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A life cycle assessment framework for large-scale changes in material circularity

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ABSTRACT

Increasing material circularity is high on the agenda of the European Union in order to decouple environmental impacts and economic growth. While life cycle assessment (LCA) is useful for quantifying the associated environmental impacts, consistent LCA modeling of the large-scale changes arising from policy targets addressing material circularity (i.e., recycled content and recycling rate) is challenging. In response to this, we propose an assessment framework addressing key steps in LCA, namely, goal definition, functional unit, baseline versus alternative scenario definition, and modeling of system responses. Regulatory and economic aspects (e.g., trends in consumption patterns, market responses, market saturation, and legislative side-policies affecting waste management) are emphasized as critical for the identification of potential system responses and for supporting regulatory interventions required to reach the intended environmental benefits. The framework is recommended for LCA studies focusing on system-wide consequences where allocation between product life cycles is not relevant; however, the framework can be adapted to include allocation. The application of the framework was illustrated by an example of implementing a policy target for 2025 of 70% recycled content in PET trays in EU27+1. It was demonstrated that neglecting large-scale market responses and saturation lead to an overestimation of the environmental benefits from the policy target and that supplementary initiatives are required to achieve the full benefits at system level.

1. Introduction

Growing awareness of the environmental impacts caused by anthropogenic activities has encouraged decision-makers to factor in environmental implications along with socio-economic aspects when deciding new policies. In recent years, circular economy (CE) has gained traction both at governmental and business levels as a solution to support economic growth while reducing environmental footprint (EC, 2015a). CE is a restorative industrial economy concept that is based on three principles: to design out waste and pollution, to keep products and materials in use, and to regenerate natural systems (Ellen MacArthur Foundation, 2017). CE aims at increasing material circularity by reducing the need for resource extraction (i.e., materials, nutrients, etc.), encouraging reuse, repair, and recycling instead of the linear "extract-use-discard" consumption, and supporting innovative business models (Bao et al., 2019; Rashid et al., 2013). Even if no univocal definition of CE exists (Geisendorf and Pietrulla, 2018) and the focus of CE legislation varies significantly in different geographical areas (McDowall et al., 2017), the majority of CE policies and CE literature focused on waste recovery and recycling to close the materials loops (Ghisellini et al., 2016; Merli et al., 2018; Morseletto, 2020). In Europe, material circularity has been implemented in particular through policy targets focusing on recycling (Morseletto, 2020), where recycling indicates "any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes" (EC, 2008).

To support the development of policy targets leading to the intended effects, assessment of the environmental consequences associated with full-scale implementation of these policies should reflect appropriate system and framework conditions (Cantzler et al., 2020). Policy targets

Abbreviations: CE, Circular economy; LCA, Life cycle assessment; PEF, Product Environmental Footprint; CAGR, Compound annual growth rate.

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for recycling are inherently intended to cause large-scale changes in society, industry, and markets, and thereby affect material and product production capacity. A large-scale change is defined as a consequence that cannot be considered "marginal" nor likely to affect only parts of a market, and thereby warrant assumptions of linearity in market responses and material supply (EC, 2010). While the environmental consequences from implementing such policy targets can be quantified with life cycle assessment (LCA) (EC, 2015b), little methodological guidance has been provided for the application of LCA to large-scale changes with market implications at meso/macro levels.

We found two main methodological challenges in LCAs aiming at evaluating CE policies and targets in relation to waste management and recycling. First, the majority of methods modeling recycling in LCA (Allacker et al., 2014; EC, 2018a; JRC-EC, 2020; Schrijvers et al., 2016) focus on individual products and divide the environmental burdens/savings associated with recycling between the product being recycled after use and the product being produced from the recycled material. The way recycling should be modeled in LCA has been debated extensively in the scientific literature (e.g., Allacker et al., 2014; Schrijvers et al., 2016), and several LCA standards (e.g., PAS 2050, ISO/TS 14067, BPX 30-323-0) offer contradictory guidance. Already in 1993, SETAC (1993) observed that recycling (and, more generally, any linked system) should be assessed by jointly modeling all involved life cycles since concerted actions between actors in different life cycles are required to reach the desired goal (i.e., recycling) within the system. This is particularly important when assessing environmental consequences associated with policy targets intended to have large-scale implications across value and supply chains, as, for example, the changes required in the European material circularity required to comply with current recycling targets.

Second, increased material recycling has been assumed to be completely absorbed by the market and to always substitute primary material (as in Andreoni et al., 2015; Gibbs et al., 2014; Hestin et al., 2015; Tallentire and Steubing, 2020), without considering potential market saturation or the scale of the impact (i.e., small-scale versus large-scale changes relative to unsatisfied demand), because all modeled systems have been assumed to be linear without accounting for the volume of recycled material. The majority of studies on non-linearity focus on the problem of modeling the upscaling of emerging and scalable technologies (Arvidsson et al., 2018; Pizzol et al., 2021). However, far less attention has been focused on the different kinds of non-linearity occurring when individual material sources or markets are limited. Exceptions to this include Andreasi Bassi et al. (2020) who quantified the risk of recycling market saturation and environmental dispersion via export, Binnemans et al. (2013) who determined whether the market could absorb a co-product of rare earth material mining, and Söderman et al. (2016) and Ekvall et al. (2016) that combined macro-economic models with LCAs.

Without addressing the effects on value-chains over several life cycles, the full consequences of a society-level policy such as recycling targets cannot be encompassed by the LCA. Simply applying individual product LCAs as support for legislative decisions on recycling and, thereby, ignoring scale effects and potential market saturation, may lead to false expectations of savings and burdens associated with new recycling initiatives. In the case of large-scale policy initiatives, this may lead to the rollout of unsubstantiated regulation, without the intended effects on environmental impacts and resource efficiency. Based on existing literature, further guidance is needed for transparent and consistent LCAs addressing the societal transition and the policies supporting material circularity.

The goal of this study is to develop a guiding framework supporting a consistent goal and scope definition for consequential LCAs of large-scale changes in material circularity. The focus is on changes induced by implementing policy targets on recycled content and post-consumer recycling rate of products and materials.

The framework i) describes how to consistently define LCAs' goal and scope to model changes in recycled content and recycling rates, ii) accounts for jointly assessing the supply and use of recycled materials, and iii) integrates future developments of market conditions and background systems, including associated effects on market saturation. The specific objectives are to: i) describe and document the individual steps and assumptions associated with the framework, ii) illustrate the implementation of the framework for a hypothetical EU-wide recycling target, and iii) put the framework in perspective by comparing it with existing alternatives and discuss its overall limitations of the framework. The framework departs from the authors' experiences with consequential LCAs (i.e., which impacts are due to a change in the system). Attributional modeling (i.e., "process-based modeling intended to provide a static representation of average conditions" (EC, 2013)) is not considered, since we agree with many researchers that the consequential approach is more appropriate for modeling system-wide changes and supporting policy decisions (Frischknecht et al., 2017; Sala et al., 2016; Zamagni et al., 2012).

2. Methodology and framework description

2.1. Framework overview

This section describes the assessment framework (Fig. 1) and focuses on those parts of the LCA we believe are most important to address, without aiming to provide comprehensive guidance on how to perform an LCA. The application of the framework is illustrated in Section 3, but several other examples of different goals, functional units, and system boundaries are provided in the Supplementary Material (SM) to support the reader. Since material circularity can be increased by leveraging both recycled content and recycling rates, the framework especially focuses on how to model changes in the recycled content and the recycling rate.

From this point forward, p*rimary materials* are materials that have been extracted from or produced from nature (i.e., fossil fuels, metal ores, forests, and plantations), while *secondary materials* are materials produced from recycling. Furthermore, this framework introduces the concept of *side policies* that is similar to the economic side policies found in Domenech and Bahn-Walkowiak (2019). In fact, since the legislation on CE is a complex constellation of policy frameworks, economic incentives, and economic side policies (Domenech and Bahn-Walkowiak, 2019), it is often not possible to quantify and isolate the impacts of introducing a single policy target independently from other legislative tools. *Side policies* are defined as other legislative tools (e.g., targets, bans, monitoring tools, economic tools) that address the same functional unit of the study. For example, the consequences of increasing recycling can vary depending on the implementation of landfilling and incineration bans, exporting or importing embargos, minimum recycled content, design requirements, incentives in sorting and recycling plants, etc.

2.2. Goal: Assessing the impact of waste policy targets

The goal of an LCA (Fig. 1, a) summarizes the reasons for carrying out a study and its intended application. The goal specifies the material circularity change under investigation, which for waste policy often translates into a target rate (e.g., "collection rate", "recycling rate", "landfilling rate", "incineration rate"). The goal also specifies the geographical and temporal scope of the LCA.

The goal is also the phase where it is clarified whether or not the impacts will be allocated between product life cycles. To assess the impacts of changes in the material circularity in a society, we recommend avoiding allocation. This is justified by the system-level approach that legislators usually take, whereby the goal is to quantify the overall consequences of a policy and not to subdivide burdens and savings of recycling between different products or stakeholders. Allocation can be avoided through the expansion of the system. In this case, the system is expanded to include the consequences in the waste management of the

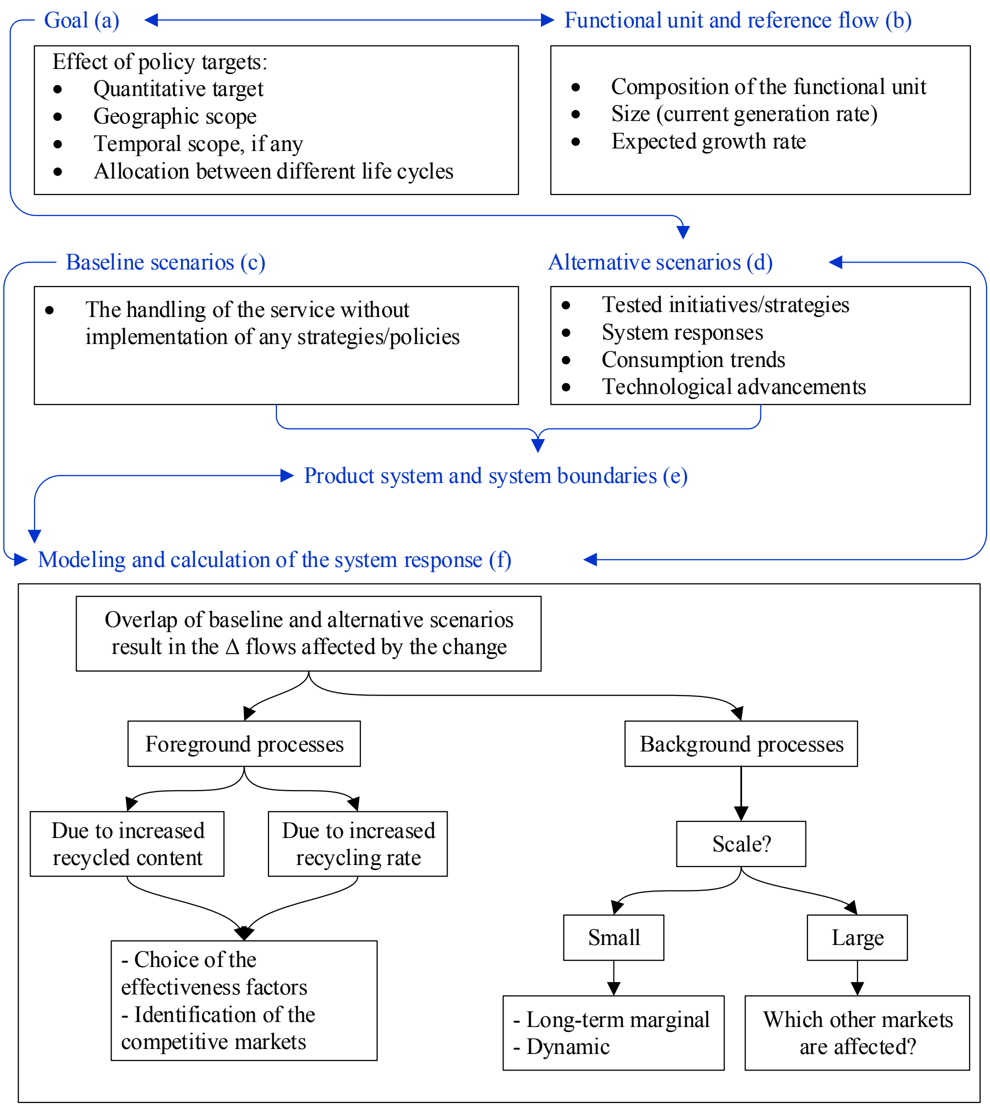

Fig. 1. Proposed framework to define the goal and scope of consequential LCAs aiming at quantifying the impacts of increasing/decreasing material circularity. The blue arrows indicate the connections between the different steps. Letters (a) to (f) indicate the steps described in more detail in Sections 2.2 to 2.6. All steps are affected in the case of temporally dynamic modeling (see Section 2.6.9). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

life cycle generating the secondary material to the system in focus and the products absorbing the secondary material generated by the system in focus (Fig. 2 and Fig. 3). Expanding the system will also show the intrinsic mutual interaction between legislation covering the secondary material market and legislation covering the waste management market.

For instance, an LCA could have the goal "To quantify the environmental impact of increasing the recycled content of PET bottles in the EU27+1 from the expected 11% to 25% by 2025, *combined with the needed changes in the waste management system to reach such recycled content*." or "To quantify the environmental impact of increasing the recycling rate of PET bottles in the EU27+1 from the expected 50% to 70% by 2025, *combined with induced reactions of the market absorbing the generated secondary material*". Here, the PET bottles are the system in focus, and the phrases in italic reveal the system expansion and the avoidance of the allocation to capture all possible effects. On the other hand, the goal "To quantify the environmental impacts of increasing the recycled content of PET bottles in the EU27+1 from the expected 11% to 25% by 2025" would indeed require allocation, because this goal does not include the reaction of the waste management system providing the secondary material (e.g., increasing in source-separation, increasing of recycling, simple diversion from other products that were before using the secondary material).

2.3. *Functional unit and reference flow*

The functional unit defines the service assessed in the LCA (Fig. 1, b) and the reference flow that fulfills the functional unit. Due to the scope of the framework, the functional unit could address a material flow such as represented by an anthropogenic consumption (e.g., "cardboard consumed in the USA in 2025) or a specific waste generated (e.g., "plastic packaging waste generated in the EU27 between 2020 and 2030").

Depending on the study, the composition of the reference flow can be subdivided between different stakeholders (e.g., waste generated from households, industry), sectors (e.g., packaging, textile), plastic polymers (e.g., PET, HDPE), colors, products (e.g., PET bottles and PET trays), or quality (e.g., low- and high-quality waste paper). Often, the composition of multi-material fractions (e.g., scrap vehicles and mixed waste) needs to be estimated or based on other data. Since this framework focuses on material circularity, we suggest collecting information on how the recyclables and secondary materials market are subdivided and then base the composition on such understanding.

Since policy targets always refer to future years (e.g., by 2035 only 10% of the municipal solid waste should be landfilled in the European Union (EC, 2018b)), the future consumption or waste generation (e.g., municipal solid waste generated in 2035 or between today and 2035)

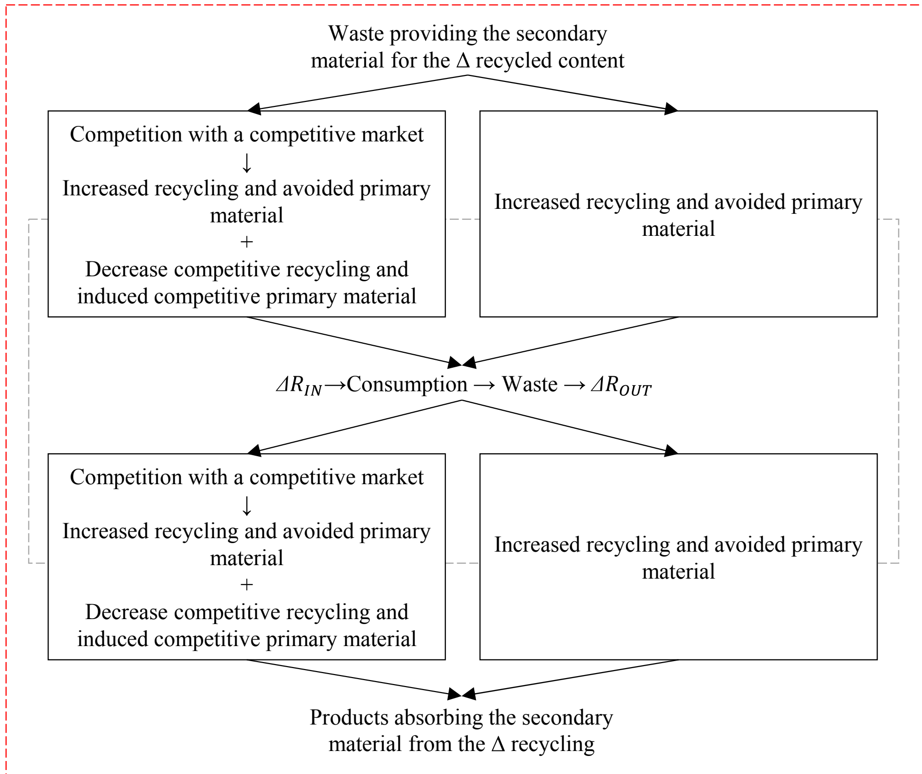

Fig. 2. System boundaries of increasing/decreasing the recycled content (R_{IN} [kg/kg or %]) and the recycling rate (R_{OUT} [kg/kg or %]) in case the goal of the study avoids (red dotted lines) or includes (gray dotted lines) allocation between different life cycles. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

has to be quantified. Assumptions on future consumption can be based on the expected annual growth rate or compound annual growth rate (CAGR), economic growth, technological innovation, material efficiencies, etc. The system can be tested for different market trends (e.g., if the consumption increases more than expected, or remains stable) that fulfill the same service to be assessed (i.e., the functional unit), and this can help evaluating the impact of changing the demand (i.e., avoiding consumption) or comparing waste prevention against end-of-pipe solutions (described in Section 2.6.3).

2.4. Baseline and alternative scenarios

We propose to calculate the impacts as the difference between the baseline scenario (no change, "frozen policy" situation) and the alternative scenarios, following best practice in impact assessment of policies (EC, 2015b).

The baseline (Fig. 1, c) has many synonyms in the literature ("reference", "business-as-usual", "current scenario", "counterfactual", "conventional scenario", "status quo"), reflecting "what would happen to the functional unit if nothing changed beyond already implemented or decided policies". This involves the definition of a baseline that evolves over time (e.g., as a result of changes in framework conditions from the implementation of existing policies) in the absence of new initiatives (EC, 2015b): e.g., involving gradual changes in collection and recycling efficiency, technology development, and energy supply. A single baseline scenario may be appropriate in the case of relatively short timeframes, but multiple baseline scenarios are advised to encompass distinct future developments of system (and framework) conditions as part of a sensitivity assessment, as in Söderman et al. (2016).

Alternative scenarios (Fig. 1, d) represent specific alternatives implementating the policy target in question, and here reflect how the full system reacts to an external "impulse" such as increasing or reducing material circularity. While the range of alternative scenarios should reflect the range and complexity of potential effects from the implementation of the policy target, it is advised that this is limited within an individual study to ensure transparency in interpretation and communication towards decision-makers.

Analysis of the economic and legislative context is thus fundamental to derive scenarios and to avoid unrealistic or impossible scenarios. Modeling efforts could benefit from collecting information on the regulation of contaminants in the waste, potential and current markets providing the secondary material (e.g., which markets could supply the secondary materials used in the functional unit?), potential and current end-markets (e.g., which markets could absorb the secondary material produced by recycling the waste), etc.

2.5. System boundaries

In the system boundaries (Fig. 1, e), all processes associated with the functional unit are identified. Fig. 2 illustrates the processes and flows that should be included when the alternative scenario has a different recycled content (ΔR_{IN} [kg/kg or %]) or recycling rate (ΔR_{OUT} [kg/kg or %]) compared with the baseline scenario.

As also reflected in other frameworks (Ekvall, 2000; Schrijvers et al., 2020), an increase in recycled content affects the demand for secondary material that can either increase recycling of the waste providing the secondary material and/or shift the use of secondary materials from other products. Similarly, an increase in the collection of material for recycling affects the supply of secondary material. Depending on market dynamics, this might increase the recycled content in some products, and/or reduce the supply of secondary materials from other sources. Decisions on recycling made in one product life cycle affect wider markets and the circularity of other products. To account for these effects, we need to evaluate the interactions between several different markets.

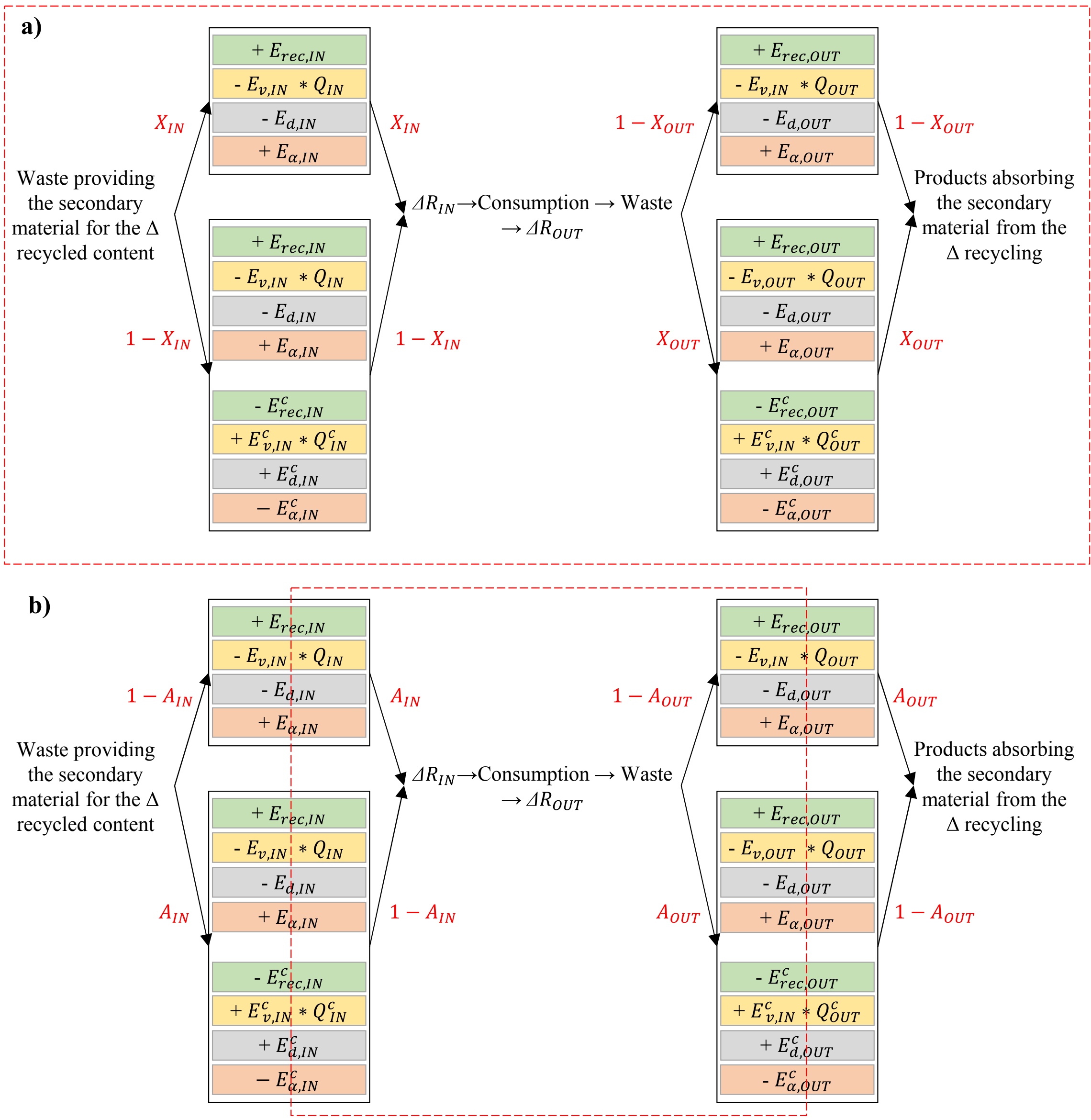

Fig. 3. System boundaries (red dotted lines) of an LCA assessing a change in the recycled content ($\Delta R_{IN} \neq 0$) and in the recycling rate ($\Delta R_{OUT} \neq 0$) displaying the affected recycling processes (green boxes), primary material production (yellow boxes), energy recovery and disposal activities (grey boxes), and downstream processes (orange boxes). a) without allocation between life cycles; b) with allocation between life cycles. The processes are explained in Sections 2.6.1 to 2.6.7, and all the terms are described in Table 1. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

The red lines in Fig. 2 show the system boundaries without allocation between different life cycles (recommended when the full impacts of a policy target are assessed), while the gray lines show the system boundaries in the case of allocation.

2.6. Modeling and calculating the environmental impacts

As described in Section 2.4, we are interested in studying the difference between the baseline system and alternative scenarios, and including only the affected processes.

The environmental impact of the difference between the baseline and alternative scenarios ΔE_{tot} can be summarized with the following equation: $\Delta E_{tot} = \Delta E_{IN} + \Delta E_{OUT} + \Delta E_{D}$, where the impacts due to a change in the recycled content (ΔE_{IN}) are summed to the impacts related to a change in the recycling rate (ΔE_{OUT}) and to a change in the demand (ΔE_{D}). All the terms presented in the equations in Sections 2.6.1 to 2.6.9 are explained in Table 1. To note that the terms describing the life cycle impact assessment are not specific (i.e., given per kg) as in other literature because of the non-linearity of this framework.

By subtracting the alternative scenarios from the baseline, it is possible to identify those processes that are small- or large-scale compared to the market of interest. Fig. 3 shows the system boundaries and the affected process of an LCA assessing a change in the recycled content ($\Delta R_{IN} \neq 0$) and the recycling rate ($\Delta R_{OUT} \neq 0$) without (Fig. 3,a) and with (Fig. 3, b) allocation.

The equations in Sections 2.6.1 to 2.6.9 can be used for both small- and large-scale changes in the foreground system, but the input data can vary according to the magnitude of the investigated change.

2.6.1. Environmental impacts due to a change in recycled content (ΔE_{IN})

Recycled content (R_{IN}) is defined here as the ratio between the input secondary material and the total input material required to make a product.

The impact of a $\Delta R_{IN} \neq 0$, if the goal avoids allocation, is described in Eq. (1) (for a unit of analysis), which shows the two possible chain effects illustrated by the two lines in the equation. All the terms of Eq. (1)

Table 1
Description of the terms used in Sections 2.6.1 to 2.6.9.

Term	Description	Unit
ΔE_{tot}	Difference between the environmental impacts of the alternative scenarios and in the baseline scenario.	$\frac{impact}{FU}$
ΔE_{IN}	Difference between the environmental impacts of the alternative scenarios and in the baseline scenario due to a change of the recycled content (R_{IN}).	$\frac{impact}{FU}$
ΔE_{OUT}	Difference between the environmental impacts of the alternative scenarios and in the baseline scenario due to a change of the recycling rate (R_{OUT}).	$\frac{impact}{FU}$
ΔE_{D}	Difference between the environmental impacts of the alternative scenarios and in the baseline scenario due to a change of the demand to fulfill the service of the functional unit.	$\frac{impact}{FU}$
Alternative scenarios with different recycled content.		
ΔR_{IN}	Difference of the recycled content in the alternative scenarios and in the baseline scenario. The recycled content is defined as the ratio between the input secondary material and the total input material.	$\frac{kg}{kg}$ or%
X_{IN}, X_{OUT}	Effectiveness factors. Number between 0 and 1.	No-dimension
$E_{rec,IN}$	Total environmental impacts of recycling[**] the waste that provides the additional input secondary material.	$\frac{impact}{FU}$
$E_{d,IN}$	Total environmental impacts of the avoided energy recovery and disposal[***] of the waste providing the additional input secondary material.	$\frac{impact}{FU}$
$E_{v,IN}$	Total environmental impacts of the avoided production of primary materials in case the increased recycled content causes a reduction of primary material in the market.	$\frac{impact}{FU}$
Q_{IN}	Ratio between the quality (quantified with technical characteristics or market values) of the secondary material produced to increase the recycled content and of the primary material used in the baseline scenario.	[*]
$\Delta E_{\alpha,IN}$	Difference between the downstream environmental impacts of the use of the input secondary material instead of the input primary material.	$\frac{impact}{FU}$
$E^{c}_{v,IN}$	Total environmental impacts of the induced production of primary materials in case the increased recycled content causes a reduction of recycling in a competing market.	$\frac{impact}{FU}$
$E^{c}_{rec,IN}$	Total environmental impacts of the avoided recycling[**] of the competing waste that was recycled in secondary material in the baseline scenario in case the increased recycled content causes a reduction of recycling in a competing market.	$\frac{impact}{FU}$
$E^{c}_{d,IN}$	Total environmental impacts of the induced energy recovery and disposal[***] of the competing waste that was recycled in secondary material in the baseline scenario in case the increased recycled content causes a reduction of recycling in a competing market.	$\frac{impact}{FU}$
$\Delta E^{c}_{\alpha,IN}$	Difference between the downstream impacts of using the competing secondary material instead of the primary material in case the increased recycled content causes a reduction of recycling in a competing market (e.g., use, transport).	$\frac{impact}{FU}$
Alternative scenarios with different recycling rates.		
ΔR_{OUT}	Difference of the recycling rate in the alternative scenarios and in the baseline scenario. The recycling rate is defined as the ratio between the output secondary material (after collection, sorting, and recycling losses) and the total generated waste.	$\frac{kg}{kg}$ or %
$E_{rec,OUT}$	Total environmental impacts of the additional waste recycling[**].	$\frac{impact}{FU}$
$E_{d,OUT}$	Total environmental impacts of the avoided energy recovery and disposal[***] of the waste being recycled in the alternative scenarios.	$\frac{impact}{FU}$
$E_{v,OUT}$	Total environmental impacts of the avoided production of primary materials in case the increased recycling rate causes a reduction of primary material in the market.	$\frac{impact}{FU}$
Q_{OUT}	Ratio between the quality (quantified with technical characteristics or market values) of the additional secondary material and of the primary material used in the baseline scenario.	[*]
$\Delta E_{\alpha,OUT}$	Difference between the downstream environmental impacts of the production of secondary material in the alternative scenarios and in the baseline scenario.	$\frac{impact}{FU}$
$E^{c}_{v,OUT}$	Total environmental impacts of the induced production of primary materials in case the increased recycling rate causes a reduction of recycling in a competing market.	$\frac{impact}{FU}$
$E^{c}_{rec,OUT}$	Total environmental impacts of the avoided recycling[**] of the competing waste that was recycled in secondary material in the baseline scenario in case the increased recycling rate causes a reduction of recycling in a competing market.	$\frac{impact}{FU}$
$E^{c}_{d,OUT}$	Total environmental impacts of the induced energy recovery and disposal[***] of the competing waste that was recycled in secondary material in the baseline scenario in case the increased recycling rate causes a reduction of recycling in a competing market.	$\frac{impact}{FU}$
$\Delta E^{c}_{\alpha,OUT}$	Difference between the downstream impacts of using the competing secondary material instead of the primary material in case the increased recycled content causes a reduction of recycling in a competing market (e.g., use, transport).	$\frac{impact}{FU}$

[*] The units of Q_{IN} and Q_{OUT} depend on how these factors are calculated.
[**] Recycling here includes collection, sorting, transportation, recycling processes, and the fate of residues from such processes.
[***] Disposal here includes incineration without energy recovery, landfilling, and environmental dispersion.

are explained in Table 1. The *effectiveness factor* X_{IN} is a number between 0 and 1 and splits the mass between the two chain effects, depending on the forecasted market reaction.

$$\begin{aligned}\Delta E_{IN} = {} & \Delta R_{IN}{*}X_{IN}{*}\left(E_{rec,IN} - E_{d,IN} - E_{v,IN}{*}Q_{IN} + \Delta E_{\alpha,IN}\right) \\ & + \Delta R_{IN}{*}(1 - X_{IN}){*}\left(\left(E_{rec,IN} - E^{c}_{rec,IN}\right)\right. \\ & \left. + \left(-E_{v,IN}{*}Q_{IN} + E^{c}_{v,IN}{*}Q^{c}_{IN}\right) + \left(-E_{d,IN} + E^{c}_{d,IN}\right) + \left(\Delta E_{\alpha,IN} - \Delta E^{c}_{\alpha,IN}\right)\right)\end{aligned} \tag{1}$$

In the first case, the increased secondary material is coupled with an increase in recycling in the previous life cycle of the waste providing the secondary material, and it is quantified by summing the direct impacts of recycling ($E_{rec,IN}$), the avoided energy recovery and disposal of waste that is now recycled ($-E_{d,IN}$), and the avoided production of primary material ($-E_{v,IN}$). Since the secondary material often has lower functionality than the avoided primary material, $E_{v,IN}$ is multiplied by a factor Q_{IN}, representing the ratio between the quality of the secondary material and of the primary material. Quality can be based either on technical characteristics (Rigamonti et al., 2020; Zink et al., 2016) or market value (Allacker et al., 2014; Schrijvers et al., 2016). Furthermore, differences in the downstream impacts between the use of secondary versus primary material are considered ($\Delta E_{\alpha,IN}$) and described in Section 2.6.7.

In the second case, the increased secondary material simply reduces the recycled content in other markets, e.g., increasing the use of recycled nutrients from food waste could simply reallocate these nutrients, rather than sourcing additional food waste from the mixed municipal waste sent to energy recovery and disposal. In this case, we are certainly avoiding the use of primary material to fulfill our functional unit ($-E_{v,IN}$), but we are also inducing the production of a competitive primary material market from which we are diverting the secondary material ($E^{c}_{v,IN}$). Note that the competitive market can be composed of different products/markets. Furthermore, we should consider the difference between the collecting/sorting/recycling processes needed for our functional unit ($E_{rec,IN}$) compared to the collecting/sorting/recycling processes required to fulfill competitive demand ($-E^{c}_{rec,IN}$), as well as the difference between avoided energy recovery and disposal ($-E_{d,INP}$) and induced energy recovery and disposal ($E^{c}_{d,IN}$). If the processes avoided and induced are the same ($E_{rec,INP} = E^c_{rec,IN}; E_{v,IN} = E^c_{v,IN}; Q_P = Q^c_{IN}, \Delta E_{\alpha,IN} = \Delta E^c_{\alpha,IN}$), increased recycling does not bring any net environmental burden/saving.

In case allocation is needed, X_{IN} becomes an allocation factor that can be based on the PEF (EC, 2013; Zampori and Pant, 2019) or other market-based methods (see Section 1.2, Supplementary Material).

More information on X_{IN}, $E_{rec,IN}$, $E_{d,IN}$, Q_{IN}, $\Delta E_{\alpha,IN}$can be found in the Sections 2.6.4 to 2.6.7.

2.6.2. Environmental impacts due to a change in the recycling rate (ΔE_{OUT})

The recycling rate (R_{OUT}), also called the end-of-life (EoL) recycling rate (Graedel et al., 2011), is defined as the ratio between secondary material after collection, sorting, and recycling losses and the waste generated.

The impact of a $\Delta R_{OUT} \neq 0$, if the goal avoids allocation, is described in Eq. (2) (for a unit of analysis), which shows the two possible chain effects illustrated by the two lines in the equation. All the terms of Eq. (2) are explained in Table 1. The *effectiveness factor* X_{OUT} is a number between 0 and 1 and splits the mass between the two chain effects, depending on the forecasted market reaction.

$$\Delta E_{OUT} = \Delta R_{OUT}{}^*\left((1 - X_{OUT})^*\left(E_{rec,OUT} - E_{d,OUT} - E_{v,OUT}{}^*Q_{OUT} + \Delta E_{\alpha,OUT}\right)\right) + \Delta R_{OUT}{}^*X_{OUT}{}^*\left(\left(E_{rec,OUT} - E^c_{rec,OUT}\right) + \left(-E_{v,OUT}{}^*Q_{OUT} + E^c_{v,IN}{}^*Q^c_{OUT}\right) + \left(-E_{d,OUT} + E^c_{d,OUT}\right) + \left(\Delta E_{\alpha,OUT} - \Delta E^c_{\alpha,OUT}\right)\right) \quad (2)$$

Similarly to the recycled content (Section 2.6.1), in the first case, the produced secondary material avoids some primary material that would have been produced without recycling ($E_{v,OUT}{}^*Q_{OUT}$), and the impacts of recycling ($E_{rec,OUT}$) are added to the avoided energy recovery and disposal ($-E_{d,OUT}$) and to the downstream impacts ($\Delta E_{\alpha,OUT}$). In the second case, additional recycling simply replaces a competitive material that is now not recycled but sent to energy recovery and disposal. Also in this case, if the processes related to the functional unit are the same as the ones in the competitive market ($E_{rec,OUT} = E^c_{rec,OUT}; E_{d,OUT} = E^c_{d,OUT};$

$$E_{v,OUT} = E^c_{v,OUT}; Q_{OUT} = Q^c_{OUT}; \Delta E_{\alpha,OUT} = \Delta E^c_{\alpha,OUT})$$

, increased recycling does not have any net environmental burden/saving. If allocation is needed, X_{OUT} becomes an allocation factor (see Section 1.2, Supplementary Material).

More information on X_{OUT}, $E_{rec,OUT}$, $E_{d,OUT}$, Q_{OUT}, $\Delta E_{\alpha,OUT}$ can be found in the Sections 2.6.4 to 2.6.7.

2.6.3. Demand (ΔE_D)

The expected future demand of the functional unit can be affected by different strategies, as in the case of waste prevention measures (e.g., a ban on plastic bags) or due to a change in consumer behavior.

The ΔE_D (see Eq. (3)) can be calculated as a sum of avoided material production, including both primary and secondary material, as described in Section 2.6.1, and of the avoided waste management stage, including both the direct burdens and savings that would have happened in the case of recycling (Section 2.6.2) in the baseline scenario.

$$\Delta E_D = \Delta D^*(E_{IN} + E_{OUT} + E_\alpha) \quad (3)$$

Note that environmentally important downstream impacts (E_α) should be considered if present (Section 2.6.7).

2.6.4. The effectiveness factors X_{IN} and X_{OUT}

As highlighted in Eqs. (1) and (2), the impacts can be quite different, dependent on the effectiveness factors X_{IN} and X_{OUT} that describe to what extent the additional recycled content and recycling rate avoid primary material production, or to what extent they simply compete with other secondary materials in the market. The difference between these effectiveness factors and the more common allocation factors is discussed in Section 4.1.

The effectiveness factors quantify the full reaction of the market and forecast the consequences of changes therein. This is a complex and uncertain task, especially in the case of large-scale changes. Due to the uncertainty surrounding these factors, several scenarios should be modeled to evaluate the sensitivity of these assumptions. We do not provide a mathematical equation on how to calculate the effectiveness factors, but we do emphasize the critical data that should be collected for their quantification.

The choice of X_{IN} can be supported by quantifying the mass available for additional recycling, e.g., how much waste is it possible to collect from the residual waste, how easily accessible is such a mass, what are the political and economic side policies that could be implemented to support such a collection (e.g., extended producer responsibility, deposit systems). Second, competing markets supplying the secondary material to fulfill the functional unit should be identified (e.g., composting could be the competing market of suppling organic fertilizers instead of the digestate). Third, the own-price elasticity of supply for the recyclable material could be used to indicate the direction of the market.

The choice of X_{OUT} should be made after collecting data on end-market (markets absorbing the secondary materials) types, size, and trends. A first estimate could be done by looking for products already advertised in the market. End-markets are important because they highlight not only which markets absorb the secondary material, but also why and under what conditions (e.g., the color and/or purity of a material). Another relevant characteristic of end-markets is the current recycled content and saturation level. The potential volume of the market can be calculated by multiplying the size of the end-market by the maximum (current or potential) recycled content. Unsaturated volume is obtained by subtracting the current use of secondary material from this potential volume. Maximum recycled content depends on several factors, such as technical limitations, quality of the secondary material, color, etc.

Another indication of short-term possible market saturation can be forecasted by looking at the response of the market, in particular to crisis conditions. Information on imported/exported waste can provide several indications on market health, the geographical location where potential environmental burdens or savings will take place, and the presence of environmental risks (e.g., environmental dispersion). For example, the recent Chinese ban on low-quality plastic and paper, and the following bans set in place by several other Asian countries, pinpointed the materials for which European recycling industry capacity was saturated or non-existent (i.e., plastic and low-quality paper). Finally, known market bottle-necks or specific weaknesses in the value chains of interest could be relevant.

2.6.5. Recycling

The impacts of recycling ($E_{rec,P}; E^c_{rec,P}; E_{rec,WM}; E^c_{rec,WM}$) include collection, sorting, transportation, recycling processes, and the fate of residues from such processes (e.g., sorting rejects). The information gathered in Section 2.6.4 helps identify these activities.

2.6.6. Energy recovery and disposal (material exiting the loop)

As mentioned earlier, disposal is defined in this paper as including incineration without energy recovery, landfilling, and environmental dispersion.

The impacts of energy recovery and disposal ($E^z_{d,y}$, being y either IN or OUT and z the market objective of the study or the competitive market c) can be calculated as in Eq. (4):

$$E^y_{d,x} = R_{inc}*\left(E^z_{coll,y} + E^z_{inc,y} - E^z_{energyinc,y}\right) + R_{land}*\left(E^z_{coll,y} + E^z_{land,y} - E^z_{energyland,y}\right) + (1 - R_{OUT} - R_{inc} - R_{land})*E^z_{other,y} \tag{4}$$

Eq. (4) shows that the total environmental impacts of the collection $E^z_{coll,y}\left[\frac{impact}{FU}\right]$ (e.g., the collection of mixed residual waste) is added to the total direct environmental impacts of incineration ($E^z_{inc,y}\left[\frac{impact}{FU}\right]$) and landfilling ($E^z_{land,y}\left[\frac{impact}{FU}\right]$), and subtracted from the avoided impact of the energy (electricity and heat) that would have been produced without energy recovery and disposal technologies ($E^z_{energyinc,y}\left[\frac{impact}{FU}\right]; E^z_{energyland,y}\left[\frac{impact}{FU}\right]$). Note that energy recovery also occurs when landfilling, for example, organic waste. The factors R_{OUT} (recycling rate), R_{inc} (% of waste sent to incineration), and R_{land} (% of waste sent to landfill) are mass factors. We also added the impacts of waste that is neither recycled, incinerated, nor landfilled ($E^z_{other,y}\left[\frac{impact}{FU}\right]$), as highlighted for environmental dispersion/littering in (Andreasi Bassi et al., 2020).

2.6.7. Downstream impacts

Downstream impacts ($\Delta E_{\alpha,IN}$, $\Delta E^c_{\alpha,IN}$, $\Delta E_{\alpha,OUT}$, $\Delta E^c_{\alpha,OUT}$) can happen every time a primary material is avoided or induced ($E_{v,IN}, E^c_{v,IN}, E_{v,OUT}$, $E^c_{v,OUT}$). They include all of the direct and indirect impacts that are caused by using secondary material instead of primary material (Schrijvers et al., 2020), such as different manufacturing processes (e.g., the need to use more additives), different impacts in the use phase (e.g., emissions from the use on land of organic fertilizers versus mineral fertilizers, or the combustion of natural gas versus biogas), and differences in waste management (e.g., incineration or landfilling of fossil plastic versus biobased plastic, or where increased use of secondary material lowers the recyclability of products).

Among the downstream indirect impacts, one could also include (market-mediated) rebound effects since the reduced demand for primary material might lead to increased consumption of other goods using that material. Macro-economic models (e.g., partial or general equilibrium models) can be used to forecast such responses of the economy (Almeida et al., 2020).

2.6.8. Background processes

LCA modeling involves several other multifunctional processes running in the background that needs to be solved with system expansion (e.g., energy generation from incineration plants, ancillary material consumption). Changes in background systems, such as electricity production, material production, etc., are often marginal, even when the foreground change is significant. To model marginal changes in background systems, we recommend using long-term marginal processes, although acknowledging that the uncertainty in actual marginal impacts is significant (Eriksson et al., 2007). Several systematic approaches exist for identifying marginal processes (Mathiesen et al., 2009; Mattsson et al., 2003; Palazzo et al., 2020; Weidema et al., 1999).

The impacts of large-scale changes in the background system are more difficult to pinpoint. System dynamics models, technology choice models, or agent-based models - in principle - can be used for this purpose (Palazzo et al., 2020). A mix of expertise on systems, technology development forecasting, market forecasting, technology cost modeling, and macro-economic models can also provide a basis for estimating large-scale impacts on background processes.

2.6.9. The dimension of time

Since assessing large-scale changes in material circularity is often used to support policy targets, which are defined for a precise future year, we incentivize LCA practitioners to model a dynamic (i.e., time-dependent) LCA instead of a static LCA, where possible. This will allow for comparing the results of different ways of transitioning to reach such policy targets. This aspect is often under-evaluated in other methods for modeling waste management and recycling.

There is no clear definition in the literature of what constitutes a dynamic LCA, and there is no consensus on how to deal with the issues of different time horizons and discounting (Lueddeckens et al., 2020), even though there is a growing interest in the topic (Sohn et al., 2020). In general, dynamic LCAs can include the dynamic modeling of goal and scope, inventory analysis, dynamic impact assessment (i.e., using time-dependent characterization factors), and interpretation with time-dependent weighted factors (Sohn et al., 2020).

Even if we do not aim at covering all of the challenges of time horizons and dynamic modeling, we suggest having a dynamic goal and scope that leads to a dynamic system inventory. In relation to policies involving circularity, the choice of a dynamic goal and scope helps investigate time-related consumption trends, policy implementation pathways (for example, constant improvements in time or fast investment in the last year), side policies and their time frames, monitoring indicators, technological improvements, and system developments. The results could also point out the years where an unwanted market response is more likely to happen. For example, since many databases provide yearly datasets, all of the processes included in Eqs. (1) to (4) should be assigned the year they are most likely to happen, and each year could have a different life cycle inventory, for example, a different energy mix composition, technological efficiencies (e.g., sorting and recycling efficiencies), consumption rates, bio-based content, etc. An example of a dynamic life cycle inventory can be found in Andreasi Bassi et al. (2020).

Such a dynamic goal and scope and system inventory could then be followed by the dynamic accounting of emissions, in order, for example, to capture the time-related effects associated with biogenic carbon flows (Brandão et al., 2013; Cherubini et al., 2011; Faraca et al., 2019; Tonini et al., 2021).

3. Application on PET tray circularity

The application of the framework is illustrated for the hypothetical evaluation of an EU-wide policy target on PET trays. PET trays are defined as all the thermoforms packaging made of PET (Petcore Europe, 2016) and represent one of the plastic circular economy bottlenecks because they currently absorb a high quantity of secondary PET while having a very low source-separation and recycling rate (Plastics Recyclers Europe, 2020). The framework is illustrated by showing the goal and scope, the life cycle inventory, and the mass balance. No life cycle impact assessment results are provided because it is beyond the scope of this paper.

3.1. Goal

The goal is “To quantify the environmental impact of implementing a new policy target increasing the recycled content of the PET trays consumed in the EU27+1 to 70% in 2025 (from the current 40%), combined with different changes in the waste management system to reach such recycled content”. The split of impacts between subsequent life cycles is not the focus of the policy-maker, as the interest is placed in a system-level assessment and in highlighting the hotspots and the risks that would prevent environmental improvements from occurring.

3.2. Functional unit

The functional unit is the “consumption of PET trays in the EU27+1 in 2025". The total PET tray demand in 2018 was 0.9Mt (Plastics Recyclers Europe, 2020), and the yearly growth rate is assumed to be 5.4% (Deloitte Sustainability, 2017; Plastics Recyclers Europe, 2020; Wood Mackenzie, 2017), meaning that, in 2025, the EU27+1 demand is expected be 1.3Mt PET trays. In total, 14% of the trays are assumed to be black (Eriksen and Astrup, 2019), meaning that only 14% of them can be manufactured from secondary flakes from colored PET bottles, while the rest can be made from clear PET bottles or PET trays.

3.3. Baseline scenario

The secondary material used in PET trays is assumed to come from a combination of three markets (56% from clear bottles, 31% from mixed colored bottles, and 13% from trays (Andreasi Bassi et al., 2020)). The recycling of PET trays back into PET trays is a very niche market, due to the very low demand, low capacity of dedicated recycling facilities, and a high percentage of multi-polymeric trays.

3.4. Alternative scenarios

To reach the desired target (i.e, 70% recycled content in 2025), more than 0.46 Mt of food-grade secondary PET granules need to be produced. Two alternative scenarios were modeled based on two potential implementation scenarios for PET packaging.

The alternative scenario I assumes no change in the PET waste management, meaning that the additional secondary granules are simply shifted from other markets that previously absorbed these (i.e., bottles and polyester).

The alternative scenario II models a consorted legislative effort to increase the overall PET recycling rate by increasing the source-separation of bottles and trays and incentivizing sorting and recycling facilities in Europe. In this case, the additional secondary material for PET trays is derived from the increased source-separation of PET bottles and PET trays. Due to the high demand for clear secondary PET (as previously mentioned), it is more likely that the PET trays would absorb secondary PET from colored bottles to fulfill the demand for black PET trays and source the remaining secondary material from PET trays.

3.5. System boundaries

Fig. 4 shows the system boundaries of the alternative scenarios I (a) and II (b).

3.6. Impacts of the PET circularity

In the alternative scenario I, the additional 0.39 Mt of secondary granules are simply shifted from the manufacturing of the competitive market to trays. Due to the high pressure on PET bottle producers who have to increase their recycled content after the EU Directive 2019/804 (EC, 2019), it is more likely that the competitive market would be polyester used in textile. If polyester is the competitive market, the recycled content of European polyester becomes almost non-existent because the new policy target would divert the secondary material from polyester producers to PET trays producers. The only processes that count in the environmental assessment would be the difference between the primary materials avoided $E^c_{v,IN} - E_{d,IN}$ (i.e., is the primary PET used in the trays the same as in polyester?) and the recycling processes $E^c_{rec,IN} - E_{rec,IN}$ (i.e., is the recycling of PET in secondary material for PET trays the same as recycling PET in secondary material for polyester?). However, the alternative scenario I could be tested to quantify the error of identifying the competing market in bottles instead of polyester. In this case, the recycled content of PET bottles would decrease from 11% in the baseline (EPBP, 2021) to 1% in the alternative scenario. Note that the different recyclability of bottles, trays, and polyester would not affect the results, since these products would be produced and handled as waste, with or without the use of a secondary material instead of a primary material.

The alternative scenario II assumes that the additional 0.39 Mt of secondary granules required to increase the recycled content of trays come from increased source-separation and recycling of PET bottles. However, only 14% of this 0.39 Mt can originate from colored bottles,

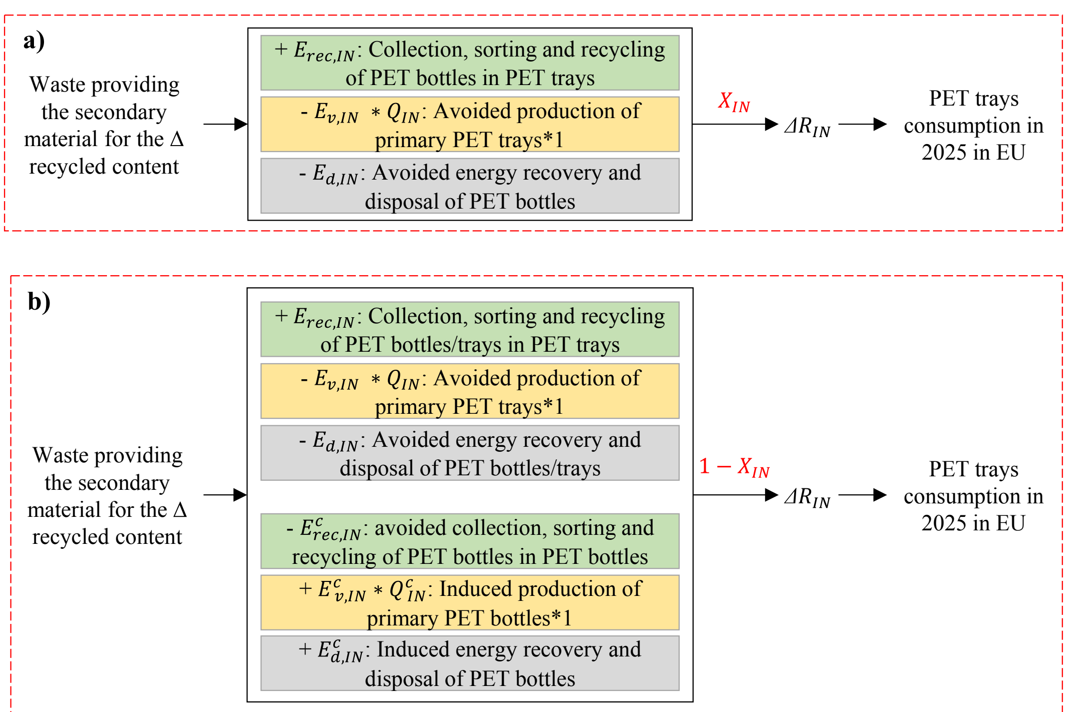

Fig. 4. System boundaries (dotted red lines) for the alternative scenarios I (a) and II (b), assuming all quality factors equal to 1 and no downstream impacts. EU indicates EU27+1. All the terms are described in Table 1. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

because only 14% of the trays are assumed to be black and can absorb this material (Eriksen and Astrup, 2019). In this alternative, the source-separation rate of bottles (excluding impurities) would have to increase from 57% in the baseline (ICIS, 2018; Plastics Recyclers Europe, 2020; Wood Mackenzie, 2017) to around 75% (depending on the type of source-separation and on the level of impurities). The increase in bottle source-separation also leads to an additional source-separation of 0.22 Mt colored bottles, 0.13 Mt of which cannot be absorbed by the tray manufacturing process, due to the constraint on the 14% maximum black trays. Colored bottles are usually used for strapping manufacturing and for black polyester; however, these are already saturated markets in Europe. Since in the baseline 25% of PET bales made from colored bottles are already exported outside EU27+1, it is unlikely that a new market able to absorb this quantity will be created in EU27+1 in the short time horizon considered herein (i.e, 2020–2025). This means that either corrective policies will be put in place to increase the share of clear material, or colored bottles will likely continue to be exported off EU27+1 or sent to energy recovery and disposal in the sorting step.

4. Discussion

4.1. Applicability of the framework

The application of the framework to an EU-policy target on PET trays showed that collection rates, recyclability, and market absorption are equally important to consider, as well as how side policies (i.e., European targets for the source separation collection rate) could be employed to avoid unintended system responses that would reduce or cancel off the environmental savings.

The framework provided additional information on the potential responses of the market. Further analyses could focus on analyzing the impacts of reducing the dependency on fossil PET through, for example, alternative bio-based feedstock. Several other alternative scenarios could be tested (e.g., changing only the collection system, increasing the recyclability of products, removing colored bottles from the market, etc.). While it was not focus of the current study, we recommend that uncertainty analysis should also be performed, to increase the robustness of results and interpretation.

4.2. Differences with other frameworks

While Eqs. (1) and (2) may appear similar to the circular footprint formula proposed by the PEF method and reported in the SM (EC, 2013; Zampori and Pant, 2019), and to the consequential LCA applied to marginal changes (Ekvall, 2000; Schrijvers et al., 2020), all of these methods require allocation between life cycles and rely on factors "A" indicating whether a secondary material market is constrained (i.e., saturated, characterized by a high offer and a low demand) or unconstrained (i.e., not saturated, characterized by a low offer and a high demand).

As described previously, we do not recommend allocation when assessing large-scale changes in material circularity (notably, waste policy targets); instead, we suggest the use of X_{IN} and X_{OUT} as parameters that expand the system to jointly assess the actions of recycling material from one life cycle and using recycled material in the next life cycle. We support the idea that, in the case of large-scale studies, modeling recycled content or waste management activities separated from each other is as analyzing the effect on applauses from "clapping with one hand" (Ekvall et al., 2021b, 2021a). However, Eqs. (1) and (2) can be transformed to include allocation, if needed. Comparing Fig. 3a with Fig. 3b, it appears clear that the LCA avoiding allocation would not give the same results as summing the different life cycles in the case of allocation.

Unlike others, our framework does not change in the case of a closed-loop. Even if the term closed-loop recycling has been utilized in several papers (e.g. Marie and Quiasrawi, 2012), it does not have a clear definition (Geyer et al., 2016) and in itself does not provide any additional information on the environmental effects connected to market demand, to the risk of market saturation, and to which materials are avoided (Andreasi Bassi et al., 2020; Geyer et al., 2016; Lonca et al., 2020). Moreover, as described in Section 2.5, it is highly unlikely that the different markets to be analyzed will coincide. Furthermore, we consider our framework to be more complete than the others, as it allows LCA practitioners to include all the overall interactions between several different markets. This is demonstrated by the higher number of processes included (e.g., downstream impacts, maintained mass balance in recycling).

To our knowledge, this is the only proposed framework for LCAs that can be applied to any size of change in material circularity, like other frameworks that focus only on marginal changes. We recommend applying the developed framework to large-scale case studies where decision-makers have an influence on the investigated policy targets and side-policies. Finally, the framework allows dynamic modeling, in particular relevant with respect to time-dependent life cycle inventories and the life cycle impact assessment stage.

4.3. Limitations of the framework

Three main limitations are found to be associated with the framework: i) the intrinsic uncertainty of consequential LCA, ii) the challenging use of macro-economic models for definition of market responses, and iii) the incompatibility of the framework for product LCAs.

First, while a common critique of consequential LCAs is uncertainty in market responses (Plevin et al., 2014; Zamagni et al., 2012), which certainly increases when addressing future consequences, we believe that such uncertainty can be captured with an appropriate set of baseline scenarios encompassing potential developments in framework conditions and selection of alternative scenarios reflecting potential effects of the investigated policies. Defining a set of relevant "what-if" or "likely" scenarios can inform decision-makers about the consequences from specific actions and initiatives, while systematic analysis of uncertainties through parameter uncertainty propagation, data quality evaluation (e.g., pedigree matrix), and sensitivity analyses on framework conditions support more robust results and interpretations thereof.

Second, the suggested use of macro-economic models (Sections 2.6.7 and 2.6.8) is often challenging due to their "coarse" disaggregation of economic sectors (Mattila, 2017). Yet, advances have been made to adapt some models, such as GTAP, to better support individual product types such as biofuels (Dandres et al., 2011; Igos et al., 2015), indicating that similar advancements are possible for other sectors such as waste management. To improve relevance for LCA, further flexibility in disaggregation within these models is required.

Third, an assessment of products or company-level impacts should follow different approaches as allocation between individual life cycles becomes essential. Relevant methods for such product-oriented LCAs already exist in the literature (Allacker et al., 2014; Ekvall, 2000; Schrijvers et al., 2016; Zampori and Pant, 2019).

5. Conclusions

This paper describes a methodological framework for the definition and modeling of large-scale consequential life cycle assessments aiming at quantifying potential environmental impacts of policy targets focusing on increasing material circularity.

Two types of scenarios are proposed, (multiple) baseline scenarios and alternative scenarios representing potential effects of an implemented policy target. The difference in environmental performance between these two scenarios represents the consequential impacts associated with the implementation of the policy. Detailed recommendations for the goal and scope phase of LCAs are provided, combined with mathematical formulations of how to calculate the environmental consequences, and are supplemented with an illustrative example of applying the framework for EU target setting of recycled content in PET trays. Compared with previous frameworks in literature, allocation of impacts between individual life cycles is not recommended to support a system-level focus reflecting the legislative scope of the policies in question. The framework can accommodate changes and systems of various sizes within the suggested calculation approach, and also allows for non-linear market responses and dynamic modeling (i.e., time-dependent) when relevant. Applying the framework on an illustrative example of increasing the recycled content of PET trays consumed in the EU27+1 from 40% to 70% in 2025 (combined with required changes in the waste management system to reach this level of recycled content), demonstrated that such a policy target has to be supported by side policies (i.e., parallel initiatives) to also increase recycling of PET trays themselves. If not, secondary material already recycled in polyester textiles and bottles are likely to be diverted to trays. The likely consequence of this is a considerable drop in recycled content of polyester to negligible levels, or for PET bottles a decrease from 11% to 1%, thereby not leading to the desired environmental impacts at system level. This illustrates that singular recycling targets may not be sufficient and that consequences throughout the full system are essential when assessing impacts from policy targets.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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The views expressed in this article are the sole responsibility of the authors and in no way represent the view of the European Commission and its services.

Appendix A. Supplementary material

The Supplementary Material includes several examples on how to practically use the framework (including goal, functional unit, reference flow, baseline, and alternative scenarios). Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2021.09.018.

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