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Abiotic resource use in life cycle impact assessment—Part II – Linking perspectives and modelling concepts

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ABSTRACT

Starting from a lack of consensus on how to consistently assess abiotic resource use in life cycle assessment, a structured approach was developed to enable a classification of perspectives on resource use, based on the so-called role of resources. Using this classification, this paper focusses on analysing links between perspectives and modelling concepts, i.e. the conceptual implementation. To analyse the modelling concepts for a selection of existing LCIA methods and other modelling approaches, the concept of the system model is introduced. It defines the relevant inventory flows to be assessed by the LCIA method, and, at the same time, to be considered in the characterization model, and how the flows and stocks of resources used to calculate the characterization factors are positioned in relation to environment (nature) and economy (technosphere). For consistency, they should be aligned with the position of inventory flows and, at the same time, reflect the perspective on resources taken by the method. Using this concept, we critically review a selection of methods and other modelling approaches for consistency with the perspectives on resource use, as well as for their internal consistency. As a result of the analysis, we highlight inconsistencies and discuss ways to improve links between perspectives and modelling concepts. To achieve this, the new framework can be used for the development or improvement of LCIA methods on resource use.

1. Introduction

Life cycle impact assessment (LCIA) methods enable a quantification of (environmental) impacts of emissions or resource extractions. Over the past two decades, life cycle impact assessment methods have received extensive attention and developed into mature methods, mostly reflecting the state of the art in the respective scientific fields. Despite these achievements and recent harmonization achievements by the Task force on mineral resources of the Life Cycle Initiative hosted by UN Environment (Task Force Mineral Resources), stakeholders may take different perspectives on the use of abiotic resources, and new methods for the assessment of resource use in LCA are still being developed. Impact categories in LCA reflect concerns regarding environmental entities worth protecting – and therefore a human interest in reducing particular impacts which negatively affect them (Steen, 2006a; Udo de Haes et al., 2003). Hence, a selection of important environmental impacts for the system to be analyzed is inherent to any LCA study, and required for practicality reasons. This is particularly important in the context of abiotic resources such as metals and minerals, which are the subject of this paper, and are here simply referred to as 'resources'[1] . A key part of the life cycle impact assessment (LCIA) phase is the characterization step, which provides criteria according to which the impact of using "resource A" versus using "resource B" is assessed. LCIA methods have traditionally been concerned with the issue that the extraction of a resource from the environment means that it cannot be extracted anymore by future generations and/or is temporarily or permanently unavailable for other purposes. So, whilst the use of the resource in itself is beneficial, it is associated with increased difficulties or challenges elsewhere in society. However, the topic is complex and the increased difficulties or challenges – or, in LCA terms, the impact paths or environmental mechanisms to be assessed, are not obvious to determine (Guinée and Heijungs, 1995; Sonderegger et al., 2017; Steen,

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[1] In general, abiotic resources are natural resources (including energy resources) regarded as non-living, such as zinc ore, crude oil or wind energy. However, this paper focuses on minerals and metals.

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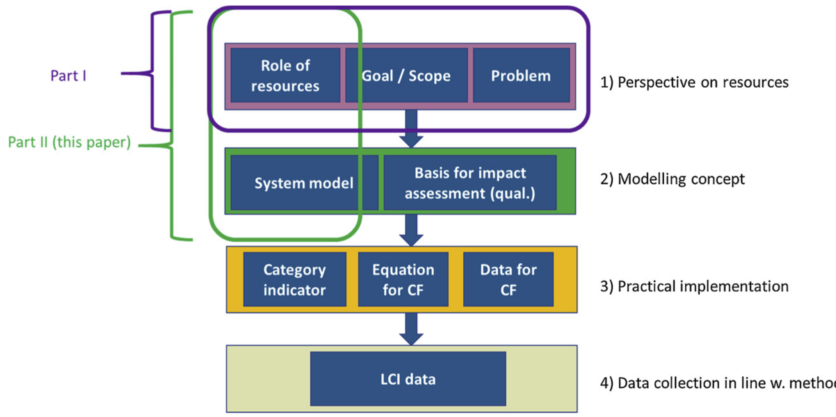

Fig. 1. Framework for analysis (and development) of LCIA methods, and other approaches to modelling the use of abiotic resources (minerals and metals), and relevant aspects in this paper (based on 1,Part I, modified).

2006b).

Outside the LCA community, different impacts from the use of abiotic resources are discussed, e.g. in the context of resource criticality (see e.g. Graedel and Reck, 2016; Dewulf et al., 2016), the circular economy (see e.g. Strothmann and Sonnemann, 2017), ecosystem services (ES) concepts (Dewulf et al., 2015), and in the context of green technologies and sustainable production and consumption (see e.g. Ali et al., 2017). These concepts all have different underlying roles, goals and problem definitions with regards to the related resource management strategies. This has implications if the concepts were to be used as a basis for the definition of the relative impacts of different resources, i.e. different results are to be expected.

For an LCIA indicator to be effective, a clear definition of the perspective taken by an LCIA method is important for transparency. The perspective on resources refers to the role and context in which the resources are seen, and the motivation behind protecting the resources. Perhaps equally important is a model that reflects the chosen perspective.

This paper is the second part of a 2-part submission. Part I reports on a consensus process which was guided by a framework and aimed at clearly defining the perspectives on resources to be used as a basis for the further development and refinement of life cycle impact assessment on resource use (Schulze et al., 2020). In *this* paper (Part II), the same framework is applied to a selection of LCIA methods and other modelling approaches for the assessment of resource use, followed by a discussion on insights from the analysis. The term ‘modelling approach’ describes LCIA methods and other new solutions to modelling resource use which are not LCIA methods as such, but rather attempt to provide solutions at LCI level. The aim of this paper is to analyse the linkages between the perspectives on abiotic resource use taken by the developers of life cycle impact assessment methods and the models they use. This is achieved by identifying the underlying model structures (system models) of existing approaches used to model the perspectives, with the help of a structured approach (framework) developed for this purpose.

With the help of the framework, we address the following questions:

(1) Which perspectives are adopted by *existing* life cycle impact assessment methods?
(2) Which perspectives are adopted for *__new__* method ideas and/or approaches to modelling impacts of resource use that are being proposed in the LCIA community, but have not been developed into full methods or modelling approaches yet?
(3) Based on the analysis with the help of ‘system models’, are the methods and modelling approaches consistent with the perspectives taken, and are they internally consistent?

Section 2 of this paper explains which part of the framework introduced in Part I of this submission (Schulze et al, 2020) is used for the analysis in *this* paper (Part II). Section 3 presents the results of the analysis, and the answers to the first two questions. In Section 4, a summary of the findings and observations is presented, and the third question is answered. Finally, Section 5 presents the conclusions from this article.

2. Methods

2.1. Introducing the framework used for the analysis

The endpoint regarding the damage to the safeguard subject ‘natural resources’ as defined in Jolliet et al. (2014) allows for a variety of interpretations regarding perspectives on resources, and an even larger number of methods/ indicators. A more specific definition of the perspective on resources can help refine the issue that is being addressed in LCIA. For this purpose, we use a framework developed in the SUPRIM project. The framework consists of (1) an overarching perspective, (2) a conceptual level (“Modelling Concept”) and (3) a practical implementation level. The aspects which are key to this paper are the perspective on resources and the modelling concept of Level 1 and 2 of the framework (Fig. 1). For a detailed explanation of the framework which this paper builds on, please refer to Part I of this submission.

2.2. Applying the framework

In this paper, the framework (Fig. 1) was used to analyse and critically review a selection of existing LCIA methods and method ideas. The analysis focused on 1) the perspective on resources taken by the developers of the respective method, and 2) the modelling concept used to address the issue. More specifically, the analysis was based on the alignment between the ‘role of resources’ and the ‘system model’. In a first step, the concept of the ’role of resources‘ from the top level of the framework is used for the classification of perspectives on resource use adopted in existing LCIA methods on resource use (Section 3.1). The role of resources answers the very basic questions about the perspective, and is therefore suitable for a classification of perspectives. From the second level of the framework, the concept of the system model‘ was used for the analysis of existing LCIA methods and method ideas (Section 3.2). The system models illustrate how the conceptual role is translated into a model. An analysis of how well the role and system model are aligned can provide an indication of how well the perspective taken by the method developers is reflected in the modelling concept, and thus whether the model is consistent with the idea it is supposed to reflect.

The analysis was based on either explicit statements made by the method developers, or implicit wordings in the documentation. The selection of methods (and other modelling approaches) contained examples of methods which assess the use of abiotic resources (minerals/ metals), and methods which include those types of resources as well as others (e.g. fossil fuels). The aim was not to analyse the whole range of available methods, but to show examples in order to illustrate and analyse the range of perspectives and modelling concepts applied in life cycle impact assessment methods for resource use today. Both established methods which are already being used by LCA practitioners and methods which are published in the peer-reviewed literature, but not yet implemented in software packages, or still at conceptual stage were included in the analysis. Methods which are still conceptual, i.e. not fully developed yet, are addressed in a separate section to highlight that the analysis of the not-yet-finalized methods should also be interpreted as tentative.

3. Results

3.1. Different perspectives on resource use

Eighteen possible perspective types can be classified according to the criteria for the definition of the 'role' of resources. They are shown in Table 1 of Part I of this submission (Schulze et al., 2020). However, not all of the possible combinations are meaningful or easy to interpret. Four of these possible combinations (A–D), (Fig. 2 *of Part I,* and Section 3.1.1 of *this*paper) are reflected in LCIA methods or conceptual method ideas – those are explained in more detail in Section 3.1.1. Although not represented in LCA, a fifth perspective type (E) is briefly mentioned in the text below since it has often been mistakenly referred to in the context of LCA and resources. Within each defined perspective type, several specific perspectives can be taken, which are not presented in this overview. Those can vary with regards to their goal and scope of resource management, and regarding the problem/ impact.

3.1.1. Perspectives types represented in LCIA methods and new method ideas

In the following sections, we present a selection of perspective types, namely types A–E (please refer to Fig. 2 of Part I of this submission, Schulze et al., 2020). The selection is based on their representation in current LCIA methods and new method ideas. At the same time, the selection also covers the key concerns associated with resource use, as determined in a stakeholder workshop with representatives from industry, academia and EU policy support (Schulze et al., 2018).

3.1.1.1. Type A perspectives. Methods of Type A perspectives share the same role of resources, based on the three criteria stakeholder, system of concern, and production system. Type A perspectives focus on humans (as 'stakeholders'), who have an interest in using the resources in the economy (as the 'system of concern'). For example, iron is used to make steel used in products in the economy. Type A perspectives focus on primary resource production, e.g. the extraction of iron ore from the environment.

The majority of existing, established LCIA methods fall into this category, which is concerned with the continued access to resources for mining from the environment by future generations and the use value they represent, or may represent, to humans (stakeholders) in the economy (system of concern). The focus is on primary production, i.e. the extraction of resources from the environment for use in the economy. In other words, secondary resource use and anthropogenic stocks are excluded.

Methods which share this perspective include (but are not limited to) ADP elements (Guinée and Heijungs, 1995; van Oers and Guinée, 2016), EDIP (Hauschild and Potting, 2005), ReCiPe (Goedkoop et al., 1998), Surplus Cost Potential (Vieira et al., 2016, 2012), ORI (Swart and Dewulf, 2013a, 2013b), Ecoindicator 99 (Goedkoop et al., 1998). They focus on the flows of resources from the environment for use in the economy, i.e. primary production. For this perspective, the most common goal is to reduce or delay the extraction from the environment as much as possible (Steen, 2006b)- either to avoid a situation in which extraction is no longer possible, or to prevent access to resources from getting more difficult. These observations are in line with the key concern regarding resources addressed by LCIA methods identified in previous studies as "the scarcity of the resource and hence the limitations in its availability for current and future generations" (Hauschild et al., 2013), or "the declining environmental provision of natural resources "(Sonderegger et al., 2017).

Type A perspectives are also commonly reflected in resource management concepts outside LCA which focus on the management of primary natural resources. Peak resources, resource depletion or scarcity are concepts which concern the management of abiotic resource stocks in the environment to ensure that future generations can still use them – (see e.g. UNEP, 2011; Henckens et al., 2016a,b). Resource management according to the concept of Sustainable Production and Consumption focuses on a reduction of primary resource use per unit of value generated – this is also referred to as resource decoupling (Bizikova et al., 2015).

Where Type A perspectives are adopted, different problem definitions are possible, with implications for how the impact of using one resource is measured and evaluated against the impact of using another. Method developers of Type A LCIA methods may have different views on what limits the continued access to those resources. This is relevant for how the relative impacts of different resource flows are modelled. Consequently, the criteria for the relative impacts between different resource flows are diverse, and include for example geological availability (EDIP), geological availability and rate of extraction (Guinée and Heijungs, 1995; van Oers et al., 2002), overall mass extracted per unit of target metal (Hinterberger and Schmidt-Bleek, 1999), marginal increase of mass (ore mined per unit of target metal, Swart and Dewulf (2013)), cost (Vieira et al., 2016; Goedkop et al., 2009), or energy per unit (of mass) mined (e.g. (Baayen, 2000; Jolliet et al., 2003).[2] As a consequence, impact assessment results from different methods are not directly comparable even if the methods share the same perspective type (e.g. Type A).

3.1.1.2. Type B perspectives. Type B perspectives are similar to Type A perspectives in that they are concerned with the continued access to resources by humans for use in the economy (stakeholder: humans, system of concern: economy). However, they differ in terms of criterion 'production system', i.e. they do not distinguish between the availability of primary resources in the environment or secondary resources in the economy. (Secondary resources are resources which have already been used in a product, or have been processed by industry (e.g. industrial scrap)). The production system comprises both the primary and secondary production of resources, and both primary and secondary production can impact the accessibility or availability of resources. To get back to the previous example, the perspective is concerned with the use of iron by humans in the economy, but does not distinguish between iron obtained through primary production from ore, and secondary production from scrap iron. Since the physical presence in the environment is no longer considered the sole decisive factor for the availability or accessibility of the resources, other criteria regarding the property of the resource are then required - e.g. the concentration of the target metal, or other economic, technical and legal criteria which determine whether a resource can be (re-) extracted. The problem definition adopted by these methods still concerns a stock which may be depleted through resource use/

[2] The methods which focus on marginal effort consider neither the size of the stock nor the extraction rate in the calculation of the indicator, but nevertheless focus on primary production, i.e. the extraction of resources from the nature to economy

extraction, but the stock is defined through the presence of useful properties which lead to the resources being viable to extract, or potentially viable to extract (e.g. the concentration of target metal in ore), rather than the presence of the resource in a specific compartment (the environment or economy). One method which explicitely incorporates both primary and secondary stocks has been/ is being developed (Schneider et al., 2015, 2011).

Furthermore, the criticality concept fits with the Type B perspectives - although it focuses on the accessibility of a resource for a specific region, company, product, etc. Recently, approaches to integrate criticality into LCA have been developed (see e.g. Bach et al., 2016; Cimprich et al., 2017; Mancini et al., 2015). Theoretically, a focus on primary resources only is not consistent with the concept: the provision of secondary resources is considered a resource management strategy to mitigate criticality. As mentioned by the method developers, socio-economic supply constraints, such as company concentration or price volatility, can exist for secondary resources as well. Consequently, the considered system is a different one altogether, in which a differentiation between environment and economy is not crucial. In practice, the developers of the methods which integrate criticality aspects into LCA have not (yet) focused on the question whether and how to consider secondary resources in the assessment. For the criticality-based LCIA method ESSENZ, the characterization is applied to primary resource flows only (Bach et al., 2016). For GeoPolRisk (Gemechu et al., 2016), characterization is applied at product level rather than elementary flow level, i.e. the characterization factors are applied to intermediate flows within the economy rather than elementary flows between the environment and the economy (Cimprich et al., 2017). However, the characterization factors are based on geopolitical supply risks of primary production rather than primary and secondary production. Cimprich et al. (2017) point out that the role of recycling, despite being relevant in supply risk assessment, is not yet reflected by the method.

3.1.1.3. Type C perspectives. Type C perspectives distinguish and consider both the instrumental value of resources in the economy and in the environment (as the 'system of concern'), as valued by humans (as the 'stakeholders'). The instrumental value of resources refers to their utility for humans (Frischknecht and Jolliet, 2016; Justus et al., 2009). Sand and gravel are the most prominent examples for this category, since their competing roles in both environment and economy are commonly recognized as valuable (Dan Gavriletea, 2017; Delestrac, 2011; Knight et al., 1999). In the environment, sand and gravel have important ecosystem functions which are compromised by the (sometimes illegal) extraction of the material from beaches or rivers. In the economy, sand and gravel are important raw materials e.g. for the production of concrete and glass. For the classification of the perspective type, only primary production is considered as the 'production system' since it covers both types of uses in both 'systems of concern'- the current use in the environment and the potential for future use in the economy. One method could be identified which adopts this perspective, albeit only for a certain type of resources, i.e. sand and gravel. The Ecological Scarcity Method (Frischknecht and Büsser Knöpfel, 2013) is based on a distance-to-target approach, i.e. the ratio of the current flow (pollutant load or resource extraction) and the "critical flow", which is determined by political targets. Since the method is based on political targets, different approaches can be taken for different resources, and the approach taken for sand and gravel differs (conceptually) from the approach taken for other resources. Sand and gravel are considered worth protecting for their use by humans in the economy (e.g. as construction material), but also for their regulating functions in the environment/ the ecosystem. For the latter, the role of gravel and sand in groundwater formation is emphasized (Frischknecht and Büsser Knöpfel, 2013).

The problem definition focuses on the presence of the resources in a certain physical form (i.e. as sand or gravel, respectively), for use in the environment and the economy. The focus is on the extraction from the environment: the loss of material from 'system of concern: environment' is 'tolerated' politically in recognition of the benefits of the material in the system of concern 'economy'.

3.1.1.4. Type D perspectives. Type D perspectives consider the in-situ functions of abiotic resources in the environment (as the 'system of concern', which are being recognized as beneficial to humans (as 'stakeholders'). For this perspective, only primary production is the relevant 'production system', since it compromises the in-situ functions of the resources extracted from the environment. This perspective is represented by CEENE and CExD, two life cycle impact assessment methods based on exergy, that do so indirectly: The CEENE method quantifies the exergy taken away from natural ecosystems, which sustains various in-situ functions in the environment (Dewulf et al., 2007). Similarly, the CExD method is based on the assessment of exergy "removal from the environment" (to the economy) (Bösch et al., 2007).

Furthermore, outside LCA, the Type D perspective is represented by the in-situ functions of abiotic resources covered by the ecosystem service (ES) concept (WRI, 2005WRI, 2005). Several researchers are working on the integration of in-situ ecosystem services into life cycle impact assessment methods (Bruel et al., 2016; Chaplin-Kramer et al., 2017; Maia de Souza et al., 2018; Zhang et al., 2010). The concept includes the following in-situ functions:

- *regulating services* within the environment, (recognized by humans as useful functions, such as flows of nutrients between different compartments of the ecosystem, e.g. between the lithosphere and biosphere), or hydrogeological process important for water quality
- *cultural services* that provide recreational, aesthetic, and spiritual benefits; and
- *supporting services* such as soil formation, and nutrient cycling.

Generally, the ecosystem services concept has originally put a greater focus on biotic resources, but a more explicit assessment of the role which abiotic resources play in or next to ES is being discussed (van der Meulen et al., 2016; Van Ree and van Beukering, 2016). Abiotic resources play a role in the ecosystem cycles through their interaction with biotic resources, and both are relevant to the ES concept. Interactions between biosphere and geosphere happen with microorganisms (see e.g. Itävaara et al., 2011) as well as other organisms (plants and animals). Geosphere-biosphere interactions play a role in sedimentation and soil formation, for example.

3.1.2. Perspectives types outside LCA

Type E perspectives focus on the intrinsic value of resources in the environment as the 'system of concern'. Intrinsic value is independent of human interest and refers to the existence of the resource itself (Frischknecht and Jolliet, 2016; Justus et al., 2009). They are included in this list to acknowledge that a non-anthropocentric perspective on resource use is theoretically possible, where the resources themselves are seen as the 'stakeholder'. Since the resources are valued in the environment, only primary production is the relevant 'production system'. The adoption of this perspective type has implications with regards to the practicability of indicator development, since all resources have the same intrinsic value, and intrinsic value independent of human interest is impossible to measure (Justus et al., 2009). Hence, in practice, this perspective is not adopted in LCA (neither in methods nor in other modelling approaches). This means that the criterion "stakeholder = human" can essentially be considered a given for those perspectives where implementation into LCIA is feasible.

3.2. System models which reflect the perspective types in LC(I)A

For the development of LCIA methods, the 'perspective types' (3.1.1) defined at conceptual level (Level 1 of the framework), based on

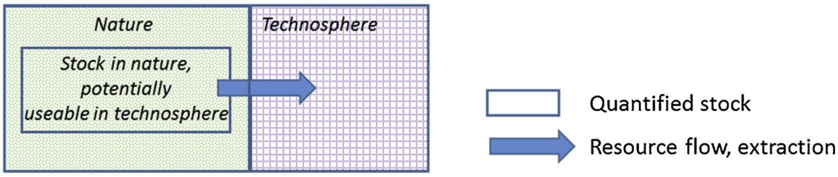

Fig. 2. System model A, (reflects Type A perspectives).

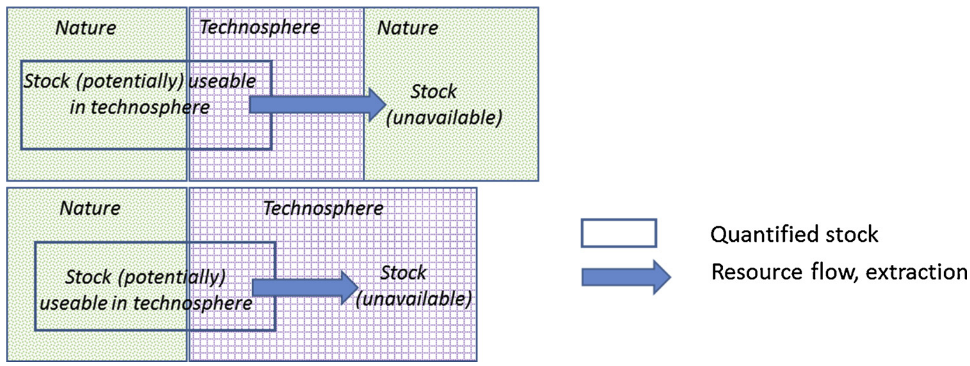

Fig. 3. System model B1 and B2, (possible system models reflecting Type B perspectives).

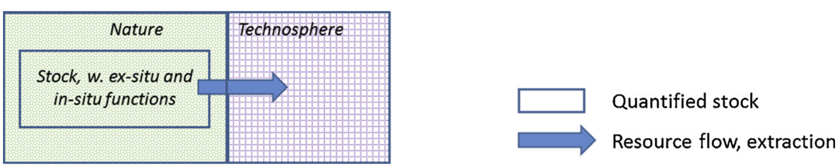

Fig. 4. System model C (Type C perspectives as modelled for sand and gravel in the Ecoscarcity method).

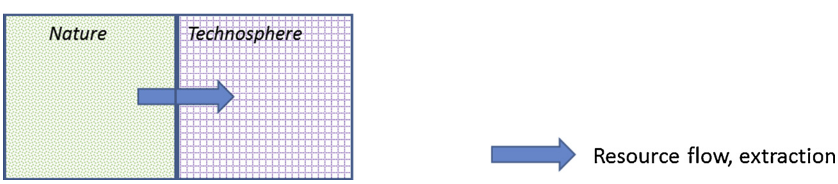

Fig. 5. System model 4 (Type D perspectives as modelled in CEENE and CExD).

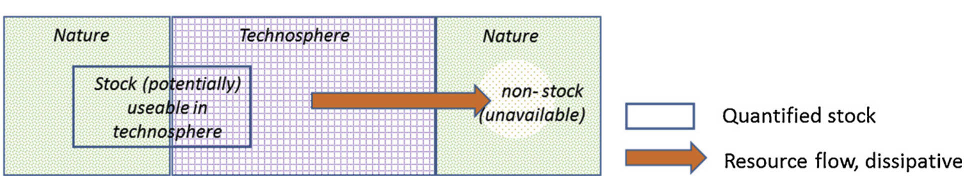

Fig. 6. System Model B, variant 3 (tentative).

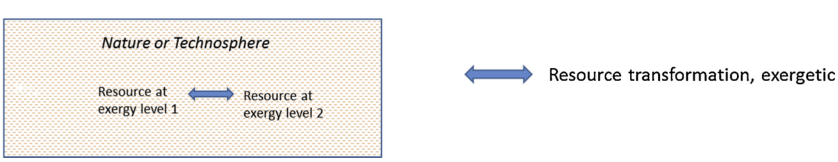

Fig. 7. System Model B, variant 4 (tentative).

the 'role of resources', are translated into "system models" (Level 2 of the framework). The term "system models" is introduced here to describe simple models which illustrate how the flows and stocks to be inventoried and characterized are positioned in relation to economy and environment (see Figs. 2–9). Traditionally, the model applied in LCA considers resources lost from a stock in the environment through extraction (i.e. a flow) to the economy (Type A perspectives, 3.1.1.1). However, as presented in Section 3.1, different perspective types are possible, and other model structures are thinkable to represent these perspectives, even if they may not fit the current model structure in LCA, where only elementary flows can be characterized. Moreover, these other perspectives and model structures have already been introduced in LCA by existing impact categories. Therefore, in the following, an attempt is made to match the perspective types described in Section 3.1 with different system models adopted in existing LCIA methods. Some approaches found in the literature try to tackle alternative perspectives at LCI rather than LCIA level (see Section 3.2.1.2, Ecological Scarcity Method for abiotic resources (other than sand and gravel). Furthermore, we attempt to translate some new LCIA method ideas (and other LCA approaches to modelling resource use) into system models. Here again, some of the suggested approaches tentatively concern the inventory rather than the impact assessment stage (System model B, variants 5 and 6 (tentative), System Model B, variant 7 (tentative). Since the goal of the suggested approaches is a common one, i.e.

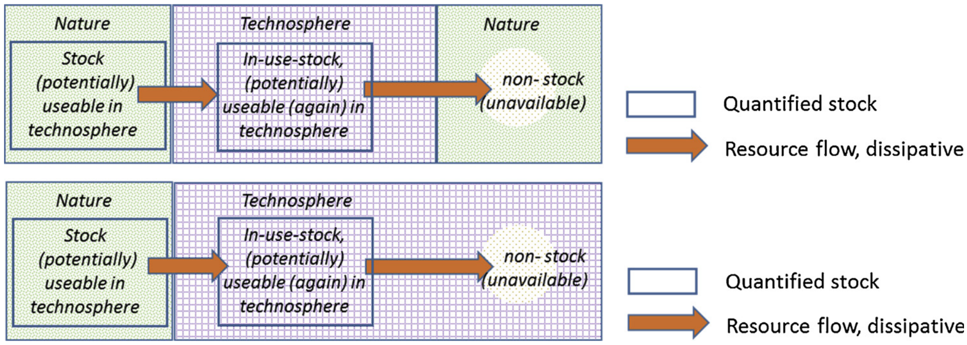

Fig. 8. System Model B, variants 5 and 6 (tentative).

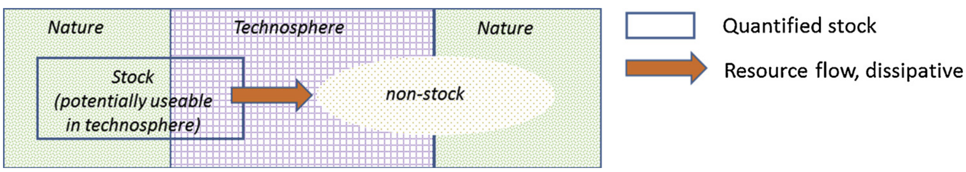

Fig. 9. System model B, variant 8 (tentative).

to reflect alternative perspectives, and the distinction between LCI and LCIA is not always presented in a clear-cut way, we intentionally include the different approaches in the same analysis.

3.2.1. Analysis of system models for existing LCA methods

3.2.1.1. System model A (reflect Type A perspectives). Most existing LCIA methods consider abiotic resources as a non-renewable stock in the environment, which is being depleted through physical extraction of resources (mining and subsequent processing) from the environment into the economy (Fig. 2, System Model A). A one-directional and non-reversible/ consumptive flow from a quantified stock in environment to the economy is modelled. "System model A", reflects the methods which adopt the "Type A perspectives" (Fig. 2, 3.1.1.1). Methods with this system model include (but are not limited to) ADP elements, EDIP and LIME (Guinée and Heijungs, 1995; Hauschild and Potting, 2005; Itsubo and Inaba, 2012; van Oers and Guinée, 2016; van Oers et al., 2002; Wenzel et al., 1997). The stock and/or flow size are used for the calculation of the category indicator (Wenzel et al., 1997). The choice of data to be used in the LCIA models as an estimate for stock quantities varies between methods, and has been explicitly debated (see e.g. (Drielsma et al., 2016; Guinée and Heijungs, 1995; Hauschild and Wenzel, 1998; Sonderegger et al., 2017). In contrast, the definition of what constitutes an extraction flow is often not explicitly stated, and has also not been a focus of discussions. The extraction of elements from the environment to the economy is usually approximated with production data, which refer to the net production of target metals rather than the overall quantities extracted from the environment to the economy (i.e. flows of material which end up in tailings, waste rock, or as emissions to the environment are not accounted for due to data limitations).

Methods that focus on future consequences of resource extraction from the environment to the economy (measured in monetary ((Goedkoop et al., 2009; Huppertz et al., 2019; Vieira et al., 2016), mass ((Swart and Dewulf, 2013b; Vieira et al., 2012) or energy units (Goedkoop et al., 1998; Müller-Wenk, 1998)), also reflect the Type A perspectives: The methods are also based on a model where resources are extracted from a stock in the environment to the economy and are thereby depleted, which forms the basis for the calculation of a future effort or cost of future use of the remaining stock. In some cases, however, the size of the stock is not considered in the calculation of the indicator, e.g. (Huppertz et al., 2019; Swart and Dewulf, 2013a)[3]

3.2.1.2. System model B (Type B perspectives). More recently, it is increasingly being discussed that resources can still be accessible after extraction as secondary (anthropogenic) stocks in the economy. It is acknowledged that stocks of resources can occur in both the environment and the economy, and that extraction from the environment to the economy does not automatically render the resources inaccessible – this rather depends on the type of transformation and destination of the resource, which determines whether the resource still remains (potentially) accessible (see also Part I of this submission).

3.2.1.2.1. AADP. Schneider et al. (2011, 2015) developed the AADP method, which reflects the Type B perspective. The Type B perspective can broadly be translated into a system model with a stock of resources in both the environment and the economy, and a flow leaving that stock to a 'space' or 'place' in the environment or the economy, where the resources are inaccessible. The Type B perspective does, however, not specify where this 'non-stock", and the corresponding flow from the stock to the 'non-stock" are to be placed in relation to the economy and the environment in the model. Taking the Type A Perspective as a starting point, Schneider et al. (2011) added the anthropogenic stocks to account for the fact that their availability can reduce the pressure on the natural stocks.

The AADP method considers a combined *stock* in the environment and the economy, as illustrated in Fig. 3. The representation of the flows in a system model is more challenging for AADP, due to some inconsistencies: The first inconsistency refers to a mismatch between the system model for the impact assessment model and the inventory model. The current method only contains characterization factors applicable to primary resource extractions (Schneider et al., 2015). Extractions from the secondary stock are not characterized. (NB: It should be noted that this could be tricky in practice, due to the prevailing structure of the existing life cycle inventory models, which consider flows across the environment-economy boundary only.) Also, for a consistent system model, only flows from the (primary or secondary stock to the 'non-stock 'should be accounted for (net extractions leading to the resources being inaccessible, or, in other words, dissipative flows). This means that not all resource flows from the environment to the economy are relevant for characterization – only those which leave the "availability space", i.e. which are no longer available in the (primary or secondary) stock after use. However, the AADP

[3] The method developed by Huppertz et al (2019) uses the estimated time before depletion in the calculation of its indicators, based on the ratio between stock size and consumption rate, although only for sensitivity analysis.

characterization factors are applied to all flows from the environment to the economy – thus, the position of the *flows* in the system model used in AADP is still the one illustrated in Fig. 2 (System Model A). The second inconsistency refers to the characterization model itself: Extractions from secondary stock (flows) are not considered in the calculation of the characterization factors, but secondary stocks are, i.e. the system model for the stocks is different to the system model for the flows.

3.2.1.2.2. Ecological Scarcity Method for abiotic resources (other than sand and gravel). A similar idea has been brought forward by Frischknecht (2014) and Frischknecht and Büsser Knöpfel (2013). It is similar with regards to the perspective /problem definition of the method presented by Schneider et al. (2011) in that it focuses on whether the elements are in a form that allows (re-) extraction. However, the model does not introduce a new life cycle impact assessment method as such. Rather, it relies on the commonly used ADP model (Guinée and Heijungs, 1995; van Oers et al., 2002). The basis for impact assessment is based on geological availability and extraction rates (System model A), i.e. the dissipative flows are not addressed at LCIA level. Instead, the idea is to take into account that fraction of a resource flow from the environment to the economy in the inventory which is destined for dissipation (Frischknecht, 2014; Frischknecht and Büsser Knöpfel, 2013). Similar to the AADP method, the system model used to derive category indicators / characterization factors (which takes into account the full extraction from primary sources) does not match the model behind the inventory, which only accounts for the fraction of the extraction flow which is destined for dissipative use. There, the calculation of the characterization factor/ category indicator is not in line with the problem definition, i.e. that the accessibility is lost through dissipative use. Furthermore, with this perspective in mind, the inclusion of resource stocks in the economy as suggested by Schneider et al. (2011) (and the losses from these stocks) would be consistent, but is missing from the Ecoscarcity method.

3.2.1.3. System model C (Type C perspectives)

3.2.1.3.1. Ecological scarcity method for sand and gravel. As outlined in Section 3.1.1.3, paragraph “System model B (Type B perspectives) Ecological Scarcity Method for sand and gravel”, for sand and gravel, the Ecoscarcity method takes a perspective type which is different to the perspective taken for other abiotic resources although they are assessed within the same impact category. The usefulness of sand and gravel in the environment (in-situ) is considered as well as their usefulness in the economy (ex-situ). In other words, both the environment and the economy are considered 'systems of concern'. However, a translation of the Type C perspectives into a system model is challenging, since it is not obvious from the outset how the double-system of concern can be accounted for in a single characterization model: Should both functions be considered equally valuable, or one prioritized over the other? Frischknecht (2014) suggest that the current extraction flow is “tolerated”, suggesting an implicit focus on the damage to the role of gravel and sand in the environment. However, the characterization factors can in principle be adjusted depending on political resource management goals, which is a characteristic of so-called distance-to-target methods. The considered production system is only primary. Characterization flows are based on the extraction of gravel and sand from the environment (alluvial gravel pits) to the economy.

The model is based on primary production and the extraction of gravel and sand from a defined stock in the environment – like the System Model A, representing the Type A perspectives (Fig. 2). However, the stock in the environment has two types of valued functions: in-situ ecosystem functions as per Type C perspectives (e.g. groundwater formation), and ex-situ functions as per Type A perspectives (use of resources in the economy) (Fig. 4).

3.2.1.4. System model D (Type D perspectives)

3.2.1.4.1. CEENE and CExD. The illustration of the Type D perspectives for CEENE and CExD looks almost the same as the illustration for perspectives of Type A and C, but without accounting for a stock: Conceptually, all three perspectives assess the removal of resources from the environment to the economy (Fig. 5). The exergy methods CEENE and CExD do not determine a stock for the calculation of the indicator for abiotic resources (metals and minerals). Instead, they define a reference state for each resource, reflecting the most common, most stable compound occurring in the environment, based on the approach of Szargut (Dewulf et al., 2007). For a (natural) resource, one considers a reference compound in the natural environment for each chemical element in the resource. These ground states are the most probable (i.e., most common in the lito-, hydro- and atmosphere) products of the interaction of the elements with other common compounds in the natural environment and typically display a high chemical stability. In order to define a reference state of each element, a reference species can be selected from the atmosphere (gaseous compounds), hydrosphere (dissolved ionic compounds), and the lithosphere (solid compounds). The exergy value of a pure reference compound is prescribed by geochemical data: its relative occurrence in the natural environment. Starting from the exergy of the reference species, the chemical exergy of any (natural) resource can be calculated through thermochemistry. More details are found in Dewulf et al. (2008). In this way, the size of the stock of a resource is not quantified, and not taken into account in the impact assessment. The use of resources is modelled as a one-directional (consumptive) flow from the environment to the economy. If a stock was accounted for with the Type D methods, in order to be in line with the perspective, it should be quantified based on the functions of the resources in the environment in-situ valued under the Type D perspectives, rather than the option to mine for resource use ex-situ (and hence the definitions of resources, reserves etc.). Similar to the system models illustrated in 1b (Section 3.1.1.1), one could also speak of a ‘non-quantified stock’ in the environment and illustrate this system model accordingly.

3.2.2. Outlook: analysis of new system models for new approaches to LCIA methods

3.2.2.1. System model B, variant 3 (tentative). Van Oers et al. (2002) and van Oers and Guinée (2016) proposed an alternative system model, based on a similar understanding of the Type B perspectives, as the models / concepts proposed by Schneider et al. (2011) and Frischknecht (2014) – i.e., whether the elements are in a form that allows (re-) extraction. Their suggested approach is based on emissions of abiotic resources from the technical system back to the environment (air, water, and soil), which are quantified at inventory stage. Hence, it is assumed that the emissions of resources, rather than their extraction from the environment to the economy, constitute the relevant flows which render the resources inaccessible for future use. The idea implies that the emission to the environment constitutes an absolute loss. Also, it implies that flows of resources from the environment to the economy are not assessed. Since in 2016 this idea was still a conceptual one, no category indicator was defined. Thus, the illustration provided in Fig. 6 is a tentative one based on van Oers and Guinée (2016).

3.2.2.2. System model B, variant 4 (tentative). A new variant of the exergy-based LCIA methods which reflects the Type B perspectives has been discussed in the course of the SUPRIM project (Dewulf, 2017, unpublished idea). For abiotic resources, previous LCIA methods based on exergy rely on exergy levels of the resources, based on the most stable and most commonly occurring compounds in the environment used to calculate a reference state. When focusing on the potential functions of resources for use in the economy (Type B perspective), it should be noted that the exergy level of a resource in the economy may well exceed that reference state, for example when pure metals or alloys are produced. The new idea therefore involves a shift of that reference state to the level of pure elements. In this way, exergy losses of abiotic resources can be modelled regardless of whether they occur in the economy or environment, and the exergy concept can be adapted to suit the Type B perspective. Hence, the stock (although not used for the calculation of the indicator, as its absolute size is not considered for the development of the characterization factor) could be considered location independent (Fig. 7). In other words, the chemical form in which the resource occurs is considered independent of whether the resource is located in the environment or the economy. Furthermore, the concept could be used to model exergy gains and losses, and they could be modelled gradually, which would make a definition of stock and non-stock unnecessary.

3.2.2.3. System model B, variants 5 and 6 (tentative). In a recent feasibility report, Zampori and Sala (2017) provide an alternative suggestion for modelling resource use from a Type B perspective. They suggest to adjust the life cycle inventory modelling in a way that resources are not considered lost (or depleted) once they have been extracted from the environment to the economy, but rather, once they are less accessible, or "dissipated to a larger extent". They suggest a tiered approach similar to the MFA-type model introduced by Ciacci et al. (2015), where flows between environment and the economy are tracked *as well as* flows between various types of stocks *within* the economy. They further suggest to differentiate at least three types of resource "stocks": available in the environment, temporarily unavailable ("in-use dissipated") in the economy, and ultimately unavailable ("dissipated") resources, here referred to as 'non-stock' resources. For the latter, it is not clear from the text in their report whether they consider the non-stock to be within the economy or in the environment. Both options are shown in Fig. 8.

The text suggests that flows within the economy (or from the economy to the environment) would be characterized *in addition* to flows from the environment to the economy. However, the authors also state that over a full product life cycle, the impact assessment result are the same as if calculated with a depletion method reflecting Type A perspectives (Fig. 2). This remark does raise a question, since it suggests that the new modelling approach would shift the assignment of the *same* overall impact from *depletion* between life cycle stages, as is usually done with allocation, although here with a focus on resource flows only. This suggests that this approach does not aim to address a different type of *impact*, based on a different understanding of the problem (the characterization model is still based on the concept of depletion).

3.2.2.4. System Model B, variant 7 (tentative). Along a similar line of thinking, another modelling approach was presented at the ISSST conference in 2016 (Laurin, 2016). It reflects the idea that the ultimate loss of a material (or the loss of its accessibility), rather than its extraction from the environment to the economy, is the ultimate problem. This idea has a different scope: it does not focus on LCIA methods and not even on abiotic resources, but rather on the allocation of burdens in inventory modelling in general. The suggestion is to allocate burdens from the life cycle stages of virgin production AND manufacturing to the landfilling process (or incineration, or any other EOL treatment that renders the material unusable/ inaccessible, for that matter). It is assumed that landfilling, rather than production, causes the demand for new virgin materials – since when landfilled (or, in more general terms, when used in a non-recoverable way), the material is lost from the system, and additional material has to be produced from virgin sources. In terms of its interpretation of the problem, this idea is very similar to the idea presented by van Oers and Guinée, 2016; van Oers et al. (2002) – although the ultimate losses could here be interpreted as happening within the economy, since the example of a landfill is given. Fig. 9 provides a tentative illustration with the 'non-stock' positioned between the economy and the environment.

4. Summary of findings, conclusion and outlook

Against a background of misunderstandings regarding the perspectives on resource use in LCIA, and the different issues associated with resource use (see e.g. Dewulf et al., 2015; Drielsma et al., 2016; Sonderegger et al., 2017), the SUPRIM project team developed a framework for a structured approach to the analysis and development of LCIA methods (Part I). This paper provides a structured analysis of different perspectives on resource use as reflected in current methods and other modelling approaches and new propositions on how to model the issues.

Next, the concept of the 'system models' is used in order to analyse whether the methods and other modelling approaches provide a clear reflection of the perspectives. The system-model-based consistency analysis relies on the idea of a simple illustration of stocks and flows positioned in relation to the background system consisting of the economy and the environment. The analysis highlights a number of different types of inconsistencies, which lead to a mismatch between the perspectives taken by the method developers and the modelling choices. In other words, the impact assessment methods or modelling approaches often do not fully reflect the perspectives. This is often due to the modelling choices, which were found to be internally inconsistent. For a consistent model, the position of the stocks and flows in relation to the environment and the economy should be clear, with the flows leaving the stocks. In some cases, the inventory models are not aligned with the impact assessment models used for the calculation of the characterization factors. In other cases, the characterization model itself do not rely on a clear system model (see Sections 3.2.1.2 and 3.2.1.3).

Based on the findings from the analysis of system models, it is likely that the use of inconsistent system models is contributing to confusions and misunderstandings when modelling of resource use in LCA, which, in some cases, have led practitioners not to assess the impacts of resource use at all. Therefore, improved consistency during the development of methods and modelling approaches is crucial to the communication and acceptance of impact assessment methods and/or modelling approaches. Where full consistency is not possible due to e.g. limitations regarding data availability – which may be the reason in the given examples – we suggest that this should be explained along with the method documentation.

It has previously been highlighted that due to a variety of perspectives and problem definitions on resource use, a single best method does not exist. Rather, methods which best fit a certain perspective can be recommended (Berger et al., 2019; Sonderegger et al., 2019). The analysis of suggested model implementation of the Type B perspective shows that even where the method developers seem to share a very similar, perhaps even identical perspective and understanding of the perspective on resource use, different system models have been proposed, as well as implementations at LCIA (e.g. van Oers and Guinée (2016) and LCI level (e.g. (Frischknecht and Büsser Knöpfel, 2013), including through allocation (Laurin, 2016). It can be expected that the different system model approaches will also lead to different LCIA results. Starting from a common perspective type (e.g. Type B), it would therefore be interesting to investigate the sensitivity of the modelling results to different approaches. Furthermore, it would be interesting to further look into the compatibility of different modelling approaches proposed at LCI and LCIA level.

Insights from this paper are twofold: First, for life cycle impact assessment methods, care should be taken to develop the methods and other modelling approaches with a system model that reflects the chosen perspective and is internally consistent, without which a transparent modelling approach and meaningful impact assessment results cannot be achieved. The SUPRIM project has taken the framework developed in Part I as a basis for a top-down method development approach, starting from a stakeholder consensus process to find an agreed-upon perspective type on resource use, then proceeding towards its consistent implementation into a conceptual, and finally, practically implemented LCIA method. Publications on the results of this work are being prepared by the project team. Where inconsistencies have been identified for methods and modelling approaches on resource use, the framework and the criteria applied in this paper can be used to overhaul or dismiss existing approaches or to further develop the conceptual ideas in future research work beyond the SUPRIM project. In that way, the analysis presented in this paper can contribute towards the development of consistent modelling approaches which reflect the intended perspectives.

Second, the paper highlighted that for the Type B perspectives, which are currently receiving more interest, different modelling approaches at LCI and LCIA level have been taken. This raises a question of how Type B perspectives can be best addressed in LCA, and whether changes to the current modelling structure, which characterizes only elementary flows, are necessary at LCI and LCIA level. The compatibility of LCI and LCIA modelling approaches also warrants further investigation.

Authors contribution statement

The project team conceptualized the ideas presented in the manuscript. The first author wrote the first draft of the manuscript and revised the text as requested by the reviewers. The co-authors contributed through the discussion of concepts and ideas, and by providing textual and verbal comments throughout the whole process.

Declaration of Competing Interest

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

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