Sustainable Production and Consumption 45 (2024) 509–524

Contents lists available at ScienceDirect

Sustainable Production and Consumption

journal homepage: www.elsevier.com/locate/spc

Review article

A comprehensive framework covering Life Cycle Sustainability Assessment, resource circularity and criticality

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A R T I C L E I N F O

Editor: Prof. Piera Centobelli

Keywords:
Critical Raw Materials (CRM)
Circular economy (CE)
Life Cycle Sustainability Assessment (LCSA)
Product system
Area of protection (AoP)
Sustainability framework

A B S T R A C T

Life Cycle Sustainability Assessment (LCSA) is a recognised methodology for analyzing and communicating the sustainability of product systems. However, the fragmented analysis of the environmental, social and economic dimensions of sustainability remains an obstacle to its wider application. In addition, the increasing demand for raw materials leads to resource availability issues that are not necessarily addressed by LCSA, but are addressed by Criticality and Circularity (here called C^{2}s) assessment of materials. Based on these methodologies, we propose an integrated comprehensive LCSA framework that includes criticality and circularity assessments (here called LC^{3}SA) and addresses the issue of inconsistencies, such as overlaps. To do this, we review the most relevant literature on the topics and analyze the scope of the LCSA and C^2 methodologies in detail. We analyze how the definition of the product system is dealt with, how the methodologies view the product system, including their similarities and differences, and what the ultimate goal of an integrated assessment is. The analysis shows that all the assessment methods look at the product system from different perspectives. That is, they use different approaches, different system boundaries and different data and information. We argue that the consistency of an integrated assessment does not necessarily depend on the alignment of these perspectives. It mainly concerns the consistent definition of the product system, including the collection of available data and information to characterizes the system. Subsequently, the definition of indicators based on the areas of protection (AoP) is key to identify, understand and track the cause-effect mechanisms to improve the chances of making conscious choices towards a more sustainable system. Based on the discussion and conclusion of the analysis of each of these features, we propose a stepwise LC^{3}SA framework. We argue that each of the steps improves the interconnectedness, consistency, credibility and transparency of the assessment by considering common modelling assumptions. However, further research is needed to address data gaps or issues in the interpretation of results.

1. Introduction

In the 1980ies, the arising imbalance between nature and the human-made environment caused by the impacts of the resource-intensive living standards of a growing population became more apparent. To counter the imbalance, it became clear we need more sustainable development to meet current and future generations' needs (WCED, 1987). Ever since, the United Nations has addressed humans' right to

Abbreviations: C^{2}s, circularity and criticality; AoP, areas of protection; CA, Criticality Assessment; CE, circular economy; cLCC, conventional life cycle costing; CRM, Critical Raw Materials; EcLCA, economic life cycle assessment; eLCC, environmental life cycle costing; EoL, end-of-life; EU, European Union; G&S, Goal and Scope; LC^{3}SA, LCSA + C^{2}s = LCA + LCC + S-LCA + Circularity + Criticality; LCA, (environmental) Life Cycle Assessment; LCC, Life Cycle Costing; LCI, Life Cycle Inventory; LCIA, Life Cycle Impact Assessment; LCSA, Life Cycle Sustainability Assessment = LCA + LCC + S-LCA; PEF, Product Environmental Footprint; R&D, research and development; SETAC, Society of Environmental Toxicology and Chemistry; S-LCA, Social Life Cycle Assessment; sLCC, Societal Life Cycle Costing; TCO, total cost of ownership; UNEP, United Nations Environment Programme.

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https://doi.org/10.1016/j.spc.2024.01.018
Received 5 July 2023; Received in revised form 19 January 2024; Accepted 21 January 2024
Available online 1 February 2024
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well-being and accessibility to a habitable planet with different initiatives, including the latest 17 Sustainable Development Goals (UN, 2015a). The Paris agreement (UN, 2015b) became a landmark when policymakers committed to join efforts to combat climate change. Yet, to be able to act, one first needs to understand the dimension and sources of the problem.

Over the last decades, the methodological developments for measuring and reporting social, economic and environmental impacts show progress based on integrated solutions (Finkbeiner et al., 2010; Popescu et al., 2021). In the European Union (EU) context, the development of the Product Environmental Footprint (PEF) represented a significant step. The methodology proposes a harmonized European approach to measure and communicate the product's environmental performance based on (environmental) Life Cycle Assessment (LCA) (EC-JRC, 2012).

LCA is the "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle" (ISO, 2006a). It is a scientifically based and standardized methodology (see ISO 14040/44, 2006a, 2006b), first proposed in 1960's and largely researched since then (Bjørn et al., 2018). There are four methodological steps: the definition of Goal and Scope (G&S), Life Cycle Inventory (LCI), Life Cycle Impact Assessment (LCIA) and the interpretation phase (ISO, 2006a). However, LCA and PEF focus on the environmental pillar of sustainability. In this sense, Life Cycle Costing (LCC) and Social LCA (S-LCA) methodologies were later on proposed to measure the social and economic impacts. Based mainly on the same methodological steps, LCC and S-LCA have different characteristics from LCA because the impacts they aim to address are not easily measurable due to data availability and variability (Hackenhaar et al., 2022b). Yet, LCA, LCC and S-LCA together make up the full portfolio of a Life Cycle Sustainability Assessment (LCSA) (Kloepffer, 2008).

The first and currently most widely accepted methodological framework for LCSA was developed by the UNEP/SETAC Life Cycle Initiative (UNEP/SETAC, 2011). Such a sustainability perspective of a product system producing goods and services allows for identifying trade-offs between environmental, social or economic impacts (UNEP/SETAC, 2011). Ultimately, LCSA aims to assess the impacts of a product system to a targeted value and/or concern called Area of Protection (AoP). AoPs are the entities to be sustained or protected. They define which impacts should be assessed and modelled (Dewulf et al., 2015a). Typical AoPs concern human health, ecosystems and natural resources (Verones et al., 2017). According to the first principle by Valdivia et al. (2021) when conducting LCSA, the AoPs and impact pathways need to be understood, including the underlying cause-effect relationships connecting inventory results to the assessed impact indicators (i.e. a quantifiable representation of the AoPs). However, there is no consensus in the literature on the AoPs in LCSA (Schaubroeck and Rugani, 2017). Besides, according to Alejandrino et al. (2021) and Valdivia et al. (2021), the shortcomings of LCSA studies include, but are not limited to: a lack of interconnectedness among the three domains of sustainability; the application of contradicting models and assumptions such as inconsistent system boundaries; lack of transparency of assumptions and data collection; lack of a clear criterion for the definition of the functional unit, system boundaries and selection of indicators; not following cause-effect chains and mechanisms leading to an endpoint; not clearly defining the target audiences and users of the LCSA results and disregard their goals, personal values, or cultural differences. At the same time, a suitable method should also have desired features such as acceptance, easiness of application, credibility, robustness and relevance (Wiedmann et al., 2009). These methodological gaps create a barrier to the wide application of LCSA for decision-making at company or policy level (Alejandrino et al., 2021).

Moreover, the Covid-19 pandemic has started an economic slowdown, with a shortage in products/materials supply globally (Karmaker et al., 2021). This emphasized a growing concern about resource scarcity. Natural resources are at the very beginning of every supply chain for a broad range of goods and services including future eco-efficient technologies (Dewulf et al., 2015b; European Commission, 2020). In light of this, companies, governments and academia have been dedicating their efforts to promote resource efficiency (Velenturf and Purnell, 2021). For instance, policy-makers in the EU have been relying on actions towards more responsible and efficient use of resources through the transition towards a more circular economy (CE) (European Commission, 2008, 2011a; Reichel et al., 2016). In 2008, the Raw Materials Initiative was launched to support the sustainable supply of raw materials in the EU and improve resource efficiency and recycling by addressing materials' criticality and circularity (Nuss and Blengini, 2018).

The dictionary defines criticality as something that is extremely important to someone. In this context, the criticality of raw materials is related to their high importance to an economic activity and the risk of supply disruption that can be associated with it (European Commission, 2017). The Criticality Assessment (CA) proposed by the European Commission (European Commission, 2011b) intends to flag the supply risk of relevant/important raw materials for the EU economy according to its needs (European Commission, 2017). It facilitates the management of the so-called Critical Raw Materials (CRM) to secure the industrial value chains by prioritizing needs and actions, e.g. creating incentives for the local production of CRM, recycling and recovery of CRM, and negotiating trade agreements with suppliers (European Commission, 2020).

Meanwhile, the transition towards a CE can foster waste prevention and reduce dependency on virgin natural resources. Although a single definition of CE is debatable (Kirchherr et al., 2017), the European Commission defines it as "a system which maintains the value of products, materials and resources in the economy for as long as possible, and minimises the generation of waste" (EC, 2024). Also called R-strategies, circularity strategies include, but are not limited to, reuse, repair, refurbish, remanufacture, repurpose and recycle. (Moraga et al., 2019; Potting et al., 2017). They are embedded within the overarching framework of ecodesign, where life-cycle thinking serves as a fundamental pillar (EC-JRC, 2011). An efficient circular system allows the production of recycled or reusable materials from what is otherwise called 'waste' that serves as secondary raw materials (Ardente et al., 2019). Secondary raw materials can also help to minimise supply risks, i.e. materials' criticality. Thus, materials circularity is an important factor to include when analyzing resource flows within sectors and products (Rigamonti et al., 2017). The transition towards a circular economy is expected to create a 'greener' and potentially less risky economy. However, a more circular system does not necessarily mean a more sustainable one (Kirchherr et al., 2017; Thomassen et al., 2024). In this sense, both circularity and criticality indicators must be integrated within a sustainability assessment framework such as LCSA to provide a better understanding of resource-related impacts or dependencies. However, the integration of such complex methodologies is not a simple task.

Circularity and criticality indicators and related methods were reviewed in terms of robustness, credibility and relevance (ORIENTING, 2021a). A few proposals exist on how to integrate those indicators into LCA or LCSA (e.g. Sonnemann et al., 2015; Mancini et al., 2016; Adibi et al., 2017; Yavor et al., 2021; Luthin et al., 2023). For example, the most recently published circular lifecycle sustainability assessment (C-LCSA) framework (Luthin et al., 2023) discussed the complexity of coupling circularity with LCSA, by indicating the adaptation needs of LCA, LCC and S-LCA to include circularity. However, both C-LCSA and previous frameworks fail to link circularity and criticality with the three sustainability domains and at the same time address the shortcomings of LCSA (listed above).

The main goal of this research is to propose an integrated comprehensive LCSA framework that includes criticality and circularity assessment and addresses how to deal with inconsistencies among assessment methods, such as overlaps. Therefore, we propose a LC^{3}SA framework, where Criticality and Circularity are the "C^2s" that complement $LC^{(1)}SA$. Based on a review of the public and grey literature on LCSA, we investigate the fundamental differences in the approaches to circularity and criticality assessment and the methods used within LCSA, i.e. LCA, LCC and S-LCA. Based on the analysis, we propose an adaptation of the widely-known framework for life cycle assessment to concisely integrate all the assessment methods. The literature review and structure of this paper are explained in Section 2.

2. Methodology

2.1. Literature review

The starting point was the literature cited in the extensive but not exhaustive review of the ORIENTING project, addressing all sustainability domains as well as criticality and circularity separately (ORIENTING, 2021a, 2021b, 2021c, 2021d, 2021e, 2021f, 2022). The European ORIENTING project (2020–2024, https://orienting.eu/) aims to develop an operational LCSA framework that considers not only environmental, social and economic aspects of a product life cycle but also material criticality and product circularity metrics. Two different approaches were taken in ORIENTING's literature review (LR): snowballing sampling and/or systematic LR. Assessment methods from industry initiatives known to ORIENTING experts were also considered (ORIENTING, 2021d). The LR was conducted between November 2020 and April 2021. The LR method for each of the domains is briefly described in SM1, while a detailed description can be found in the ORIENTING reports stated above.

2.2. Analysis of the state of art

As the philosopher Aristotle coined, "the whole is greater than the sum of its parts". Therefore, in order to propose a coherent integrated assessment framework, it is necessary to understand each of its parts and how they interact. Grounded on the existing literature, the scope of the methodologies of C^2s and LCSA are analyzed in detail in terms of their life-cycle perspective, the definition of the product system, how the methodologies look at the product system, including their similarities and differences, and what the ultimate goal of an integrated assessment is. In Sections 3, 4 and 5, each of these aspects is analyzed and discussed.

In Section 3, we analyze the relationships between a product system, the nature- and the human-made environment and summarize the findings in a comprehensive product system diagram. In Section 4, we analyze how the impacts and effects are measured, quantitatively and/or qualitatively, by distinct indicators and methods in the environmental, social, economic, criticality and circularity assessments. In addition, we discuss the similarities and differences of the methodologies. Sustaining that the ultimate goal of LCSA is to communicate the impacts on an AoP (see Section 1), in Section 5 we analyze and discuss Section 5 which impact pathways are considered by which methodology and in which way. Finally, based on the findings described in the previous sections, we propose a stepwise LC^3SA framework, described and discussed in Sections 6 and 7, respectively.

The framework does not prescribe which approach to take or which methods or indicators to use.[1] Rather, it guides practitioners to make modelling choices in a consistent way when setting up a comprehensive LC^3SA. In addition, integrating the LC^3SA results into a single score is out of scope. This and other challenges related to the application of the framework are highlighted and discussed (Section 7).

[1] Best available approaches for all elements of the LC3SA have been identified in ORIENTING against a set of sub-criteria (ORIENTING, 2021b). A full guide to the ORIENTING LCSA framework is forthcoming.

3. In-depth analysis of a product system and the associated flows and implications

LCSA has several conceptual and practical challenges. To understand the overall benefits and limitations of the integration of circularity and criticality assessment into LCSA, we first analyze what the aims are of LCA, LCC and S-LCA when assessing a product system and how they do it.

During the G&S definition, the first phase of the LCA, the intended application, the reasons for carrying out the study and the intended audience are specified to define the goal. The scope addresses the goal by describing the product system and its boundaries, defining both the foreground as well as the background systems; product system functions and the corresponding functional unit (used as reference unit for data compilation and comparison of results); allocation procedures; impact indicators and their impact assessment methodology; data sources and quality requirements; and all assumptions and limitations of the study. Following the definition of the G&S, relevant data on inputs and outputs of each unit process in the product system is collected and calculated during the LCI phase. All inputs and outputs are then related to potential impacts through the characterization modelling at the LCIA step. The interpretation of the results might lead to conclusions and recommendations or reviews of the other methodological steps. Since LCA is standardized (ISO, 2006a, 2006b), it was considered as a reference for the analysis of the integrated assessment.

In LCC and S-LCA, a similar methodological approach is taken (Hunkeler et al., 2008; UNEP, 2020; UNEP/SETAC, 2011), however, no standardization exists for these domains yet. Because of the different characteristics of assessment methodologies used in the single domains, an alignment of the G&S is not straightforward. For example, as highlighted by Swarr et al. (2011) regarding the complementary analysis of LCC and LCA, the goal and scope definition is similar to both studies, but some cut-off criteria may differ among them. Yet, the approaches should have the same reference point which allows for establishing a common functional unit (or functional equivalent) to communicate impacts: a consistent definition of the product system.

Current product systems are complex: they are embedded in a globalized economy, with variable and dynamic geographical, socio-economic and cultural characteristics and associated environmental impacts (Hackenhaar et al., 2022b). These characteristics influence all methodological steps of the LCSA studies. This section presents an in-depth analysis of the relationships between the product system and the complex context in which it is inserted to provide a common ground for the analysis of each of the approaches. Terms and definitions used throughout the text can be found in SM2.

3.1. Identifying the relationships between a product system, the nature- and human-made environment

As shown in Fig. 1, the human-made environment, or *technosphere*, represents the part of the natural environment, or *ecosphere*, that has been modified by humans for use in human activity (Schaubroeck et al., 2013). Humans are distributed across *different regions* of the world (represented by grey circles), consisting of communities here called *locations*. These locations are smaller portions of regions distinguished by cultural, political and societal organizations (represented by black circles). In the different locations, activities occur (here called *processes*, represented by blue parallelograms) and are part of the product system under study. Each process has inputs entering the system and outputs leaving it. These flows represent exchanges between the natural and human-made environment (represented by straight yellow and green arrows), or within the human-made environment (represented by straight blue and wavy grey arrows). Energy and natural resources, products, co-products, wastes and releases are the physical flows (straight blue arrows) and constitute inputs and outputs, in the narrow sense (UNEP/SETAC, 2011). In a broad sense, non-physical socio-

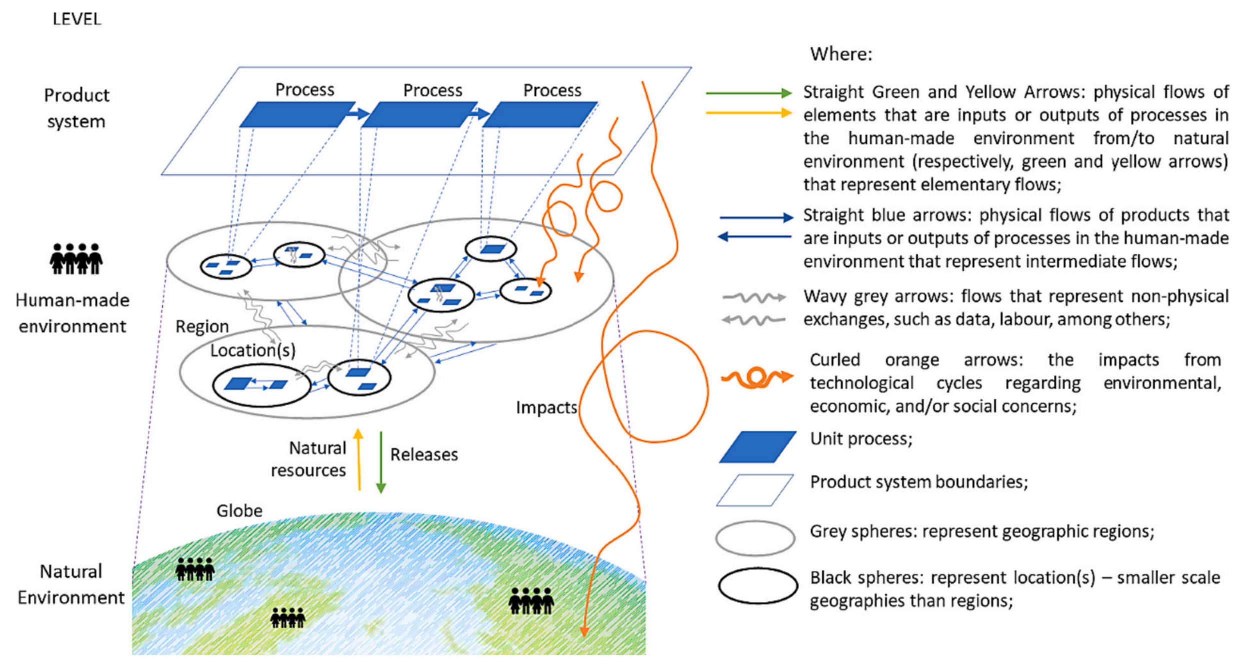

Fig. 1. Contextualization of the product system within the natural environment (planet) and the human-made environment (people and economic prosperity).

economic flows exist and are also needed for the functioning of each process. They are, for example, the work and wages, represented by wavy grey arrows (Goedkoop et al., 2020).

The exchange of both physical and non-physical flows in the product system leads to impacts (represented by curled orange arrows), that are burdens or benefits impacts to the AoPs (ISO, 2015). These impacts happen in human-made and natural environments through various cause-effect chains. When the impacts of each process in a supply chain of a product are integrated, they represent the total contribution, direct and indirect, of a product system to sustainability impacts (Di Noi et al., 2020).

Fig. 1 also shows the complexity of cause-effect mechanisms in multi-dimensional, multi-geoscale and multi-size impacts between multiple stakeholders (Taelman et al., 2020a, 2020b). They are multi-dimensional because of the different environmental, social and economic nature of impacts. They are multi-geoscale because they reflect the different geographical locations of the sources (point of release or extraction of raw materials) of these impacts. They are multi-size because of the different magnitude of the impacts from micro (local, e.g., noise, fair wages or raw material prices) to macro (regional or global, e.g., climate change, poverty or raw material prices) scales. They also can impact different groups of stakeholders such as workers, consumers or society (UNEP, 2020).

In conclusion, processes take place in different locations and utilize social, economic and natural resources to fulfil the needs of people in the form of products through time. LCSA aims at translating those complex dynamics into quantifiable flows to estimate the impact of human activities of the *product system*, represented by the grey contour parallelogram, on the environment, society and economy. The following section details the representation of these dynamics in a comprehensive product system scheme.

3.2. Analysis of product system in function of cause-effect modelling

A product system analysis potentially considers all the variables of the complex dynamics explained in Section 3.1. Which variables are accounted for in the analysis depends on the scope of the study and the definition of the system boundaries. The differences in the definition of the product system boundaries are one of the main challenges in LCSA (Fauzi et al., 2019). The assessment methodologies might not have the same system boundaries if they do not have the same perspective of the system being studied (Hunkeler et al., 2008). Due to differences in the nature of the analysis of the environmental, social and economic domains, a consistent but not necessarily identical system boundary is key (Alejandrino et al., 2021). Hence, Fig. 2 attempts to detail the product system in terms of a flow scheme which potentially identifies all flows and associated impacts of all kinds, being environmental, social, and economic. In this paper, for the sake of comprehensiveness, we consider the foreground system to contain all relevant processes in the life cycle of a given product from cradle-to-grave, i.e. from the extraction of resources to end-of-life (EoL) of the product.

First, the product system is split into a foreground and background system. The foreground system consists of (1) the product supply chain, (2) the use stage; and (3) the EoL stage. The foreground system does not only exchange directly with the natural environment but also with various other value chains within the human-made environment that are present in the background system. Within the foreground system, the product supply chain can be further divided into three core processes: the primary production delivering (primary) raw materials, the manufacturing-stage 1 delivering components, or intermediate products, and the manufacturing-stage 2 delivering final products. The product supply chain also relies on a production environment with its infrastructure and also involves product design and research and development (R&D). The use stage normally has the longest timespan in the product system and often relies on operational energy and/or water as well as other maintenance processes (CEN, 2011). The EoL stage can be further subdivided into three phases. An eventual hibernation phase, also called hoarding, might happen when the materials or products no longer in-use can be considered at stock in the technosphere. That can, for example, last a few days in the case of house solid waste, to years, in case of laptops forgotten in a deposit. After hibernation, products will be subject to waste treatment or product recovery (Godoy León et al., 2020). At the product recovery stage, the various R-strategies can be employed. If no EoL recovery strategy is implemented, the product is to be labelled as 'waste' and undergoes a waste treatment process.

Processes in the foreground and background systems utilize *(a) natural resources, including land transformation and occupation* and create different (b) *releases to the air, soil or water* (Edelen et al., 2018;

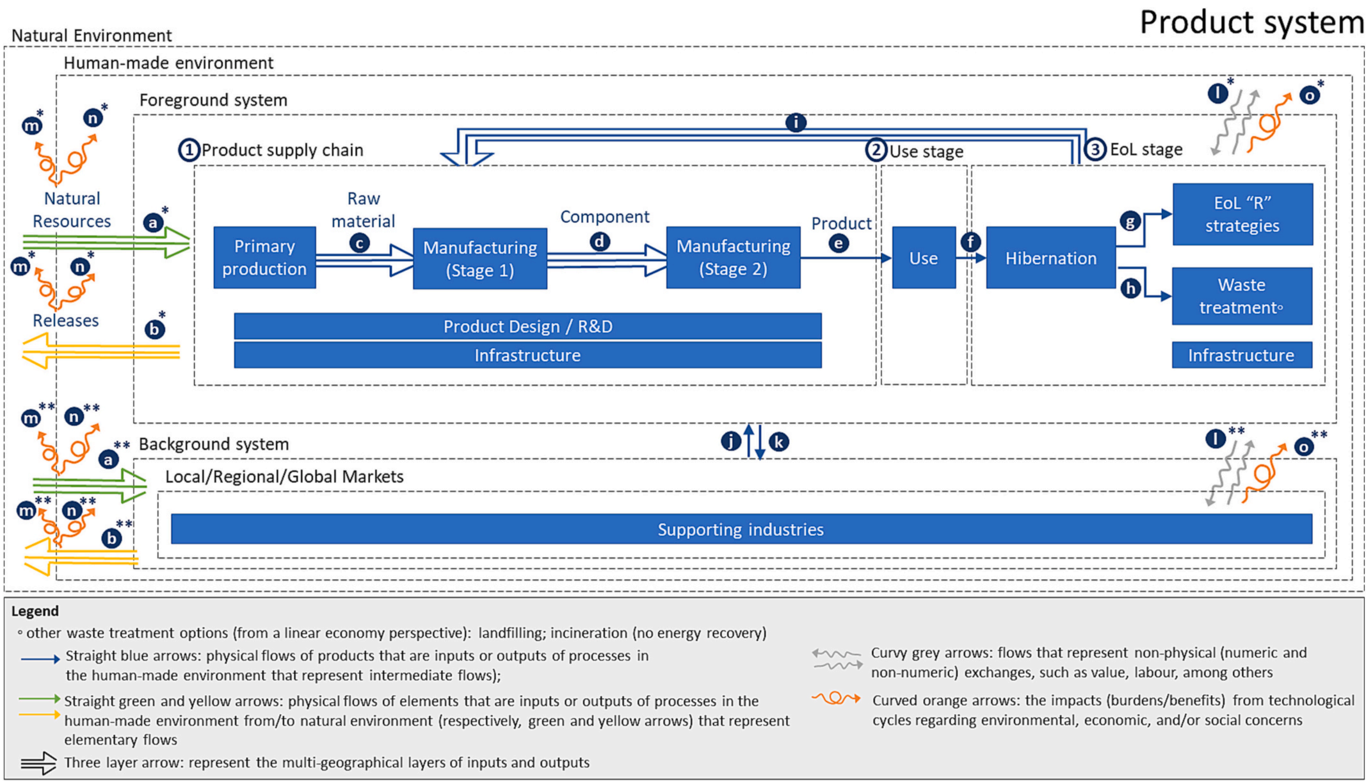

Fig. 2. Detailed product-system analysis based on life cycle stages of (1) production (represented by the product supply chain), (2) use stage and (3) end-of-life (EoL) for foreground and background systems. The flows represent: (a) natural resources (including land transformation and occupation); (b) releases to the air, soil and water; (c) processed raw materials; (d) components; (e) processed products; (f) used products; (g) products/materials to be recovered; (h) wasted products; (i) recycled or reusable products/components/materials; (j) products exchanged from the background; (k) products exchanged to the background; (l) non-physical flows; (m) impacts of consumption of natural resources or releases to the natural-environment; (n) impacts of consumption of natural resources or releases to the human-made environment; (o) impacts of consumption of non-physical flows to the human-made environment. * represents flows in the foreground; ** represents flows in the background. Note that co-products and waste streams are not represented for each process but are acknowledged. The figure represents only flows allocated to the product understudy. For the sake of readability, (l) are represented once for foreground and once for background at the interface of the foreground and background systems and the human-made environment, but these flows are considered to happen for each of the processes.

Hauschild et al., 2018). Within the human-made environment, products are exchanged in the form of *(c) processed raw materials*, *(d) components*, *(e) products*, *(f) used products*, *(g) products/materials to be recovered*, *(h) wasted products* or *(i) recycled or reusable products/components/materials* within or among locations and regions through the life cycle of the product. The background system provides various inputs to the product system, e.g. *(j) utilities and auxiliaries*, whereas it also receives outputs from the foreground system, e.g. *(k) co-products*. These *(a)* to *(k)* flows represent physical flows that can be measured quantitatively based on physical features such as mass, volume and units.

Several *(l) non-physical flows* are also associated with activities in the product system e.g. flows of knowledge, labour value, data flow about products and consumers, or money (UNEP/SETAC, 2011). They are present in both foreground and background systems. Non-physical flows are either quantified by convention, such as wages in local currency, or are considered quantifiable although they cannot be given an exact value, such as perceived value or happiness. So, these can be semi-quantitative data, such as indices, or qualitative data to which numeric values can potentially be assigned, meaning that they are indexable (UNEP, 2020).

The exchange of both physical and non-physical flows initiates cause-effect mechanisms that are also represented in Fig. 1. Furthermore, natural resource consumption and dispersion of releases have an impact on the natural-environment, represented by *(m)*, and the human-made environment, represented by *(n)*, both in foreground and background. The *(o) impacts from non-physical flows on the human-made environment* are represented at the interface of the human-made environment, also both in foreground and background systems. Which flows and associated impacts are analyzed, depends on the nature of the assessment: environmental, social or economic, as well as criticality and circularity. For the sake of understanding, all *(l)* socio and economic flows and *(o)* impacts related to the product system are represented in Fig. 2 in an integrated form. A literature-based laptop example that helps exemplifying and revising the potential processes and flows of a product system can be found in SM3.

4. Analysis of the product system in function of the assessment methodologies

After visualising the product system with the identification of potential flows and impacts, we can analyze which flows and the potential associated impacts are within the scope of the respective assessment methodologies, being LCA, S-LCA, LCC, materials' criticality and circularity. This analysis aims to facilitate the identification of potential overlaps and double counting when the assessment methodologies are brought together in a conceptual integrated framework. The impact pathways related to each method are not addressed in this section but in Section 5.1.

4.1. LCA

LCA assesses, in a systematic way, the environmental impacts of the product system by assigning LCI results to impact categories (ISO, 2006a). These impacts are associated with the total inputs and outputs, e.g. the use of *(a) natural resources* consumed and (b) *releases* emitted by the product system in the LCI. These consumed natural resources and releases are termed as elementary flows, which are exchanged between the natural environment and the technosphere. Important are the allocated portions of the flows *(a)**, (b)***, *(a)** and* (b)** to the product system under study as it may be part of a broader 'co-product' system with multiple products delivered. Moreover, the *(a)**, (b)***, *(a)** and (b)*** to be accounted for will depend on the system boundaries determined in the G&S step. As defined by ISO 14040 (ISO, 2006a), LCA addresses the potential impacts from cradle-to-grave, i.e. from raw material extraction to EoL. However, other boundaries are also possible: cradle-to-gate studies that are focused on the product supply chain in Fig. 2; gate-to-gate studies that are focused on the manufacturing-stage 2 of the product in Fig. 2; cradle-to-cradle studies, accounting for secondary use of *(i)* in Fig. 2 as inputs; or studies focused on specific parts of the life cycle, such as waste management or components of a product (see Table of definitions in SM2 and ISO (2006a, 2006b)). As for dealing with multi-functionality, allocation should be avoided (ISO, 2006a). Where necessary, allocation can be based on physical relationships such as material or energy content and, where this is not possible, on economic values (ISO, 2006a). In this case, particular attention should be paid to the life cycle of products and co-products and, consequently, to the quantity of reference units in production (ORIENTING, 2021c).

The potential environmental impacts *(m)* and *(n)* are the result of these *(a)* and *(b),* foreground and background flows affecting the natural environment and/or the human-made environment. Different impact assessment methods might be used for calculating the indicators *(m)* and *(n)*. The choice of indicators and associated methods varies according to the G&S of the study being in turn influenced by geophysical features of the cause-effect mechanisms.

4.2. S-LCA

S-LCA uses methodological features from LCA, such as the four methodological steps, combined with social sciences methods. It converts social and socio-economic aspects into a *(o)* social performance profile for a product (Goedkoop et al., 2020; UNEP, 2020). Two main approaches exist: the Reference Scale approach (RE S-LCA) and the Impact Pathway approach (IP S-LCA). The first aims to describe a product system and its social performance or social risk, while the second aims to characterize the social impacts as a consequence of the product system. In both S-LCA approaches, identifying and selecting stakeholder groups is key. The indicators describe impacts that directly affect the people, i.e. stakeholders of the product system. The stakeholder categories can be, but are not limited to: workers, local communities, value chain actors such as suppliers, consumers, children, and society (UNEP, 2020). Similar to LCA, the categories to be analyzed depending on the G&S of the study, and a materiality assessment can support the stakeholders' selection. In S-LCA, materiality assessment means the selection of topics that are relevant due to their impact on stakeholders and/or on the product system (Goedkoop et al., 2020; UNEP, 2020).

The *(o)* social impacts are assessed through impact categories, such as working conditions and socio-economic repercussions, and subcategories, such as probability of child labour or wealth distribution. The RE S-LCA makes use of subcategories, while the IP S-LCA defines the impact categories and their impact pathway to AoPs (UNEP, 2020). The definition of categories will be linked to the selected stakeholder groups according to significant themes or attributes relevant to those groups. The subcategories are assessed through inventory indicators (UNEP, 2020). The inventory indicators are social flows which reflect social conditions, vis-a-vis different stakeholder groups. Social flows are variables that provide quantifiable information about a particular life cycle stage or process (UNEP, 2020). Here, they are called non-physical flows (see Section 3). For example, they can represent a salary or a number of accidents at the workplace. In Fig. 2, the *(l) non-physical flows* are the representation of the social flows as defined by the UNEP (2020). They can be expressed (semi-) quantitatively as well as qualitatively.

The *(l)* flows to be accounted for in S-LCA should normally be considered from cradle-to-grave but limitations, e.g. data, of the study may require a different boundary setting. In this case, the system boundary can be determined based on the interaction among stakeholders. Such as in LCA, allocation should be avoided through the subdivision of processes and the collection of specific data for each of the co-products (UNEP, 2020). Alternatively, it can be based on causal links established by the activity variable associated with the social flow, i.e. to include the consequential relationships of activity and the related aspect to human well-being (UNEP, 2020).

4.3. LCC

Historically, LCC is not a homogeneous approach but was tailored to specific sectors regarding which costs were distinguished and how they were referred to (Neugebauer et al., 2016). Different LCC approaches exist. They are distinguished by Hunkeler et al. (2008) into conventional (cLCC), environmental (eLCC) and societal (sLCC). It is not the aim of this research to discuss all the fundamental differences between these approaches. Instead, we highlight two of those differences. First, the reference unit utilized in eLCC is the functional unit, similar to LCA, while in cLCC, it is an item or a unit of a product, and in sLCC it is the total system under study (Hunkeler et al., 2008). Conducted from a single stakeholder perspective, cLCC typically does neither consider the EoL phase, nor the use phase, whereas eLCC and sLCC evaluate the entire life cycle. Regarding the system boundaries, the approaches might consider only internal costs, i.e. directly reflected in product price, or partially or fully include costs in terms of monetized externalities as well (Arendt et al., 2020). cLCC only accounts for internal costs; eLCC accounts for internal costs and external costs expected to be internalized. sLCC accounts for all internal and external costs. Hence, while cLCC lacks information on the EoL and does not match the scope of LCSA, sLCC leads to double counting when brought together with LCA and S-LCA into an integrated sustainability framework. Meanwhile, eLCC results are complementary to LCA (Hunkeler et al., 2008).

Cost data represents the monetary flows within a system, e.g. the cost of the raw material for a laptop. They are indicated in Fig. 2 by the *(l)* non-physical flows. During the LCI phase, money flows are converted to have the same unit, i.e. in terms of currency and base year. As a result, no characterization of inventory data is needed. During the impact assessment, discounting and potentially equity weighting can be applied before aggregation. The aggregated cost data is a direct measure of financial impact (Swarr et al., 2011). In this sense, the monetary values, without discounting and equity weighting, can be considered as inventory accounting indicators similar to social flows which are inventory indicators in S-LCA. LCC results can albeit be interpreted in terms of cost per process in the product system or aggregated by stakeholder groups (Hunkeler et al., 2008). In terms of the allocation procedure, companies' data on costs and revenues, for example in design and waste management, are often aggregated at the organizational level (Hunkeler et al., 2008). Due to the economic context, it is usually preferable to rely on economic allocation based on market price or revenue rather than physical criteria (Hunkeler et al., 2008; ORIENTING, 2021c).

Yet, these LCC approaches alone do not consider the *(o)* impacts of the *(l)* money flows associated with the product system e.g. on the economy. As an alternative, Neugebauer et al. (2016) proposed the connection between usually detached economic aspects such as costs and impact categories and developed an impact pathway-based approach called Economic LCA (EcLCA). Since the development of EcLCA is not yet mature in terms of applicability, it is beyond the scope of this research to consider this further. The potential contributions of this study to the development of the impact pathway of economic indicators to the AoPs are discussed in Section 5.1.

4.4. Raw materials' criticality

A sufficient non-interrupted supply of resources is key to sustain the product system. The availability of resources in the natural environment can be assessed in LCA e.g. concerning the potential impact of depletion of biotic and abiotic resources (Guinée, 1995). In addition, there is a potential supply disruption within the technosphere. The complex socio-economic dynamics that create supply risks and vulnerability to the operation of product systems are the subject of Criticality Assessment (CA) (Schrijvers et al., 2020). It identifies raw materials that are essential in a supply chain or an economy and whose availability is at risk from an economic and geopolitical perspective (Dewulf et al., 2016; Di Noi et al., 2020). Yet, some methods look into both geological availability (natural environment) and availability within economic systems (technosphere) (Graedel et al., 2012). Existing approaches do not necessarily take a life cycle perspective.

CAs have historically been developed to analyze supply risks of essential raw materials in the economy, e.g. the USA (National Research Council, 2008) and the EU (European Commission, 2011b). When analyzed from a product system perspective, CA has been used to demonstrate the effects of economy-wide phenomena on smaller systems such as technologies or companies (Bach et al., 2016; Gemechu et al., 2016; Yavor et al., 2021). There can be risk factors of physical/technical/geological nature, such as concentration of resources or depletion time of resources stocks and reservoir; of economic/strategic/market nature, such as the concentration of mining and refining companies, the sudden growth of demand, import dependence or commodity prices; regulatory/social nature, such as conflict minerals; and of political stability/governance nature, such as political instability (Dewulf et al., 2015b; Vogtländer et al., 2019). The supply risk factors considered depend on the stakeholders' perception. The accounted factors represent the relationship between the demanding stakeholder (a company, a region – e.g. EU - or the global market) and the suppliers (a company's suppliers, a region's supplier – e.g. EU's suppliers - or global market suppliers). The scope is determined by the method utilized. For example, the European Commission's CA method (EC-CA) considers the demand and importance of raw materials to the EU's economy and the potential risks of supply disruption according to EU suppliers or global suppliers, depending on data availability.

One of the main differences in the scope of CA when compared to LCA is that resource related indicators in LCA estimate the impact of a product and its system on resource availability (depletion) in the environment. CA, on the other hand, estimates the impact of resource availability in the supply chain on a product system. In this sense, CA methods are so-called "outside-in" models (Sonderegger et al., 2020). Regarding the system boundaries, considering that criticality indicators cover potential "outside-in" impacts, the boundary is often limited to "cradle-to-gate". EoL can be considered when applying recovery strategies as a supply risk mitigation factor, e.g. regional recycling rates within the system in the EC-CA methodology (European Commission, 2017). For the creation of company-specific indicators, a "gate-to-gate" boundary can be applied. A "cradle-to-grave" boundary would also be feasible when the company's recycling and reuse rates are available (Santillán-Saldivar et al., 2021; Yavor et al., 2021).

CA methods can take into account data of different flows: the extraction of the natural resources, such as total *(a)* raw materials content in ore bodies; the primary production or refining of primary raw materials, i.e. in a *(c)* "ready-to-use" form; their market uses as part of *(d)* components and *(e)* products; and the EoL, e.g. secondary raw materials coming from EoL mitigating criticality. Within the traditional CA approaches, this information is considered metadata about the economy-wide dynamics in which the product system is inserted, i.e. it is information about the *(a**)* resource consumption, *(l**)* socio-economic flows and *(o**)* economic and geopolitical impacts of flows in the background system. When analyzing it from a microeconomic perspective, in addition to the macroeconomic dynamics, the raw material flows in the product supply chain *(c)* to *(e)* and in the EoL *(i)* to *(k)* are included.

4.5. Circularity

As introduced in Section 1, CE is an economic system that preserves the value of products, materials, and resources for an extended duration while minimising waste generation (EC, 2024). However, as mentioned above, it is not the CE that is evaluated as a whole, in terms of their sustainability impacts through a comprehensive LC^{3}SA, but individually or jointly applied circular strategies. Therefore, the term circularity is preferred here. Circularity can be interpreted as an approach to improve the prolonged and/or cyclical use of materials, which in this definition includes hibernation (Moraga et al., 2019; ORIENTING, 2021a). This can be achieved, for instance, by making use of secondary resources instead of primary ones. This in turn requires to deliver outputs at a product's EoL valuable to the economy through a proper R-strategy, avoiding the generation of waste (Moraga et al., 2019; Saidani et al., 2019). Besides, design for redundancy, multi-functionality and use intensification of products such as product service-systems can also be considered to promote CE. Circularity assessments, therefore, look at the product system to identify its circularity based on the adopted strategies.

Different CE indicators have been proposed to measure the circularity of various strategies' implementation. Moraga et al. (2019) classified three types of CE indicators in terms of scope of analysis (renamed here for the sake of understanding): Type 'one' and type 'two' indicators are based on mass flows of the product system. They can be called CE inventory-based indicators and are distinguished into those that are based on the technological life cycle with an LC perspective, e.g. in-use occupation of material in a specific product cycle (kg.year) (Moraga et al., 2021) and those without life cycle perspective, e.g. Recycling Rates in % (Graedel et al., 2011),. The type 'three' indicators account for impacts of more circular production systems at the level of associated impacts e.g. reduction of environmental impacts by employing LCA methodological aspects. These can be called CE impact-based indicators based on cause-effect modelling (Moraga et al., 2019). For example, the Sustainable Circular Index (SCI) utilizes social, economic and environmental indicators together with material flow indicators to measure circularity (Azevedo et al., 2017). CE assessments can follow similar system boundaries as in LCA: cradle-to-grave, cradle-to-gate, gate-to-gate or cradle-to-cradle depending on the G&S of the assessment. They can also measure multiple levels of strategy application: micro, i.e. products, components, materials; *meso*, i.e. businesses, industrial symbiosis; or macro, i.e. cities, regions, nations (Saidani et al., 2019).

Inventory-based CE indicators can account for the physical flows *(a)* and *(c* to *k)* depending on the scope of the study. Impact-based CE indicators do not necessarily make use of these material flows but rather use indicators that rely on the results from LCA, LCC and/or S-LCA by computing *(m)*, *(n)* and/or *(o)* to quantify the sustainability gains associated to certain circular economy strategies, e.g. recycling versus re-use. Hence, the CE-impact-based indicators do have a substantial risk of overlapping with the other assessment methods when brought together into an LC^{3}SA.

4.6. Analysis of overlaps and the risks of double counting

Table 1 summarizes the flows and implications that are typically accounted for in each of the assessment methodologies. For a proper classification, further subdivisions need to be made. Concerning LCC, it is important to differentiate between eLCC, cLCC and sLCC. The S-LCA is divided into RS S-LCA and IP S-LCA. For criticality, a differentiation is made between micro and macro-level indicators. Lastly, inventory-based CE indicators are to be distinguished from impact-based ones. The flow *(o)* represents impacts of socio-economic nature coming from relationships of *(l)* non-physical flows. Besides, in the product system presented in Fig. 2, for *(l)* non-physical flows, a semi-quantitative

Table 1
Main input flows directly accounted for in the assessment by LCA, S-LCA, LCC (cLCC, eLCC, sLCC), criticality (macro and micro-level indicators) and circularity (inventory-based and impact-based indicators) in a life-cycle-based product-system, used for impact characterization. Legend: x = flow data used; x = data interpreted in eLCC, but analyzed according to LCA.

	Framework/flows	LCA	S-LCA		LCC			Criticality		Circularity	
			Reference scale	*Impact pathway*	*cLCC*	*eLCC*	*sLCC*	*Macro*	*Micro*	*Inventory-based*	*Impact-based*
Physical flows (elementary and product flows)											
a	*Natural resources*	x							x	x	
b	Releases to air, soil and water	x									
c	Processed raw materials								x	x	
d	Components								x	x	
e	Processed (and sold) products	x	x	x	x	x	x	x	x	x	x
f	Used products									x	
g	Recovered products									x	
h	Wasted products									x	
i	Recycled or reusable products/components/materials								x	x	
j	Products obtained from the background								x	x	
k	Co-products transferred to the background								x	x	
Non-physical flows											
l	Non-physical flows (numerical or non-numerical flows)		x	x	x	x	x	x**	x**		
Impacts											
m	Impacts due to the consumption of natural resources or releases to the natural-environment					x	x				
n	Impacts of consumption of natural resources or releases to the human-made environment		x	x		x	x				x
o	Impacts of consumption of non-physical flows to the human-made environment		x	x			x	x**	x**		x

*Foreground only.
**Background only.

assessment is at least required to allow the measurement of all product system impacts and to allow comparisons among results from the different methodologies. They can be associated with *(c)* to *(k)* physical flows.

Six main conclusions can be drawn from the analysis in Table 1. First, product flow data (*(d)* to *(k)*, noting that micro-level criticality indicators do not cover all product flows, see below) is needed for inventory-based circularity indicators and micro-level criticality indicators as well as *(a)* resources flow data. Product flow data is inventoried in LCSA to quantify materials and product flows in the product system but are not directly part of the analysis in the LCA, LCC or S-LCA. While for some CE inventory-based indicators the product flow data in the use-phase is relevant, e.g. for calculation of dissipation of materials during use or lifetime of the product, this information is disregarded for the revised micro-level criticality indicators (Hackenhaar et al., 2022a).

Second, CE inventory-based indicators, as well as macro and micro-level criticality indicators, share the consideration of *(a)* resource flows used within LCA. There is no issue of double counting as long as the data is used to analyze the impacts of different natures. Using the same flows for indicators of different nature represent an opportunity because of effort sharing. Hence, the effort needed to add circularity and criticality indicators to LCSA can be facilitated.

Third, data on *(b)* releases to air, soil and water, e.g. CO_2 emissions to air, are only used directly in LCA. They are also indirectly considered in the internalization of externalities in LCC. When internalized, costs are considered in eLCC and sLCC. The releases that affect human well-being are included in S-LCA. Those might be considered by the *(m)* and *(n)* impact flows.

Fourth, non-physical flows are the core data for S-LCA, LCC and criticality methodologies, although usually not represented in the product system schemes in LCSA. The non-representation of such flows potentially leads practitioners to a fragmented data collection of physical and non-physical flows. In this sense, the particular connections between data and information about the product system, are lost and the interpretation of overlaps and trade-offs is hampered.

Fifth, not only elementary, product and non-physical flows are used as sources of information for the assessment methodologies, but also the impact indicators here represented by impacts. This creates risks of double counting. While the characterization modelling of the impacts is discussed in Section 5, here it is important to acknowledge that *(m)* and *(n)* represent impacts computed in LCA that can appear in LCC as externalities, e.g. costs of air pollution through taxes (Hunkeler et al., 2008) and in S-LCA as part of impacts felt by stakeholders, e.g. pollutants in the workplace (UNEP, 2020). In the case of eLCC, LCA results are not used in the assessment but interpreted alongside cost indicators e.g. in an eco-efficiency analysis (Ciroth, 2009). In contrast, the use of sLCC is debatable in LCSA due to the risk of double counting when internalization occurs (Bachmann, 2013; Neugebauer et al., 2016). Thus, the results of LCA impact indicators can be considered as input data for LCC and S-LCA with a risk of double counting. Similarly, impact-based circularity indicators make use of other LCSA results regarding their impacts on the human-made environment. If integrated into an LCSA framework, CE impact-based indicators lead to double counting. For example, the SCI (see Section 4.5) will overlap with the indicators from LCA (water depletion), S-LCA (accidents at work) and LCC (value generated).

Due to their "outside-in" nature, criticality indicators also use the information on the socio-economic impacts of the product system on the human-made environment. However, this information comes from the background (market systems), and temporal variability plays an important role in the discussion of overlaps. While the LCA, LCC and S-LCA analyze the snapshot of the product system with its short- to long-term impacts on the world, criticality indicators use the information on how environmental and socio-economic states might impact the product system before the product can be produced. In this sense, criticality indicators do not generate double counting with other LCSA indicators. However, if aggregated, inventory-based circularity and micro-level criticality indicators can create double counting of the benefits of R-strategies when producing secondary raw materials. Therefore, they should be carefully interpreted as complementary to LCA, LCC and S-LCA results. The main challenge is to select or design the criticality and circularity methods and indicators such that double counting with LCA, S-LCA or LCC is avoided.

Sixth, criticality methods are mainly differentiated by the scale of the study that is often limited to the availability of data. That is, ideally micro CA methods would produce more assertive results about criticality for a product or organization using case-specific data. For that, a thorough analysis of the supply chain would be needed. However, data accessibility, e.g. due to confidentiality, might be an issue. Therefore, macro CA methods are often the chosen alternative. When neither of the approaches creates double counting with other indicators in LCSA, the choice of the most adequate method to be used can be defined according to the goal and scope of the LC^{3}SA assessment.

5. From inventoried flows to areas of protection

The physical and non-physical flows are the starting point of various cause-effect chains, potentially leading to environmental, social or economic impacts. These flows can also be used to assess criticality and circularity. As discussed in Section 4.6, the "causes", i.e. the flows analyzed, might generate overlaps that do not necessarily lead to double counting if the "effect" expressed by the impact indicators is different. To provide reliable results in terms of sustainability impacts, one needs to have a good understanding of the impact pathways to link the physical and non-physical flows in the inventory to impacts on AoPs (Valdivia et al., 2021). As introduced in Section 1, AoPs, or areas of concern, are the safeguard objects of sustainability (Schaubroeck and Rugani, 2017). They represent what we want to protect, and are therefore the final point in the cause-effect chain. Therefore, a clear understanding of the AoPs is fundamental to starting an LCSA (Valdivia et al., 2021). It is only by knowing the question one wants to answer that one can look for an answer. Thus, the definition of the AoPs guides the choice of indicators in an LCSA.

Criticality and circularity are of a different nature than LCA/LCC/S-LCA: they are rather strategic indicators (Mancini et al., 2016). Besides, they are not conceived in a complete cause-effect logic towards an AoP level or sustainability domain. Therefore, a comprehensive framework also integrates the analysis of the impact pathways for those methodologies and the potential overlap at the AoPs. For that, we analyze the existing literature on LCSA impact pathways from inventoried flows to AoP and propose how to further connect the criticality and circularity indicators to a revised LC^{3}SA AoP.

5.1. Cause-effect chain specifications and AoPs in LCSA

A non-exhaustive review based on the LCSA-related frameworks shows that there is still no consensus on the AoPs (Dewulf et al., 2015a, 2015c; EC-JRC, 2011; Gaasbeek and Meijer, 2013; Goedkoop et al.,

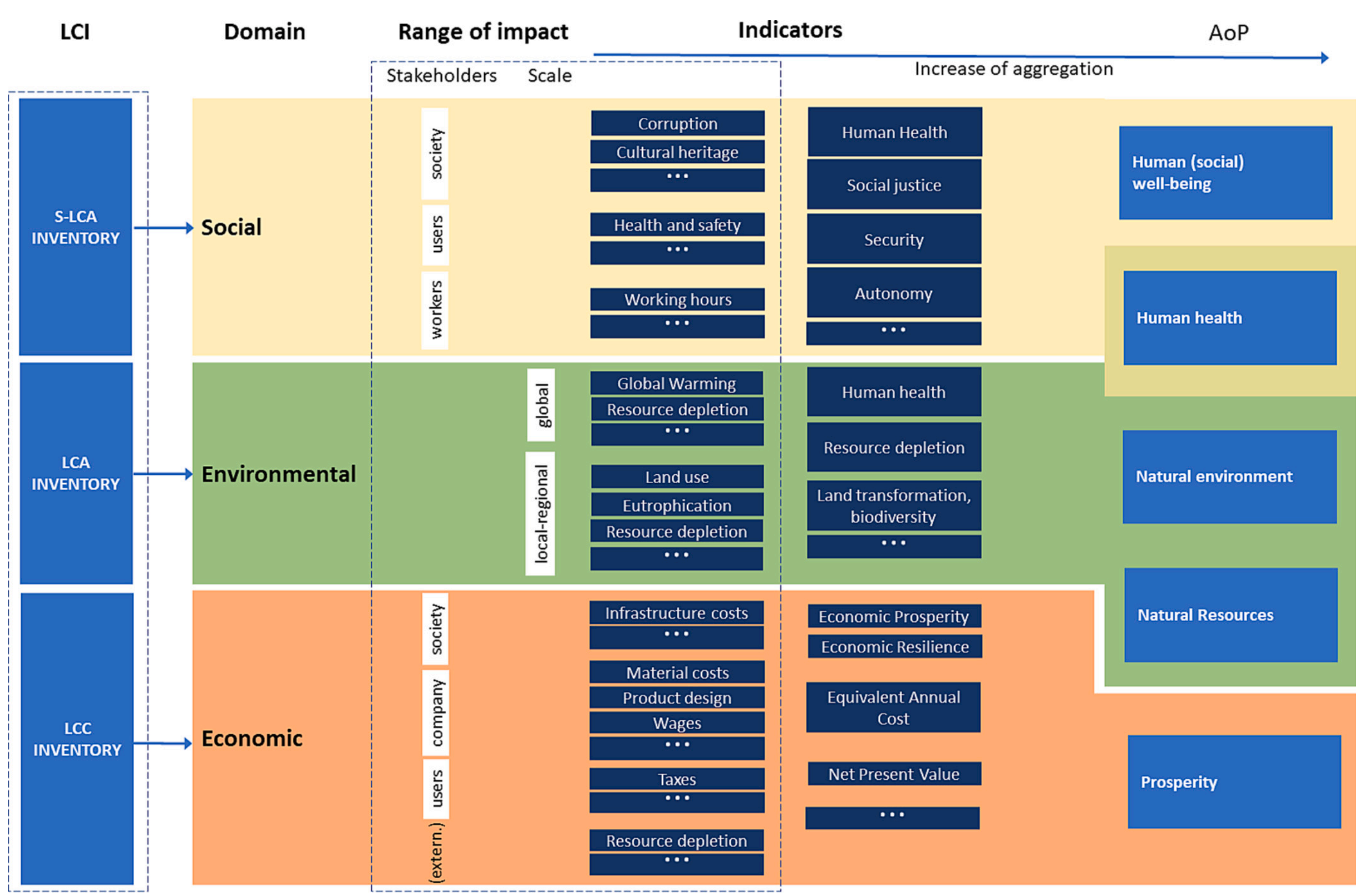

Fig. 3. Non-exhaustive exemplification of the impact pathways of the different LCSA domains from the inventory flows to the Areas of Protection based on the literature (adapted from EC-JRC (2011), Gaasbeek and Meijer (2013), Goedkoop et al. (2020), Guinée et al. (2002), Huijbregts et al. (2016), Neugebauer et al. (2016) and UNEP (2020)).

2020; Guinée et al., 2002; Huijbregts et al., 2016; Neugebauer et al., 2016; Schaubroeck and Rugani, 2017; Taelman et al., 2020a, 2020b; UNEP, 2020) (see SM4 for details). Fig. 3 summarizes the results from the review by exemplifying and distinguishing the cause-effect chain pathway according to different approaches in the literature. The indicators shown in the figure are illustrative examples for each of the midpoint and endpoint indicator categories, stakeholders and scale of impacts.

Within LCA, the impact pathways connect inventory flows to the so-called midpoint and endpoint environmental impact categories in the middle and final stages of the cause-effect chain, respectively. These environmental impacts are multi-size and can reach a local, regional or global environment. At the end of the cause-effect chains, the impacts can be positioned at the level of AoP. The impact pathways are established and can be quantified through impact assessment methods such as CML (Guinée et al., 2002) and ReCiPe (Huijbregts et al., 2016), among others (EC-JRC, 2011). Well-established and accepted AoPs in LCA are Human health, Natural environment and Natural resources (EC-JRC, 2011, 2012; UNEP/SETAC, 2019). Due to practical reasons (i.e. the availability of relevant models), the use of midpoint indicators is usually preferred (Hauschild et al., 2013). However, endpoint indicators ease the interpretation of results as they reduce the number of impact indicators at the intermediate cause-effect chain level to the number of AoPs distinguished (Schaubroeck and Rugani, 2017).

From S-LCA frameworks, the endmost AoP concerning social issues is Human (or social) well-being. According to the UNEP (2020), this social AoP could be split into two: Health and Safety, and Happiness. On the other hand, Goedkoop et al. (2020) determined the concept of Human Well-being as AoP per stakeholder group: for workers, it is job satisfaction and engagement; for local communities, it is healthy communities; for product users, it is the provision of well-being per se; and for small-scale entrepreneurs, it is the livelihood.

The AoPs considered in LCC are less clear, although “Prosperity” appears in multiple LCSA studies that consider AoPs (Gaasbeek and Meijer, 2013; Schaubroeck and Rugani, 2017; Taelman et al., 2020a, 2020b). LCC results can also be interpreted according to inventory indicators which are based on costs (ORIENTING, 2021c). However, cause-effect chain connecting such a cost indicator to the AoP “Prosperity” is in general not quantified. Moreover, the cost elements to consider also differ according to the stakeholders such as industry, users, society and policymakers. Neugebauer et al. (2016) note that the causal chains are not yet established for LCC impact categories. They propose that micro-level economic indicators could be linked to macro-level economic indicators in an LCSA. For example, the endpoint indicators “Economic resilience” and “Economic Prosperity” could directly represent the AoPs “Economic stability” and “Wealth generation”, respectively (Neugebauer et al., 2016). The framework is not yet widely applied. Therefore, it does neither appear in Fig. 3, nor is it considered further in the proposed LC^{3}SA framework.

As mentioned before, impacts can be multi-dimensional, i.e. an AoP represents the endpoint of one or more environmental, social and economic midpoint impact categories (Gaasbeek and Meijer, 2013; Taelman et al., 2016, 2020b). For example, the LCA framework from Guinée et al. (2002) and the EC-JRC (2011) consider that environmental impacts can cause damage to Human Health. However, Human Health is also a concern of S-LCA and can be considered a social AoP (Bachmann, 2013; Goedkoop et al., 2020; UNEP, 2020).

The multi-dimensional nature of several impacts may cause a potential overlap between these AoPs that are considered to be part of LCA, LCC or S-LCA alone. This potential overlap results from similarities and differences in goal and scope definition, inventory and impact modelling from each of these methodologies. Furthermore, several arguments could be drawn from the relationship between LCSA methodologies. For example, one could argue that from an anthropocentric point-of-view “Human well-being” is the primary goal of sustainability, and other AoPs are rather connected to this societal concern in a supportive role (Schaubroeck and Rugani, 2017). As highlighted by Dewulf et al. (2015c), although there might be an overlap between some AoPs, the impact pathway starts from different causes.

To avoid any potential overlap of sustainability dimensions, the PROSUITE framework (Blok et al., 2013; Gaasbeek and Meijer, 2013) defined five AoPs as impact categories that are not exclusively defined as environmental, social or environmental, but cover all domains of sustainability in an integrated manner. These five AoP (called in the project “major impact categories”) are “Human Health”, “Social well-being”, “Prosperity”, “Natural Environment” and “Exhaustible Resources”. Gaasbeek and Meijer (2013) propose to normalize, weight and aggregate the indicator results into five AoP scores. This way of assigning and grouping midpoint indicators to AoPs (with or without aggregation) is considered here as a solution for handling the multi-dimensional nature of impacts within LCSA. Given their scientific relevance and robustness for the purpose of our work, we adopted the five AoPs proposed by the PROSUITE framework, while adapting the AoP “Exhaustible/Natural Resources”. This is further discussed in the following section.

5.2. Revising the AoP “Exhaustible/Natural Resources”

There is not yet a consensus on how to address resource use in LCA, despite the variety of robust contributions in the field in the last years (Bachmann, 2013; Pradel et al., 2021; Sonderegger et al., 2020). Depletion is a representation of the effect of resource consumption that can affect both the natural environment and the human-made environment. In the natural environment, resources are connected to the supply of ecosystem services to both human kind and other species. But from the anthropocentric perspective, the current depletion of the natural environment may prevent future generations to fulfil their needs. The assessment of the geophysical accessibility of materials using depletion-based methods remains the predominant approach employed in product-level LCA studies. For example, PEF recommends the characterization of mineral resource depletion (in kilograms of antimony equivalent), under the AoP “Natural Resources” (Zampori and Pant, 2019). The extent of the impact assessment concerning the AoP “Natural Resources” is debatable given that resource use encompasses economic, environmental, social, and technical aspects (Dewulf et al., 2015b).

According to the Global Guidance on Environmental Life Cycle Impact Assessment Indicators (UNEP/SETAC, 2019), the use of approaches that consider the impacts of resource use in terms of the accessibility for future generations is recommended. However, events such as pandemics and regional wars show that the socio-economic availability of resources to the product system is also relevant, as well as the increased rate of recovery and valorisation of secondary materials. Therefore, to tackle resource use from a multi-dimensional point, we recommend expanding the AoP “Natural resources” to an AoP “Resources”, which does not only include the “natural resources”, but also the secondary raw materials derived from CE strategies such as dismantling or decomposition of products in their EoL (Dewulf et al., 2015b). Defined this way, the efficiency of the use of raw materials can be better captured considering the economy-wide dimension and changes resulting from a CE system.

The AoP “Resources” could integrate not only indicators from LCA but also the impact pathways of circularity indicators. However, to limit double counting, type three CE indicators (see Section 4.5) should be avoided. Circularity indicators to be used in an LC^{3}SA are inventory-based ones, focused on the analysis of flows. These are primarily accounting indicators. They cannot directly be related to any pillar of sustainability, yet relationships to environmental (use of resources) and economic (CE at a macro level) dimensions exist. In this sense, the given proposal of the multi-dimensional AoP “Resources” would suit the connections of the effect of circularity in the protection of resources.

Furthermore, criticality can be linked to different domains of sustainability too. CA might consider e.g. depletion of mineral and metal stocks (environmental dimension), cost increases in the product system due to shortage of supply (economic dimension) and/or socio-political issues such as corruption and political stability (social dimension) (Hackenhaar et al., 2022a; Sonnemann et al., 2015). On one hand, at an endpoint level, it can be argued that CA methods envisage to analyze first and foremost the potential of increased production costs for companies and increased product prices for consumers (André and Ljunggren, 2021; Mancini et al., 2016; Sonderegger et al., 2020). In this sense, having criticality indicators analyzed within LCSA could be interpreted as an economy-wide indicator affecting the AoP "Prosperity". It does not necessarily mean that criticality indicators internalize supply risks that are reflected in LCC. However, the monetization of supply risk has already been proposed in the literature (Santillán-Saldivar et al., 2022). On the other hand, CA methods and indicators that focused mainly on raw materials availability or accessibility (e.g. indicators proposed by Graedel et al., 2012) could be connected to AoP "Resources".

5.3. LC^{3}SA indicators and AoP

Based on the discussions in Section 5.1 and 5.2, we propose five AoPs to be considered in an LC^{3}SA: AoP "Natural environment", AoP "Human health", AoP "Human well-being", AoP "Prosperity", and AoP "Resources". Fig. 4 shows the generic scheme of indicators and impact pathways of an LC^{3}SA where these AoPs are tackled (a comprehensive scheme of an LC^{3}SA framework based on literature studies of laptops can be found on SM3). Note that the term "indicators" is used here to address both impact categories and subcategories, at midpoint and endpoint, accounting and impact indicators (see SM5 for the differentiation between accounting and impact indicators). Some of the most commonly used indicators in the literature were used as examples to touch upon all AoPs but they are not necessarily the most relevant ones. They are used in the figure to illustrate the possible choice of indicators based on the laptop example in SM3. Thus, special attention should be given to aligning the selection of indicators with the G&S.

6. Introducing an integrated LC^{3}SA framework

Inspired by the ISO 14040/44 (ISO, 2006a, 2006b) diagram of the LCA phases and the learnings from the literature presented in this manuscript, we propose the LC^{3}SA framework, shown in Fig. 5. The framework aims to provide step-by-step guidance on how to conduct an integrated assessment that allows to include all environmental, social, economic and material circularity and criticality indicators. We detail the 4 phases of the LC^{3}SA and the relevant steps in each phase, highlighting how circularity and criticality assessments and indicators are considered. The approach allows for an integrated modelling at an early stage of the assessment, improving the consistency of the assessment and results. It is based on the state-of-the-art of methods and is ready to be used as it relies on currently available tools.

Such as in LCA, the first step is to define the G&S of the study. The importance of materials' criticality and circularity in the product system and the interest in conducting such assessments should be clear in the definition of G&S. The second step consists of the mapping of the product system (e.g. using the detailed product system diagram of Fig. 2), the definition of system functions and FU. While mapping the product system, relevant information about stakeholders, geographies and suppliers in the value chain should also be gathered and connected

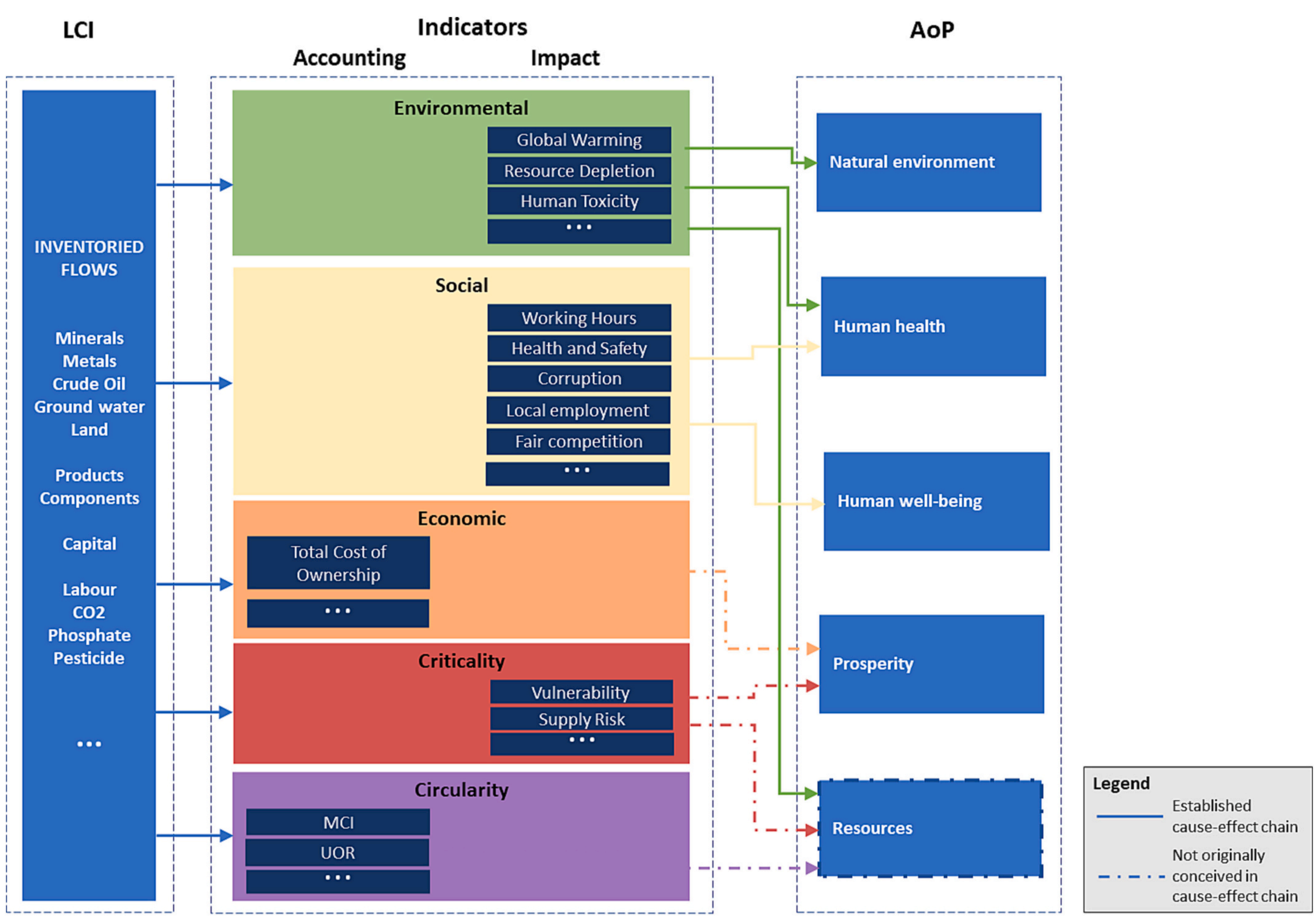

Fig. 4. Generic scheme of indicators and impact pathways of LC^{3}SA, integrating the environmental, social, economic, criticality and circularity indicators connected to the AoPs of "Natural environment", "Human health", "Human well-being", "Prosperity" and "Resources".

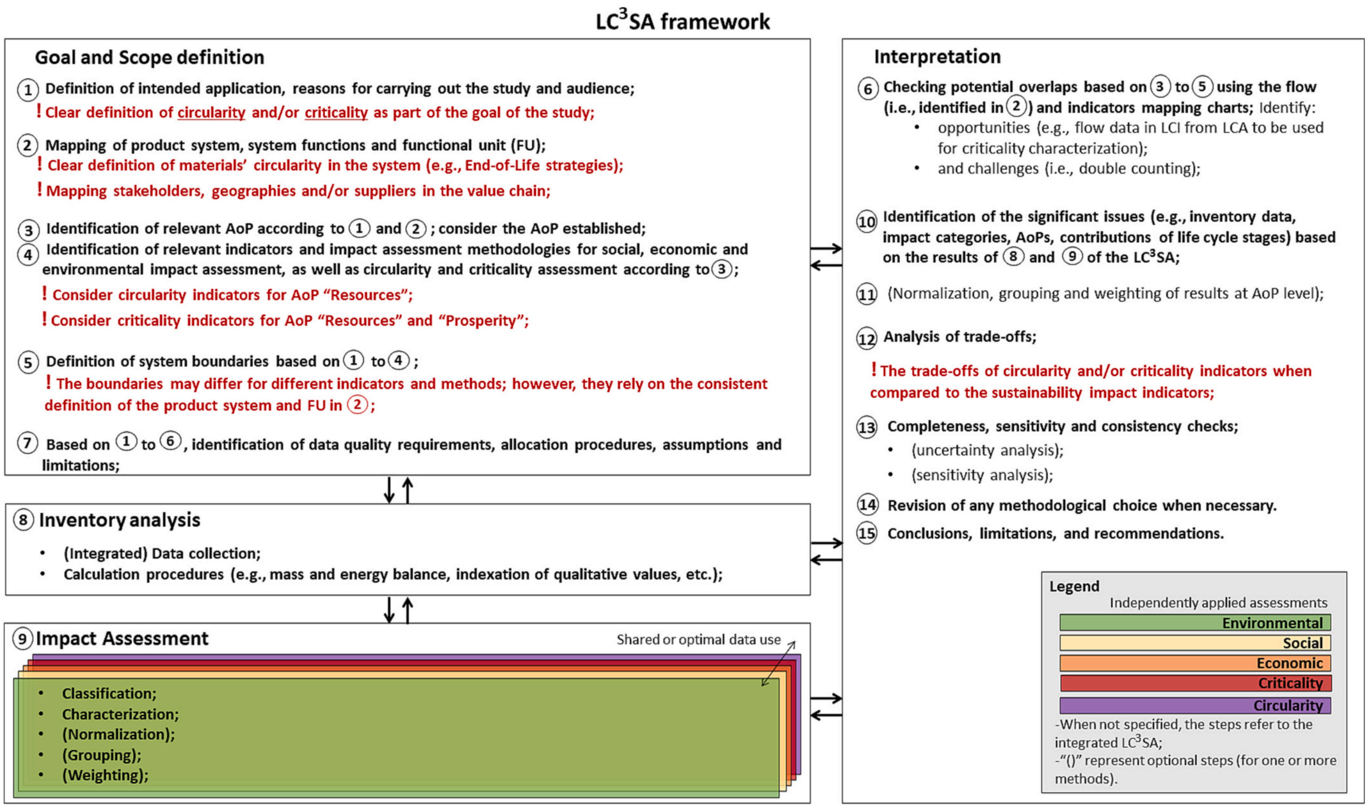

Fig. 5. LC^{3}SA framework integrating the environmental, social and economic impact indicators from LCSA and material's circularity and criticality indicators for resource efficiency.

to the life cycle stages. Similarly, the circular strategies applied to the product system and the materials' flow should be clear at this step.

Following the principles of UNEP/SETAC (2011), the framework considers the definition of AoPs as the starting point for the definition of the type of indicators and methodology of the assessments. Thus, in step three, relevant AoPs shall be identified.

In step four, the accounting and/or impact indicators to be included in the LC^{3}SA shall be identified in a first joint analysis of impact pathways to the relevant AoPs. A first recommendation to identify impact pathways to AoPs is that (see Fig. 4, Section 5.3): if AoP "Natural environment" is considered relevant, then environmental indicators which cause-effect chain leads to damage at this AoP should be considered; similarly, if AoP "Human health" is relevant, environmental and/or social indicators might be considered; if AoP "Human well-being" is relevant, social indicators should be considered; if AoP "Prosperity" is relevant, economic and/or criticality indicators might be considered; if AoP "Resources" is relevant, environmental, criticality and/or circularity indicators might be considered. Indicators should then be assigned to the AoP of relevance for further interpretation of results.

As discussed in Section 3, different methodologies and approaches may differ in system boundaries. Yet, they should rely on the consistent definition of the LC^{3}SA product system. Once the indicators and their assessment methods are identified, the system boundaries of the product system can be defined in step five. A first consistency check of the LC^{3}SA is proposed in step six. Using mapping charts such as those presented in Table 1 (see also SM3), one can analyze the potential overlaps of flows, chosen indicators and their methods. If pertinent, based on the identified opportunities and challenges, a revision of methodological choices to strength the consistency of the LC^{3}SA can be made. After that, in step seven, one can identify data requirements (i.e. different data types with different – quantitative, semi-quantitative and qualitative – data quality and metadata), allocation procedures, assumptions and limitations of the LC^{3}SA study.

At step eight, data should be collected in an integrated form. That is, all the flow data that can be shared among assessment methodologies is collected at once, linking the physical and non-physical flow data to each life cycle stage. Any necessary calculation procedure should be done and reported at this step.

At step nine, the impact assessment is performed. Current LCSA tools do not allow a joint application of the assessment methodologies. Thus, step nine is differentiated for the independent calculation and classification of indicators. Yet, it facilitates an optimal data use among the assessments. In line with ISO 14040/44 (ISO, 2006a, 2006b), normalization, grouping and weighting are optional. However, if aggregation of results is relevant for the study, it is discouraged at the impact assessment level, but recommended in step eleven, where the results can be aggregated at the AoP level.

Based on the results of previous steps, one can interpret the results. In step ten, the significant inventory data, impact categories, AoPs and life cycle stages should be identified. In step eleven, the normalization, grouping and weighting of results at the level of AoPs are optional. If aggregation is relevant, attention should be paid to the indicators that might be connected to more than one AoP according to the G&S definition. They should be assigned to the most pertinent AoP to avoid double counting.

Regardless of the use of single scores at the AoP level, in step twelve the trade-offs among indicators should be analyzed. In step thirteen, completeness, sensitivity and consistency checks are recommended. If needed, revision of the LC^{3}SA steps should be made (step fourteen) before conclusions can be drawn and recommendations given (step fifteen).

7. Discussion

LCSA is complex and still needs to be further adapted to meet the demands of LCSA practitioners, industry players, suppliers, consumers and policy makers (Backes and Traverso, 2021). In Section 3.2, we discussed that the inconsistent definition of system boundaries is a challenge in LCSA (Fauzi et al., 2019) and therefore also in an LC^{3}SA. However, consistency does not depend on the definition of a uniform system boundary (Alejandrino et al., 2021), but on the definition of a consistent product system where the information used for each assessment is interrelated. In the analysis of potential overlaps between assessment methods in Section 4.6, we discussed that both non-physical flows and impacts are also used as information for the characterization of indicators in LCC, S-LCA, Criticality and Circularity. For example, impact-based circularity indicators use other LCSA results, which can lead to double counting. In the case of criticality, however, time variability plays an important role in the discussion of overlaps, as the assessment can generate feedback information to each other. In addition, inventory data, which are not used to characterize impacts in LCSA, are key to characterizing criticality and circularity. They also use resource flows as in LCA, which doesn't necessarily lead to double counting if the indicators analyze impacts of different types. The release flows considered in LCA can also be considered indirectly in LCC and SLCA, depending on the approach and indicators chosen. The summarized findings as presented in Sections 3, 4 and 5 emphasize both the need for transparent, credible and integrated data collection in an LC^{3}SA and the need to identify complementary indicators at an early stage of the assessment. These serve as a basis for the developed LC^{3}SA framework.

The inclusion of circularity and criticality indicators in LCSA risks creating additional controversy or double counting. This can be avoided if carefully selected or designed. LC^{3}SA results are intended to clarify the effectiveness of actions towards a more sustainable and resource-efficient product system. Keeping this in mind, the motivation for carrying out an LC^{3}SA rather than an LCSA needs to be clear when defining the goal (step 1 in Fig. 5). For the proposed integrated methodology, the benefits and challenges inherent to LCSA and adding circularity and criticality assessments were considered.

In an attempt to address the shortcomings (Alejandrino et al., 2021; Valdivia et al., 2021) and desired features (Wiedmann et al., 2009) of LCSA, presented in Section 1, we have proposed an LC^{3}SA framework. The framework addresses potential overlaps, double counting, the interpretation of multi-dimensional results and data requirements in a transparent definition of G&S and of a concise product system. Steps 1 to 5, for example, address the definition of the goal and scope taking into account the multi-dimensional nature of the assessment and ensuring that the purpose of an integrated analysis can be met from the outset, thereby increasing robustness. With steps 3 and 4, we emphasize the need to identify, understand and follow the cause-effect mechanisms before starting the impact assessment. This improves the chances of making conscious decisions towards a more sustainable system.

As part of the interpretation phase, step 6 draws attention to the harmonization of the LC^{3}SA model. This can improve both ease of application by optimizing data usage and credibility by reducing double counting. Similarly, steps 8 and 9 suggest optimal use of human resources in conducting the assessment and optimal use of data. The following steps are not unique to this framework, except for step 12, dealing with trade-offs within an integrated assessment. Their consideration improves the robustness of decision making.

Each step aims to improve the interconnectedness and consistency of the assessment by reaching common modelling assumptions. Similarly, a stepwise framework supports a more conscious study design and increases transparency as each step is designed to complement the others. However, it does not eliminate complexity, data gaps or issues of interpretation of results. These are challenges of operationalization that are briefly discussed below.

A product system consists of multiple processes. Each of these processes differs in terms of geographical, temporal or technological contexts (Hackenhaar et al., 2022b). Not all of the underlying LC^{3}SA methodologies need to specify these complex contexts in the same way or detail. As a result, information will be collected for the description of the same process for different LC^{3}SA methodologies in different ways. Gathering the inventory independently for the different dimensions can therefore lead to inconsistencies in results. The integrated LC^{3}SA framework proposes data and information be jointly inventoried (step 8 of the LC^{3}SA framework) or interpreted (step 6 and 10 to 15 of the LC^{3}SA framework).

However, this represents the first challenge of operationalization of the integrated framework since current LCSA tools, such as software, spreadsheets and databases, do not fully integrate environmental, economic and social assessments or allow interoperability between tools (ORIENTING, 2022). While tool developers deal with the challenge, scenario analysis, and prospective or dynamic models are potential solutions. For instance, one of the aimed targets of the European Innovation Partnership in Raw Materials is to develop a fully dynamic LCSA model that can evaluate mid- to long-term scenarios for the sustainable supply of raw materials (ORIENTING, 2021a).

Data unavailability and quality are another big challenge in LCSA, criticality and circularity studies (ORIENTING, 2021f; Santos et al., 2022). Unlike LCA results, which are assumed to be more stable, LCC and S-LCA, as well as circularity and criticality indicators, are more influenced by short- and medium-term changes, e.g. market prices and socio-geopolitical context (ORIENTING, 2021a). For example, cost data can be more volatile than physical flows, gathered in different currencies and reflecting different periods, often requiring recalculations and updates for the sake of consistency across the full inventory (Swarr et al., 2011). In S-LCA and CA, it is acceptable to use generic, i.e. non-site or non-company specific, data or economy-wide data from databases (UNEP/SETAC, 2011). This can be related to the maturity or complexity of the approaches as well as to the nature of both assessments i.e. LCC, S-LCA and CA rely on the collection of data and quantification of mainly non-physical flows. As mentioned in Section 4.6, although those flows cannot be given an exact value, the estimation of the potential impact is based on the assignment of numerical values to qualitative information (semi-quantification) (UNEP, 2020). Thus, it is also important to understand that the concept of "reliability" of data might have different "weights" for each LC^{3}SA methodology. The differences in data quality requirements may be dealt with through transparency in reporting, as well as using consistency checks.

The interpretation of integrated results is another challenge of operationalization. In this research, we propose that indicators are first assigned to the relevant AoP to make more assertive decisions. However, the aggregation of results into AoP scores is optional. It can be desired in sustainability assessments to facilitate communication of results for a broader audience, without expertise in the field (Gaasbeek and Meijer, 2013; ORIENTING, 2021f). However, the ISO 14040/44 standards (ISO, 2006a, 2006b) consider that both normalization and weighting necessary for aggregation include value choices that bring subjectivity to the results. It is out of the scope of this research to discuss the suitability of specific aggregation methodologies. However, we recommend the aggregation methodology selected be compatible with both qualitative and (semi-)quantitative indicators, considering the different nature of the indicators discussed in this research. Moreover, if results are aggregated, they should also allow visualization of trade-offs and both burdens and benefits.

Another important note about interpretation is that trade-offs are a consequence of comprehensive sustainability assessments and their inherent interconnectedness (Valdivia et al., 2021). Knowing their existence enables more informed decisions. For example, the product system of more environmental-friendly technologies can lead to high criticality scores. At the same time, circularity measures can solve high resource scarcity and criticality issues. Similarly, socially beneficial actions reduce social impacts but can also help reduce criticality. This means that LC^{3}SA topics are inevitably cross-cutting or have interrelated issues. That is precisely the reason why they may be assessed and evaluated in an integrated way. Attention needs to be paid that an integrated approach does not lead to double counting and potential compensation. Presumably, it could also lead to more assured actions since decision-makers would be aware of potential trade-offs between different ends (expressed as AoPs in LC^{3}SA terms): if we know the trade-offs, we can better discuss and assess what we want to maintain, and at what costs/burdens.

The proposed LC^{3}SA framework is an attempt to integrate LCSA, criticality and circularity in a transparent and comprehensive form. While the challenges of operationalization were identified, the framework applicability still needs to be tested quantitatively (and improved). Future research should focus on testing the framework, as well as further developing a consistent way of interpreting LC^{3}SA results in an aggregated form.

8. Conclusions

The development of a 'greener' and more sustainable economy in Europe seems closer to becoming a reality as the EU consolidates science-based policies. Although the literature proposes robust life cycle based methodologies for sustainability assessment, the fragmented analysis of environmental, social, economic and resource availability issues remains a gap. In this study, we explored the definition of each methodological step to propose a comprehensive LC^{3}SA framework in line with the principles for conducting an LCSA according to the UNEP Life Cycle Initiative (UNEP/SETAC, 2019), also based on the developments of the ORIENTING project (ORIENTING, 2022). We analyzed the literature on LCA, LCC, S-LCA, circularity and criticality in relation to the definition of a consistent product system, how the methodologies look at the product system, identified the challenges associated with integrated assessment and what the ultimate goal of integrated assessment is. The analysis highlighted three main features that are addressed by the proposed LC^{3}SA framework.

First, we suggest that when conducting an LC^{3}SA, the application of LCA, S-LCA, and LCC, as well as criticality and circularity assessments should follow their respective inherent methodologies, but a common and comprehensive product system should be defined. This allows a clearer definition of the goal and scope, data requirements and limitations of the study. Consequently, potential overlaps can be anticipated and identified as opportunities, e.g. fewer human resources needed for data gathering, or challenges, e.g. the risks of double counting. Furthermore, we suggest that the analysis of results should not distinguish between domains, but rather relate results to multidimensional AoPs. This would allow for an integrated interpretation of synergies of results, reduce problems of double counting and manage trade-offs between pillars of sustainability.

Second, we propose to communicate impact indicators linked to their associated AoP. We suggest that the AoP "Exhaustible/Natural Resources" should be revised into an AoP "Resources" to include the assessment of natural/primary and secondary resource use. This also enables the introduction of resource related indicators on circularity and criticality of materials.

Third, based on these analyses, we propose a way to include criticality and circularity indicators in LCSA impact assessment, developing an LC^{3}SA framework. The inclusion of those indicators meets the needs of both businesses and the global economy to understand and manage resource use while carefully considering sustainability impacts. We suggest that circularity fits the AoP "Resources" as it can be used to communicate the maximisation of resource use, from reducing waste streams to avoinding virgin materials. Furthermore, criticality indicators could fit the impact pathway of the AoP "Prosperity" or the revised AoP "Resources".

We highlight that the LC^{3}SA framework is a first attempt to integrate circularity and criticality into LCSA but its applicability needs to be tested, i.e. through mathematical modelling. Bringing all the elements together should add value without adding too much complexity to LCSA.

Supplementary data to this article can be found online at https://doi.org/10.1016/j.spc.2024.01.018.

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.

Acknowledgements

The work was conducted in the frame of the ORIENTING project that has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 958231. The authors also acknowledge the financial support received from the Flemish administration via the Circular Economy Center. This publication contains the opinions of the authors, not that of the Flemish administration. The Flemish administration will not carry any liability with respect to the use that can be made of the produced data or conclusions. The authors are grateful to Hanna Pihkola for her valuable suggestions offered during the revision of this manuscript.

References

Adibi, N., Lafhaj, Z., Yehya, M., Payet, J., 2017. Global Resource Indicator for life cycle impact assessment: applied in wind turbine case study. J. Clean. Prod. 165, 1517–1528. https://doi.org/10.1016/j.jclepro.2017.07.226.

Alejandrino, C., Mercante, I., Bovea, M.D., 2021. Life cycle sustainability assessment: lessons learned from case studies. Environ. Impact Assess. Rev. 87, 106517 https://doi.org/10.1016/j.eiar.2020.106517.

André, H., Ljunggren, M., 2021. Towards comprehensive assessment of mineral resource availability? Complementary roles of life cycle, life cycle sustainability and criticality assessments. Resour. Conserv. Recycl. 167, 105396 https://doi.org/10.1016/j.resconrec.2021.105396.

Ardente, F., Latunussa, C.E.L., Blengini, G.A., 2019. Resource efficient recovery of critical and precious metals from waste silicon PV panel recycling. Waste Manag. 91, 156–167. https://doi.org/10.1016/j.wasman.2019.04.059.

Arendt, R., Bachmann, T.M., Motoshita, M., Bach, V., Finkbeiner, M., 2020. Comparison of different monetization methods in LCA: a review. Sustainability (Switzerland). https://doi.org/10.3390/su122410493.

Azevedo, S.G., Godina, R., Matias, J.C. de O., 2017. Proposal of a sustainable circular index for manufacturing companies. Resources 6, 1–24. https://doi.org/10.3390/resources6040063.

Bach, V., Berger, M., Henßler, M., Kirchner, M., Leiser, S., Mohr, L., Rother, E., Ruhland, K., Schneider, L., Tikana, L., Volkhausen, W., Walachowicz, F., Finkbeiner, M., 2016. Integrated method to assess resource efficiency – ESSENZ. J. Clean. Prod. 137, 118–130. https://doi.org/10.1016/j.jclepro.2016.07.077.

Bachmann, T.M., 2013. Towards life cycle sustainability assessment: drawing on the NEEDS project's total cost and multi-criteria decision analysis ranking methods. Int. J. Life Cycle Assess. 18 (9), 1698–1709. https://doi.org/10.1007/s11367-012-0535-3.

Backes, J.G., Traverso, M., 2021. Life cycle sustainability assessment—a survey based potential future development for implementation and interpretation. Sustainability (Switzerland) 13. https://doi.org/10.3390/su132413688.

Bjørn, A., Owsianiak, M., Molin, C., Hauschild, M.Z., 2018. LCA history. In: Hauschild, M., Rosenbaum, R., Olsen, S. (Eds.), Life Cycle Assessment. Springer, Cham. https://doi.org/10.1007/978-3-319-56475-3_3.

Blok, K., Huijbregts, M., Roes, L., Van Haaster, B., Patel, M., Hertwich, E., Wood, R., Hauschild, M.Z., Sellke, P., Antunes, P., Hellweg, S., Ciroth, A., Harmelink, M., 2013. A Novel Methodology for the Sustainability Impact Assessment of New Technologies Report Prepared Within the EC 7th Framework Project Project no: 227078 Project Acronym: PROSUITE Project Title: Development and Application of a Standardized Methodology for the PROspective SUstaInability assessment of TEchnologies.

CEN, E.C.F.S, 2011. EN 15.978:2011 - Sustainability of Construction Works - Assessment of Environmental Performance of Buildings - Calculation Method.

Ciroth, A., 2009. Cost data quality considerations for eco-efficiency measures. Ecol. Econ. 68, 1583–1590. https://doi.org/10.1016/j.ecolecon.2008.08.005.

Dewulf, J., Benini, L., Mancini, L., Sala, S., Blengini, G.A., Ardente, F., Recchioni, M., Maes, J., Pant, R., Pennington, D., 2015a. Rethinking the area of protection "natural resources" in life cycle assessment. Environ. Sci. Technol. 49, 5310–5317. https://doi.org/10.1021/acs.est.5b00734.

Dewulf, J., Mancini, L., Blengini, G.A., Sala, S., Latunussa, C., Pennington, D., 2015b. Toward an overall analytical framework for the integrated sustainability assessment of the production and supply of raw materials and primary energy carriers. J. Ind. Ecol. 19, 963–977. https://doi.org/10.1111/jiec.12289.

Dewulf, J., Mancini, L., Blengini, G.A., Sala, S., Pennington, D., 2015c. Social Issues in Classical and Social LCA: From Identification of Overlaps to an Integrated Framework.

Dewulf, J., Blengini, G.A., Pennington, D., Nuss, P., Nassar, N.T., 2016. Criticality on the international scene: quo vadis? Res. Policy 50, 169–176. https://doi.org/10.1016/j.resourpol.2016.09.008.

Di Noi, C., Ciroth, A., Mancini, L., Eynard, U., Pennington, D., Blengini, G.A., 2020. Can S-LCA methodology support responsible sourcing of raw materials in EU policy context? Int. J. Life Cycle Assess. 25, 332–349. https://doi.org/10.1007/s11367-019-01678-8.

EC, European Union Publications Office, 2024. Circular economy. Glossary term. Accessed in May 2024 at. http://publications.europa.eu/resource/cellar/c06ece73-c7b4-11eb-a925-01aa75ed71a1.

EC-JRC, 2011. ILCD Handbook: Recommendations for Life Cycle Impact Assessment in the European Context, first ed. Publications Office of the European Union, Luxembourg (doi:10.278/33030).

EC-JRC, 2012. Product Environmental Footprint (PEF) Guide, vol. 154. Eropean Commision Joint Research Centre.

Edelen, A., Ingwersen, W.W., Rodríguez, C., Alvarenga, R.A.F., de Almeida, A.R., Wernet, G., 2018. Critical review of elementary flows in LCA data. Int. J. Life Cycle Assess. 23, 1261–1273. https://doi.org/10.1007/s11367-017-1354-3.

European Commission, 2008. The Raw Materials Initiative — Meeting Our Critical Needs for Growth and Jobs in Europe.

European Commission, 2011a. COM(2011) 571 Roadmap to a Resource Efficient Europe.

European Commission, 2011b. Tackling the Challenges in Commodity Markets and on Raw Materials.

European Commission, 2017. Methodology for Establishing the EU List of Critical Raw Materials. Publications Office of the European Union, pp. 1–25. https://doi.org/10.2873/769526.

European Commission, 2020. Study on the EU's List of Critical Raw Materials (2020). European Commission, Luxembourg. https://doi.org/10.2873/11619.

Fauzi, R.T., Lavoie, P., Sorelli, L., Heidari, M.D., Amor, B., 2019. Exploring the current challenges and opportunities of Life Cycle Sustainability Assessment. Sustainability (Switzerland). https://doi.org/10.3390/su11030636.

Finkbeiner, M., Schau, E.M., Lehmann, A., Traverso, M., 2010. Towards life cycle sustainability assessment. Sustainability 2, 3309–3322. https://doi.org/10.3390/su2103309.

Gaasbeek, A., Meijer, E., 2013. Handbook on a Novel Methodology for the Sustainability Impact Assessment of New Technologies.

Gemechu, E.D., Helbig, C., Sonnemann, G., Thorenz, A., Tuma, A., 2016. Import-based indicator for the geopolitical supply risk of raw materials in life cycle sustainability assessments. J. Ind. Ecol. 20, 154–165. https://doi.org/10.1111/jiec.12279.

Godoy León, Blengini, G.A., Dewulf, J., 2020. Cobalt in end-of-life products in the EU, where does it end up? - The MaTrace approach. Resour. Conserv. Recycl. 158, 104842 https://doi.org/10.1016/j.resconrec.2020.104842.

Goedkoop, M.J., de Beer, I.M., Harmens, R., Saling, P., Morris, D., Florea, A., Hettinger, A.L., Indrane, D., Visser, D., Morao, A., Musoke-Flores, E., Alvarado, C., Rawat, I., Schenker, U., Head, M., Collotta, M., Andro, T., Viot, J.-F., Whatelet, A., 2020. Product Social Impact Assessment Handbook 5.0.

Graedel, T.E., Allwood, J., Birat, J.P., Buchert, M., Hagelüken, C., Reck, B.K., Sibley, S.F., Sonnemann, G., 2011. What do we know about metal recycling rates? J. Ind. Ecol. 15, 355–366. https://doi.org/10.1111/j.1530-9290.2011.00342.x.

Graedel, T.E., Barr, R., Chandler, C., Chase, T., Choi, J., Christoffersen, L., Friedlander, E., Henly, C., Jun, C., Nassar, N.T., Schechner, D., Warren, S., Yang, M. Y., Zhu, C., 2012. Methodology of metal criticality determination. Environ. Sci. Technol. 46, 1063–1070. https://doi.org/10.1021/es203534z.

Guinée, J.B., 1995. Development of a Methodology for the Environmental Life-Cycle Assessment of Products: With a Case Study on Margarines. CML, Leiden.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Klejin, R., de Koning, A., van Oers, L., Sleeswijk, A.W., Suh, S., Udo de Haes, H.A., de Brujin, H., van Duin, R., Huijbregts, M.A.J., 2002. Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Kluwer Academic Publishers, Hardbound.

Hackenhaar, I.C., Alvarenga, R.A.F., Bachmann, T.M., Riva, F., Horn, R., Graf, R., Dewulf, J., 2022a. A critical review of criticality methods for a European Life Cycle Sustainability Assessment. In: Procedia CIRP. Elsevier B.V., pp. 428–433. https://doi.org/10.1016/j.procir.2022.02.071

Hackenhaar, I.C., Babí Almenar, J., Elliot, T., Rugani, B., 2022b. A spatiotemporally differentiated product system modelling framework for consequential life cycle assessment. J. Clean. Prod. 333 https://doi.org/10.1016/j.jclepro.2021.130127.

Hauschild, M.Z., Goedkoop, M., Guinée, J., Heijungs, R., Huijbregts, M., Jolliet, O., Margni, M., De Schryver, A., Humbert, S., Laurent, A., Sala, S., Pant, R., 2013. Identifying best existing practice for characterization modeling in life cycle impact assessment. Int. J. Life Cycle Assess. 18, 683–697. https://doi.org/10.1007/s11367-012-0489-5.

Hauschild, M.Z., Rosebaum, K.R., Olsen, S.I., 2018. Life Cycle Assessment-Theory and Practice. Springer, p. 550.

Huijbregts, M., Steinmann, Z.J.N., Elshout, P.M.F.M., Stam, G., Verones, F., Vieira, M.D. M., Zijp, M., van Zelm, R., 2016. ReCiPe 2016, vol. 194. National Institute for Public Health and the Environment. https://doi.org/10.1007/s11367-016-1246-y.

Hunkeler, D., Lichtenvort, K., Rebitzer, G., 2008. Environmental life cycle costing. In: Environmental Life Cycle Costing. https://doi.org/10.1201/9781420054736.

ISO, 2006a. ISO 14040:2006 Environmental Management — Life Cycle Assessment — Principles and Framework (Geneva, Switzerland).

ISO, 2006b. ISO 14044:2006 Environmental management — Life cycle assessment — Requirements and guidelines (Geneva, Switzerland).

ISO, 2015. ISO 14001:2015 Environmental Management— Requirements with Guidance for Use (Geneva, Switzerland).

Karmaker, C.L., Ahmed, T., Ahmed, S., Ali, S.M., Moktadir, M.D.A., Kabir, G., 2021. Improving supply chain sustainability in the context of COVID-19 pandemic in an emerging economy: exploring drivers using an integrated model. Sustain. Prod. Consum. 26, 411–427. https://doi.org/10.1016/j.spc.2020.09.019.

Kirchherr, J., Rieke, D., Hekkert, M., 2017. Conceptualizing the circular economy: an analysis of 114 definitions. Resour. Conserv. Recycl. 127, 221–232. https://doi.org/10.1016/j.resconrec.2017.09.005.

Kloepffer, W., 2008. Life cycle sustainability assessment of products (with comments by Helias A. Udo de Haes, p. 95). In: International Journal of Life Cycle Assessment. Springer Verlag, pp. 89–95. https://doi.org/10.1065/lca2008.02.376.

Luthin, A., Traverso, M., Crawford, R.H., 2023. Circular life cycle sustainability assessment: an integrated framework. J. Ind. Ecol. 00, 1–18. https://doi.org/10.1111/jiec.13446.

Mancini, L., Benini, L., Sala, S., 2016. Characterization of raw materials based on supply risk indicators for Europe. Int. J. Life Cycle Assess. 23, 726–738. https://doi.org/10.1007/s11367-016-1137-2.

Moraga, G., Huysveld, S., Mathieux, F., Blengini, G.A., Alaerts, L., Van Acker, K., de Meester, S., Dewulf, J., 2019. Circular economy indicators: what do they measure? Resour. Conserv. Recycl. 146, 452–461. https://doi.org/10.1016/j.resconrec.2019.03.045.

Moraga, G., Huysveld, S., De Meester, S., Dewulf, J., 2021. Development of circularity indicators based on the in-use occupation of materials. J. Clean. Prod. 279 https://doi.org/10.1016/j.jclepro.2020.123889.

National Research Council, 2008. Minerals, Critical Minerals, and the U.S. Economy, Minerals, Critical Minerals, and the U.S. Economy. https://doi.org/10.17226/12034.

Neugebauer, S., Forin, S., Finkbeiner, M., 2016. From life cycle costing to economic life cycle assessment—introducing an economic impact pathway. Sustainability 8, 428. https://doi.org/10.3390/su8050428.

Nuss, P., Blengini, G.A., 2018. Towards better monitoring of technology critical elements in Europe: coupling of natural and anthropogenic cycles. Sci. Total Environ. 613–614, 569–578. https://doi.org/10.1016/j.scitotenv.2017.09.117.

ORIENTING, 2021a. (D1.4) Critical Evaluation of Material Criticality and Product-related Circularity Approaches. Authors: T. M. Bachmann, I. C. Hackenhaar, R. Horn, M. Charter, F. Gehring, R. Graf, S. Huysveld, R. A. F. Alvarenga, M. Cordella, F. Riva, J. van der Kamp. EU Horizon 2020 project ORIENTING (GA No 958231).

ORIENTING, 2021b. (D1.6) Best Available Approaches for LCSA. Authors: T. M. Bachmann, M. Bianchi, M. Cordella, R. Harmens, S. H. Hong, R. Horn, S. Huysveld, A. Zamagni. EU Horizon 2020 project ORIENTING (GA No 958231).

ORIENTING, 2021c. (D1.3) Critical Evaluation of Economic Approaches. Authors: Bianchi, M., Bachmann, T.M., van der Kamp, J., Riemer, M., Riva, F., Preat, N., Taelman, S.E., Alvarenga, R.A.F. EU Horizon 2020 project ORIENTING (GA No 958231).

ORIENTING, 2021d. (D1.2) Critical Evaluation of Social Approaches. Authors: Harmens, R., Goedkoop, M. EU Horizon 2020 project ORIENTING (GA No 958231).

ORIENTING, 2021e. (D1.1) Critical Evaluation of Environmental Approaches. Authors: Horn, R., Ha Hong, S., Knüpffer, E., Alvarenga, R.A.F., Boone, L., Preat, N., Behm, K., Pajula, T., Pihkola, H. EU Horizon 2020 project ORIENTING (GA No 958231).

ORIENTING, 2021f. (D1.5) Critical Evaluation of Sustainability Integration Approaches. Authors: Huysveld, S., Taelman, S.E., Hackenhaar, I.C., Pihkola, H., Goedkoop, M., Isasa, M., Zachi, L., Kujanpää, L. EU Horizon 2020 project ORIENTING (GA No 958231).

ORIENTING, 2022. (D2.3) LCSA Methodology to be Implemented in WP4 Demonstrations. Authors: Pihkola, H., Cordella, M., Katri, B., Horn, R., Zamagni, A., Zanchi, L., Harmens, R., Sonderegger, T., Cadena Martinez, E., Bachmann, T. M., van der Kamp, J., Bianchi, M., Riva, F., Fehrenbach, D., Moraga, G., Hackenhaar, I. C., Sanchez Moreno, L., Isasa, M., Taelmann, S. E., Kujanpaa, L. Technical report. EU Horizon 2020 project ORIENTING (GA No 958231).

Popescu, I.S., Hitaj, C., Benetto, E., 2021. Measuring the sustainability of investment funds: a critical review of methods and frameworks in sustainable finance. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2021.128016.

Potting, J., Hekkert, M., Worrell, E., Hanemaaijer, A., 2017. Circular Economy: Measuring Innovation in the Product Chain - Policy Report, vol. 42. PBL Netherlands Environmental Assessment Agency.

Pradel, M., Garcia, J., Vaija, M.S., 2021. A framework for good practices to assess abiotic mineral resource depletion in Life Cycle Assessment. J. Clean. Prod. 279, 123296 https://doi.org/10.1016/j.jclepro.2020.123296.

Reichel, Almut, De Schoenmakere, Mieke, Gillabel, Jeroen, European Environment Agency, 2016. Circular Economy in Europe: Developing the Knowledge Base. Publications Office.

Rigamonti, L., Falbo, A., Zampori, L., Sala, S., 2017. Supporting a transition towards sustainable circular economy: sensitivity analysis for the interpretation of LCA for the recovery of electric and electronic waste. Int. J. Life Cycle Assess. 22, 1278–1287. https://doi.org/10.1007/s11367-016-1231-5.

Saidani, M., Yannou, B., Leroy, Y., Cluzel, F., Kendall, A., 2019. A taxonomy of circular economy indicators. J. Clean. Prod. 207, 542–559. https://doi.org/10.1016/j.jclepro.2018.10.014.

Santillán-Saldivar, J., Cimprich, A., Shaikh, N., Laratte, B., Young, S.B., Sonnemann, G., 2021. How recycling mitigates supply risks of critical raw materials: extension of the geopolitical supply risk methodology applied to information and communication technologies in the European Union. Resour. Conserv. Recycl. 164, 105108 https://doi.org/10.1016/j.resconrec.2020.105108.

Santillán-Saldivar, J., Gemechu, E., Muller, S., Villeneuve, J., Young, S.B., Sonnemann, G., 2022. An improved resource midpoint characterization method for supply risk of resources: integrated assessment of Li-ion batteries. Int. J. Life Cycle Assess. 27, 457–468. https://doi.org/10.1007/s11367-022-02027-y.

Santos, A., Carvalho, A., Barbosa-Póvoa, A., 2022. A methodology for integrating the characterization factors uncertainty into life cycle assessments. Sustain. Prod. Consum. 33, 1018–1030. https://doi.org/10.1016/j.spc.2022.08.018.

Schaubroeck, T., Rugani, B., 2017. A revision of what life cycle sustainability assessment should entail: towards modeling the net impact on human well-being. J. Ind. Ecol. 21, 1464–1477. https://doi.org/10.1111/jiec.12653.

Schaubroeck, T., Alvarenga, R.A.F., Verheyen, K., Muys, B., Dewulf, J., 2013. Quantifying the environmental impact of an integrated human/industrial- natural system using life cycle assessment; a case study on a forest and wood processing chain. Environ. Sci. Technol. 47, 13578–13586. https://doi.org/10.1021/es4046633.

Schrijvers, D., Hool, A., Blengini, G.A., Chen, W.Q., Dewulf, J., Eggert, R., van Ellen, L., Gauss, R., Goddin, J., Habib, K., Hagelüken, C., Hirohata, A., Hofmann-Amtenbrink, M., Kosmol, J., Le Gleuher, M., Grohol, M., Ku, A., Lee, M.H., Liu, G., Nansai, K., Nuss, P., Peck, D., Reller, A., Sonnemann, G., Tercero, L., Thorenz, A., Wäger, P.A., 2020. A review of methods and data to determine raw material criticality. Resour. Conserv. Recycl. https://doi.org/10.1016/j.resconrec.2019.104617.

Sonderegger, T., Berger, M., Alvarenga, R., Bach, V., Cimprich, A., Dewulf, J., Frischknecht, R., Guinée, J., Helbig, C., Huppertz, T., Jolliet, O., Motoshita, M., Northey, S., Rugani, B., Schrijvers, D., Schulze, R., Sonnemann, G., Valero, A., Weidema, B.P., Young, S.B., 2020. Mineral resources in life cycle impact assessment—part I: a critical review of existing methods. Int. J. Life Cycle Assess. 25, 784–797.

Sonnemann, G., Gemechu, E.D., Adibi, N., De Bruille, V., Bulle, C., 2015. From a critical review to a conceptual framework for integrating. The criticality of resources into life cycle sustainability assessment. J. Clean. Prod. 94 https://doi.org/10.1016/j.jclepro.2015.01.082.

Swarr, T.E., Hunkeler, D., Klöpffer, W., Pesonen, H.L., Ciroth, A., Brent, A.C., Pagan, R., 2011. Environmental life-cycle costing: a code of practice. Int. J. Life Cycle Assess. 16, 389–391. https://doi.org/10.1007/s11367-011-0287-5.

Taelman, S.E., Schaubroeck, T., De Meester, S., Boone, L., Dewulf, J., 2016. Accounting for land use in life cycle assessment: the value of NPP as a proxy indicator to assess land use impacts on ecosystems. Sci. Total Environ. 550, 143–156. https://doi.org/10.1016/j.scitotenv.2016.01.055.

Taelman, S., Sanjuan, D., Tonini, D., Wandl, A., Dewulf, J., 2020a. REPAiR D4.4: Definitive Framework for Sustainability Assessment.

Taelman, S., Sanjuan-Delmás, D., Tonini, D., Dewulf, J., 2020b. An operational framework for sustainability assessment including local to global impacts: focus on waste management systems. Resour. Conserv. Recycl. 162, 104964 https://doi.org/10.1016/j.resconrec.2020.104964.

Thomassen, G., Peeters, E., Van Hee, N., Noeth, E., Du Bois, E., Boone, L., Compernolle, T., 2024. The environmental impacts of reusable rice packaging: an extended comparative life cycle assessment. Sustain. Prod. Consum. 45, 333–347. https://doi.org/10.1016/j.spc.2024.01.014.

UN, 2015a. Transforming Our World: The 2030 Agenda for Sustainable Development. United Nations.

UN, 2015b. Paris Agreement. United Nations, pp. 1–25.

UNEP, 2020. Guidelines for Social Life Cycle Assessment of Products and Organizations 2020.

UNEP/SETAC, 2011. Towards a Life Cycle Sustainability Assessment (doi:DTI/1412/PA).

UNEP/SETAC, 2019. Global Guidance on Environmental Life Cycle Impact Assessment Indicators.

Valdivia, S., Backes, J.G., Traverso, M., Sonnemann, G., Cucurachi, S., Guinée, J.B., Schaubroeck, T., Finkbeiner, M., Leroy-Parmentier, N., Ugaya, C., Peña, C., Zamagni, A., Inaba, A., Amaral, M., Berger, M., Dvarioniene, J., Vakhitova, T., Benoit-Norris, C., Prox, M., Foolmaun, R., Goedkoop, M., 2021. Principles for the application of life cycle sustainability assessment. Int. J. Life Cycle Assess. 26, 1900–1905. https://doi.org/10.1007/s11367-021-01958-2.

Velenturf, A.P.M., Purnell, P., 2021. Principles for a sustainable circular economy. Sustain. Prod. Consum. 27, 1437–1457. https://doi.org/10.1016/j.spc.2021.02.018.

Verones, F., Bare, J., Bulle, C., Frischknecht, R., Hauschild, M., Hellweg, S., Henderson, A., Jolliet, O., Laurent, A., Liao, X., Lindner, J.P., Maia de Souza, D., Michelsen, O., Patouillard, L., Pfister, S., Posthuma, L., Prado, V., Ridoutt, B., Rosenbaum, R.K., Sala, S., Ugaya, C., Vieira, M., Fantke, P., 2017. LCIA framework and cross-cutting issues guidance within the UNEP-SETAC Life Cycle Initiative. J. Clean. Prod. 161, 957–967. https://doi.org/10.1016/j.jclepro.2017.05.206.

Vogtländer, J., Peck, D., Kurowicka, D., 2019. The eco-costs of material scarcity, a resource indicator for LCA, derived from a statistical analysis on excessive price peaks. Sustainability (Switzerland) 11. https://doi.org/10.3390/su11082446.

WCED, W.C. on E. and D, 1987. Report of the World Commission on Environment and Development: Our Common Future.

Wiedmann, T., Wilting, H., Lutter, S., Palm, V., Giljum, S., Wadeskog, A., Durk, N., 2009. Development of a methodology for the assessment of global environmental impacts of traded goods and services. In: Final Report, ISBN 978-1-84911-121-8.

Yavor, K.M., Bach, V., Finkbeiner, M., 2021. Adapting the ESSENZ method to assess company-specific criticality aspects. Resources 10. https://doi.org/10.3390/resources10060056.

Zampori, L., Pant, R., 2019. Suggestions for updating the Product Environmental Footprint (PEF) method (Issue EUR 29682 EN, JRC115959). Publications Office of the European Union. https://doi.org/10.2760/424613.