appendix A11 in Supporting Information S2). For water consuming products, the efficiency of water use can vary as well (here: modeled analogously to energy efficiency, between 8,000 L for A2041 and 17,000 L for D). A Weibull distribution function is used in the literature to define the lifetime of energy using products (Parajuly, Habib, & Liu, 2017). The expected usage duration in the base case results from the mean defined by the product group specific shape and scale parameters published by Wang, Huisman, Stevels, and Baldé (2013) (2.2 and 13.9, respectively, for new washing machines). PfR products may have required repair or may have still been functional before entering the second life, but in either case, they have already been in service for a certain time span. We therefore assume a shorter lifetime for PfR products and adapt the scale parameter accordingly (reduction of 25% compared to the new product, resulting in a scale parameter of 10.4). The average lifetime amounts to 12.3 years for new products, and 9.2 years for PfR products (for detailed data see Appendix A9 in Supporting Information S2). Since no reliable data on lifespan profiles for PfR products and future efficiency improvements is available, we rely on assumptions and test these in the sensitivity analysis.

2.4 | Modular life cycle assessment

An attributional LCA for generic new and reused washing machines in the German market is conducted (capacity of 7 kg, weight of 70 kg), as previously described by Boldoczki et al. (2020). Because the study aims for a modular design, the life cycle inventory is then split into several small processes corresponding to the life cycle phases modeled in the MFA (see Figure 2). Each process in the LCA is modeled separately with its own inputs and is therefore independent of the preceding or subsequent processes. The spatial system boundary for the MFA itself comprises Germany, which means that the use phase and intranational transports are modeled for Germany. The production of washing machines as well as replacement parts for the PfR may however include international transports and processes that are only available as the global average. In the following the functional unit of each module (process P1–P5 of Figure 2) is briefly described, detailed information as well as life cycle inventories are provided in Appendix A2 in Supporting Information S1.

The functional unit of P1 comprises the production of new washing machines and transport from manufacturer to retailer to consumer. While washing machines become more efficient over time, this does not necessarily imply considerable changes in production since a reduction of water and energy consumption is mainly achieved by an improvement in the interplay of the four parameters washing temperature and duration, spinning dynamics, and chemicals (Personal communication with Tschöpe S., *Director Production Engineering Miele*, September 3, 2020). Nevertheless, experience from the industry suggests that not only products become more efficient, but also production processes. It could, however, also be argued that improvements in efficiency demand an increase in production impacts. To account for both theories, we introduce one sensitivity analysis in which production impacts decrease for more efficient products and one for which impacts increase.

For P2 the average use of a washing machine in European households for 1 year is modeled, which equals 220 washing cycles (Boyano et al., 2017). It is assumed that electricity and water consumption depend solely on the energy efficiency, no difference between new and reused products is modeled. In the base case, an increase of renewable sources is assumed (see Appendix A10 in Supporting Information S2). Previous studies show a strong influence of the energy mix on the environmental impacts during the use phase, therefore, the energy mix is part of the sensitivity analysis (Baxter, 2019; Boldoczki et al., 2020). Impacts of P3 (EoL) are assumed to be neglectable, as this process only comprises the sorting of products between P4 and P5.

PfR (P4) comprises examination, cleaning and, if necessary, repair of the products. Impacts of the examination and cleaning are assumed to be neglectable. For the repair, a typical, financially economical repair scenario for washing machines including transport from the collection point to a primary treatment facility for WEEE and to the consumer is modeled. Since 14.5% of the products can directly be reused (see Table 1), we only account for 85.5% of the impacts of repair. When a product enters the EoL phase and is not assigned to PfR it is directed toward recycling (P5), the respective data is complemented as described in Appendix A2 in Supporting Information S1 and the assumptions for the parameters and calculations are given in Appendix A12 in Supporting Information S2. In the base case impacts of the processes P1 (Production), P4 (PfR) and P5 (Recycling) are static and do not change during the years due to data limitations. This assumption is further investigated in the sensitivity analysis for P1. Since processes P4 and P5 only contribute marginally to the overall impacts, they are excluded from the sensitivity analysis. Impacts of P2

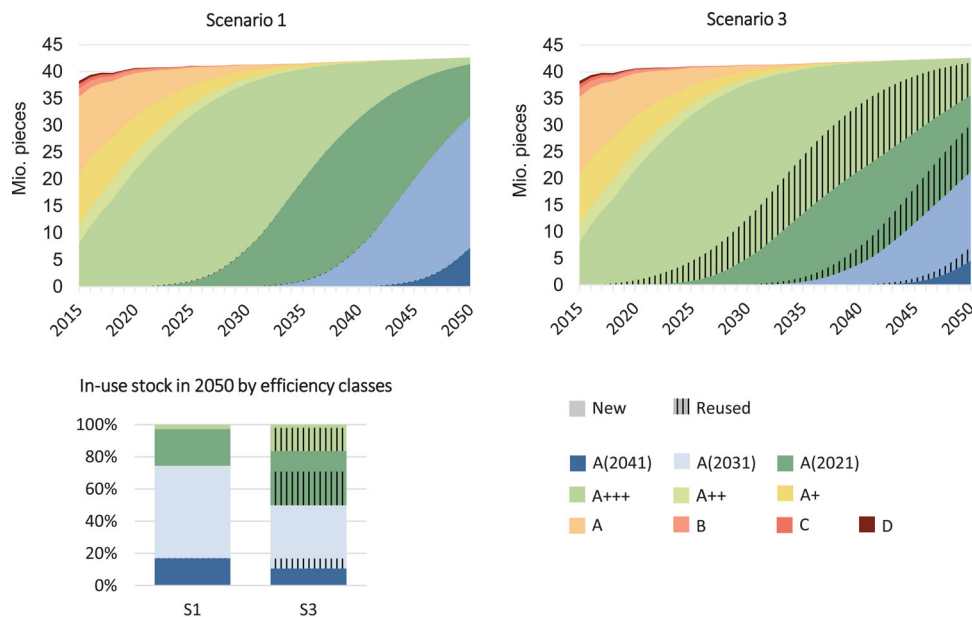


FIGURE 3 Stock of washing machines by efficiency classes from 2015 to 2050 (top) and for 2050 (bottom) for scenarios 1 (BAU) and 3 (strong increase in PfR target). Underlying data used to create this figure can be found in Appendix A14 in Supporting Information S2

(Use) are dynamic as they change every year in dependency on the energy mix. Data on the environmental impacts for each process and year are provided in Appendix A13 in Supporting Information S1.

Life cycle inventories are modeled with the SimaPro 9.1 software and the Ecoinvent 3.6 database (Ecoinvent Centre, 2019). The life cycle impact assessment is carried out using ReCiPe 2016 (H) v1.1 (Huijbregts et al., 2017). While we calculate results for all 18 ReCiPe midpoint categories, the evaluation is focused on the categories *climate change* (CC), *terrestrial ecotoxicity* (TE), *human carcinogenic toxicity* (HT), *mineral resource scarcity* (MRS), and *water consumption* (WC). The cumulated impacts are also evaluated for the ReCiPe endpoints *human health* (HH), *ecosystem quality* (EQ), and *resource availability* (RE). This set is complemented with the aggregated *cumulative energy demand* (CED) (V. 1.11), which includes six categories (each three non-renewable and renewable).

3 | RESULTS

The results of the model depend on product stocks and flows and LCA-based environmental impacts. The results are described for the default parameters. In the sensitivity analysis, changes in the results due to single parameters are evaluated.

3.1 | Product stocks and flows

Figure 3 gives a graphical representation of the composition of the product stocks for each year from 2015 to 2050, the respective data is presented in Appendix A14 in Supporting Information S2. Naturally, a higher share of PfR leads to a higher share of reused products in stock. During the first years, the PfR target differs only marginally between scenarios 1 (S1) and 3 (S3). From 2020 onward, the difference between the targets in the two scenarios increases and so does the difference in the stocks. Whereas in S3 reused products comprise 18% of the total stock by 2030, it is close to zero in S1. By 2050, this difference is even more obvious. A higher share of reused products also implies a higher share of older and less efficient products in stock. This is evident if the stock in 2050 is compared (Figure 3, bottom). In S1, the stock almost exclusively comprises products rated A(2021) or better, whereas in S3 still 16% of washing machines rated A+++ or worse are in stock. The share of reused products amounts to 0.4% for S1 in contrast to 42% for S3.

Whereas in S1 almost all products end up in recycling, in S3 close to 44% are reused in 2030. These products will be part of the in-use stock during the subsequent years. In S1, the stock of the subsequent years will be filled with new and therefore more efficient products. This has two main implications: First, in S1, more products need to be manufactured than in the other scenarios and second, the in-use stock of S1 is more efficient. These two implications have controversial effects on the environmental impacts. Whereas manufacturing generates impacts, a more efficient in-use stock diminishes environmental impacts. MFA alone can only show changes in the product portfolio, but to assess the environmental implications of different PfR targets, LCA is necessary.

3.2 | Environmental impact assessment

Figure 4 shows the composition of annual impacts in the categories *climate change* (CC), *terrestrial ecotoxicity* (TE), and *human carcinogenic toxicity* (HT) for S1 and S3, the respective data is presented in Appendix A15 in Supporting Information S2. These impact categories are selected for a detailed analysis, because they represent a domination of impacts by the use phase, by production, and by neither of both, respectively. Impacts of S2 are almost identical to S1, a comparison of all three scenarios is included in Appendix A4 in Supporting Information S1 (), a comprehensive analysis of all impact categories follows (see Figure 5). Despite an increase in in-use stock, impacts do not increase over time, due to efficiency gains of the products and an increase of renewable energy sources. In the beginning, impacts of all scenarios are equal, since the reuse quotas are identical (0.5%). In 2015, CC amounts close to 7 billion kg CO₂-eq. By 2050, the impacts will decrease to between 3.5 (S3) and 3.6 (S1) billion kg CO₂-eq. Even when the reuse targets between the scenarios differ strongly, the difference in CC is only marginal. This can be attributed to the fact that impacts of CC mainly arise in the use phase. The avoided impacts of production due to a high reuse quota are compensated by the higher impacts in the use phase due to more inefficient products in stock. Depending on year and scenario, impacts of production account for between 11% and 29% of the yearly impacts. Both impacts of PfR and benefits of recycling only contribute marginally (<1%), but it can be seen that the impacts due to PfR increase with increasing reuse in S3, while impacts of production decline simultaneously. Nevertheless, even for high reuse quotas this decline does not generate benefits concerning CC.

A different picture shows for TE, which is dominated by the production phase. A decline in production due to increased reuse therefore leads to reduced total impacts. Whereas BAU (S1) leads to 31 billion kg 1,4-dichlorobenzene eq. (1,4-DCB eq.), a reuse target of 86.6% (S3) results in 23 billion kg 1,4-DCB eq. by 2050. In S3 the reuse target is already 44% by 2030, therefore many reused products are in stock in the subsequent years. Once products are disposed of (on average after 9.2 years), they cannot be reused again. Even if the reuse target is almost 90% by 2040, the actual share of reuse over all EoL products never exceeds 60% during the research period. In the current case, the share of reuse peaks in 2040, in the subsequent years many already reused products are discarded and the demanded in-use stock is satisfied with new products. This explains the increase in production and therefore in impacts in S3 from 2040 onward. HT shows slightly more differences between the scenarios and is not as strictly dominated by either production or use phase.

Water consumption (WC) and CED behave similarly to CC and are dominated by the use phase. An increased reuse leads to more inefficient products in stock and results in minor positive or negative effects for these categories. (MRS behaves similar to TE and therefore shows larger differences between the scenarios. The respective graphs are included in Appendices A4 and A5 in Supporting Information S1.

For further investigation, we cumulate the impacts arising from 2015 to 2050 for each impact category (respective data is provided in Appendix A16 in Supporting Information S2). Figure 5 shows the cumulated impacts of each scenario in relation to S1 (BAU), including the endpoints *human health*, *ecosystem quality*, and *resource availability* (Figure 6, right). This enables an aggregated view on the environmental implications of different reuse targets.

S3 sets the highest reuse targets. This scenario is most beneficial in 18 out of 19 impact categories, only WC results in slightly higher impacts (<1%) than BAU. Other impacts decrease up to 26% (land use, LU). The investigation of S3 shows that a strong increase in reuse targets leads to minor changes (up to +/-5% compared to BAU) for seven impact categories, including CC, WC, and CED. Savings of more than 10% are generated in other seven impact categories, including TE and MRS. On average, this scenario generates savings of 8.9%. Taking aside the marginally negative effect on WC, we can confirm the assumption that the implementation of a PfR target (up to nearly 90%) reduces environmental impacts since the avoided burden of manufacturing of new products exceeds the impacts of additional resource consumption during the use of older, less efficient reused devices. The same findings result on an endpoint level. The latest proposal of a 5% reuse target results in savings between 0.1% (WC) and 2.6% (LU), averaging to 1.0%. On an endpoint level, savings of 0.8% (HH), 2.2% (EQ), and 0.5% (RE) can be realized.

3.3 | Sensitivity analysis

The sensitivity analyses show whether changes in parameters, for which no reliable data sources are available, influence the resulting environmental impacts. In the following, the effects of the energy mix, efficiency of products, impacts of production and lifetime of PfR products are explored (details are described in Appendix A17 in Supporting Information S2).

- A constant energy mix based on the composition of 2015, instead of an increase of renewable sources (as in the base case), is modeled.
- A decrease of energy efficiency gains from 12% to 6% for future efficiency classes instead of a constant decrease of 12% in the base case, and no decrease in water consumption for all future efficiency classes, instead of a one-time decrease of 15% between A+++ and A2021 is modeled.
- In the base case, the scale parameter for PfR products is reduced by 25% to model a shorter lifetime of reused products. For the sensitivity analysis, a reduction of 50% and no change compared to new products is tested.

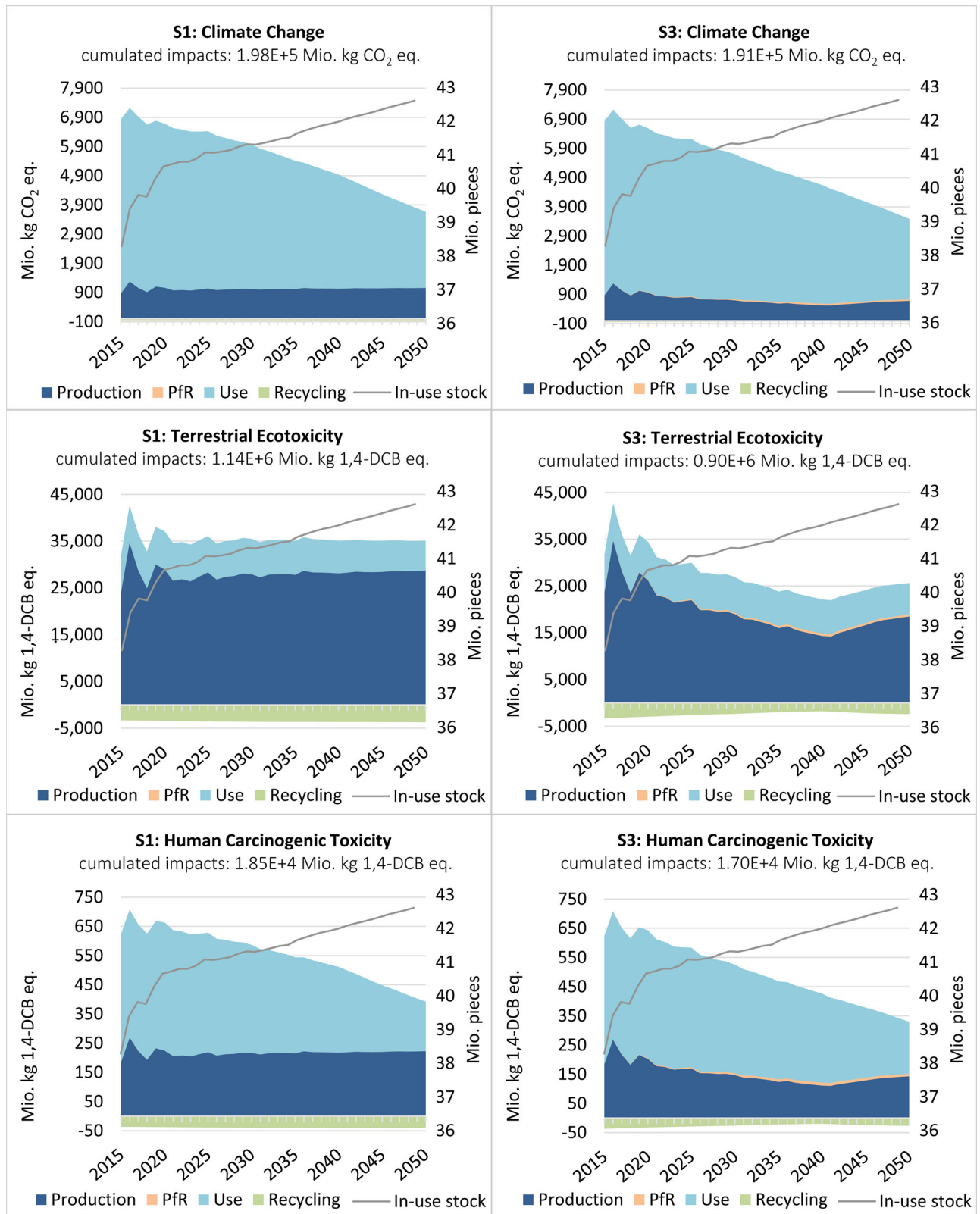


FIGURE 4 Impacts per life cycle phase for climate change, terrestrial ecotoxicity, and human carcinogenic toxicity from 2015 to 2050 for scenarios 1 (BAU) and 3 (strong increase in PfR target). Underlying data used to create this figure can be found in Appendix A15 in Supporting Information S2

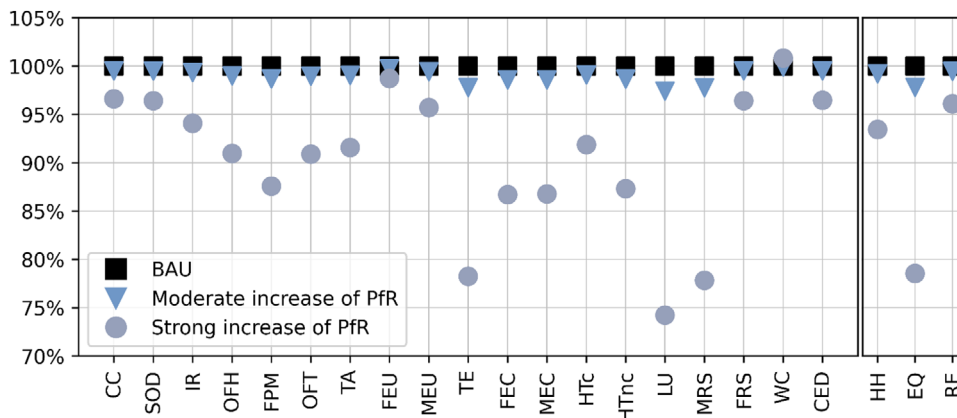


FIGURE 5 Cumulated environmental impacts (from 2015 to 2050) for scenarios 1 (BAU), 2 (moderate increase in PfR target), and 3 (strong increase in PfR target). Note: CC, climate change; SOD, stratospheric ozone depletion; IR, ionizing radiation; OFH, ozone formation, human health; FPM, fine particulate matter formation; OFT, ozone formation, terrestrial ecosystems; TA, terrestrial acidification; FEU, freshwater eutrophication; MEU, marine eutrophication; TE, terrestrial ecotoxicity; FEC, freshwater ecotoxicity; MEC, marine ecotoxicity; HTC, human carcinogenic toxicity; HTnc, human non-carcinogenic toxicity; LU, land use; MRS, mineral resource scarcity; FRS, fossil resource scarcity; WC, water consumption; CED, cumulative energy demand; HH, human health; EQ, ecosystem quality; RE, resource availability. Underlying data used to create this figure can be found in Appendix A16 in Supporting Information S2

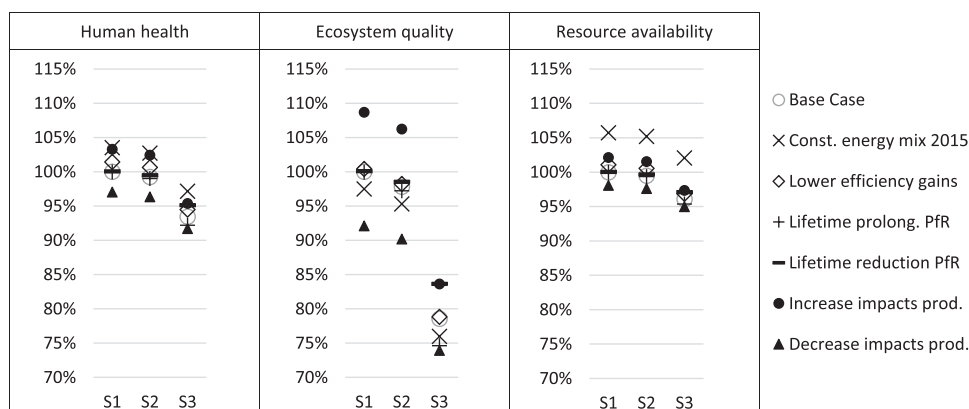


FIGURE 6 Results of sensitivity analysis for the endpoints *human health*, *ecosystem quality*, and *resource availability*. Underlying data used to create this figure can be found in Appendix A17 in Supporting Information S2

- An increase of impacts of production by 10% between an efficiency class and the next better one and a decrease of impacts by 10% between the classes is modeled instead of constant production impacts.

Figure 6 shows the cumulated impacts for the sensitivity analyses for each scenario on endpoint level in comparison to the base case impacts of S1 (100%).

Assumptions concerning efficiency gains and the expected lifetime of reused products only have a minor influence on the total impacts, whereas the energy mix and changes in production influence the absolute impacts more strongly. But more important is the comparison of the three scenarios for each sensitivity analysis. For example, for *ecosystem quality*, the increase of impacts of production leads to an increase of absolute impacts of nearly 9% in S1 and 6% in S3, but impacts in S3 are still 23% less than in S1, and therefore in the same range as in the base case (savings of 21%). So even if the absolute impacts change, different assumptions concerning the production do not lead to different results (S3 still more beneficial than S1). The same holds true for the energy mix. The retention of a constant energy mix instead of an increase in renewable sources leads to an improvement in *ecosystem quality*. This can be attributed to lower impacts for the midpoint *terrestrial ecotoxicity*. If on the contrary the energy mix of 2050 (63% renewable energy sources) is modeled as constant, impacts in *human health* and *resource availability* decrease by about 20%, while impacts in *ecosystem quality* remain constant (see Appendix A6 in Supporting Information S1). This emphasizes the leverage of the energy mix to reduce environmental impacts and shows the necessity to consider future trends in energy generation in a dynamic approach.

4 | DISCUSSION AND CONCLUSION

In this article, we assess the Circular Economy measure of setting reuse targets for WEEE toward its environmental sustainability on midpoint and endpoint level by combining dMFA and LCA. We apply this approach to the case study of washing machine reuse in Germany and assess impacts between 2015 and 2050 for different reuse targets. This addresses the need for a quantitative long-term assessment in the evaluation of CE activities.

As a consequence of a reuse target, the product portfolio changes over time. The resulting product stocks and flows are calculated in a dynamic stock model. In a second step, we include LCA data to assess the environmental implications of the resulting product portfolio during all life cycle phases. The simultaneous consideration of future reduction in water and energy consumption, changes in the energy mix and in-use stock allow to model future development as accurately as possible. The cumulated impacts over a 35-year period show only a marginal difference between “business as usual” (reuse target of less than 1%) and a reuse target of 5% as currently discussed. The achievable savings range between 0.1% (WC) and 2.6% (LU). The results demonstrate that reuse does not lead to a considerable reduction of environmental impacts, but neither to an increase. While Kondo and Nakamura (2004) imply losses in employment induced by the lifetime extension of products, other studies show positive economic and social implications of reuse (job creation, accessibility of cheap products) (González, Rodríguez, & Pena-Boquete, 2017; O’Connell, Hickey, & Fitzpatrick, 2012; Pini et al., 2019), and the results of this study confirm that those are not in conflict with environmental goals.

If a strong increase in the reuse target (up to almost 90%) is modeled, a more differentiated picture occurs. While in 6 out of 18 midpoints as well as the CED still only marginal changes occur (+/− <5% in comparison to BAU), the categories TE and MRS (both 22%) and LU (26%) indicate a reduction potential. This shows that a consideration of single impact categories is insufficient for evaluating the manifold environmental implications that CE incentives may entail. If only CC is measured, as is often done, the cumulated impacts suggest that an increase of PFR has almost no effect (reduction of 3%). The results reveal that PFR might not be the solution if the target value is a reduction of CO₂ equivalents in order to limit climate change, but in a discussion about the future availability of raw materials or the health of ecosystems, reuse has a potential of risk reduction.

As previously discussed by Graedel (2019), MFA results are valuable for policy purposes, but “their policy utility will be no greater than the accuracy and timeliness of their assessments.” The current approach comprises a rather long research period, since the considered Circular Economy measure only takes effect in the long term. This approach, as other efforts to assess future development, is therefore subject to data limitations, particularly in terms of future changes in the energy mix (e.g., faster transition to renewable sources due to fossil-fuel phase-out) and production and recycling processes. Moreover, the present model does not consider storage times as an immediate discard of a product is assumed. Subsequent efforts should be focused on obtaining more comprehensive data on technological advances. Further applications of the methodology could analyze other product groups, different geographical scopes, or additional Circular Economy measures.

The combination of a dynamic stock model and LCA delivers detailed information on the implications of policy decisions in multiple impact categories. It could be assumed that a reuse quota of more than 80% leads to significant environmental savings. The application of this hybrid approach shows that this is not necessarily the case. For CC as one of the most commonly discussed impact categories, reuse has rarely an impact. It is therefore necessary to include detailed environmental assessments in an evaluation of CE incentives. If the environmental sustainability of an economy is solely assessed by its circularity (in the form of recycling or reuse rates) important information is missing (Haupt & Hellweg, 2019). This study shows that more advanced tools, such as LCA, are necessary to enable a more nuanced assessment of the sustainability of CE incentives.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

MODEL SCRIPT AND DATA AVAILABILITY

We also provide the complete hybrid dynamic stock model, including the input data (Excel) and the model script (Python) in a GitHub repository, which can be accessed via: <https://github.com/sandraboldoczki/Hybrid-DSM-washing-machine-reuse>.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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