An instrumental value-based framework for assessing the damages of abiotic resources use in life cycle assessment

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Abstract

Introduction & literature review. Abiotic resources are extensively used in industrialised societies to deliver multiple services that contribute to human wellbeing. Their increased extraction and use can potentially reduce their accessibility, increase competition among users and ultimately lead to a deficit of those services. Life cycle assessment is a relevant tool to assess the potential damages of dissipating natural resources. Building on the general consensus recommending evaluating the damages on the instrumental value of resources to humans in order to assess the consequences of ressources dissipation, this research work proposes a novel conceptual framework to assess the potential loss of services provided by abiotic resources, which when facing unmet demand, can lead to a deficit to human users and have consequences on human wellbeing.

Results. A framework is proposed to describe the mechanisms that link human intervention on the resources in the accessible stock to competition among users. Users facing the deficit of resource services are assumed to have to pay to recover the services, using backup technologies. The mechanisms that are proposed to be characterized are dissipation and degradation. Data needed to later operationalize the framework for abiotic resources are identified. It also proposes a framework at the life cycle inventory level to harmonize life cycle inventories with the current impact assessment framework to fully characterize impacts on resource services. It regards ensuring mass balances of elements between inputs and outputs of life cycle inventory datasets as well as including the functionality of resource flows.

Discussion and Conclusions. The framework provides recommendations for the development of operational life cycle impact assessment (LCIA) methods for resource services deficit assessment. It establishes the impact pathway to damage on the area of protection "Resource Services", data needed to feed the model and recommendations to improve the current state of life cycle inventories to be harmonized with the LCIA framework.

Keywords: resource services, functionality, dissipation, occupation, LCIA, MFA

1. Introduction

There is a growing concern about abiotic resources dissipation and our current unsustainable use of resources (de Wit et al. 2020; Charpentier Poncelet et al. 2022b; Graedel and Miatto 2022). Life cycle assessment (LCA), framed by an international standard (ISO 2006a, b), is increasingly used by decision-makers to determine paths toward sustainability (Sonnemann et al. 2018; Sala et al. 2021). It accounts for all the activities across the entire value chain of products and quantifies the flows of substances emitted into the environment or resources used, the so-called elementary flows, related to a functional unit. There is a growing consensus in the LCA community that the resources dissipation has to be characterized in term of loss of resources instrumental value (Verones et al. 2017; Sonderegger et al. 2019; Berger et al. 2020). The degradation of resources properties (without loss of matter) that leads to a loss of instrumental value, e.g. through the contamination of tramp elements in alloys (Reijnders 2016; Daehn et al. 2017; Fayomi et al. 2017; Daigo et al. 2021), should also be characterized.

However, currently no operational impact assessment methodology allows to quantify adequately the deficit of resources instrumental value due to resources dissipation and degradation (see the review of Berger et al. (2020) complemented with our analysis of recently published methods in section 2.4 and supporting information of this article). Most of the existing models characterize resources extraction (Berger et al. 2020). As further developed in section 2.4, there is a lack of consistency between most life cycle impact assessment models to evaluate the anthropocentric instrumental value deficit due to the dissipation of the abiotic resources.

The aim of this article is to develop a conceptual framework to assess the deficit of the instrumental values of resources and its consequences on human wellbeing. To reach it, we need to:

- Describe the cause-effect chain for the assessment of the potential deficit of services provided by abiotic resources (impact assessment framework);
- Propose an approach to map all the services provided by each resource and all the resources providing each service (with equivalency factors between resources providing the same service);
- Propose a harmonized life cycle inventory framework with a compliant interface with the impact assessment framework.

2. Literature review

2.1 The anthropocentric instrumental values of resources

The quantity and diversity of consumed fossils, metals and non-metallic minerals significantly increased in less than a century (Greenfield and Graedel 2013; Carmona et al. 2017; Fernandez 2017; Krausmann et al. 2017, 2018). These resources can be managed by using different strategies. The first strategy focuses the efforts on reducing utilities required for transformation of resources—such as water and energy—while the second regards the efforts for reducing the quantity of resources used, by designing recyclable products or products with a longer lifespan

(Carmona et al. 2017). These two strategies do not regard the contribution of resources to human wellbeing. If we want to do so, it is necessary to understand the purpose behind the extraction and transformation of resources. The third strategy goes beyond the paradigm of seeing resources (or materials) as products and focuses on the optimisation of the provision of the services behind them (Carmona et al. 2017). Material services can be defined *as those benefits that materials contribute to societal wellbeing, through fuels and products (regardless of whether or not they are supplied by the market) when they are put to proper use* (Carmona et al. 2017). Kalt et al. (2019) distinguish energy services from energy carriers, such as mobility vs. transport fuels or illumination vs. electricity. Energy carriers are abiotic resources, such as fossil fuels directly used as transport fuels or metals used in the infrastructure to produce electricity, e.g., in solar panels. Whiting et al. (2021) proposed a resource service cascade framework that expands the relationship between resource use and their contribution to human wellbeing (HWB) beyond solely energy services. This framework considers that resources interact with ecological and socioeconomic processes and provide them a potential service, also called a function. Once the latter benefits a human end-user, it is recognized as a resource service (RS) contributing to HWB. The link established by Whiting et al. (2021) between satisfied needs and their contribution to HWB is aligned with the definition of HWB by Breslow et al. (2016) as *a state of being with others and the environment, which arises when human needs are met […] and when individuals and communities enjoy a satisfactory quality of life*. The concept of resource service thus regards the utility of resources to humans, i.e., their instrumental values (Sonderegger et al. 2017). The latter depends on the level of functionality of resources (Sonderegger et al. 2017) to fulfill a service to humans. Resources services deficit is not directly proportional to resources dissipation as some services can be fulfilled by a single specific resource only (for example phosphorous cannot be replaced for fertilization in agriculture) whereas other services can easily be fulfilled by alternative resources (for example copper can easily be replaced in plumbing lines), meaning a systemic model is needed to understand to what extent the dissipation of a resource contributes to a service deficit. The framework for such a model is developed in the present paper.

2.2. Main features: How are resource services impacted?

Resource services can be affected by multiple mechanisms, either by direct human interventions such as the addition of alloying elements (Reijnders 2016; Fayomi et al. 2017) or by physical mechanisms such as corrosion (Cinitha et al. 2014). More generally, a loss of functionality could result from the modification of some key properties of the resource, which results in a loss of instrumental value that will negatively affect human wellbeing if human needs are unmet. As an example, aluminium cast alloys cannot be used in aerospace applications (Rambabu et al. 2017; Van den Eynde et al. 2022). Resource services can also be restored. As an example, aluminium cast alloys can be upcycled to primary aluminium (Lu et al. 2022), which can then be used in aerospace application (Van den Eynde et al. 2022). Resources can also end up in sinks or stocks that are not accessible to future users due to geopolitical, social, economic, technical and environmental constraints and can then be considered as being dissipated (Beylot et al. 2020b; Dewulf et al. 2021). Resources in-use can also lead to resources services deficit. While in-use, resource services are provided to human users (Pauliuk and Müller 2014; Haberl et al. 2017, 2019). However, at the same time, these resources are not available to other users (Beylot et al. 2020c).

Competition between human users occur when the demand exceeds the supply coming from the accessible stock. Such competition happened during the rare earth elements crisis in 2011 (Fernandez 2017). Users of resources facing the inaccessibility of the services fulfilled by resources try to adapt by e.g., improving recycling, opening new extraction sites or substituting the inaccessible resource with functionally equivalent resources, as it was done during the rare earth elements crisis (Sprecher et al. 2017). This illustrates the fact that the transformation and use of resources contribute to the loss the services they provide to humans and may lead to unmet demand, i.e. resources services deficit. Therefore, it is key to develop tools to guide decision-making regarding the assessment of the accessibility of resource services, in order to sustain and potentially maximize human wellbeing.

2.3 How are resource services considered in life cycle assessment?

The life cycle impact assessment (LCIA) stage characterizes the potential impacts on the so-called areas of protection through multiple environmental pathways of elementary flows emitted or extracted all along the life cycle and aggregated in the life cycle inventory (LCI). An area of protection (AoP) defines what safeguard subjects we would like to sustain or protect and which impacts should be assessed and modeled (Bare and Gloria 2008).

In LCI databases, only extraction flows from the ecosphere into the product system boundary within the technosphere and emission from the latter back to the ecosphere are modelled. The quantity moving from one sphere to another is then analyzed disregarding the functionality of the substance. Therefore, most LCIA models only characterize extraction flows from the ecosphere, which contributes directly to the depletion of geological stock. However, this does not adequately represent the loss of resources services as the resource may still be accessible for future uses once it is in the technosphere. Many authors consider dissipation as being the actual cause of resource inaccessibility, but most models fail to characterize it adequately (Berger et al. 2020). Beylot *et al.* (2020b) defines *dissipative flows of abiotic resources [as] flows to sinks or stocks that are not accessible to future users due to different constraints. These constraints prevent humans to make use of the function(s) that the resources could have in the technosphere. The distinction between dissipative and non-dissipative flows of resources may depend on technological and economic factors, which can change over time*. In this regard, the approaches that consider dissipation require differentiating elementary flows in the life cycle inventory implying additional efforts in data collection and harmonization with LCIA (Zampori and Sala 2017; Beylot et al. 2020b). Beylot et al. (2020c) suggested, in a time horizon of 25 years, that *any flow of resources to i) environment, ii) final waste disposal facilities and iii) products in use in the technosphere (with low functionality) shall be reported as dissipative*. Owsianiak et al. (2021) proposed an approach based on the resulting concentration of the dissipative flow in an environmental compartment (air, soil, water) to qualify whether a resource is dissipated or not (using USEtox 2.1 to quantify if the concentration in the receiving soil compartment is higher or not than with the average element concentration in the upper continental crust). Other parameters could be considered such as economic recovery costs, entropy parameters (Beylot et al. 2020b) and accessible energy to extract the resources.

Today there is a consensus to evaluate the anthropocentric instrumental value of resources to humans under the "Natural resources" AoP in the LCIA framework (Verones et al. 2017; Berger et al. 2020). This consensus emerged within the UNEP Life cycle initiative from an extensive debate during the last decades and builds on an initial proposition from Sonderegger et al. (2017) suggesting to move from the intrinsic value – defined by Verones et al. (2017) as the value given for the sake of the existence of resources in itself – to the instrumental values – defined by Verones et al. (2017) as the value that resources have because of their utility to human. The protection of instrumental value of resources is also seen as the relevant approach in the management of primary raw materials (Schulze et al. 2020a, b). Beyond the life cycle initiative consensus, Charpentier-Poncelet et al. (2022a) introduce an additional nuance to the notion of resources instrumental value by distinguishing the exchange value (i.e. the economic value in the technosphere) from the use value, both being part of the instrumental value of resources. It can also regards the non-use values (Turner et al. 2003) such as heritage value, that are intangible (Kareiva et al. 2011). However, until very recently, LCIA models focusing on resource inaccessibility did not take into account the instrumental value of resources for humans (Sonderegger et al. 2019; Berger et al. 2020).

2.4 How do current LCIA methods address the instrumental value loss of natural resources?

Several modeling perspectives can be identified from the literature to characterize the life cycle impact assessment of resource inaccessibility as highlighted by different literature reviews (Klinglmair et al. 2014; Rørbech et al. 2014; Dewulf et al. 2015; Drielsma et al. 2016; Sonderegger et al. 2019; Pradel et al. 2020; Charpentier Poncelet et al. 2022a). We conducted a review (available in Supporting information 1) on the consistency of those LCIA methods to evaluate the anthropocentric instrumental value of the natural resources as per the LCIA framework of Verones et al. (2017). Hereby, we analyze the innovative methods that either characterize dissipative flows or integrate aspects such as resources services demand or resources substitution (or both) in more detail.

Steen (1999, 2016) proposed to characterize resource extracted and emitted into the environment on the basis of the market price of potential substitutes. Then, this method aimed at characterizing the cost to pay by human users to access to the functionality of the resource. However, the substitution of the resource was limited to one service of the resource (e.g. energy for fossils) and the demand of those services is ignored.

De Bruille (2014) proposed an exploratory approach to assess the loss of functional values of dissipated abiotic resources related to competition between human users. The demand of resources was integrated over an infinite horizon of time, using a constant resource demand based on the 2003-2012 period. The substitutability of each resource is integrated into the model (is the resource substitutable?) but not the substitution (by which resource is it substituted and in which quantity?). Therefore, the implementation of substitution is flawed since no mass-based equivalency between resources for a given function is used. The method does not account for the increased consumption of substitutes. Moreover, the characterization factors (CFs) of some resources are aggregated, e.g., Rare Earths Elements (REEs) have the same CF. Those CF are applied to the extraction flows, but with an underlying assumption that a fraction of this flow will be recycled hence not dissipated (using the average recycling rate based on the 2003-2012 period).

The environmental dissipation potential (EDP) method (van Oers et al. 2020) has been applied on emissions of resources to the environment instead of extraction flows, therefore it characterizes dissipation. However, their characterization factors are based on extraction rate (of 2020) and not on dissipation rate, assuming "that primary extraction at present equals the very-long-term emission to the environment". This equivalence between extraction and long-term dissipation is debatable. Dewulf et al. (2021) considers that in-use stocks are not available to other users which lead to additional extraction.

The Lost Potential Service Time (LPST) and the Average Dissipation Rate (ADR) methods (Charpentier Poncelet et al. 2021, 2022b) are based on the dissipated quantity of a resource that, as a result, no longer provide any service to humans. The approach of Charpentier Poncelet et al. (2021) takes a material pool perspective and uses dynamic MFA to estimate average dissipation rates and lost potential service times. It is applied to the extraction flows of the LCI and thus does not differentiate the dissipation of the specific product system under study. The LPST method translates the impact of dissipation into a potential loss of service over a given period of time while the ADR method assesses the average quantity that would need to be extracted every year to compensate the dissipation of the resource occurring over the same year. These two latter methods do not enable the assessment of the consequences of dissipation on resource services deficit, as they are not linked with the demand of resource services.

Beylot et al. (2020a) and Charpentier-Poncelet et al. (2022c) proposed to characterize the impact of dissipated flows of resources on the basis of their market price. However, even if there are numerous examples of metals with high values in terms of the functions they provide, which are correlated to the high prices that economic actors are willing to pay for these functions, Watson and Eggert (2020) stated that the market price is only partly connected to the consumption and functionality of resources ("*The results suggest that energy requirements in production explain 43% of observed variation in metal prices, crustal abundance 21% (…)*").

Even if recent progress were done to assess impacts of resources dissipation, none of these methods allows yet to consider in a consistent way the resources use in the technosphere (and their substitutability for some services), their dissipation, the degradation of their quality and the way all those mechanisms together influence the deficit of services – even though these mechanisms are well documented in the literature (Gerst and Graedel 2008; Nakajima et al. 2010; Nakamura et al. 2012; Ciacci et al. 2015, 2016; Reijnders 2016; Daehn et al. 2017; Watari and Yokoi 2021; Tonini et al. 2022).

3. Results: Resource services deficit assessment framework

The proposed framework complies with the definition of the AoP of "Natural Resources" from the UNEP life cycle initiative by Verones et al. (2017) as "the instrumental values of natural resources" and by Berger et al. (2020) as "the potential to make use of the value that mineral resources can hold for humans in the technosphere", which it aims at operationalizing. We decided to rename this AoP "Resource Services" (RS) in order to explicitly exclude the intrinsic value of resources. We consider, in the present framework, that the instrumental values of natural resources correspond to the services provided by resources both in the ecosphere and in the technosphere to humans. Therefore, we consider the AoP "Resource Services" as a subset of an AoP "Human Wellbeing". This framework is built for all abiotic resources i.e., metals, fossils, non-metallic minerals, all considered being resource (single resource of the earth crust or aggregated resources into materials). It is fundamental to include materials, such as alloys, since their functionality depends on their chemical composition (see e.g. end uses of different aluminum alloys in Van den Eynde et al. (2022)).

Building on the framework of Stewart and Weidema (2005) and the consumption-competition-adaptation approach of De Bruille (2014), we develop a framework for assessing the life cycle impacts of resource consumption in terms of the cost for society to recover the service lost. A glossary is developed to facilitate the understanding of our framework and avoid confusion on different terms since there is no consensus on their scope and definition in the literature.

Glossary

The reserve base and the anthropogenic accessible stock (intermediate grey boxes in figure 2) are defined as the stocks of resources that are already accessible or may become easily accessible soon using economically viable technological improvements, in both the ecosphere and the technosphere respectively, using current technologies and without additional extraction costs. This definition builds on the definition of reserve base of van Oers et al. (2002), which is limited to the ecosphere, by also encompassing the resources within the technosphere. The anthropogenic accessible stock may include, in particular, a fraction of the hibernating stock (defined by Schneider et al. (2011) as the amount of resource that is not used anymore but has not been discarded yet) and the landfilled material that may become economically recyclable, but it excludes the stock in use (not accessible as long as it remains in use).

Remarks: We use the word "accessible" and not "available" since accessibility is located at the intersection of the concepts of "availability" and "approachability", the latter being defined as the ability of being easy to deal with (Mueller et al. 2017). *Therefore, the approachability depends on several exogenous parameters such as technological improvement (TI) and geopolitical, social and environmental constraints (GSEC). TI can enable the use of a larger amount of the functional resource without additional extraction costs. The reserve base can also change with GSEC such as a lack of social acceptability of resource extraction and transformation (10.3% of mining projects reported in the Environmental Justice Atlas were cancelled due to socio-environmental conflicts* (Scheidel et al. 2020)*) or the constraint on environmental externalities* (Jowitt et al. 2020) *such as greenhouse gas emissions (see, e.g., the case study of Watari et al.* (2021)*). Hence the approachability evolves in time and there is to define a time horizon to be able to identify the part of the resources that have a reasonable potential for becoming accessible within this "planning horizon".*

The ultimate stock (darker grey boxes in figure 2) corresponds to the amount of a resource that is ultimately available in both the ecosphere and the technosphere. The original definition of ultimate stock (Guinée 1995; Schneider et al. 2015), i.e., the total estimated amount of a resource in the Earth crust, is expanded to encompass the total amount of resource in the technosphere as well. The ultimate stock includes the reserve base.

A backup technology is a technology that enables the recovery or extraction of a resource from the ultimate stock, but requires additional costs compared to the costs associated with extraction from the reserve base. This definition agrees with the proposition of Stewart and Weidema (2005).

The dissipation of a resource happens when a resource is widely dispersed or confined in the ecosphere or the technosphere, and the current available technologies and prospects do not allow for its easy recovery, thus having access to a sufficient amount of energy, within a reasonable timeframe. This means that the dissipation corresponds to the loss of resource accessibility. The dissipation can occur over the entire life cycle of the resource. All of the resource services are considered to be lost with dissipation.

Remark: The terms "easy recovery" and "reasonable timeframe" may seem vague, mainly due to absence of consensus in the literature of how to distinguish the reserve base from the ultimate stock. This fuzzy boundary is discussed in section 4.1. Our definition of dissipation is compliant with the one of Beylot et al. (2020) since they consider that dissipative flow corresponds to "flows to sinks or stocks that are not accessible to future users due to different constraints » (Constraints which may be economical or technological) : this corresponds exactly to what we define as the loss of accessibility.

The restoration of a resource qualifies a functionality gain (i.e., the inverse of resource degradation).

The degradation of a resource qualifies its **functionality loss**, i.e., the loss of its potential to fulfill one or several services. Resource degradation can potentially reduce the provision of services it provides. It corresponds to a loss of performance, to the reduction in some (or all) of the key properties of the resource that is functional for a given service. Part of the instrumental value is then considered to be lost.

Remark: The loss of functionality without necessarily a loss of matter was addressed by Stewart and Weidema (2005) but not defined as degradation. Our definition of quality loss greatly differs from the one of Steinmann (2019), *which defines quality as the ratio between energy demand for secondary production and primary production, i.e., the relative energy effort.*

The downcycling of a resource regards a resource reprocessed from waste that is suitable for fewer services than that it was potentially able to provide before.

Remark: This definition is inspired from the definition of functional downcycling of Helbig et al. (2022).

3.1 Cause-effect chain to assess the consequences of the use of resources on the services they provide

The framework proposed in this article evaluates the potential impact of the deficit of services provided by abiotic resources rather than the inaccessibility of the resources themselves. As depicted in Figure 1, assessing the deficit of services provided by the resource to humans implies a cause-effect chain that links a human intervention on a resource to a potential damage on one AoP. Human interventions regard transfer of a stock (accessible or ultimate stock) to a product system or from a product system to a stock, which leads to environmental impacts because of dissipation, degradation or restoration. The fate links the human intervention on resources to a loss of service that leads to an increased competition between users. The demand of resource services is integrated in the fate model framework. The exposure model links competition with resource service deficit, i.e. an unmet demand of services for unadapted users. It is assumed that unadapted users have to pay the cost to adapt to the deficit of service.

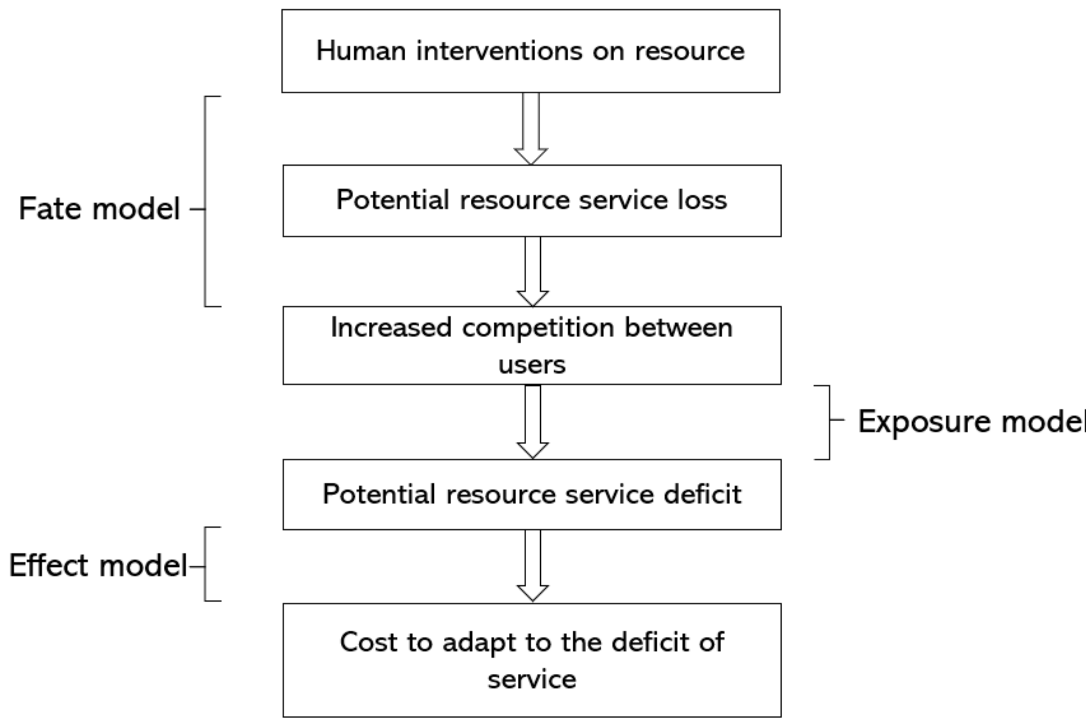

Figure 1. Conceptual framework describing the cause-effect chain assessing the loss of services provided by resources, leading to a deficit for users and additional cost to pay to recover the service.

3.1.1 The fate model framework: linking human interventions to increased competition between users

The fate model assesses the extent to which human interventions affect the competition between users for the accessible stock, represented by the reserve base plus the anthropogenic accessible stock. Resources in the accessible stock are functional for different services, therefore the reserve base is a stock of services. Different subsets of the accessible stock correspond to various levels of functionality of a resource; a "high quality" subset of the accessible stock can be used to fulfill all services (high functionality) while another subset of lower functionality is likely to fulfill fewer services.

To assess the loss of a potential service i (instead of the inaccessibility of a resource A) one needs to evaluate the inaccessibility of the subset of the reserve base of resource A that is functional for service i. One also needs to consider which other accessible resources can deliver service I and therefore easily substitute resource A (i.e. is accessible at an equivalent cost to fulfill service i). In figure 2, the accessible stock subsets of resources A and B, both functionals for service I, are represented by the white boxes. It represents the total accessible stock for service i. All human interventions influencing this total easily accessible stock for service i (white boxes) are shown as arrows on figure 2 and are further described below. Any marginal additional human intervention (arrows) may contribute to increasing or decreasing the accessible stock of service i (sum of white boxes) in the future. This requires integrating the demand of resources services. Our fate model framework allows modeling this marginal contribution linking an intervention to an increase in competition between users, in regard to the demand of those services.

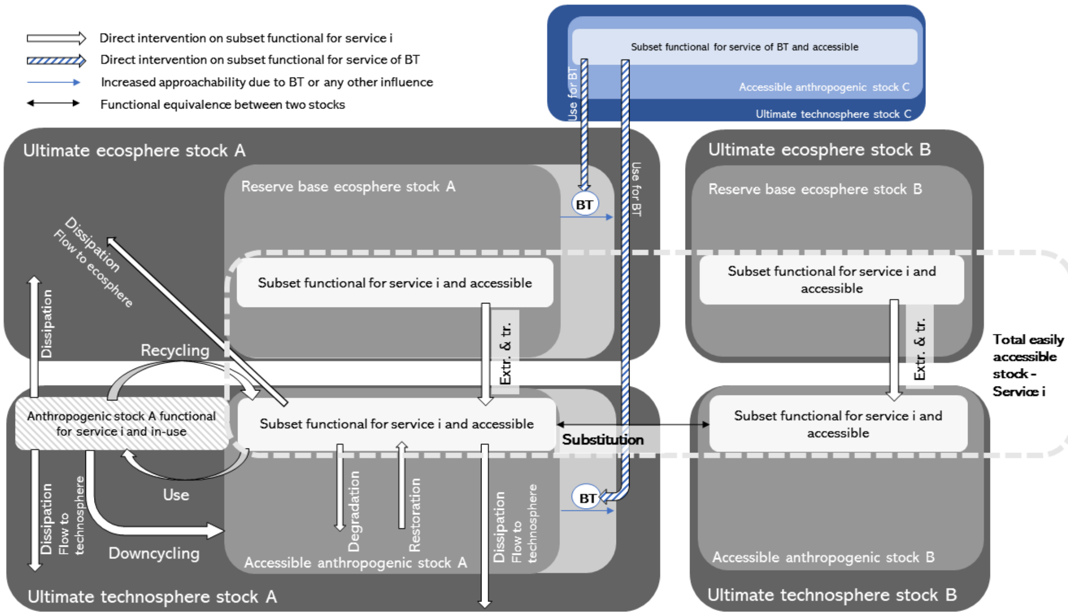

Figure 2. Fate (in grey) and exposure/effect (in blue) model framework linking human interventions on resources A, B and C (dissipation, degradation, restoration, etc. are represented by white arrows) to the increase in competition between users (fate) and the subsequent potential deficit of service i leading to the use of a backup technology (exposure/effect). The white, black and blue arrows are only presented for resource A for simplification, but also apply similarly to resources B and C. Extr. & tr. = Extraction and transformation; BT = backup technology. See text below for explanation.

The reserve base stock functional for service i can be reduced by dissipation or degradation. A resource returning to the technosphere reserve base with a lost functionality is considered as downcycled, which is degradation occurring at the end of the use phase[1]. When all the resource functionalities are lost, it is dissipated, i.e. no longer accessible for any functionality. Dissipated flows to the ecosphere can end in soil, air or water. Dissipation can also occur within the technosphere, e.g., alloying elements that are unrecoverable. The dissipation of a resource A into another resource B, through contamination, can also lead to the degradation of the contaminated resource. As an example, an excessive copper contamination in steel prevent steel use in reinforcing bars in buildings (Daehn et al. 2017). The degraded reserve base functional for service i cannot provide the service i but can still be functional for other services. The amount of the resource entering into use in a product system is considered as being part of the ultimate stock. While it contributes to providing service i for the users of the product system, it also deprives other users from using it for other services. The subset of the reserve base functional for service i can also be functional for other services and then be used by other users (a more detailed

[1] For simplification in figure 2, the functionality is focused on the service i. However, the subsets could be functional for other services, then that is why the downcycling arrows comes back to the reserve base (only the service i is lost but not all the other services).

representation of this share of resources between services is available in supporting information 2).

When there is competition for resource A (i.e. there is not enough of A for all the A users) and there is a resource B accessible for the and functional for the service i (i.e. available at an equivalent cost than resource A for this specific service) substitution between the two resources may occur to supply the service i. One needs to determine the amount of resource B that is needed to replace resource A for the exact same service i. The mass ratio of resource B over resource A is termed the functional substitution factor. If substitution is not possible (or if the demand for i exceeds the subsets of all reserve base of all the alternative resources functional for service i), the service i is lost (i.e. the users of resource A for service i will be considered as unadapted deprived users once the total easily accessible stock for service i will be depleted). Some services may require a combination of resources instead of a single resource. In such a case, the service i is lost as soon as the substitution is no longer possible for one of the resources needed. Hence the fate model allows to calculate dynamically at which moment in the future the service i is lost and which fraction of the users of service i will be deprived at that moment.

3.1.2 The exposure model framework: From competition to resource service deficit

Ideally, the exposure model would calculate the share of the unadapted deprived users that are unable to pay or to have access to resources for the backup technology (i.e. currently loosing a service) and the share of users able to adapt through a backup technology (i.e. the recovery cost to the service deficit). Nonetheless, this exposure is likely to happen far in the future. Then, it is very challenging to model a realistic evaluation of the share of unadapted users unable to recover a service lost. Therefore, we decided to use the simplified assumption that all unadapted users facing deficit of service will potentially adapt and pay for recovery costs (section 2.1.3), hence considering a 100% exposure for this impact pathway. Therefore, our model will assess the cost society would have to pay to maintain 100% of services to protect the human wellbeing (and not the real consequences of resource services deficit, which will be in many situations an absence of adaptation and a loss of wellbeing).

3.1.3 The effect model framework: From the potential service deficit to recovery cost of resource service lost

The effect model assesses the cost that society should pay to recover a lost service using a backup technology (BT), but also other indirect consequences on the accessibility of other resources, like resource C in figure 2, which are needed for the backup technology. Building on the approach of Stewart and Weidema (2005) – but considering as "backup technologies" all the intermediary technological increases in intensity between the current one and what Stewart and Weidema define as the backup technology – we consider that different BTs will be required to compensate for the deficit of services. This will allow, by combining fate, exposure and effect, to assess the service deficit in term of additional cost the society has to pay to adapt to this deficit and maintain human wellbeing. As an example, downcycling of aluminium 6000 series (Al6000) to cast alloy (e.g. 319) through a high contamination of silicon can result in a potential loss of telecommunications service (e.g. provided by a satellite since Al6000 are used in aerospace applications (Van den Eynde et al. 2022)). If the accessible stock is not sufficient to meet the demand, the additional cost for the society has to pay can either be to restore other subsets of reserve base (e.g. through upcycling) or to extract bauxite in the ultimate stock and transform it into Al6000.

Remark: There is a feedback loop between the exposure and effect assessment of resource A and the fate model of resource C that will be used in the backup technology (backup technologies are not only more costly in term of money but may be more intensive in term of resources and energy use, hence influencing the deficit of other resource services).

3.2 Identification of end-uses and services of resources for humans in the technosphere

The evaluation of the deficit of resource services requires the exhaustive identification of services fulfilled by each resource (e.g. is service i equal to mobility, storing energy or conditioned a living space?). It is also necessary to specify at which level of resolution the service has to be defined. Material Flow Analysis (MFA) or Substance Flow Analysis (SFA) are tools that may be used as data sources to identify the end-uses of resources (Brunner and Rechberger 2016; Graedel 2019). Then, the resource service classification of Whiting et al. (2021) can be used to map final end-uses (e.g. vehicles) with services (e.g. mobility). A service answers to human needs and has benefits for human wellbeing. Once the services associated with each resource are listed, mapping with human needs can be established, i.e., subsistence, protection, creation, identity, affection, participation, understanding, leisure and freedom, using the human needs matrix (Max-Neef and Ekins 1992). Figure 3 illustrates the links between resource use and answer to human needs.

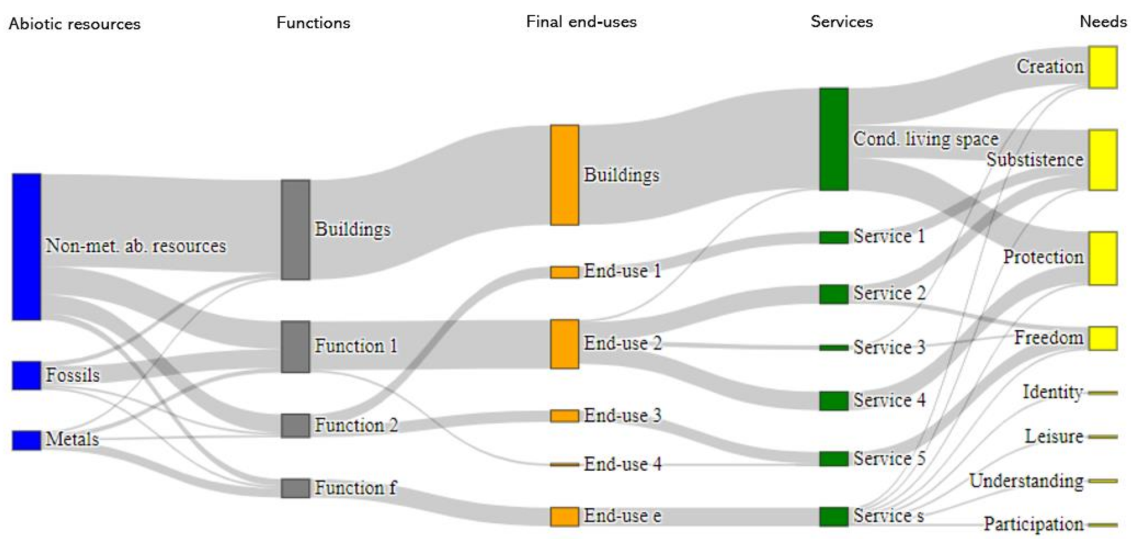

Figure 3. Illustrative example of need-service-function relations of abiotic resources. Flow values are fictive. Cond. = conditioned.

The quantification of a service fulfilled by a resource (or a material) is based on the performance of the resource (or material) in the key properties. A material, like an alloy, is made of several resources that contribute to the performance in the key properties. The key properties can be of different natures (electrical, chemical, mechanical, acoustic, thermodynamic, etc.). As an example, aluminium is used in different components of a vehicle such as chassis, engine, doors and exhaust parts. A specific quantity of aluminium is required to fulfill the technical performance of a given component ($kg_{Al}/component_i$). This (and other components) is required to build a vehicle ($\sum components_i/vehicle$) that fulfill the service of mobility (vehicle/person*kilometer). By quantifying the amount of aluminum needed per each component of a vehicle and the lifetime of the vehicle (and its occupancy) one can determine how much aluminium is required to fulfill a mobility service (kg_{Al}/person*kilometer). A similar reasoning can be applied for alloying elements ($kg_{alloying\_element}/kg_{Al}$) and all other elements within the different components of a car.

Some resources or materials have unique properties for given services for which no alternatives are available. In such specific cases, it is necessary to distinguish service fulfillment at the service level and not at the end-use level. As illustrated in figure 3, end-use 1 only fulfills service 1. Any resource or material functionally usable for end-use 1 will fulfill service 1, so the equivalence between resources or materials can be done at the end-use level. However, end-use 2 can fulfill three different services. If all resources or materials used in end-use 2 are functionally equivalent for services 2, 3 and 4, then we can calculate equivalence at the end-use level. If not, the distinction should be done at the service level. Hence, the resolution at which we need to go to evaluate the substitutability varies depending on the service considered.

3.3 Harmonized interface between LCIs and current framework

In our framework, we consider the accessible stock to consist in a combination of anthropogenic and geological stock. This means that extraction flows from the ecosphere to the technosphere are not sufficient to be able to quantify the impact of resource use. We also need to consider some economic flows and to consider that materials are part of our resources stocks. To characterize the impacts of a change in resources functionality[2], it is necessary to have (i) additional information on the functionality of intermediary and elementary flows and (ii) characterization of the interactions within the technosphere, i.e., intermediary flows of resources. This means that both elements (e.g., aluminium, copper) and materials (aggregated resources such as aluminium alloys) should be characterized.

Regarding the first point, as pointed out by Stewart and Weidema (2005) – and it does not seem to have evolved since 17 years –, current LCI databases (e.g. ecoinvent, thinkstep) neglect the change in functionality of resources. The different quality levels of resources that correspond to different levels of functionality should enable the differentiation of different elementary flows, as proposed for water resource by Boulay et al. (2011a). These intermediary flows with functionality information will enable us to characterize the services loss occurring with degradation at each life cycle step. For the second point, we could build on the approach developed by Zampori and Sala, (2017) and Beylot et al. (2020) that would enable to consider flows of mineral resource flows in technosphere in LCI datasets by adding a label "in technosphere". This could be used to label flows from and to the accessible anthropogenic stock (AAS). Aggregated resources such as alloying elements should also be integrated as intermediary flows into LCI datasets as illustrated by the output flow in figure 4d. So far only the name of the intermediary flow is given (e.g.,

[2] Additional change of the computational structure of LCA is needed for integrating the feedback loops (as already proposed by Weidema et al. (2018)), due to the use of backup technology (BT) in our framework. However, the implementation of feedback loops requires knowing the inventory of resources used for BTs which is mostly unknown at the time being.

aluminium scrap) without the alloying element composition (De Wachter 2021), which is needed not only to respect the mass balance, but also to characterize functionality changes. Using the knowledge on services provided by resources (using tools described in section 2.2) will enable to inform services provided by resources flows in LCIs. Finally, missing dissipative flows to the ecosphere and the technosphere need to be added in existing LCI databases, as previously called by Beylot et al. (2020c, a, 2022). Resource flows should be classified as dissipative on the basis of their accessibility (as defined in the glossary section). Our approach therefore may benefit from the work of Beylot et al. (2020c) and of Owsianiak et al. (2021) as long as their dissipation definition is aligned, which is quite the case for Beylot et al. (2020c) but may need an adaptation in the case of Owsianiak et al. (2021). In fact, this latter approach considers an emission as a dissipative flow if the accumulation from global anthropogenic activities to the soil in the long term does not exceed the average concentration in the upper continental crust. It may not correspond to what we define as "inaccessibility", which is for us the driver of dissipation. In other words, if the resulting concentration in the upper crust is higher than the average, this does not necessarily mean that the resource is approachable. The content after the dash in arrows labels of elementary and intermediary flows (Figure 4) is what we propose to integrate into LCIs.

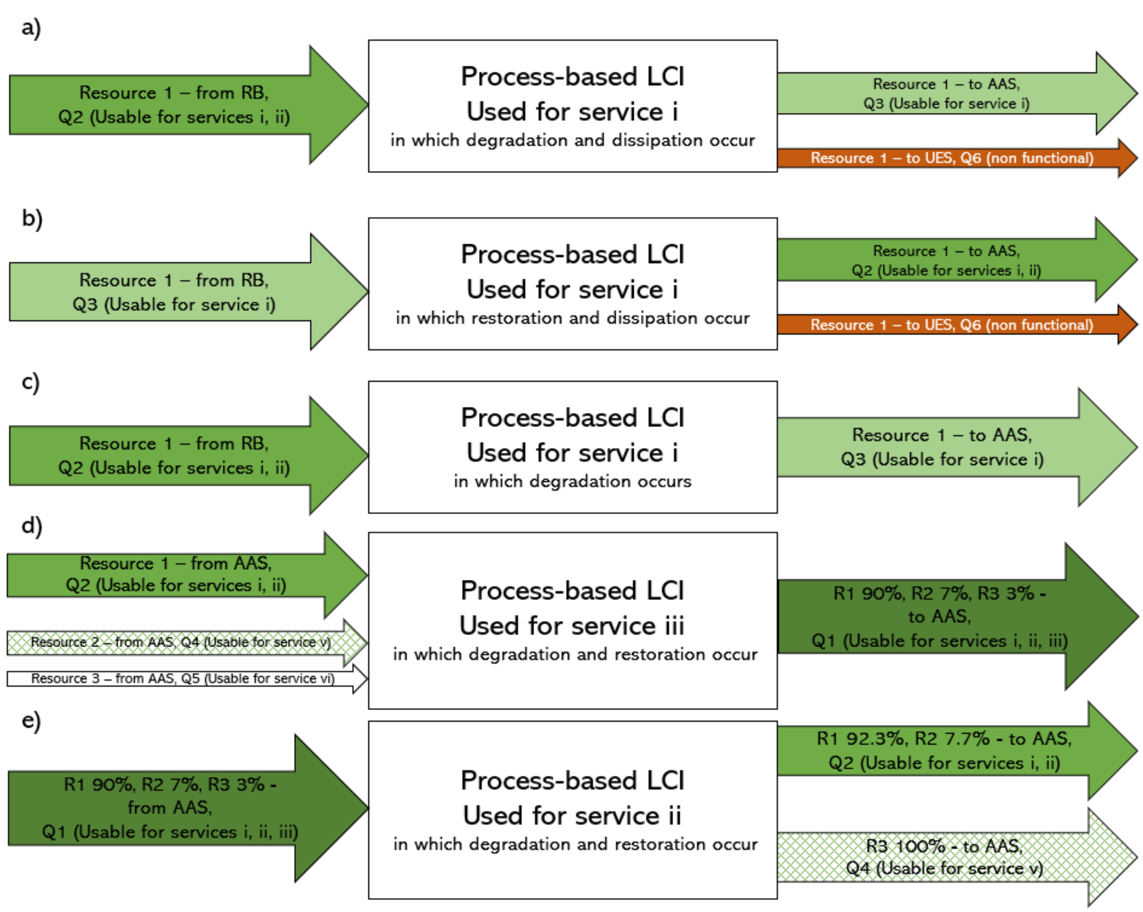

Figure 4. Overview of inventory flows that are required to characterize variations in resource functionality in the proposed framework. R = resource; RB = reserve base; AAS = accessible anthropogenic stock; UES = ultimate ecosphere stock. Qi (i ϵ [1,6]) corresponds to different

categories of quality. Flows directly coming from the ecosphere or emitted to the ecosphere are elementary flows, otherwise they are intermediary flows. Please note that the numbers and the sizes of the arrows reported in this illustrative example are fictive. Each color corresponds to a different inventory flow (the different shades of green represent different functionality levels for resources R1, R2 and R3, orange arrows represent dissipative flows).

In figure 4a, the process requires resource 1 from the reserve base of high-quality resources able to fulfill services i and ii. Since degradation and dissipation occur, two output flows leave the process. The dissipated quantity implies a loss of the two services and only service ii is lost in the other flow because of degradation (the light green arrow). In figure 4b, partial restoration occurs instead of partial degradation, while 4c illustrates the degradation of the resource with no dissipated flow, implying a loss of service without loss of matter. Figure 4d illustrates the fact that when resources are aggregated, such as alloys, the functionality of the output flow can differ from the functionality of each input flow. Figure 4e illustrates the need to have new intermediary flows in the technosphere in which resources are aggregated, such as alloys, that have different levels of functionality. Both degradation and restoration occur since service iii is lost but service iv is gained.

LCI datasets should enable the quantification of the amount of a resource entering into the product system as well as the amount leaving it, as proposed by Beylot et al. (2020c) and De Wachter (2021), along with the associated functionality level. The difference between input and output flows will enable the characterization of the loss of functionality in a process based LCI dataset, as presented in the following equation:

$$IS = \sum_r (I_{e,r} * CF_{I_{e,r}} + I_{t,r} * CF_{I_{t,r}}) - (O_{e,r} * CF_{O_{e,r}} + O_{t,r} * CF_{O_{t,r}}) \quad (1)$$

Impact score (IS) is calculated with input $\mathbf{I_{e,r}}$, $\mathbf{I_{t,r}}$ and output $\mathbf{O_{e,r}}$, $\mathbf{O_{t,r}}$ of resource r (or materials) from ecosphere (e) and technosphere (t) having a certain functionality multiplied by their respective characterization factors $\boldsymbol{CF_{I_{e,r}}}$, $\boldsymbol{CF_{I_{t,r}}}$, $\boldsymbol{CF_{O_{e,r}}}$ and $\boldsymbol{CF_{O_{t,r}}}$ to calculate the total impact on the deficit of resource services" (ie the deficit of services provided by resources induced by a product system through the use of resources over its life cycle).

4 Discussions

4.1 Models and data for the fate model

This section introduces to models and data sources needed for the fate model framework to be operational. As of now, none of the resource reserve base is depleted at the global level (Mudd 2021), then the mechanisms affecting the size of the reserve base are likely to generation potential damages in the future. Moreover, the deficit of resource services depends on the demand of those latter (a depleted reserve base without demand does not result in any deficit). It is therefore necessary to understand the evolution of the demand on resource services over time. To do so one could rely on integrated assessment models (IAMs) that build on narratives describing the plausible future world, such as the so-called Shared Socioeconomic Pathways (SSPs) (Van Vuuren et al. 2012; O'Neill et al. 2014, 2017). The SSPs are a "scenario framework used by the climate change research community in order to facilitate the integrated analysis of future climate impacts, vulnerabilities, adaptation, and mitigation" (Riahi et al., 2017). It tackled the issue of "the longstanding problem of incompatibility of scenarios across studies"(Fishman et al. 2020) by proposing consistent scenarios. Five narratives describe alternative socio-economic developments and are used in IAMs that address possible technology mixes and future costs of climate change mitigation by generating scenarios for the future industrial system (Pauliuk et al. 2017). SSP1 describes a gradual but pervasive shift towards a more sustainable path. SSP2 projects an evolution of societies without a marked shift from historical trends. SSP3 is characterized by regional rivalry and international fragmentation while environmental concerns are not prioritized, and consumption remains materially intensive. SPP4 describes a world with high inequalities and SSP5 is a techno-optimistic pathway with continuous economic growth and increasing energy consumption. SSPs should be used with caution since important criticisms of overestimation of economic growth and CO_2 emissions of SSP3 and SSP5 have been adressed (Burgess et al. 2020; Pielke and Ritchie 2021; Pielke Jr et al. 2022). Since long-term decoupling between energy consumption and economic growth seems impossible (Huang et al. 2008; Haberl et al. 2020; Hickel and Kallis 2020; Andrieu et al. 2022) and the likeliness of all-oil liquids peak in the next 15 years (Delannoy et al. 2021a; Laherrère et al. 2022) and gas before 2050 (Delannoy et al. 2021b), the plausibility of scenarios with continuous economic growth in the 21st century seems low, as pointed out by Steckel et al. (2013). Therefore, additional efforts are needed to have access to a wider range of post-growth scenarios (Hickel et al. 2021; Keyßer and Lenzen 2021; Lenzen et al. 2022), building upon the propositions of Nieto et al. (2020) and Bodirsky et al. (2022) which focus only on energy infrastructures and the food sector, respectively. Other scenarios focused on sufficiency such as the Low Energy Demand (Grubler et al. 2018) and the Decent Living Energy (Millward-Hopkins et al. 2020) should be considered. These different scenarios being "educated guess" on what our future resource consumption needs might be, they will allow to assess an uncertainty on the characterization factor. Other parameters such as accessible energy, recovery and recycling rates or functional substitution factors will also be considered in the uncertainty analysis.

To evaluate the potential deficit of resources services, one should assess if the available stock is sufficient to meet demand. If there is a mismatch, we should assess the contribution of mechanisms to such mismatch (e.g. the contribution of a kg of resource dissipated to the deficit of services). It is therefore necessary to quantify the future stocks, occupation, degradation, restoration and dissipation of resources. One option is to couple MFA models (Wang et al. 2018; Godoy León et al. 2020; Helbig et al. 2020, 2021; Yokoi et al. 2021) with IAM models to quantify prospective stocks and flows of resources over their life cycle, as Yokoi et al. (2021) did for six metals. The coverage of resources can be extended to other abiotic resources. MFA enable to trace stocks and flows of resources over their life cycle, which are needed to assess their fate. Strengths and weaknesses of each MFA model with regards to our fate model is presented in Table S1: some of them are more compatible to our fate model framework than other, but none is fully compatible and some adjustments will be needed. Another option is to couple physical input-output tables (pIO) (Wieland et al. 2021) that trace resource stocks and flows in the entire economy (i.e., including waste flows) with IAMs (Wiebe et al. 2018; Beaufils and Wenz 2021)[3] although additional efforts are needed to integrate dissipative flows to the environment in pIO

[3] Scenario-based projections of economic IO tables.

tables. Coupling MFA with IAMs (option 1) has the advantage of allowing the assessment of specific resource services, with the downside of having limited coverage of resource services. Coupling pIO with IAMs (option 2) has the strength to cover all commodities (end-uses) of the economy, but at a high aggregation level of commodities that may limit the characterization of services[4]. Those two options will be analysed in term of feasibility and data availability when operationalizing the framework. To do the calibration of the model such as the ultimate reserve, the occupation time of resources in the technosphere, the recycling rate of resources and functional substitution factors. Some of the potential sources for those data can be found in supporting information 2 section 3.

The difficulties in reliably quantifying the reserve base, the accessible anthropogenic stock and ultimately extractable resources have been discussed extensively (Jowitt and McNulty 2021; Winterstetter et al. 2021) and especially in the LCA community (Drielsma et al. 2016; Northey et al. 2018). Drielsma et al. (2016) citing Graedel et al. (2011) mention that the extractable global resource is challenging since it depends on "demand and technology develop[ment]" and that resource status in mines could switch from by-product to main product. Nevertheless, as of now, more than half of metals extracted from the lithosphere are by-products of host metals (Nassar et al. 2015) or fossils (Månberger 2021) and some, like indium, are likely to remain by-product in the future (Werner et al. 2017). Thus, the reserve base of these by-products should be estimated on the basis of the future demand of host elements. It requires to calculate the ratio of by-product to main-product production, extending the coverage of elements reported by McNulty and Jowitt (2021). Explicit thresholds could be proposed to clarify the boundary between the reserve base and the ultimate stock. Owsianiak et al. (2021) proposed to use the average concentration of elements in the upper crust. An alternative option would be to consider the thermodynamic limit in terms of energy consumption intensity (see for example (Calvo et al. 2016; Magdalena et al. 2021)).

To the best of our knowledge, we are not aware of any study estimating the amount of accessible stock functional for specific services that would allow the differentiation between different subsets of the reserve base, represented by the white boxes in the figure 2. First, the relationships between different resources (or materials) and their end-uses need to establish. This is the case for aluminium alloys (Hatayama et al. 2007; Van den Eynde et al. 2022) and steel alloys (Dworak and Fellner 2021; Dworak et al. 2022). Additional research efforts are required to fill in this knowledge gap for other resources. Second, several resources can be functional for the same service, but not with the same quantity. Then, the functional substitution factor between one resource and another needs to be calculated. It depends on the performance of the substitute in the key properties, that shall be the same or better to the ones of the substituted resource to reach full functional equivalence. If the service fulfilled by resources A and B depends on a unique key property, the functional substitution factor can be calculated as the ratio of the mass of resource B needed to fulfil the service over the mass of resource A needed to fulfil the service, similar to the method of Rigamonti et al. (2020). If multiple key properties of the resources are essential to fulfill the service, the functional substitution factor should be determined as the highest value

[4] e.g., the commodity "motor vehicles, trailers and semi-trailers" in Wieland et al. (2021) is highly aggregated and does not enable the distinction between services of vehicles and trailers, which are different.

among the mass ratios of resources B and A for each key property required to fulfill the service i. If the functional substitution factor cannot be established, there might a partial loss of performance in the key properties and therefore a partial loss of service i. The extent of the loss of service can be estimated using the scoring function of Demets et al. (2021). This function scales from 0 to 1 the functional equivalence of a substitute depending on the performance of the material in the key property considered.

4.2 Data for the effect model

The recovery cost of lost services could be evaluated by the cost of the backup technology. For instance, desalination (Qasim et al. 2019) and deep-sea mining technologies (Sharma 2011) are used to expand, respectively, the reserve base of drinkable water and of many metals including lithium. However, most of the backup technologies to recover other resource services are so far still unknown. Prospective modelling may be required to assess their future economic cost.

4.3 Elementary and intermediary flows in life cycle inventories (LCIs)

The operationalization of this framework partially depends on the integration of additional information: the functionality level of intermediary and elementary flows of resources in LCIs. This may be a challenge: some LCIA method that would have needed additional efforts at the inventory level to be fully operational exist, while these efforts have never been done. This is the case for the 17 different categories of water elementary flows proposed by Boulay et al. (2011a) that have not been implemented in existing LCI databases. Therefore, it may be a challenge to convince LCI developers to add functionality information for the abiotic resources that the current framework is intended to cover. However, before the additional needed information is added to life cycle inventory databases, at least it will be possible to propose some default characterization factors with the corresponding uncertainty related to the lack of information on resources quality (highlighting the need to collect those additional data and allowing to prioritize data collection in a parsimonious way).

There is ongoing work by Life Cycle Initiative experts (Laurent et al. 2021) to "provide recommendations and mitigation measures for a coherent connection between LCI and LCIA, accepted by all different stakeholders". The use of distributive version control could enable to improve the current content of LCI databases within a research community, as highlighted by Mutel (2021). We intend to be part of this effort of integration of additional information in intermediary and elementary flows.

In addition, further information on intermediary flows should be added in life cycle inventories, like in the ecoinvent database (Wernet et al. 2016). Currently, there is no correct mass balance of alloying elements in intermediary flows entering and leaving elementary processes (Beylot et al. 2020c; De Wachter 2021). An increased effort should be made in specifying the composition of alloying elements. Finally, given the vast number of different alloys (Graedel and Miatto 2022), it becomes urgent to expand the coverage of alloys in the ecoinvent database as expressed by Althaus and Classen (2005).

4.4 Complete the coverage of impact pathways

Aside from the pathway covered in this article, a complementary impact pathway, that is out of the scope of this framework, might be addressed in future research. It consists of assessing the damages on humans that are unable to adapt to the deficit of resource services. The damages can be assessed in terms of economic cost but also on human health. This pathway is covered by some life cycle impact assessment models such as the damage to human health from water deprivation due to water use (e.g., (Boulay et al. 2011b; Debarre et al. 2022)) or from food provision losses due to land use (e.g., (Ridoutt et al. 2019)). Finally, biotic resources (e.g., wood) could also be covered in our framework, therefore requiring integrating the renewability rate and regeneration time of stocks.

5 Conclusions

This research paper provides a novel LCIA framework to assess the potential damages due to the deficit of resource services, i.e., the deficit of the instrumental values of abiotic resources and its consequences on human wellbeing. It describes the mechanisms (degradation and dissipation) that affect the accessible stock of resources that lead to damages expressed as the additional cost to users to adapt to the deficit of resource services. We identify potential models and data sources to operationalize the framework. To harmonize LCIs with this novel impact assessment framework, the resource functionality is a data that should be integrated in intermediary and elementary flows. This framework is aligned with the recommendations of the global guidance of UNEP-SETAC Life Cycle Initiative (Jolliet et al. 2018) to address the damages on the instrumental values of natural resources rather than the inaccessibility of the resources. It still needs to be translated in an operational method that could enhance the sustainable management of the deficit of resource services, potentially affecting human wellbeing.

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Data availability statement

Data sharing is not applicable to this article.

Conflict of interest

The authors have no conflict of interest to declare.

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