Resources, Conservation & Recycling Advances 21 (2024) 200203

Contents lists available at ScienceDirect

Resources, Conservation & Recycling Advances

journal homepage: www.sciencedirect.com/journal/Resources-Conservation-and-Recycling-Advances

Advancing environmental assessment of the circular economy: Challenges and opportunities

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ARTICLE INFO

Keywords:
Circular economy
Lifecycle assessment
Environmental assessment
Data uncertainty, Meso-level circular economy, Macro-level circular economy

ABSTRACT

The lifecycle assessment (LCA) framework is widely applied to comprehensively evaluate and improve the environmental performance of a circular economy (CE). The advances and application of LCA has been primarily restricted to evaluate the environmental performance of the CE at a micro-level, wherein the CE is implemented for a single product system.

However, the CE can be operationalized at two broader levels: the meso–level (for eco-industrial parks) and the macro-level (for a city, state, or nation). Six methodological challenges emerge when applying LCA to a meso- or macro-level CE and remain unaddressed in the existing literature. This includes: selecting a relevant system boundary and functional unit, addressing data paucity and uncertainty, accounting for stakeholder behavior, assessing the trade-offs from renewable energy (RE) use, accounting for manufacturing and technology evolution, and quantifying displacement and rebound. This article proposes potential solutions and research priorities to address the above challenges.

Introduction

The Circular economy (CE) focusses on transitioning from a historically dominant "linear economy," one in which product life cycles typically follow a "take-make-use-dispose" pattern, to an economy which eliminates waste and pollution, regenerates nature and extends the lifetime of products and materials in their highest functional state. (Ellen Macarthur Foundation 2022) CE is increasingly adopted by governments (European Commission 2020; Bleischwitz et al., 2022; Canadian Circularities 2022; Geng et al., 2019; Finnish Government 2021; Wang et al., 2022) and industry (Ellen Macarthur Foundation 2022) to address a range of sustainability issues such as climate change, (Wang et al., 2022; Serrano et al., 2021) transition to renewable energy (RE) systems, (Heath et al., 2022; Mendoza et al., 2022; Heath et al., 2022) decrease the environmental footprint of manufacturing processes, (Singh et al., 2021; Acerbi and Taisch, 2020) decelerate the loss in biodiversity and ecological functions, (Ali et al., 2018) and alleviate supply constraints for critical materials. (Babbitt et al., 2021)

The CE is increasingly popular as it draws upon well-established principles of industrial ecology, (Wiprächtiger et al., 2023) waste management and systems thinking. (Reike et al., 2018; Winans et al., 2017; Gertsakis and Lewis, 2003; Ghisellini et al., 2016) Specifically, the CE principles of closing the material loop by using material waste from one product in another product, minimizing the material and energy consumption, as well as the waste generation over the life cycle of a product, transitioning to renewable energy sources, and maximizing product service life are well established in industrial ecology. (Wiprächtiger et al., 2023; Saidani et al., 2020) Additionally, CE is popular as it is accessible to practitioners and industry, (Saidani et al., 2020; Lüdeke-Freund et al., 2018) can generate an environmental and economic benefit by decreasing the material intensity and waste associated with economic activity, (World Economic Forum 2022) and can generate social benefits such as increased employment, health benefits, and decreased exposure to pollution. (Andooz et al., 2023) The embrace of CE by policy makers (European Commission 2020; Geng et al., 2012) and industry has motivated research on identifying and classifying indicators (e.g., End-of-Life Recycling Rates, resource duration indicator) (Saidani et al., 2019; Moraga et al., 2019; Parchomenko et al., 2019;

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https://doi.org/10.1016/j.rcradv.2024.200203

Available online 15 January 2024

Padilla-Rivera et al., 2021; Galatti and Baruque-Ramos, 2022; Pitkänen et al., 2023; Luthin et al., 2023; Niero et al., 2021) to quantify the material, environmental and social benefit from the CE. CE indicators provide an objective measure and rapid guidance and feedback (Alaerts et al., 2019; Franklin-Johnson et al., 2016) to assist stakeholders such as policy makers, conservationists, technologists and corporations in aligning conservation strategies, product designs, government policy and corporate performance with CE principles.

With limited guidance on how to select and apply the various CE, a stakeholder (e.g., a product manufacturer) might selectively choose indicators that narrowly benefit their own needs and marketing. (Pauliuk, 2018; Boyer et al., 2021) Circularity indicators may not always serve as a proxy for environmental impacts, (Boldoczki et al., 2021) unless validated by a robust environmental assessment. (Harris et al., 2021; Kravchenko et al., 2020) A sole reliance on circularity indicators may result in companies engaging in "circular [green] washing." (Harris et al., 2021)

In response to the skepticism on the magnitude of the environmental gains from a CE, (Boldoczki et al., 2021; Skene, 2017; Corvellec et al., 2021) there have been calls to integrate the CE and measures of circularity with lifecycle assessment (LCA). (S. Singh et al., 2021; Kravchenko et al., 2020; Haupt and Zschokke, 2017; Peña et al., 2021; Rigamonti and Mancini, 2021; Brändström and Saidani, 2022) LCA is well suited and widely applied (Corona et al., 2019) to assess a CE as it accounts for the energy and material flows and emissions across the full life cycle of a product. This aligns well with quantifying the environmental impact from reducing the material and energy intensity of products through CE strategies. An LCA identifies environmental burden shifts between life cycle stages which can help evaluate if CE strategies applied in one life cycle stage of a product can shift burdens to another product stage. (Peña et al., 2021; Corona et al., 2019) Additionally, LCA, which can identify burden shifting from one environmental impact category to another, can complement CE indicators which typically evaluate the CE performance based on only a single issue (e.g., material recovery rates) and not multiple issues (e.g., climate and toxicity impacts from a CE). (Rigamonti and Mancini, 2021; Helander et al., 2019) LCAs have been applied to CEs of several products such as tires, (Lonca et al., 2018) photovoltaics (PV), (Heath et al., 2022; Ravikumar et al., 2016; Ravikumar et al., 2020; Rajagopalan et al., 2021) lithium-ion batteries, (eath et al., 2022) wind turbines, (Ghosh et al., 2022) cups, food and packaging systems, (Rufí-Salís et al., 2021; Associates, 2006; Lighart, 2007; van der Harst and Potting, 2013; van der Harst et al., 2014; Potting and van der Harst, 2015; Miller, 2020; Biganzoli et al., 2018; Sazdovski et al., 2021) carbon dioxide utilization (Ravikumar et al., 2021; D. Ravikumar et al., 2021) and appliances. (Bracquené et al., 2020) Nevertheless, the application of LCA is largely limited to CE at a product-level or a micro-level (Harris et al., 2021; Roos Lindgreen et al., 2021; Kirchherr et al., 2017; Merli et al., 2018; Walzberg et al., 2021) due to the similarity in the goals in evaluating an individual product in a linear economy and in a CE context (SI Fig. 2). For example, LCAs have been applied to products in a linear economy to identify hotspots to improve the environmental performance and guide R&D, (Wender et al., 2014; Hellweg and Milà i Canals, 2014) compare the environmental performance of competing alternatives, (Ravikumar et al., 2018) determine burden shifting across the different life cycle stages or one environmental impact to another, (Czyrnek-Delêtre et al., 2017) and quantify the environmental consequences when a product is introduced in a market (i.e., consequential analysis). (Whitefoot et al., 2011) These examples are similar to the application of LCA to a CE at a micro-level to identify environmental hotspots and recommending R&D strategies for improve a CE implementation, (Ravikumar et al., 2016) compare the environmental performance of various CE strategies, (Rajagopalan et al., 2021; Chen, 2019; Joensuu et al., 2022; Richa et al., 2017) quantify the trade-offs between the different lifecycle stages when implementing a CE for a product, (Lonca et al., 2018) and quantify the consequential environmental impact of obtaining a product through a CE. (Buyle et al., 2019)

However, a CE can be operationalized at two levels broader than the micro-level: the meso-level wherein circularity is implemented for eco-industrial parks or firms within geographic proximity, and the macro-level wherein circularity is implemented at a city, state or nation. (Kirchherr et al., 2017; Merli et al., 2018; Walzberg et al., 2021) With increased focus on scaling the CE to the level of cities, provinces and the nation, (European Commission 2020; Bleischwitz et al., 2022; Canadian Circularities 2022; Geng et al., 2019; Finnish Government 2021; Wang et al., 2022) there is a need to apply LCAs and ensure positive environmental outcomes from a meso- or macro-level CE.

An LCA of a meso- or macro-level CE is not straightforward and methodological challenges emerge from expanding the system boundary of the LCA to account for an increased number of product systems and the diversity in their functional units. For instance, increased closed loop recycling of polyethylene terephthalate (PET) bottles might be environmentally beneficial at a micro-level but not at the macro level, when the system boundary of the LCA includes the whole PET market. (Lonca et al., 2020)

A macro-level CE consists of diverse stakeholders interfacing with different technology systems which are rapidly evolving and require a complex mix of material and energy inputs from diverse sources (e.g., renewable and fossil sources) and generate potentially recoverable wastes. Furthermore, the application of LCA to quantify the environmental impacts of the macro-level CE requires data (e.g., quantities and type of material and energy flows).

With an increase in stakeholders and potential conflicts of interests in how the CE has to be implemented (e.g., corporations opposing efforts to reduce virgin plastic production and instead promoting plastic recycling), (Scientific American 2023) there is a need to account for stakeholder behavior when conducting the LCA for a meso- and macro-level CE. (Mah, 2021)

The LCA of a macro-level CE should account for the evolution in technology, manufacturing and energy mixes. The environmental gains of a macro-level CE which recovers and reuses materials from an older product will dynamically decrease as manufacturing improvements reduce the material intensity of products. For example, the environmental benefit from recovering solar grade silicon (SOG-Si) from a PV panel and reusing it to produce concrete (Fernández et al., 2011) progressively decreases as the SOG-Si content in PV panels declines. (Hallam et al., 2022) Furthermore, the LCA should account for the foregone environmental gains if extending the lifetime and continued use an older generation product with a lower technical efficiency (e.g., refurbish) offsets a newer and a technologically improved product. (Gutowski et al., 2011)

Materials recovered from a CE will not always displace virgin materials. (Yang, 2016) Additionally, the economic gains resulting from recovering and reusing a product in a second life can result in increased consumption of new products (i.e., rebound effect). (Zink and Geyer, 2017) The type of materials displaced, and the rebound effect need to be accounted for in an LCA for a macro-level CE.

Replacing fossil energy with RE sources is increasingly recommended to improve the environmental performance of a macro-level CE (e.g., recycling CO_2 from manufacturing sector into fuels or chemicals). (Katelhon et al., 2019) However, with multiple competing needs and constrained supplies of RE (RE represents 13 % of the global primary energy consumed in 2021), (British Petroleum 2022) there is a need for LCAs of macro-level CEs to account for the environmental trade-offs between the use of constrained RE supplies between CE and non-CE applications.

The paucity of data on the energy, material and waste flows, which is required to quantify the environmental impacts of a product, is a recurring concern in LCA. (Hetherington et al., 2013; Zargar et al., 2022) This concern is further exacerbated in a macro-level LCA wherein the number and type of products increases and the likelihood of product owners and manufacturers sharing proprietary and commercially sensitive data decreases. (Luoma et al., 2023)

To assess if LCAs of macro-level CEs account for the above challenges, we conducted a review of the published literature (SI Section S1 and S2). A search in the Scopus database for LCAs of CE returned a total of 141 studies out of which 81 studies were included for further analysis. None of the 81 studies quantitatively accounted for the emergent challenges highlighted above.

In this work, we present a framework that includes material and energy flows, information flows and stakeholder coordination, which impact the environmental outcomes of a macro-level CE, discuss the methodological challenges mentioned above in applying LCA to a macro-scale CE, identify research priorities to address the challenges and, thereby, advance the application of LCA to assess the environmental performance of a macro-level CE.

Materials and methods

Advancing environmental assessment for a meso- and macro-level CE

Fig. 1 provides a framework to assess the environmental performance of the CE at the meso- or macro-level (upper half, grey box) and the CE at the product of micro-level (lower half, orange box). The framework also depicts the goals of the macro-level CE which includes minimizing the material and energy intensity of providing societal functions through product systems, transitioning to renewable sources of materials and energy, minimizing waste generation and enhancing the environmental justice (Purvis et al., 2023; Berry et al., 2021) outcomes when transitioning to a CE. The environmental performance of the CE at the meso- or macro-level based is influenced by three key factors - energy and material flows, information flows and the planning and coordination of the CE – which are explained below.

Material and energy flows: At a macro-level, the environmental performance of CE is driven by the aggregate effects of the virgin and secondary flows of energy and materials consumed and the waste emitted by the multiple product systems, which comprise the CE and provide a mix of key societal functions of housing, nutrition, mobility, services, consumables, communications and healthcare. (Harris et al., 2021) The extraction of virgin energy and materials and the waste generated across the manufacture (m), use and end-of-life (lower half, orange box) stages of an individual product system have an associated life cycle environmental burden. In contrast, secondary material and energy flows recovered through various CE strategies applied to the multiple product systems have the potential to generate an environmental benefit by displacing virgin extraction or decreasing the material and energy intensity of providing a service. There is an emerging consensus in the literature that there are 10 possible strategies (or 'R' strategies) to implement a CE across the life cycle stages of an individual product system (or micro-level): R0 – Refuse, R1 -Rethink, R2 – Reduce, R3 – Reuse, R4 – Repair, R5 – Refurbish, R6 – Remanufacture, R7 – Repurpose, R8 – Recycle, and R9 – Recover. (Reike et al., 2018; Morseletto, 2020; Potting et al., 2017; Reike et al., 2022) [Note: refer the section titled "*Quantifying displacement and rebound*" for further discussions on quantifying the potential environmental benefit from displacement of virgin materials and energy flows.]

Information flows: Tracking and understanding the flow of information is vital for creating and improving CE ecosystems. Material and energy flows are shaped by decisions made by key stakeholders including the material and energy industries, original equipment manufacturers (OEMs), consumers, end-of-life managers, regulators, and investors. Their decisions are guided and influenced by information and data ranging from material attributes to prices and regulatory drivers. Market signals and regulations influence material and energy flows between multiple products systems and can decrease the material and energy intensity of extracting functions from product systems and inform decisions on improving the material and energy efficiencies of industrial processes. (Neligan et al., 2022) For example, the information on the magnitude, type and location of waste PV panels and lithium ion batteries (LIB) is required to optimally level and geospatially locate recycling infrastructure to minimize the economic and environmental cost of recycle products. (G.A. Heath et al., 2022) As a result, the

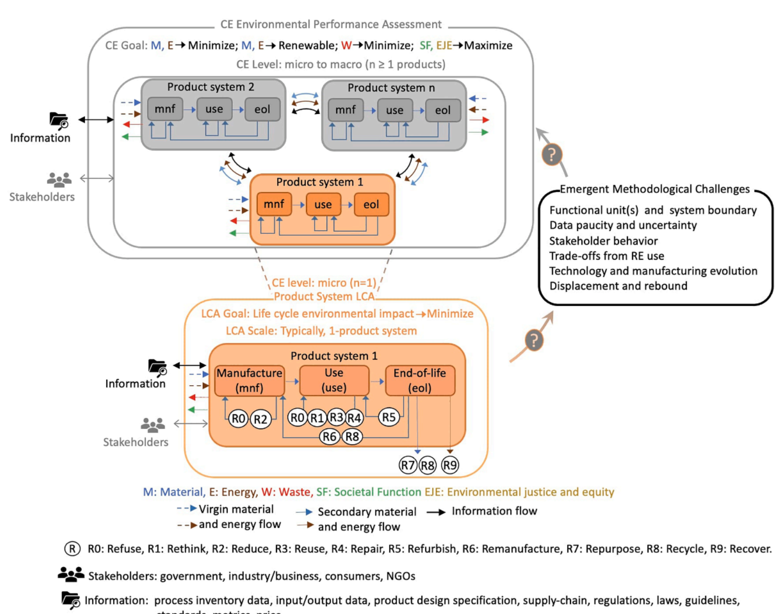

Fig. 1. Conceptual framework to assess the environmental performance of the CE. The CE is seen as a collection of interconnected product systems (upper half) that collectively require material and energy inputs (blue and brown arrows), emit waste (to land, water or air, red arrow), necessitate information inflows and outflows (black arrows), receive inputs, and provide feedback to stakeholders and provide societal functions (green arrow). LCA of a micro-level CE (lower half) requires elevated attention to product specific flows for the different life cycle stages and the different CE (or R) strategies. The same environmental impact categories of an LCA can be used to assess the environmental performance of the CE at a meso- or macro-level and for the provision of societal functions. The box on the right presents the emergent methodological challenges in applying LCA when expanding the scope of the analysis from the product to the meso- or macro-level CE. Societal functions include housing, nutrition, mobility, consumables, services, healthcare, and communication.

information flows impact the flow of energy and materials between multiple product systems and, thereby, has to be accounted for by the LCA when evaluating the environmental performance of a macro-level CE. Other types of information include industry association guidelines standards and specifications. [Note: refer the section below on *"Addressing data paucity and uncertainty"* for discussion on resolving challenges associate with information availability when applying LCA to a macro-level LCA.]

CE planning and coordinating: The planning, design and implementation of an environmentally successful CE at a macro-level requires leadership and coordination in designing the CE and enhancing collaboration between the different stakeholders in the CE with potentially conflicting goals.

There are multiple key stakeholders involved in setting the overall goal and implementing the macro-level CE (hereafter the "*owners of the CE*"). The implementation will require coordination between various stakeholders such as organizations owning a diverse portfolio of products, materials, energy and infrastructure systems (e.g., reverse logistics and waste management), customers who consume the service from the products and adopt CE strategies (e.g., repair a product, (Terzioğlu, 2021) consume remanufactured instead of new products), (Hazen et al., 2017) and planners and entities making policy and regulations. With diverse products and multiple stakeholders, the complexity and difficulty in coordination increases when the implementation of a CE broadens from a micro to a macro-level. (World Economic Forum 2022) The coordination includes facilitating the flow of material, components and energy, sharing of infrastructure between different product systems to maximize environmental benefits, (Lehtoranta et al., 2011) increasing awareness of the overall environmental goals of the CE (Kuo and Chang, 2021; Liu and Bai, 2014) and the role of various stakeholders in achieving them, incentivizing actions towards the goals of the CE through policy and creating and sharing information between stakeholders that is vital to sustain the CE. (Munaro and Tavares, 2023) For instance, a mapping of industrial symbiosis development in Europe has highlighted the importance of knowledge sharing and collaboration for projects to be successful (Domenech et al., 2019) and non-profits in Italy have focused on knowledge sharing through various channels (e.g., cultural events, urban regeneration projects, repair cafes) to promote various CE strategies. (Ghisellini and Ulgiati, 2020)

Another concern for a macro-level CE is that the overall goal of the CE may not align with the environmental and economic interests of the individual stakeholders. (Fischer and Pascucci, 2017) For example, consider a scenario for a CE for carbon dioxide (CO_2) wherein CO_2 emitted by the manufacturing sector is converted into chemicals using RE generated by the electricity sector. RE is required to ensure a climate benefit is generated when converting the CO_2 into chemicals. (Ravikumar et al., 2021; Ravikumar et al., 2020) However, the implementation of a CE for CO_2 emitted from the manufacturing sector does not align with the climate goals of the electricity sector. The RE can generate a greater climate benefit for the electricity sector than the manufacturing sector when it is supplied to the grid to displace fossil electrcity. (Ravikumar et al., 2021; Ravikumar et al., 2020)

Potential approaches to align stakeholder interests with the overall goal of the CE include providing incentives through policies, (Katrakis et al., 2021) incentivizing the recovery and use of secondary sources over economically lower cost virgin sources of material through taxes on material extraction (Söderholm, 2011) and waste incineration, (De Weerdt et al., 2020) incentivizing environmentally efficient industrial supply chains by mandating the procurement of less environmentally intensive materials, (City of Toronto 2018; Zhu et al., 2011) supporting business model requiring collaboration between stakeholders, (Santa-Maria et al., 2021) and developing infrastructure to support CE operations. (Lieder and Rashid, 2016; Scheepens et al., 2016) Recent examples show that establishing working groups consisting of diverse stakeholders (e.g., government, business, academia, not-for-profits) (Schumacher and Green, 2021) to identify synergies and novel opportunities, resolve conflicts, formulate new CE strategies, and exchange information is a promising approach to improve coordination and planning for a macro-level CE. (OECD 2021)

Emergent methodological challenges for LCA at a meso- and macro-level circular economy

The box on the right in Fig. 1 lists the methodological challenges that emerge when expanding the scope of an LCA from evaluating the environmental performance of a micro-level CE (lower half) to a meso- or a macro-level CE (upper half).

Selecting a relevant system boundary and functional unit(s): Circular systems can introduce notable challenges in the LCA scope definition phase. Consider, for example, an electric vehicle designed with CE principles: it may contain aluminum components intended for recycling back into auto components; plastics designed for easy separation and recycling as a secondary material in an unknown application; and a battery that, after its useful life in the mobile application, retains sufficient function to serve in a less demanding stationary application. (Schulz et al., 2020) In assessing this product system, one could expand the system boundaries (as is often done in LCA to avoid allocation) to also include all secondary use systems and define a functional unit that incorporates not only the primary function of transportation but also functions derived from the second life of different components (e.g., battery 2nd life) and the recycled material supply functions, as has been proposed previously. (Niero and Olsen, 2016) This approach can quickly become intractable as the CE expands and additional products become interconnected through circular material flows. Such an approach would likely require cut-off criteria like current LCA system definition methods that could be based on the level of influence on key environmental indicators of subsequent "connected systems". Boundaries could initially be set to encompass the product systems that have the largest material and energy couplings.

An alternative approach is to broaden the scope of analysis from primarily product-level to questions related to sector or even economy-wide levels, as has been proposed for life cycle sustainability assessment frameworks (Guinee et al., 2011; Onat et al., 2017) and suggested in a CE context. (Harris et al., 2021) Efforts to date to evaluate the environmental impact at the meso- or macro- level have largely utilized cross-sectoral linkages provided by environmentally extended input-output analysis (EIOLCA) or Multi-Regional Input-Output (MRIO). (Aguilar-Hernandez et al., 2019; Tisserant et al., 2017) With additional development of methods and databases, this approach holds the potential to evaluate the influence of circular design and business strategies on the environmental performance of the broader economy at the sectoral, national, or even global level. There is a notable gap in the current understanding of the linkages between system levels (i.e., micro- to macro-), and further research is needed. (Harris et al., 2021) A framework based on societal function or needs (e.g., housing, nutrition, mobility, consumables, services, healthcare, and communications) can be a foundation for understanding interactions between system levels and tracking and monitoring a CE. (Harris et al., 2021) Another limitation of using EIOLCA is input-output tables are monetary-based and, therefore, do not fully represent actual physical transitions in the economy. (Singh et al., 2021) Hence, one way to improve insights gained from EIOLCA in the CE context would be the creation of hybrid or physical input–output models. (Singh et al., 2018) In addition, assessments involving EIOLCA, MRIO, or conventional LCA for that matter, rely on current (or past) relationships between systems and sectors, so forecasting future responses to the introduction of CE approaches is limited. Realizing such predictive power would require deepening frameworks and models by incorporating additional market economic, behavioral, and physical constraint relations (e.g., planetary boundaries, resource availability).

Addressing data paucity and uncertainty: Considerable attention has been paid to addressing the methodological challenges in conducting an LCA due to data scarcity (Hetherington et al., 2013; Zargar et al., 2022) and the uncertainty. (Wender et al., 2014; Ravikumar et al., 2018; Cucurachi et al., 2016) This includes the lack and uncertainty in the bill-of-materials data for novel and emerging technologies which can impede the application of LCA and the ability of LCA to inform decisions on selecting an environmental preferred option from multiple alternatives or identifying the most effective strategy to improve the environmental performance of a technology. (Whitefoot et al., 2011) In a CE, the data challenges are exacerbated with the widening of the system boundary to include multiple products and the additional material, energy and waste flows (Esbensen and Velis, 2016) between the various product systems and the ecosystem.

There has been limited efforts to identify, categorize and rank by importance the different types of information in a macro-level CE. (G.A. Heath et al., 2022) To the best knowledge of the authors, there has been no effort to quantify the impact of information availability on operationalizing a CE and the resulting environmental impact.

Despite recent initiatives, (GS1 2022) there are no formal mechanism or guidelines to increase data transparency and address the concerns of manufacturers and suppliers on providing access to potentially proprietary, confidential or commercially sensitive product or process information. Existing approaches such as the aggregation of data to conceal confidential or commercially valuable information, (Kuczenski et al., 2016) the provision of data by industry organizations or third parties, (World Steel Association 2022; Athena Sustainable Materials Institute 2022) and leveraging public-private partnerships to create and host inventory data across multiple products and sectors (UNEP 2016) can be leveraged to increase transparency and access to data required for assessing the environmental performance of macro-level CE.

There has been an increased adoption of digital technologies (e.g., radio frequency identification (RFID), internet of things (IOT)) to collect and analyze real-time data for monitoring product health, (P et al., 2021; Deriche et al., 2019) share information between various stakeholders (Lawrenz and Leiding, 2021) and sort and track waste (Magrini et al., 2021) which are integral to the CE. (Pagoropoulos et al., 2017; Liu et al., 2022) The blockchain technology emerges especially promising to improve trust among stakeholder, enhance knowledge sharing and track material flows. (Rejeb et al., 2023) Providing public access to data (data.europa.eu 2022) generated by such technologies in publicly funded CE initiatives or publicly owned industrial entities, (Kuo and Chang, 2021) developing guidelines to collect and generate the data, and standardizing technical platforms to share the generated data by digital technologies across the supply chain for the product systems can also help address issues of data paucity for CE of multiproduct systems. (Accenture 2021) Moreover, facilitating public access to data (Kauppila et al., 2022) can potentially help evaluate if the best practices for a CE at a micro-level can be tailored, extended and applied to a CE at the macro-level.

Computational techniques such as mechanistic models (Vunnava and Singh, 2021) and environmentally extended input–output analysis (EEIOA) (Aguilar-Hernandez et al., 2018) have shown promise in evaluating the environmental outcomes from a macro-level CE by leveraging multisectoral data. However, such efforts are limited to specific CE strategies (e.g., recycling, (Vunnava and Singh, 2021) lifetime extension (Donati et al., 2020)) and sectors (e.g., agriculture (Vunnava and Singh, 2021)) and are more relevant to analysis at the aggregated sectoral level. The collection of multi-sectoral data is expensive and requires continual updates to account for new product and technology releases in the market. (Walzberg et al., 2021) Further methodological advances are required so that computational techniques can evaluate a broader set of CE strategies beyond recycling and environmental impact categories other than global warming potential (GWP) (e.g. land use, toxicity), quantify environmental impact at a more disaggregated macroeconomic level (e.g., product system level) and assess the impact of using dated multi-sectoral data on assessing environmental performance of CE.

Accounting for the impact of stakeholder behavior: Stakeholders can influence a circular economy by refusing a product, (Zambrano-Monserrate and Alejandra Ruano, 2020) adopting services instead of purchasing products (i.e., rethink), extending the lifetime extension of a product (i.e., reduce), (McNeill et al., 2020; Woody et al., 2020) reusing a product, (Walzberg et al., 2022; Walzberg et al., 2021) impacting the sorting and disposal of waste, (Nainggolan et al., 2019) purchasing refurbished (Harms and Linton, 2016; Sharifi and Shokouhyar, 2021) and remanufactured (Singhal et al., 2019) products and waste materials, (Bleicher, 2020) and recycling a product. (Yin et al., 2014; Botetzagias et al., 2015) Identifying and addressing stakeholder concerns and incentivizing stakeholder participation in a CE, therefore, can increase the likelihood of realizing an environmental benefit from a CE. (Khan et al., 2020)

However, existing literature investigating the impact of stakeholder behavior are limited to micro-level CEs (e.g., wind, (Walzberg et al., 2022) PV, (Walzberg et al., 2021) hard-disk drives, (Walzberg et al., 2022) plastics (Khan et al., 2020)). There are interdependencies between different product systems in a circular economy and stakeholder behavior impacting the CE for one product system can simultaneously promote or impede the CE strategy for another product system. For example, the increased adoption of rideshare by customers, (Pew Research Center 2019) which corresponds to the CE strategy of reduce, in combination with the increased adoption of battery electric vehicles (BEV) by rideshare providers (Uber 2022; Lyft 2020) will increase the sales of BEV. Consequently, there will be a potential increase in the volumes of LIB at the end-of-life from the BEVs, which can drive the CE strategy of repurpose when the recovered LIBs are used for energy storage on the grid. There is a need for further research on how stakeholder interaction with a product system impacts the material and energy flows and CE strategies for other product systems and, thereby, influences the overall environmental outcome from a macro-level CE.

A potential approach is to leverage existing methods in socio-technical analysis and complex systems science (Walzberg et al., 2021) such as systems dynamics (SD), (Asif et al., 2016; Franco, 2019) agent based modeling (ABM) (Lieder et al., 2017) and discrete event simulation (DES). (Charnley et al., 2019) An exploratory analysis using SD, ABM or DES can determine how various stakeholders interactions with a product system in a CE can impact the energy, material and waste flows for other product systems in a multi-product system. For example, in a multiproduct CE where materials recovered from a spent PV panel are reused in cement, (Fernández et al., 2011) pavement (Sandanayake et al., 2022) or batteries, (Sim et al., 2023) the use of ABM can help identify how stakeholder behavior can be incentivized to increase the return of spent PV panels (Walzberg et al., 2021) and, thereby, increase the potential to recover and recycle materials. The change in the material and energy flows can be integrated with an LCA (Wu et al., 2017) to quantify the overall environmental impact of a macro-level CE. Other approaches such as using partial equilibrium models (PEM) or computable general equilibrium (CGE) can also account for the impact CE systems have on the economy. The exploratory analysis can identify key stakeholders and behaviors which have the most significant impact on the environmental benefit from the multiproduct CE system, thereby, help design interventions to incentivize such behavior. Beyond application, there is a simultaneous research need to address the methodological challenges that arise when leveraging the above methods to account for stakeholder behavior of multiproduct CE systems. Costs and efforts increase due to additional data collection, and validation requirements which are necessary to ensure that the models and assumptions accurately reflect real-world conditions. (Rand, 2019) There is a need for transparent reporting of the limitations as the findings and insights based on applying such socio-technical methods may not be directly generalized to different geographies and markets. (Hansen et al., 2019) A potential approach to address data paucity for socio-technical and complex systems science modeling of a CE is to leverage existing methods in LCA which were developed to address the lack of and high degree of uncertainty in data in LCA. (Ravikumar et al., 2018; Cucurachi et al., 2016; Wender et al., 2017; Heijungs et al., 2019; Cucurachi et al., 2021)

Assessing the trade-offs from using RE for CE: A transition from fossil to renewable sources of energy is a fundamental strategy to transition to a CE. (Sikdar, 2019) Circular economy solutions for carbon capture and utilization (CCU), (Deutz and Bardow, 2021; Fajardy et al., 2019) production of chemicals (Hoppe et al., 2018) and fuels (Liu et al., 2020) are environmentally feasible only when low-carbon RE is used as the source of energy. Despite the rapid acceleration in installations, RE represents only 13 % of the primary energy consumed globally, (British Petroleum 2022) served only 50 % of the global increase in electricity demand in 2021 and 2022, (International Energy Agency 2021) and may be insufficient to meet the demand from various applications. Furthermore, RE is expected to contribute 27 to 66 % of the global energy requirements by 2050. (International Renewable Energy Agency (IRENA) 2018) As a result, in the current circumstance and for a decade or more into the future, there is a limited supply of RE. The constrained supply of RE needs to be allocated between CE solutions and other competing applications such as direct supply to the grid to decrease reliance on fossil electricity, decarbonize transportation systems, (U.S. Department of Energy (DOE) 2023) generate green hydrogen and decarbonize manufacturing and heat production. (U.S. Department of Energy (DOE) 2022) For example, recent studies have shown that using RE to recycle CO_2 into chemicals and fuels is environmentally suboptimal as the RE can generate a greater climate benefit when supplied to the grid to displace fossil electricity. (Ravikumar et al., 2021; Ravikumar et al., 2020) As a result, there is a need at least for the next decade to continually assess if the use of RE for a macro-level CE strategy may forgo an opportunity to use the RE in an alternate application with a greater environmental benefit. (Sternberg and Bardow, 2015) This issue is more important at the macro-level wherein the magnitude of RE use, the corresponding environmental trade-off and constraints on RE availability increases (e.g., RE required for converting captured CO_2 into commercially valuable products increases when CCU is implemented globally for several gigatons of worldwide CO_2 emissions).

Accounting for technology and manufacturing evolution: By retaining the functional properties of the product system or components of a product system, "inner loop" CE strategies such as remanufacture, refurbish and repair enable a second life for a legacy product system. However, inner loop CE strategies are typically implemented at the end of the technical or functional lifespan of a product system - a timeframe during which a newer and technically improved product will most likely be available on the market. (Devoldere et al., 2009; Kim et al., 2003) The environmental assessment of inner loop CE strategies should, therefore, account for the forgone environmental benefit when a newer product with improved technical features and performance is substituted by a legacy product. (Kwak and Kim, 2016) For example, there is a rapid increase in the technical efficiency for PV panels. (National Renewable Energy Laboratory 2023) Refurbishing legacy solar PV panels after a typical functional lifespan of 30 years, while being beneficial from the standpoint of extending the lifetime of a product, may substitute a more efficient and newer PV panel with a capacity to generate more RE.

Similarly, the environmental assessment of the outer loop CE processes, which recover material from a legacy product, need to account for the reduced or changed material needs in a newer product. A reduced or changed material need would mean the secondary materials recovered from a legacy product will be reused at a reduced quantity in the new product or reused in an alternate application. When the material cannot be reused in a product system from which it is recovered, the LCA should include a robust scenario analysis to quantify the variation in the environmental impact over the possible set of alternate product system (s) in which the recovered material can potentially be reused. For example, the quantity of solar grade silicon (SOG-Si) used in the manufacture of crystalline silicon (c-Si) PV panel is declining. (Hallam et al., 2022) Simultaneously, the purity requirements for SOG-Si are becoming more stringent in newer generation of PV modules. The SOG-Si recovered from an older generation PV panel will more likely be reused in an alternate application (e.g., in concrete (Fernández et al., 2011) or in aluminum-silicon alloys for automotive and aerospace applications (Mathur et al., 2020)) or used as raw material for SOG Si production than be directly used in a new c-Si PV panel. (Heath et al., 2020) In either case, the environmental benefit will likely be lower than direct reuse in a crystalline Si PV panel as the functional properties and the high purity and embodied energy of the SOG-Si wafer is lost.

As a result, there is a need for the LCA of the CE to be temporally sensitive and continually account for technical improvements and material changes that occur between an older and a newer generation of a product system. When primary industry data is not available, approaches such as learning curves, (Hutchby, 2014; Rubin et al., 2015) industry-specific data, and patent data (Singh et al., 2021; Triulzi et al., 2020) can be leveraged to forecast and account for the improvements in technical performance and material use.

Quantifying displacement and rebound: It is a common assumption when modeling end-of-life in LCA that recycled materials displace primary production on a 1:1 basis. In reality, actual displacement is determined mainly by market forces, and secondary material could partially displace primary production, a material of a different kind (including a different recycled material), or no material at all. (Yang, 2016) In addition, material losses in the collection, purification and re-use processes may make a 1:1 substitution between secondary and primary materials infeasible.

The rebound effect also reduces the displacement of primary production wherein an increase in production or consumption efficiency – driven by CE activities such as recycling – is offset by economic effects that can increase production and consumption levels directly or elsewhere in the economy. (Zink and Geyer, 2017) For example, a rebound effect from imperfect substitution in the used-phones market was observed when 18 % customers in a survey used a refurbished phone as a second phone while keeping a new phone as their primary phone. (Makov and Font Vivanco, 2018)

Due to the above reasons, the full displacement assumption in LCA is largely invalid and systematically underestimates the environmental impacts of recycling product systems. (Zink and Geyer, 2017) Methods such as PEM (Zink et al., 2016) and CGE (Peng et al., 2019) have been applied to incorporate market-based information on displacement rate in environmental performance assessment of a CE. Quantitative guidelines have been proposed (Vadenbo et al., 2017) to systematize the reporting of assumptions on the displacement effect in LCAs, and CGE and PEM methods have been applied to quantify the displacement effect for recycling of individual materials (e.g., aluminum, (Zink et al., 2017) and copper (Ryter et al., 2022)). Despite these advances, further research is needed to assess the validity of these existing methods in PEM and CGE to a multi-product system and for non-recycling CE strategies (e.g., refurbish, remanufacture) wherein whole components and product systems are recovered instead of individual materials. Compared to a recycling only scenario, the number of price signals, policy incentives and entities participating in non-recycling CE scenarios (e.g., refurbishers, remanufacturers) increases which will further increase the burdens of data collection and may make the methods mentioned above economically and computationally prohibitive to account for the displacement effect. Moreover, additional complexities are introduced as market incentives for recovery and recycling of individual materials from product systems may conflict with policy incentives to recover the component or product system as a whole. As a result, there is a need for a robust and prospective assessment of the displacement and rebound effects for CE strategies beyond recycling and for whole components and products. Given that the data on the supply and demand for components and second-life products will most likely be confidential or proprietary, there is a need to explore analytical techniques that minimize data requirements (Difference-in-Differences (Palazzo et al., 2019)) and improve mechanisms to enhance trust with industry partners to make data available (Note: refer section titled "*Addressing data paucity and* *uncertainty*”).

Illustrative example of challenges in applying LCA for a macro-level CE

The challenges associated with applying LCA to evaluate the environmental performance of a multiproduct macro-level CE are depicted using an illustrative example in Fig. 2. The CE consists of five interconnected product systems: LIB, automobiles, the electricity grid, PV and CCU. At the end of functional life in a battery electric vehicle (BEV), the LIB can be repurposed for energy storage applications in a grid (R7: LIB to grid). (Ahmadi et al., 2014; Ahmadi et al., 2015) The material waste from manufacturing a PV panel (silicon kerf) can be recycled and used in the manufacture of LIB (R9: PV to LIB) (Wagner et al., 2019) or materials recycled from end-of-life of a PV panel can be reused in manufacturing a new PV panel (R9: PV to PV). (Ravikumar et al., 2016; Ravikumar et al., 2020) Similarly, materials recovered from recycling a spent LIB can be reused in manufacturing a new LIB (R9: LIB to LIB). (Redwood Materials 2022; Ecobat 2022)

The RE generated from a PV panel is utilized to recycle the CO_2 emitted from electricity generation from fossil sources into e-fuels, (Grim et al., 2022) which can be used as transportation fuel (R9: CCU to fuel). Moreover, the stakeholder behavior can contribute to the CE. Improved charging practices by the LIB user (i.e., the owner of the BEV) can help increase the lifetime of the LIB. (Woody et al., 2020) Thereby, the material requirement for energy storage over the lifetime of the LIB is decreased (R2: Improved charging). Similarly, the stakeholder can drive a product-as-a-service (PSS) through increased adoption of ride-share of automobiles (R1: Rideshare) and this behavior combined with increasing share of BEV in the vehicle fleet of rideshare providers (Uber 2022; Lyft 2020) can increase the demand for LIBs.

When conducting an LCA, the system boundary and the choice of an appropriate functional unit will have to account for multiple services being provided in the CE. This includes energy stored by LIB (kWh stored), electricity generated by the PV (kWh), transportation (passenger-miles), energy content of the e-fuel (MJ), the mass of the materials being recycled (kg), and the primary materials being displaced by the recycled material (kg).

Conducting an LCA to quantify the environmental benefit from recycling wherein materials recovered from PV are used in manufacturing new LIB (R9: PV to LIB) or PV (R9: PV to PV) will require material and energy inventory data for the recycling process and the types and purity of the materials being recovered. However, given that many PV recycling technologies are in the early stages of technology development, there will be a lack of data to conduct a robust LCA. (Heath et al., 2022; Heath et al., 2020) Moreover, there is a lack of comprehensive data on the waste flows from PV and LIB and the material composition of the various designs of PV and LIBs that are required to determine the environmental benefit from the recovering materials (e.g., the environmental benefit from offsetting primary material extraction processes). (Heath et al., 2022; Heath et al., 2020)

The LCA for the CE should also account for how stakeholder behavior can impact the environmental outcome. An increasing adoption of rideshare instead of personally owned cars (R1: Rideshare) and the increasing share of BEV in the fleet owned by rideshare providers can decrease transportation emissions. Similarly, the improved charging behavior of the owner of the BEV (R2: Improved charging) can increase the lifetime of the LIB and thereby decrease the environmental impact of the energy stored and delivered by the LIB over its lifetime.

The application of the LCA to this CE will need to account for the foregone environmental benefit from alternate uses of the RE being utilized for CCU. For example, the RE can generate a greater climate benefit if it is supplied to a fossil-fuel intensive electricity grid than when it is utilized for CCU. (Ravikumar et al., 2020)

The environmental impact of the illustrated CE will change with

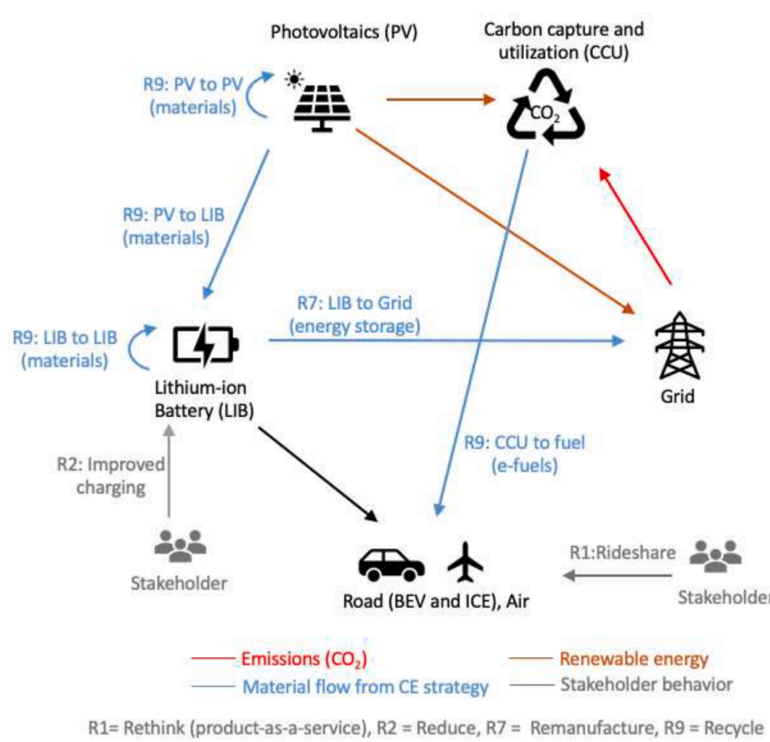

Fig. 2. An illustrative example of a macro-level CE consisting of five product systems: lithium-ion battery (LIB), photovoltaics (PV), carbon capture and utilization (CCU), the grid, and automobiles (battery electric vehicles (BEV) and internal combustion engines (ICE)). R9: PV to LIB – materials recovered from spent PV are used in manufacturing LIB. R9: PV to PV – materials recovered from spent PV are used in manufacturing PV. R7: LIB to Grid - LIB from electric vehicles are repurposed for energy storage on the grid. R9: LIB to LIB – materials recovered from spent LIB are used in manufacturing LIB. R9: CCU to fuel – CO_2 is recycled into e-fuels to be used as transportation fuel. R2: Improved charging –user’s charging behavior facilitates LIB longevity. R1: Rideshare – user behavior that increases adoption of ridesharing trips (product-as-a-service).

Table 1
Summary of the potential solutions and research priorities to address the emergent methodological challenges in applying LCA to CE at a macro-level.

Emergent methodological challenge in applying LCA to CE at a macro-level	Potential solutions/ Recommendations	Research priorities
Selecting a relevant system boundary and functional unit for a complex network of interconnected product systems in a CE	- select the system boundary and functional unit based on societal functions, which includes multiple composite functions (Alaerts et al., 2019; Harris et al., 2021) - leverage mechanistic models, (Vunnava and Singh, 2021) sectoral connections from EIOLCA or MRIO (Aguilar-Hernandez et al., 2018; Donati et al., 2020; Wiebe et al., 2019; Towa et al., 2021)	- collect and update data (primary material and energy inventory, secondary materials, component flows, waste flows) at different geographic resolutions and between product systems - characterize environmental impacts beyond GWP
Addressing data paucity and uncertainty	- aggregate data to conceal confidential information, (Kuczenski et al., 2016) - source data from industry organizations or third parties, (World Steel Association 2022; Athena Sustainable Materials Institute 2022) - strengthen public-private partnerships to obtain and publish inventory data (UNEP 2016) - leverage existing methods in LCA (e.g., global sensitivity, moment independent sensitivity analysis) to quantify the impact of data uncertainty on the results of the environmental performance of the CE - leverage the blockchain technology to promote trust, enhance knowledge sharing, and track material flows (Rejeb et al., 2023)	- develop standards to aggregate and conceal data and enhance trust for industry partners who are sharing confidential data. - support and fund initiatives (e.g., funding from public (Europian Union 2020; Environmental Protection Agency (EPA) 2022) and the private sector (Bocconi University; Ellen MacArthur Foundation; Intesa Sanpaolo 2021)) to foster partnerships between diverse stakeholders (e.g., government, industry, third parties, certifiers) (Schumacher and Green, 2021) to create data for CE strategies at different levels, which can be widely and freely accessed (data.europa.eu 2022) - utilize digital technologies (e.g., RFID to monitor product health (P et al., 2021; Deriche et al., 2019), internet of things for waste tracking (Magrini et al., 2021), and blockchain for knowledge sharing) (Rejeb et al., 2023) to create and share real-time data which can inform CE strategies
Accounting for stakeholder behavior	- leverage existing methods in socio-technical analysis like systems dynamics (SD), (Asif et al., 2016; Franco, 2019) agent based modeling (ABM) (Lieder et al., 2017) and discrete event simulation (DES) (Charnley et al., 2019) to determine how stakeholder interaction with a product system impacts the material and energy flows and CE strategies for other product systems - identify key stakeholders and behaviors having the most significant impact on material and energy flows and thereby the overall environmental impact of the CE for the multiproduct system	- address the limitation arising from additional data and validation requirements due to an increased number of product systems in a CE - develop and promote the adoption guidelines in socio-technical assessments of CEs to increase transparency when reporting assumptions on agent and their behaviors, shortcomings such as the limitations when generalizing insights and findings (e.g., across geographies)
Accounting for constraints in RE availability for CE strategies	- quantify the environmental trade-offs in using RE in CE versus non-CE applications. This can be based-on quantifying and ranking RE use based on the environmental benefit generated from alternate uses in different CE and non-CE applications. (D. Ravikumar et al., 2021; Katelhon et al., 2019; D. Ravikumar et al., 2020)	- prioritize RE use in CE strategies in which they generate a greater environmental benefit than other competing CE strategies or non-CE applications
Incorporating temporal sensitivity to account for technological improvements and change in materials for product systems	- quantify trade-offs between the life cycle environmental benefit of using a new product system with improved technical performance and extending lifetime of an older refurbished/ repaired/ remanufactured product system with a lower technical performance. - quantify the life cycle environmental impact from emerging open-loop pathways to reuse recovered materials in alternate product systems when closed loop recycling is not possible	- use of learning curves, (Hutchby, 2014; Rubin et al., 2015) industry-specific data, patent data (A. Singh et al., 2021; Triulzi et al., 2020) - monitor scientific and commercial literature to identify emerging open-loop pathways to use recovered materials in alternate product systems (e.g., reusing materials from lithium-ion batteries in catalysts, (Shen et al., 2019) lubricants) (Parikh et al., 2019)
Quantifying the impact of displacement and rebound effect	- leverage partial equilibrium models (PEM) or computable general equilibrium (CGE) to quantify the displacement effect. (Zink et al., 2017; Ryter et al., 2022)	- Extend CGE and PEM methods to a multi-product system and for non-recycling CE strategies (e.g., refurbish, remanufacture) in a macro-scale CE

technology evolution. For example, as the CO_2 intensity of the electricity mix of the grid decreasing across many regions, (Skone et al., 2019) the likelihood of realizing a greater climate benefit by utilizing RE for CCU rather than on the grid increases. Similarly, the opportunity to realize an environmental benefit from recycling silicon kerf loss from PV manufacturing into LIB (R9: PV to LIB) will decrease as the amount of solar-grade silicon being used in PV (Oreski et al., 2021) and the silicon kerf generated decreases with PV manufacturing improvements. (Kumar and Melkote, 2018; Henley et al., 2011)

In addition, an LCA to assess the environmental impact from increased adoption of ride share should account for potential reductions in the environmental benefit due to the rebound effect. Increased availability of ride-sharing options can result in a shift from a less to a more climate intensive transportation mode (i.e., public transport to ride-share in cars). (Coulombel et al., 2019; Yin et al., 2018)

Discussion

Research priorities to advance the assessment and enhance the environmental performance of a macro-level CE

The environmental assessment of a macro-level CE offers exciting opportunities for novel research to address the said emergent challenges. We summarize key emergent methodological challenges, recommendations, and research priorities to improve the environmental sustainability of the CE at a macro-level in Table 1.

Conclusions

The CE can be implemented at three levels – micro-level, meso-level and macro-level. The application of LCA to evaluate the environmental impacts of a CE, has been largely limited to the micro-level. In this work, we have identified six emergent methodological challenges that limit the application of LCA to evaluate the environmental performance of a meso- or macro-level CE. These challenges include - selecting a relevant system boundary and functional unit for a network of product systems, addressing data paucity and uncertainty, accounting for stakeholder behavior, assessing the trade-offs from RE use, accounting for manufacturing and technology evolution, and quantifying displacement and rebound.

Selecting the system boundary and functional unit based on societal functions, which includes multiple composite functions and leveraging mechanistic models, sectoral connections from EIOLCA or MRIO can better help applying LCA to a macro-level CE. Data paucity can be addressed by sourcing data from industry organizations or third parties, aggregating data to conceal confidential information, leveraging technology platforms (e.g., the blockchain technologies) to enhance knowledge sharing and track material flows. The impact of data uncertainty on the results of the environmental performance of the macro-scale CE can be quantified by leveraging methods available in global sensitivity analysis. Adopting existing methods in socio-technical analysis like SD, ABM and DES in LCAs can help to determine how stakeholder interaction with the multiple product systems impacts the various material and energy flows in a macro-level CE strategy. The various RE uses can be ranked based on the environmental benefit generated in the various CE and non-CE applications (in which the RE can be potentially utilized). RE use in a macro-level CE is environmentally justified only if it generates a greater environmental benefit than in other competing non-CE applications. To account for manufacturing and technological improvements, the LCA for macro-level CEs should quantify trade-offs when an older product (e.g., refurbished) with a lower technical performance offsets a new product with an improved technical performance. Available methods such learning curves can be leveraged to prospectively simulate technological improvements in products when conducting an LCA of a macro-level CE. To better account for the displacement effect in an LCA for a macro-level CE, we recommend leveraging and extending existing methods in leverage partial equilibrium models (PEM) or computable general equilibrium (CGE).

It is important to note that the proposed solutions to advance LCA need to be complemented with analysis to account for other equally important considerations when transitioning to a macro-level CE – economic and social impacts, (Calzolari et al., 2022) resilience (Kennedy and Linnenluecke, 2022) and risk. (Ethirajan et al., 2020) Economic and social impacts are equally relevant as environmental merit when implementing a macro-level CE. (Niero et al., 2021) Furthermore, conflicts may arise when the social, economic and environmental benefits and costs are incurred inequitably by different stakeholders in the macro-level CE. For example, given that current supplies of RE are constrained, the use of RE for CCU is economically beneficial for manufacturers who generate products from the captured CO_2 but may prevent stakeholders in the electricity sector from realizing an environmental benefit by decarbonizing the grid.

A topic for future research is to identify mechanisms to anticipate and minimize conflicts that may emerge in a macro-level CE and, thereby, foster collaboration across multiple stakeholders with diverse interests. (Köhler et al., 2022; Calicchio Berardi and Peregrino de Brito, 2021) To this end, methods in multi-criteria decision analysis (MCDA) (Deshpande et al., 2020) can be leveraged to help stakeholders anticipate and explore the economic, environmental and social benefits and costs they will incur when participating in a macro-level CE. (J. Walzberg et al., 2021)

Another topic for future work is developing metrics and methods to measure the risk (Yang and Li, 2010; Li et al., 2017) and resilience (Fraccascia et al., 2017; Chopra and Khanna, 2014) impacts for increasingly complex network of entities participating in the macro-level CE. This is especially important as there is a lack of consensus and conclusive evidence if risk reduces and resilience improves (Kennedy and Linnenluecke, 2022; Gebhardt et al., 2022) for the various firms and supply chain entities participating in a macro-level CE. For example, a macro-level CE can increase the exposure to supply shocks (e.g., from extreme weather events, pandemics) (Fraccascia et al., 2020) for manufacturers who have pursued strategies to reduce material redundancies (e.g., by closing material loops) and, thereby reduced diversity in their suppliers (e.g., a virgin raw material producer).

CRediT authorship contribution statement

Dwarakanath Ravikumar: Conceptualization, Methodology, Formal analysis, Investigation, Writing – original draft, Writing – review & editing, Visualization. **Gregory A. Keoleian:** Conceptualization, Methodology, Writing – original draft, Writing – review & editing, Visualization. **Julien Walzberg:** Writing – review & editing. **Garvin Heath:** Writing – review & editing. **Martin C. Heller:** Conceptualization, Writing – original draft.

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.

Data availability

No data was used for the research described in the article.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.rcradv.2024.200203.

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