Assessment of abiotic and biotic resource use for products and regions in the context of sustainable development

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Vanessa Bach

geb. in Berlin

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Promotionsausschuss:

Vorsitzende: Prof. Dr. Claudia Fleck

Gutachter: Prof. Dr. Matthias Finkbeiner

Prof. Dr. Guido Sonnemann

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Abstract

Resources are the basis for a thriving industrial and technological development and therefore for prosperity of present and future generations. Thus, the demand for resources has been increasing during the last decades, leading to various challenges society is facing today. These challenges include (physical and socio-economic) availability of abiotic and biotic resources and raw materials, pollution of the environment as well as social impacts associated with resource extraction and use. To tackle these challenges several national and international strategies and measures are established to manage resource use more efficiently and sustainably. To evaluate the success of these strategies and measures, methodologies are required to comprehensively assess resource use and related impacts.

Existing methods show several shortcomings with regard to the assessment of resource use on product and regional level. First, most of them assess resource use by mass-accounting indicators. The concept of decreasing the amount of resource use is a starting point, but cannot adequately reflect relevant aspects for resource use like supply chain restrictions or environmental and social impacts. Methods going beyond the mass of the used material were developed in the last few years. By considering the static depletion of resources or the extracted energy, the assessment of resource use improved significantly, but still does not properly account for all aspects, e.g. relevant supply chain restrictions. Further, often only abiotic resource use are addressed, whereas biotic resource use and associated impacts are not taken into account. However, to identify trade-offs of different resource use types they have to be evaluated together in a consistent way.

This thesis provides four methodologies to improve the assessment of abiotic and biotic resource use and their implications on product (ESSENZ and BIRD) and regional level (SCARCE) as well as an approach to combine meth-

odologies assessing different resource types to accomplish a consistent evaluation of different resource types. An overall framework for the three methods ESSENZ, BIRD und SCARCE including relevant dimensions and categories considering all three sustainability dimensions is provided. Maturity and applicability of existing indicators is evaluated first, by establishing a criteria set with four main criteria and up to five sub criteria. Then, they are combined with newly developed indicators to establish integrated methodologies. Further, an approach is provided to combine methodologies assessing different resource types to accomplish a consistent evaluation of different resource types.

The integrated method to assess resource efficiency (ESSENZ) is established for the assessment of abiotic resources and raw material use on product level, considering overall 21 indicators to quantify the three dimensions (physical and socio-economic) availability, environmental impacts and societal acceptance in all supply chain stages. In order to determine socio-economic supply chain restrictions a new approach is developed, considering geopolitical, political and regulatory aspects, such as political stability and trade barriers, affecting resource extraction and use. Based on the life cycle assessment (LCA) approach according to ISO 14040/44 eleven categories and associated characterization factors (CFs) are established using the distance-to-target approach. Further, screening indicators are established to evaluate the societal acceptance of resources and raw materials with regard to compliance with social and environmental standards (social norms as well as laws and regulations). They reflect social and environmental impacts of resource extraction and use, which could lead to boycott from consumers.

To assess the availability of terrestrial biotic resources and raw materials in product systems along the supply chain a comprehensive methodology (BIRD) is established, which includes 25 indicators to quantify the five dimensions physical constraints, socio-economic constraints, abiotic constraints, social constraints and environmental constraints. For the evaluation of socio-economic, social and environmental constraints, the approaches applied within ESSENZ are adapted. To identify physical constraints of biotic resources extracted from the environment, the biotic resource availability indicator is established, taking into account the existing stock, replenishment rate, extraction, and the sensitivity of the extracted species (via the threatened species index). For biotic raw materials, which are extracted from the technosphere, the physical availability is determined by accounting for replenishment rate and anthropogenic stocks. In order to determine abiotic constraints phosphorus availability, water availability, land availability and natural disasters are taken into account. The latter is determined by establishing the natural disaster risk indicator for droughts and floods on country level.

In order to allow a consistent assessment, an 8-step approach is proposed to combine assessment methodologies of different resource use types in a consistent way and thus, enables the evaluation of abiotic and biotic resource and raw material use. This approach is applied to ESSENZ and BIRD leading to a combined methodology considering five dimensions with 25 categories and indicators, able to assess product systems, which use abiotic (fossils, metals and minerals) and biotic resource use and raw materials.

ESSENZ and BIRD are established to assess resource use of products and therefore promote resource efficiency on micro-economic level. However, to implement resource efficiency into existing strategies also the macro-economic level has to be addressed. Thus, the SCARCE methodology is developed to assess abiotic resource use on regional level, providing 25 indicators for the two dimensions criticality, consisting of the sub-dimensions (physical and socio-economic) availability and vulnerability, as well as societal acceptance. For the dimension societal acceptance new indicators compared to ESSENZ and BIRD are identified. To enhance the assessment of compliance with social standards, the additional aspects small-scale mining, geopolitical risk and torture are taken into account. For the assessment of compliance with environmental standards additionally the sensitivity of the local biodiversity and water scarcity are considered.

Several case studies are carried out to demonstrate the applicability of the developed methods and to confirm the need for a comprehensive assess of resource use on micro and macro level. ESSENZ is applied in e.g. case studies for smart phones and cars, BIRD in a case study for biofuels produced from rapeseed and soybean and SCARCE for the case study of Germany.

The applicability of the methodologies is further enhanced by providing CFs and indicator results for 36 metals and four fossil raw materials for the dimensions socio-economic availability and societal acceptance in ESSENZ and the dimensions availability and societal acceptance within SCARCE. Further, for the SCARCE indicators data is provided for over 200 countries.

The assessment of abiotic and biotic resource use on product and regional level is improved significantly by establishing four scientifically robust yet applicable methodologies, which consider multiple aspects of resource use in all three sustainability dimensions.

Keywords: abiotic and biotic resources, resource use, supply restrictions, criticality, life cycle assessment, societal acceptance, sustainability

List of core publications

1. V. Bach, M. Berger, M. Henßler, M. Kirchner, S. Leiser, L. Mohr, E. Rother, K. Ruhland, L. Schneider, L. Tikana, W. Volkhausen, F. Walachowicz, M. Finkbeiner (2016): **Integrated method to assess resource efficiency – ESSENZ**, Journal of Cleaner Production, 137 (2016), 118–130

   DOI: 10.1016/j.jclepro.2016.07.077

2. V. Bach, M. Berger, M. Henßler, M. Kirchner, S. Leiser, L. Mohr, E. Rother, K. Ruhland, L. Schneider, L. Tikana, W. Volkhausen, F. Walachowicz, M. Finkbeiner (2016): **Messung von Ressourceneffizienz mit der ESSENZ-Methode – Integrierte Methode zur ganzheitlichen Bewertung**, Springer, Berlin/Heidelberg, ISBN 978-3-662-49263-5, the final publication is available at Springer via: www.springer.com/de/book/9783662492635 (open access, only in German)

   DOI: 10.1007/978-3-662-49264-2

3. V. Bach, M. Berger, N. Finogenova, M. Finkbeiner (2017): **Assessing the Availability of Terrestrial Biotic Materials in Product Systems (BIRD)**, Sustainability, 9(1), 137

   DOI:10.3390/su9010137

4. V. Bach, M. Berger, S. Forin, M. Finkbeiner (2017): **Comprehensive approach for evaluating different resource types** – case study of biotic and abiotic resource use assessment, Ecological Indicators 87C (2018) pp. 314-322

   DOI: 10.1016/j.ecolind.2017.12.049

5. V. Bach, N. Finogenova, M. Berger, L. Winter and M. Finkbeiner (2017): **Enhancing the assessment of critical resource use at the country level with the SCARCE method – case study of Germany**, Resources Policy - The International Journal of Minerals Policy and Economics

   DOI: 10.1016/j.resourpol.2017.07.003

List of abbreviations

AADP	Anthropogenic stock extended abiotic depletion potential
ADP	Abiotic resource depletion potential
BDP	Biotic resource depletion potential
BIRD	Approach to determine the availability of terrestrial biotic materials in product systems
BMUB	Engl: Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety (German: Bundesministerium für Umwelt, Naturschutz, Bau und Reaktorsicherheit)
BRDI	Biotic resource depletion index
CEENE	Cumulative Exergy Extraction from the Natural Environment
CExD	Cumulative exergy demand
CFs	Characterization factors
CML-IA	Impact assessment methodology provided by Centrum voor Milieukunde (engl.: Institute of Environmental Sciences) of Leiden University
DMC	Domestic material consumption
EPI	Environmental Performance Index
ESP	Economic resource scarcity potential
ESSENZ	Integrated method to assess resource efficiency
GDP	Gross domestic product
HDI	Human Development Indicator
HRI	Human rights indicators
LCA	Life cycle assessment
LCC	Life cycle costing
LCIA	Life cycle impact assessment
LCSA	Life cycle sustainability assessment
MIPS	Material input per unit of service
OECD	Organization for Economic Co-operation and Development
PEF	Product Environmental Footprint

ProgRess	German Resource Efficiency Program
RE	Resource efficiency
ReCiPe	Life cycle impact assessment methodology by PRé Consultants, Institute of Environmental Sciences of Leiden University, Department of Environmental Science of Radboud University Nijmegen and National Institute for Public Health and the Environment (Netherlands)
SCARCE	Approach to enhance the assessment of critical resource use on country / regional level
SETAC	Society of Environmental Toxicology and Chemistry
SHDB	Social Hotspot Database
SLCA	Social life cycle assessment
SRSP	Social resource scarcity potential
TMR	Total Material Requirement
UNEP	United Nations Environment Programme
VDI	Engl.: The Association of German Engineers (German: Verein deutscher Ingenieure)

Table of contents

1. Introduction

1.1 Motivation

Resources present the basis for a stable economy adding to the wealth of present and future generations (Meadows et al. 2004; European Commission 2005). In recent decades, the demand for many resources has increased significantly (as shown in **Figure 1**) due to their growing importance for industrial and technological development (Gordon et al. 2006; Behrens et al. 2007; European Commission 2013).

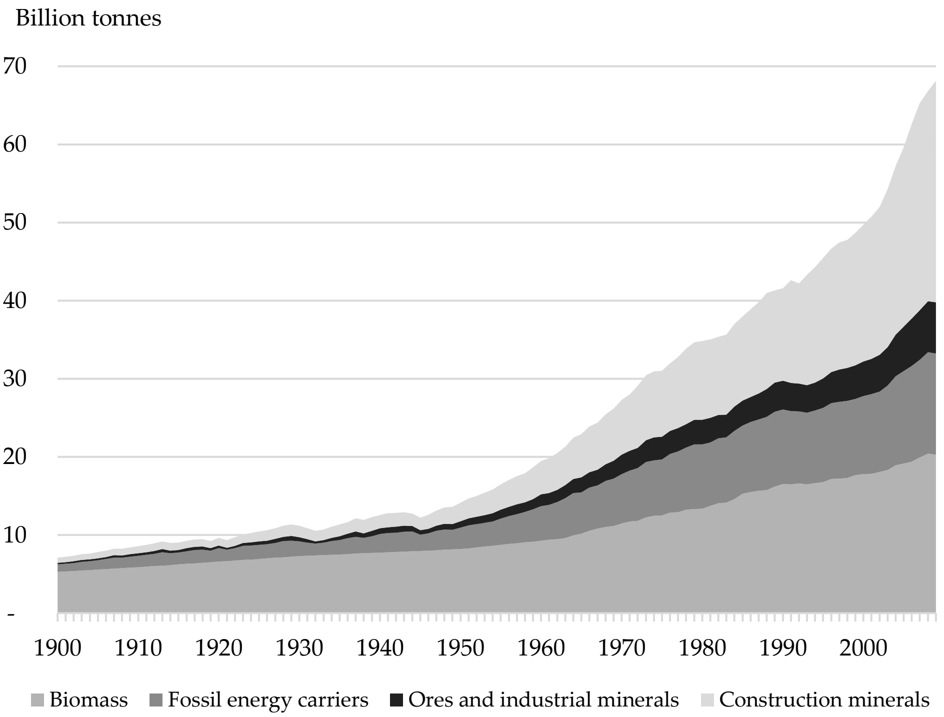


Figure 1: Global material extraction by four material categories, 1970–2010, million tonnes; based on (European Environment Agency 2009)

This led to a steadily growing awareness of their strategic importance (Angerer et al. 2009; European Commission 2011a). Considering the growing world population and rapid industrialization of emerging nations like

China and India the demand for resources is most likely to further increase (Gordon et al. 2006; United Nations Environment Programme 2010). The decoupling approach ("doing more with less") was introduced in the 70ies as one of the first strategies to enhance the efficient management of resources (UNEP 2014; UNEP 2017). Resource use is put in relation to an economic value, usually the country's gross domestic product (GDP) (UNEP 2011). This principle – also referred to as resource efficiency (RE) – is applied in several international strategies, e.g. sustainable development goals (United Nations 2015) as well as national strategies, e.g. Germanys national sustainability strategy (Federal Governance of Germany 2012).

Next to the overall demand of resources used, a growing number of materials is needed for the production of high-tech and clean-tech products (Kleijn and Gerardus 2012; Simmons et al. 2012; Schneider 2014). For example, more than 60 metals are involved in current energy grids (Simmons et al. 2012).

Thus, recently strategies focus on the availability of resources (e.g. the raw materials initiative of the European Commission (2014a)). The availability of resources is influenced by their physical occurrensce as well as by socio-economic factors. The physical occurrence refers to the availability of resources within nature (ore that can be mined) as well as in the anthroposphere (materials that can be recovered from the technosphere (urban mines)). The high amounts of resource mined in the last decades, led to a decreasing availability in nature. Thus, the recovery of resources used within products is an important factor for guaranteeing safe resource supply (UNEP 2014). By retrieving resources from products, the overall resource availability is increased, because retrieved materials (secondary materials) can be used to supplement primary materials (especially for metals, the quality of primary and secondary materials is often similar (Blomberg and Söderholm 2009; Neugebauer and Finkbeiner 2012; Pauliuk et al. 2012)). Further, the recovered materials are available within the region they are retrieved, e.g. rare earth, which are naturally occurring in few countries like China and the US can be recovered in other countries like Germany, where products containing rare earth (e.g. smartphones) are used (UNEP 2012a; UNEP 2013). This way, countries without own natural resource stocks are less dependent on ore producing countries.

Geopolitical, political and regulatory factors influence the availability of resources in every supply chain stage as well. For example, for materials like Coltan, mined in politically unstable countries like the Congo, where armed conflicts, corruption and other injustices occur, the possibility to extract resources or to establish rules and regulations supporting mining operations, might be limited. Impaired political stability as well as other socio-economic factors can lead to availability constraints of resources and raw materials. To ensure that specific resources are available for regions and companies and thus also products, purchasing strategies are frequently implemented by companies and nations (Ferretti et al. 2013).

According to the *Strategy on the sustainable use of natural resources* (European Commission 2005) the environment itself is defined as a resource, which needs to be protected as well. In the last decades the pollution of natural resources like water and soil rose significantly, leading to increasing damage of more and more ecosystems and associated ecosystem services (Costanza et al. 1997; Millennium Ecosystem Assessment 2005; UNEP 2010). Ecosystems services represent the part of the environment human well-being relies on. For example, plants remove pollutants from the atmosphere and therefore regulate the air quality. If these systems are disturbed and do not work properly, damage to human well-being is likely to occur. Several strategies on international level (e.g. Montreal Protocol to phase out ozone depleting substances (United Nations Ozone Secretariat 1987)) as well as national levels (e.g. Germanys Federal Pollution Control Act (Bundes-Immissionsschutzgesetz) (Federal Govrnance of Germany 2010)) are implemented to reduce environmental pollution.

Social aspect caused by the extraction of resources as well as due to their use in the supply chain have to be taken into account. Considering social impacts is imperative to ensure human well-being. Social impacts have become more relevant for consumers purchasing products during recent years. For example, media reports on social grievances like occurrence of child labor during mining operations, led to boycotts by consumers (Kannan 2014; The Guardian 2015). The implementation of the Dodd-Frank Wall Street Reform and Consumer Protection Act (commonly referred to as Dodd–Frank Act) was one of the fist measures set up to force manufactures to disclose labor conditions in mineral extraction (111th United States Congress 2010; Young 2015). Within the European Union a due diligence self-certification for metals and their ores was established to improve socially acceptable labor conditions during mining operations (European Parliament 2014).

Growing economic activity and population calls for a sustainable use of the earth's limited resources. Thus, resource use in line with the requirements of sustainable development is one of the most important goals to be achieved by current and future generations. Sustainability can only be accomplished, when the three aspects (also called pillars) environment, economy and society are taken into account (Finkbeiner et al. 2010). To guarantee that the needs of current and future generations are met (United Nations 1987), access to resources (this applies to the direct use of resources and raw material in production processes as well as indirect use of the environment and its ecosystem services) have to be secured (Lindeijer et al. 2002; Kleijn and Gerardus 2012).

Thus, the increasing demand of resources leads to several challenges for society today:

- (Physical and socio-economic) availability of materials (due to overall demand as well as complexity of products)
- Pollution of the environment and accompanying loss of ecosystem services
- Social conditions associated with resource extraction and use

To tackle these challenges, several strategies and measures are established to efficiently manage resource use and corresponding impacts. These strategies and measures have to be evaluated by adequate assessment methods to guarantee their successful implementation. The development of such methods is the overall goal of this thesis (see chapter 1.4).

1.2 Definitions

As several definitions of the term “resource” exist, it is necessary to clearly define how the term is used in this thesis. First, a summary of existing definitions is given. Then the definition used in this thesis is presented.

In **Box 1** an excerpt of existing definitions for the term “resource” is presented, considering relevant sources on international (United Nations Conference on Sustainable Development (United Nations 2012a)) and national level (German Resource Efficiency Program (ProgRess) (Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety (BMUB) 2012)) as well as in a political (Germanys national sustainability strategy (Federal Government of Germany 2012)) and scientific context (assessing resource use in life cycle assessment (LCA) (Lindeijer et al. 2002; Sonderegger et al. 2017)). Some of these sources clearly define the term “resource”, whereas others use the term without a distinctive definition. For the latter the context in which the term is applied is therefore described.

It is obvious that significant differences between the proposed definitions exist, e.g. terminology (some refer to resources while others use the term natural resources) or inclusion of resource types (some include the environment; others focus on raw materials only).

To put theses definitions into context , an overview is presented (see **Figure 2**) with regard to the term “resource” considering the definitions presented in Box 1 as well as additional sources (e.g. European Commission (2011b), Graedel et al. (2013), Schneider (2014), Dewulf et al. (2015), Schneider et al. (2016a), Drielsma et al. (2016a), Fanning and O’Neill (2016) and Buchert et al. (2017)).

Box 1: Excerpt of existing definitions for the term "resource"

"Natural resources are extractable entities with implications for their present but mainly future availability." (Lindeijer et al. 2002)

"Natural resources [include] raw materials such as minerals, biomass and biological resources; environmental media such as air, water and soil; flow resources such as wind, geothermal, tidal and solar energy; and space (land area)." (European Commission 2005)

In Germanys national sustainability strategy a definition for the term resource is not provided. Within the chapter addressing natural resources the following resource types are addressed: land, biological diversity, wild fish as well as agricultural products and forestry (Federal Government of Germany 2012)

Within the German Resource Efficiency Program (ProgRess) (BMUB 2012) resources include water, air, raw materials, ground/area and biological diversity.

Within the outcome document of the United Nations Conference on Sustainable Development titled "The Future We Want" a distinction is made between resources and ecosystem. It is referred to resources in the context of water, mining, genetic diversity, fishery and energy (United Nations 2012a).

"Resources are aspects of the natural world that have the capacity to produce goods and services that contribute to human welfare." (UNEP 2017)

"Natural resources are material and non-material assets occurring in nature that are at some point in time deemed useful for humans." (Sonderegger et al. 2017)

"*Resources — both energy and materials, renewable and non-renewable, and water and land — are fundamental to human wealth creation, development, health and well-being*." (UNEP 2017)

Within existing definitions "resources" are divided into different resource types, usually into abiotic resources, biotic resources, water and land. This thesis focuses on abiotic and biotic resources only. Thus, the resources water and land are not further addressed as they represent separate research fields (UNEP 2012b; ISO 14046 2014; UNEP 2016). In recent years several new methodologies were published to assess the resource water (Pfister et al. 2009; Berger et al. 2014; Sonderegger et al. 2015; Boulay et al. 2016) as well as for land (Koellner et al. 2013; Ioannidou et al. 2015; van den Berg et al. 2016; Gomiero 2016).

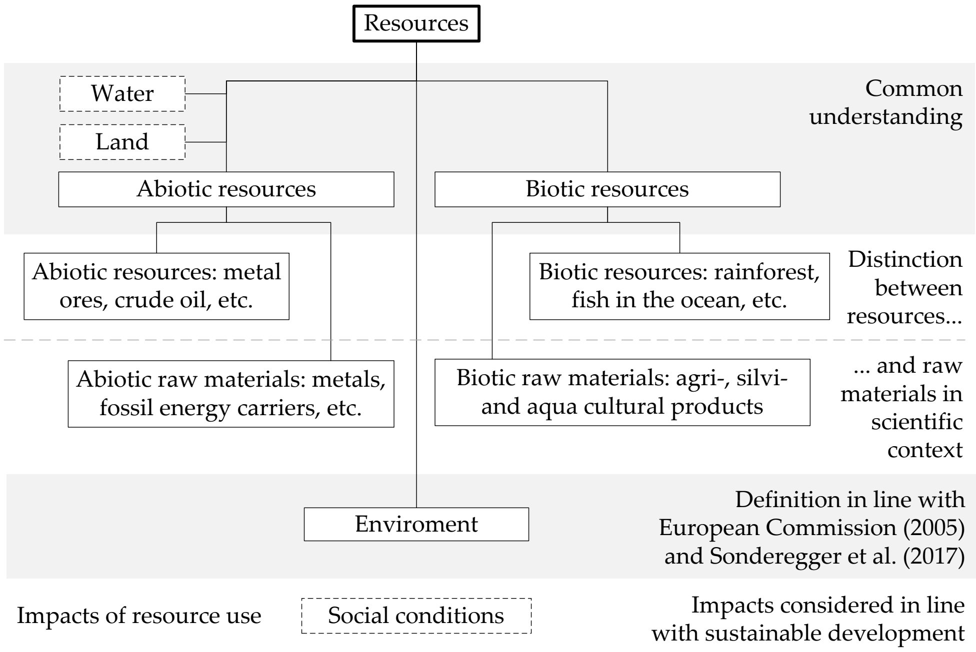


Figure 2: Overview of "resource" definition aspects

In the scientific context, e.g. in LCA (Lindeijer et al. 2002), a distinction is made between resources and raw materials. Resources originate in the environment (e.g. ores, rainforest, wild fish), whereas raw materials are part of the technosphere (e.g. metals, agricultural products such as maize, fish from farms). This division is currently discussed in a working group addressing abiotic resource use (with focus on minerals) set up by the United Nations

Environment Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) and might be revised in the future (Berger and Sonderegger 2017). However, as no final decision is made yet, terminology will be described within this thesis the way it is currently used. A distinction between resources and raw materials is not observed in publications in the policy context.

It can be seen that the definitions in Box 1 often include the environment as a resource as well, especially for definitions in the policy context, e.g. (European Commission 2005). Further, with regard to the use of resources, which has to be addressed in the context of sustainable development, also social aspects need to be included (Schneider 2014; Dewulf et al. 2015; Buchert et al. 2017). Contrary to the inclusion of the environment as a resource of itself, social aspects are considered as additional constraints to availability or as impacts due to resource extraction and use, but not as a resource as such.

Within this thesis the definition by (Sonderegger et al. 2017) is applied: *"Natural resources are material and non-material assets occurring in nature that are at some point in time deemed useful for humans."* This includes material assets as well as the environment. For abiotic and biotic resources a distinction is made between resources and raw materials according to Lindeijer et al. (2002). Additionally, social aspects related to resource extraction and use are taken into account to achieve an assessment of resources in line with sustainable development.

Further definitions of terms applied in the thesis and the corresponding publications can be found in the chapter Glossary.

1.3 Existing methods

In this chapter, existing methods for the assessment of abiotic and biotic resource and raw material use are introduced. They are structured with respect to the following aspects (see **Figure 3**):

- First, methods are divided regarding the considered **resource type**: As shown in chapter 1.3 resources are commonly divided into abiotic resources, biotic resources, water and land. The focus of this thesis is on the assessment of abiotic and biotic resources. Thus, Figure 3 does not include methods for the assessment of water and land.
- Next, assessment methods for of abiotic and biotic resources are divided considering the **addressed levels:** micro (product), meso (company) and macro (regional) level (based on existing categorizations, e.g. Behrens et al. (2007), Huppes and Ishikawa (2009) and Giljum et al. (2011)). Depending on the addressed area differences in the assessment methods exists (these are addressed later in this chapter). This thesis focuses on the assessment of abiotic resources on product and regional level and of biotic resources on product level. Thus, the company level (methods are provided by e.g. Graedel et al. (2012), Bensch et al. (2015) and Association of German Engineers (VDI) 2016)) as well as methods for the assessment of biotic resources on regional level (method is provided by Oakdene Hollins (2014) is not addressed. As the main focus of this thesis is on product level, existing methods to assess resource and raw material use on micro level are introduced in more detail.
- The assessment on micro and macro level is further divided into the **areas:** availability, environment and society. This structure is chosen, because existing methods in general as well as specific for resources usually address one of these areas. Each of these areas can be seen as separate research fields, which usually only overlap when a comprehensive assessment is carried out, as done within this thesis. This is

shown on micro level, where assessment methods for environment and society address abiotic as well as biotic resources.

- The addressed areas are further divided into different **types of methods**, where every method type addresses a different issue(s) with regard to resource and raw material use. These are e.g. depletion or future costs (availability), e.g. climate change or acidification (environment) or e.g. working conditions or human rights (society). These types of methods are further addressed in this chapter.
- Last, for each method type some **examples of methods and indicators** are given. Some aspects are quantified by more than one indicator, e.g. regional criticality, whereas for some aspects one indicator is sufficient, e.g. inventory level. Methods covering more than one aspect, e.g. CML-IA also exist. The most relevant methods and indicators are further described in this chapter.

For assessing the availability of abiotic resources on product level existing methods are often divided into future effort, thermodynamic accounting, scarcity and depletion, criticality and supply risks as well as inventory approaches (type of method). Future effort methods like Eco-Indicator 99 (Goedkoop and Spriensma 2001), IMPACT 2002+ (Jolliet et al. 2003) or *surplus ore potential* (Vieira et al. 2017) are based on the assumption that ore grades will decrease in the future and therefore increasing effort (e.g. in energy and costs) to extract resources will be necessary. Thermodynamic approaches like *cumulative exergy extraction from the natural environment* (CEENE) (Dewulf et al. 2007) or *cumulative exergy demand* (CExD) (Bösch et al. 2007) are applied to determine the efficiency of resource and raw material use in the supply chain. Scarcity and depletion methods (e.g. *abiotic resource depletion potential* (ADP) (Guinée et al. 1993; van Oers et al. 2002), *anthropogenic stock extended abiotic resource depletion potential* (AADP) (Schneider et al. 2011; Schneider et al. 2015) or the *environmental development of industrial products* method (Hauschild and Potting 2005)) usually consider the extraction of the resource in comparison with their availability and therefore determine the scarcity of a resource or their depletion. The most reliable and also applied ADP indicator is the one using the baseline approach (Lehmann et al. 2015), which considers ultimate reserves. The ADP indicator can also be determined based on the reserve base (part of resource, which has reasonable potential for becoming economic within planning horizons (United States Geological Survey (USGS) 2015)) or the economic reserve (part of the reserve base, which can be currently extracted) (Guinée et al. 2002; van Oers et al. 2002).

However, ADP does not account for mineral stocks, which are transferred into the technosphere, but assumes a fixed natural resource stock, which is depleted. As this stock cannot be renewed within human lifetime, it is assumed that extracted resources are no longer available for human use and considered as "lost". However, by transferring them into the technosphere, they can be used for production after their retrieval (Drielsma et al. 2016b; Drielsma et al. 2016a). The AADP indicator by Schneider et al. (2011) and Schneider et al. (2015) addresses this issue by taking anthropogenic stocks into account. Further, the idea to account for the dissipation of materials along the supply chain is introduced to measure resource depletion (e.g. by Zimmermann and Gößling-Reisemann (2013) and van Oers and Guinée (2016)). Dissipated resources are so finely dispersed in the environment that they cannot be extracted in the future and are therefore no longer available for human use.

The approach developed by Graedel et al. (2012) was the first one to introduce the assessment of supply restrictions of abiotic resources and raw materials on product level. Overall eight aspects are considered and indicators are provided for quantification. Several of these indicators, e.g. worldwide governance indicators, is implemented into other methodologies as well. The method also provides indicators to measure the vulnerability of companies and regions. Further, weighting factors are provided to aggregate the indicator results to one overall result for each material.

The economic resource scarcity potential (ESP) developed by Schneider et al. (2016b) was the first approach with the target to implement supply restriction into LCA. ESP takes 10 categories with corresponding indicators into account. CFs are established based on the distance-to-target approach (Müller-Wenk et al. 1990; Frischknecht et al. 2009) and range between zero and one. The results of all ten categories are then aggregated and presented as a single value. The integrated method to assess resource efficiency (hereinafter referred to as ESSENZ) (Bach et al. 2016a), which is part of this thesis, can be considered as the successor of the ESP approach.

The approach developed by VDI (2016) was developed during the same time as ESSENZ and a continuous exchange took place. Thus, the considered aspects match the ones in ESSENZ. Differences occur with regard to the applied indicators and their calculation, because the calculated indicator values are scaled from 0 to 1. The methods also provides indicators to measure the vulnerability of companies.

The geopolitical supply risk (GeoPolRisk) method, developed by (Gemechu et al. 2016), determines the supply risk of abiotic resources based on the political stability of importing countries/producers for 12 importing countries and is applied in a case study of European manufactured electric vehicles (Gemechu et al. 2017). This approach was further developed by Helbig et al. (2016), assuming domestic production to be risk free as domestic production is not subject to supply restrictions. Cimprich et al. (2017) integrate vulnerability aspects as proposed by Sonnemann et al. (2015), applying a cause-effect mechanism (characterization model) by introducing the "inside-out" (potential environmental impacts of resource and raw material use) and "outside in " (potential supply restrictions on resource and raw material use) framework and proposing not to account for the mass of the used material.

Further, the use on inventory indicators like mass are often applied for the assessment of resources and raw materials of products, especially in studies not carrying out a LCA. A prominent mass indicator is Material Input Per Service unit (MIPS) developed by the Wuppertal Institute (Ritthoff et al.

2002). The mass of the used resources and raw materials is simply summed up over the supply chain of the considered product.

Impacts of pollution on the environmental can be quantified by applying the LCA methodology (ISO 14044, 2006). Several impact assessment methods are available to determine environmental impacts. Commonly addressed categories are climate change, acidification, eutrophication, smog and ozone depletion (Lehmann et al. 2015), applying the CML-IA (Guinée et al. 2002) or ReCiPe method (Huijbregts et al. 2017). Further, new methods for quantifying these as well as other categories are introduced regularly (e.g. Boulay et al. (2016), Frischknecht et al. (2016), Bach et al. (2016c) and Huijbregts et al. (2017)).

The assessment of social impacts like human rights or working conditions is usually carried out by applying the social life cycle assessment (SLCA) method (UNEP 2009; Benoît et al. 2010). It evaluates social impacts of products in relation to the different stakeholder groups affected by the manufacturing, use and disposal of the product and provides indicators for quantifying these issues, e.g. unemployment statistics by country. Only few case studies were published in recent years (e.g. Lehmann et al. (2013), Macombe et al. (2013) and Martínez-Blanco et al. (2014)) due to the complexity (e.g. collection of inventory data, lack of quantitative cause-effect chain) of carrying out an SLCA study. Currently, improved indicators for measuring social impacts are introduced, e.g. fair wage indicator developed by Neugebauer et al. (2015a), tackling some of these challenges. Due to these existing challenges, the use of resources is so far not comprehensively addressed by any SLCA case study. Thus, Schneider (2014) developed a screening indicator – the social resource scarcity potential (SRSP) - taking child labor, forced labor and high conflict zones occurring in the mining industry into account. These screening indicators apply data of the Social Hotspot Database (SHDB) (Norris et al. 2013). The SHDB is currently the most comprehensive database providing data for several sectors and regions related to social conditions. Within ESSENZ an enhanced approach based on Schneider (2014) for abiotic resources is introduced, which is further refined in the ESSENZ+ methodology (Bach et al. 2017c). Further, for biotic resources and raw materials screening indicators are introduced in BIRD based on the approach shown in ESSENZ.

Exiting methodologies for the availability of biotic resources can be divided into scarcity and depletion, inventory level, criticality based and thermodynamic accounting approaches. So far existing methods predominately focus on scarcity and depletion of wild fish (e.g. Libralato et al. (2008), Ziegler et al. (2011) and Langlois et al. (2012)). A first approach to assess biotic resource depletion is proposed by Heijungs et al. (1992). The biotic resource depletion potential (BDP) is based on ADP and is further developed within BIRD to establish the biotic resource depletion indicator (BRDI). Another approach was recently published by Crenna et al. (2017), proposing to use the renewability rate as the category indicator. The publication is further discussed in chapter 4.2 of this thesis. On inventory level the amount of biotic resources can also be summed up by applying the MIPS approach (Ritthoff et al. 2002). A first approach to determine criticality based aspects is introduced by VDI (2016). Most existing approaches based on thermodynamic accounting provide CFs for abiotic and biotic resources, e.g. CEENE (Dewulf et al. 2007; Alvarenga et al. 2013).

For assessing the availability of abiotic resources on regional level often material-flow based indicators are applied, e. g. domestic material consumption (DMC) or total material requirement (TMR) (van der Voet 2005), which can be applied for abiotic as well as biotic resources. For abiotic resources the indicator DMC is set in relation to the GDP and represents the lead indicator of the European resource efficiency scoreboard (Scoreboard 2013). Approaches developed in recent years (e.g. Morley and Eatherley (2008), Knašytė et al. (2012), European Commission (2014b), Bastein and Rietveld (2015), Hatayama and Tahara (2015), Buchert et al. (2017) and Blengini et al. (2017)) go beyond the mass of the considered resource and raw material and determine criticality of abiotic resources (potential supply restrictions of raw materials and vulnerability of regions to potential supply restrictions). A thorough overview of existing criticality methodologies for the regional level can be found in the publication by Bach et al. (2017b). This thesis also provides a methodology (SCARCE) for the regional level to assess criticality of abiotic resources as well as environmental and social implications.

For the assessment of environmental impacts like climate change, acidification and eutrophication a variety of different methods exist. For example, in Europe the critical load method is often used to determine acidifying impacts of atmospheric nitrogen and sulphur depositions on the environment (Kruit et al. 2014). Further, indices like the Environmental Performance Index (EPI) are established to determine the environmental performance of a nations policies considering aspects like air quality, unsafe sanitation, fish stocks, protected areas and carbon dioxide emissions (Yale Center for Environmental Law & Policy 2014). Further, methods to determine criticality of resources also include environmental impacts specific for resource and raw material use, e.g. by applying life cycle impact assessment (LCIA) methods (Buchert et al. 2017).

To assess social impacts on regional level indicators and methods address aspects like working conditions or human rights, e.g. Human Development Index (HDI) (UNEP 2016), Indicators of Employment Protection by Organisation for Economic Co-operation and Development (OECD) (2013) and the human rights indicators (HRI) by United Nations (2012b). Existing methods to determine social implications of resource and raw material use address health impacts applying LCIA methods (Bensch et al. 2015) or by taken into account aspects addressed in SLCA e. g violent conflicts, working conditions and corruption of the extracting country (Dewulf et al. 2015; Buchert et al. 2017). SCARCE builds on these approaches to evaluate social impacts of abiotic resource and raw material use on regional level.

Depending on the considered level, differences in the assessment methods exists concerning:

i) considered inventory

ii) aspects to be taken into account as well as

iii) design of indicators and characterization factors (CFs).

i) With regard to the inventory data, all resources used to produce, use and recycle the considered product have to be taken into account on product level. Databases like ecoinvent (Ecoinvent 2016) and GaBi (Thinkstep 2016) support modeling the life cycle of products and also provide data for resource and raw material use. However, there are still some lacks in database for assessing the socio-economic availability of resources and raw materials. For the assessment on regional level import data have to be collected (how this can be achieved is explained in the publication by Bach et al. (2017b), which is part of this thesis).

ii) The aspects considered for the assessment vary depending on the addressed level, e.g. on regional level, dependence of imports is taken into account, which is not relevant on product level.

iii) With regard to the design of the applied indicators and CFs differences occur as well. The assessment on product level is carried out in line with the LCA approach (ISO 14044 2006), which means that the indicators are designed to fulfill the requirements of CFs (for more details see publication by Bach et al. (2016a), which is part of this thesis). For the assessment on regional level, the applied indicators are designed differently, e.g. they are not multiplied with the mass of the used products.

Based on this overview of existing methods, gaps are presented in the following chapter.

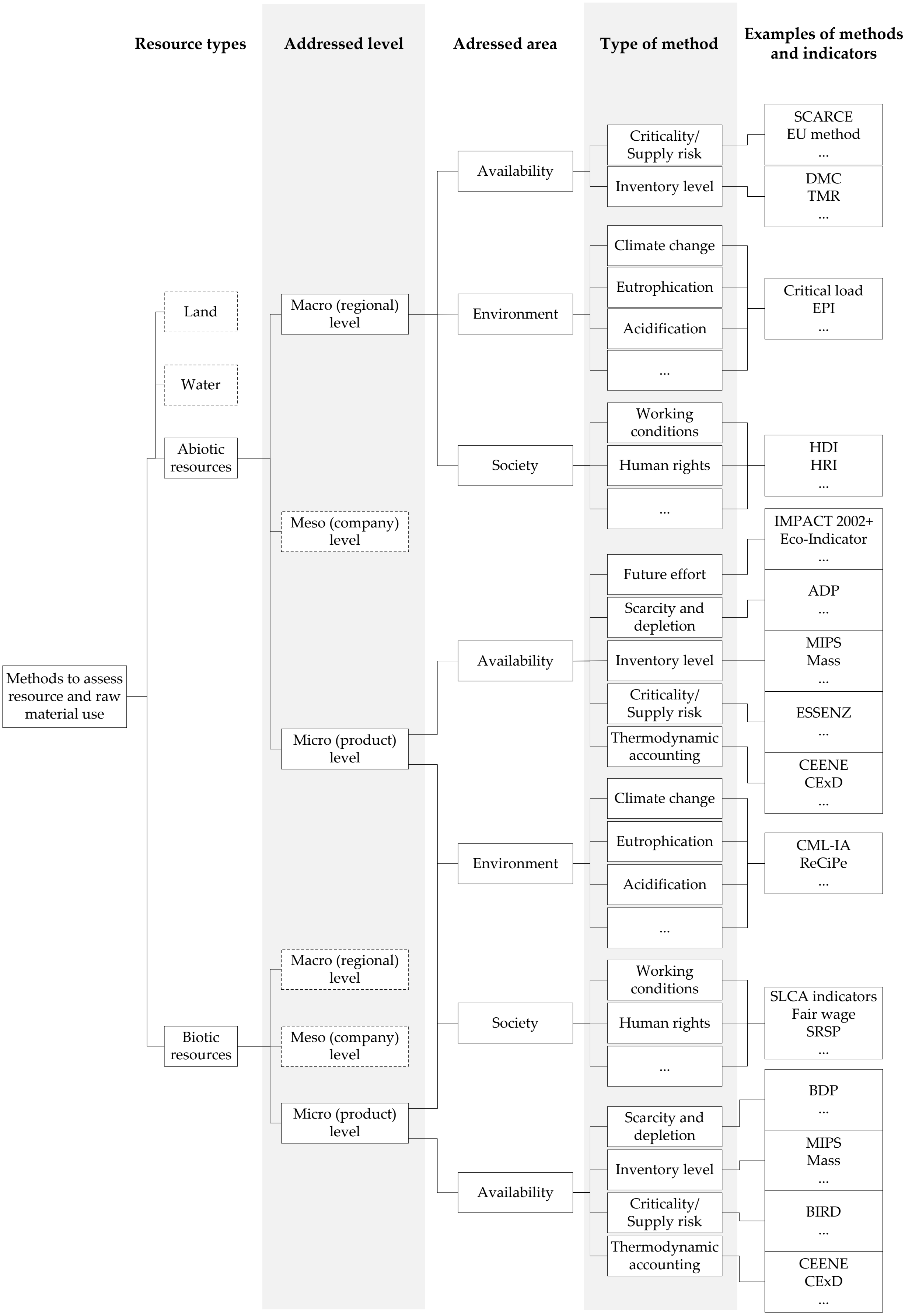


Figure 3: Overview of existing methods to assess resource and raw material use

1.4 Gaps and challenges

Several challenges still exist with regard to the assessment of resource and raw material use on product and region level. In this chapter, main gaps and challenges are introduced and the contribution of this thesis to tackle (part of) these challenge is outlined.

One of the main challenge is that the assessment of resource and raw material use is often still based on mass, even though more sophisticated, impact oriented methods already exist. The importance of the mass used in products and economies is irrefutable, however, not the only aspect relevant with regard to resource and raw material use. By assessing resource and raw material use with mass-accounting indicators only, neither the availability (or criticality) of materials, nor the environmental and social impacts of product systems and economies are taken into account. Aggregating all materials based on their mass further suggests that they are interchangeable. This is not the case as availability and environmental impacts differ depending on the resource and raw materials used. As argued in the introduction even though the basic idea of using fewer resources is good (chapter 1.1), the assessment of resource and raw material use should be carried out in line with sustainable development. Thus, next to the mass of the resource, also economic, environmental and social aspects need to be addressed.

For measuring physical resource availability of abiotic resources several indicators are available (see chapter 1.3). The most applied indicator is the ADP indicator (baseline approach – based on ultimate reserves) (Guinée et al. 1993; van Oers et al. 2002). Recent discussions indicate that the dissipation of abiotic resources should be included in the assessment, because it better reflects the real depletion of abiotic resources. However, anthropogenic stocks, which play a vital role in the availability of materials, are not considered in the dissipation approach. As so far inventory data with regard to anthropogenic stocks are missing, the validity of existing indicators like AADP is limited.

Approaches to address supply restrictions of abiotic resources on product level (Graedel et al. 2012; Gemechu et al. 2016; Helbig et al. 2016; Schneider et al. 2016b; Cimprich et al. 2017) have several shortcomings:

- Not all relevant supply chain restrictions are considered, e.g. price fluctuations
- Spread of the CFs is too small to balance out the mass of the material used in the product system. When CFs are multiplied with the mass, it is the only factor dominating the results
- ESP applies the distance-to-target approach by Müller-Wenk et al. (1990) and Frischknecht et al. (2009), but determines the target values based on a small sample of people and stakeholder groups
- Specific production locations have to be known to include regional supply risks and vulnerability aspects. Often these cannot be determined due to lack of data. This is especially challenging for products consisting of several different components like cars
- Weighting scheme proposed by Graedel et al. (2012) is rather arbitrary and results vary significantly depending on the applied weighting factors (Nassar et al. 2012)
- Approach by Cimprich et al. (2017) does not take into account the mass of the resources used in the product system and therefore neglects an important information with regard to resource and raw material use.

Existing methods assessing supply restrictions on product level published by the year 2015 are further examined in the ESSENZ publication (supplementary material) in chapter 3.1. Further, there are several methods published around the same time as ESSENZ or later (developed by Gemechu et al. (2016), Helbig et al. (2016) and Cimprich et al. (2017)). These could not be included in the evaluation or the development of ESSENZ and are therefore discussed further in chapter 4.2.

Simultaneously, with the development of methods addressing supply restrictions and criticality on product level, discussions began, whether these aspects should be addressed within LCA or rather life cycle sustainability assessment (LCSA) and which area of protection is impacted. On the one hand one could argue that addressing these aspects within LCSA instead of LCA would be more adequate as supply risk and criticality aspects go far beyond environmental impacts currently considered within LCA (Schneider 2014; Sonnemann et al. 2015; Drielsma et al. 2016b). On the other hand, addressing supply restrictions and criticality within LCA case studies would be suitable, because elementary flows (mass of raw material), which are needed to determine supply restrictions are already collected in the life cycle inventory phase (Mancini et al. 2016). Moreover, also current LCIA methods used to assess resources (e.g. surplus ore potential (Vieira et al. 2017) or AADP (Schneider et al. 2015)) take an anthropocentric perspective, which is beyond environmental impacts. With regard to the addressed AoP. Schneider (2014) proposed to establish a new area of protection called *resource provision capability for human welfare*, whereas Sonnemann et al. (2015) proposes to address supply restrictions and criticality within the existing area of protection *natural resources*.

Databases for carrying out the assessment of resource and raw material use on product level are available covering a wide range of various products and materials (Ecoinvent 2016; Thinkstep 2016). However, these databases cannot be applied for the assessment of socio-economic availability. As databases apply economic allocation to assign metal contents from mixed ores to metal datasets, the amount of metals given in the inventory results does not reflect the physically present metal content (this is discussed in more detail in the publication by Bach et al. (2016a), which is part of this thesis).

To measure environmental impacts several impact assessment methods are available for application in LCA. As new methods are constantly being developed, claiming to improve existing methods by refining the cause-effect chains, discussions are ongoing, which methods are the most mature and reliable ones (the categories climate change and ozone depletions are exceptions, because methods are available, which are commonly seen as reliable)

(Joint Research Centre 2011; European Commission 2014c; Frischknecht et al. 2016). Further, for some categories unfortunately reliable methods are still lacking, e.g. for biodiversity, land use as well as toxicity.

To assess social impacts on product level, SLCA is applied (UNEP 2009). However, several challenges exist with applying the approach due to e.g. limited inventory data and challenges in data collection (Martínez-Blanco et al. 2014). Further, applicable and valid impact assessment methods to determine social impacts are missing (Lehmann et al. 2013; Neugebauer et al. 2014). Thus, applying SLCA is currently still challenging and often not practicable. Therefore, the SRSP was introduced by Schneider (2014). Within this approach, social hotspot indices for child labor, forced labor and high conflict zones are set in relation with benchmarks below which social impacts can be seen as not relevant. However, defining small fractions of child and forced labor as neglectable is questionable, because all child and forced labor should be prevented. Within this thesis an improved approach is presented in the publications (Bach et al. 2016a; Bach et al. 2017b; Bach et al. 2017c; Bach et al. 2017a).

For the assessment of biotic resource and raw material use only few methods exist (see chapter1.3). Thus, several gaps are present for a comprehensive assessment of biotic resource use. These challenges include the assessment of biotic resource depletion, because existing approaches do not take into account all relevant aspects beyond renewability rate, e.g. vulnerability of the extracted species. Further, potential availability restrictions along the supply chain as well as social and environmental constraints are not considered comprehensively. Proposed indicators do not cover all relevant aspects and are also not designed to be multiplied by mass (thus cannot be integrated into LCA). Further, they only address terrestrial species, but not aquatic ones, e.g. algae.

For the assessment of abiotic and biotic resource use on regional level the resource efficiency indicator DMC/GDP is often applied (Giljum and Polzin 2009; Scoreboard 2013; Schneider et al. 2016a). Even though the idea of using fewer resources per obtained value is good, high uncertainties are associated with material flow indicators due to errors associated with the used databases, lacking data and model choices (Patrício et al. 2015). Further, as already addressed above for the assessment on product level by only measuring the mass of the used resources other relevant aspects are not taken into account. Within the last years serval methodologies for abiotic resources and one method for biotic resources were introduced going beyond the mass and taking into account relevant aspects such as criticality and environmental impacts. However, these methods still have some shortcomings. None of these methods takes the physical availability into account. This includes the geological deposits as well as the anthropogenic stocks, which could reduce the dependence on imports significantly. Data for geological deposits is available and should therefore be addressed in assessment methods. Including anthropogenic stocks is more challenging, because data is missing.

For the assessment of criticality, not all relevant aspects are taken into account and suitable indicators for the assessment of supply chain restrictions (e.g. price fluctuations) are therefore missing. Further, next to metals and minerals fossil fuels are not considered (further existing shortcomings are addressed in detail in the publication of Bach et al. (2017b), which is part of this thesis). Only the approach developed by Oakdene Hollins (2014) asses the criticality of biotic resources by taking into account two aspects and applying them to four materials. An adequate indicator to measure biotic resource depletion on regional level is lacking.

Environmental impacts are often not taken into account when assessing the use of resources on country level. Even though methodologies and data are often provided by the country itself, overarching indicators and data to assess all countries in a consistent way are lacking for most environmental impacts (e. g. biodiversity loss). Further, next to the country aspects, the individual impacts of the considered resources are not taken into account (e.g., aluminum production leads to higher climate change impacts compared to steel production). Occurrence of accidents during mining operation, e.g. leaking tailing ponds, cannot be quantified, because data is not available.

Further, the applied indicators for the assessment of criticality, environmental impacts as well as social conditions are often only determined considering the global production mix, but not the import mix to determine regional specific impacts, which reflect the supply risk of a region more precisely

Societal aspects (e.g. geopolitical risk) are often not taken into account when assessing resource and raw material use on country level. Bensch et al. (2015) only consider health aspects, whereas Buchert et al. (2017) take into account violent conflicts, working conditions and corruption of the extracting country. However, country based indicators are determined only for the three countries with the highest global production, because indicators applicable for all countries are missing. Thus, countries with smaller production amount but potentially higher social violations are not accounted for.

Existing methods for assessing resource and raw material use on product as well as regional level mostly consider metals and minerals, whereas bio based materials and fossil raw materials are barley taken into account. For a comprehensive evaluation of resource and raw material use, all resource types have to be included into one assessment. Otherwise tradeoffs cannot be identified e.g. by using biofuels instead of fossil fuels, certain agricultural products like rapeseed or soybean are needed. Their cultivation and use leads to several environmental impacts (e.g. land and water use), which have to be compared with the impacts due to fossil fuel extraction and burning to achieve a fair evaluation of the overall resource use.

Out of all these introduced gaps, solutions are provided within this thesis for the following challenges:

- For the availability assessment of abiotic resources
    - On product level
        - availability of biotic resources: mass based approaches are applied, which neglect criticality aspects, environmental impacts and social issues
        - with regard to the assessment of supply restrictions of abiotic resources
            - relevant supply restrictions are not accounted for
            - CFs cannot be applied as defined in ISO 14040/44
            - Mass of the resource and raw material used is not taken into account
            - target values are not based on large enough sample of stakeholder groups
            - fossil raw materials are not accounted for
    - On regional level
        - mass based approaches are applied, which neglect criticality aspects, environmental impacts and social issues
        - physical availability is not taken into account
        - with regard to the assessment of supply restrictions
            - relevant supply restrictions are not accounted for
            - fossils fuels are not considered
- For the assessment of environmental impacts
    - On product level
        - approach to adequately determine maturity and applicability of assessment methods
    - On regional level
        - overarching indicators are missing to assess all countries in a consistent way
        - aspects specific for certain resources are not considered

- For the assessment of impacts on society
  - On product level
    - social aspects within SRSE are only considered above a certain benchmark
  - On regional level
    - impacts on society are not determined for all ore producing countries, because applicable indicators are missing
- For the availability assessment of biotic resources on product level
  - mass based approaches are applied, which neglect criticality aspects, environmental impacts and social issues
  - not all relevant aspects are considered for biotic resource depletion assessment
  - not all relevant supply restrictions are taken into account for the assessment of biotic resources
- abiotic and biotic resources cannot be assessed and compared in a consistent way

The following gaps are not addressed within this thesis:

- For the availability assessment of abiotic resources
  - On product level
    - existing indicators do not adequately account for resource depletion, because dissipation effects and anthropogenic stocks are not taken into account
    - with regard to the assessment of supply restrictions of abiotic resources
      - inventory data from LCA databases cannot be used to determine supply restrictions
      - production locations of all intermediate products are not known
      - area of protection is not defined

        - consideration of vulnerability aspects on product level
        - weighting of indicators to achieve one final results
    - On regional level
        - anthropogenic stocks are not taken into account
- For the assessment of environmental impacts
    - On product level
        - assessing loss of biodiversity and toxicity impacts
    - On regional level
        - Mining accidents are not taken into consideration
- For the assessment of impacts on society
    - On product level
        - low applicability of SLCA approach, e.g. due to lack of inventory data
    - On regional level
        - country and company specific data is missing
- For the assessment of biotic resources
    - On product level
        - existing indicators do not adequately account for resource depletion, because dissipation effects and anthropogenic stocks are not taken into account
        - supply restrictions of aquatic biotic resources have to be considered
- abiotic and biotic resources cannot be assessed and compared in a consistent way
- approaches to assess abiotic and biotic resource use on meso level have to be improved

2. Goal and research targets

In this chapter, the goal of the thesis is introduced and specified by four research questions (chapter 2.1). Further, the core and complementary publications included in the thesis are listed (chapter 2.2) and their connection to the research questions is shown by introducing more specific research targets (chapter 2.3).

2.1 Goal and research questions

The overall goal of this thesis is to enhance the assessment of abiotic and biotic resource use (excluding water and land) and raw material use considering associated impacts in all three sustainability dimensions on product and regional level.

The overall goal is specified in more detail by introducing the following four research questions, which are further specified by research targets in chapter 2.3.

1) **How can the assessment of abiotic resource and raw material use on product level be enhanced towards a more holistic sustainability assessment?**

   As addressed in chapter 1, several relevant aspects are not yet considered in current assessment methods addressing resource and raw material use and associated impacts on product level. Thus, within this thesis a methodology is introduced enhancing the assessment of abiotic resource and raw material use going beyond existing approaches by including additional aspects and providing approaches for quantification (for details see chapters 2.3 and 3.1).

2) **How can the availability assessment of terrestrial biotic resource and raw material use on product level be enhanced?**

As addressed in chapter 1, only few aspects with regard to biotic resource and raw material use and associated impacts were so far assessed on product level. Thus, within this thesis a methodology is introduced which enhances current methods by including additional aspects and providing approaches for quantification (for details see chapters 2.3 and 3.2).

3) **How can the assessment of abiotic and biotic resource and raw material use be made more consistent?**

As addressed in chapter 1.3, existing methodologies mostly focus on one resource type only. An integrated and consistent assessment methodology for both abiotic and biotic resources and raw materials is lacking. Thus, within this thesis an approach is proposed how existing methodologies can be combined to assess different resources types in a consistent way. This approach is applied to two of the methodologies of this thesis leading to a combined methodology able to assess abiotic and biotic resource and raw material use on product level in a consistent way (for more details see chapters 2.3 and 3.3).

4) **How can the assessment of abiotic resource and raw material use on regional level be improved?**

As addressed in chapter 1, even though several methods exist for the assessment of resource and raw material use on regional level some important aspects are still missing. Thus, within this thesis a methodology to further enhance the assessment of resource and raw material use on regional level based on the methodology established for the product level is introduced (for more details see chapters 2.3 and 3.4).

2.2 Publications of thesis

This chapter lists the publication of the thesis. First, the core publications of the thesis are listed. Then, the complementary publications are presented. The core publications answer the above mentioned research questions (for more details see chapters 2.3 and 3), whereas the complementary publications only relate to some of the aspects raised by the research questions (for more details see chapters 2.3 and 3.5).

Overall five core publications are included in this thesis:

I. V. Bach, M. Berger, M. Henßler, M. Kirchner, S. Leiser, L. Mohr, E. Rother, K. Ruhland, L. Schneider, L. Tikana, W. Volkhausen, F. Walachowicz, M. Finkbeiner (2016): **Integrated method to assess resource efficiency – ESSENZ**, Journal of Cleaner Production, 137 (2016), 118–130
DOI: 10.1016/j.jclepro.2016.07.077

II. V. Bach, M. Berger, M. Henßler, M. Kirchner, S. Leiser, L. Mohr, E. Rother, K. Ruhland, L. Schneider, L. Tikana, W. Volkhausen, F. Walachowicz, M. Finkbeiner (2016): **Messung von Ressourceneffizienz mit der ESSENZ-Methode - Integrierte Methode zur ganzheitlichen Bewertung**, Springer, Berlin/Heidelberg, ISBN 978-3-662-49263-5
The final publication is available at Springer via https://www.springer.com/de/book/9783662492635, (open access, in German)
DOI: 10.1007/978-3-662-49264-2

III. V. Bach, M. Berger, N. Finogenova, M. Finkbeiner (2017): **Assessing the Availability of Terrestrial Biotic Materials in Product Systems (BIRD)**, Sustainability, 9(1), 137
DOI: 10.3390/su9010137

IV. V. Bach, M. Berger, S. Forin, M. Finkbeiner (2017): **Comprehensive approach for evaluating different resource types** – Case study of abiotic and biotic resource use assessment methodologies**,** Ecological Indicators 87C (2018) pp. 314-322
DOI: 10.1016/j.ecolind.2017.12.049

V. V. Bach, N. Finogenova, M. Berger, L. Winter and M. Finkbeiner (2017): **Enhancing the assessment of critical resource use at the country level with the SCARCE method – Case study of Germany,** Resources Policy - The International Journal of Minerals Policy and Economics
DOI: 10.1016/j.resourpol.2017.07.003

In addition, the following six publications support the overall framework of the thesis:

i. V. Bach, M. Finkbeiner (2016): **Approach to qualify decision support maturity of new versus established impact assessment methods—demonstrated for the categories acidification and eutrophication**, International Journal of Life Cycle Assessment, 22 (3), pp 387–397
The final publication is available at Springer via:
DOI: 10.1007/s11367-016-1164-z

ii. A. Lehmann, V. Bach, M. Finkbeiner (2015): **Product environmental footprint in policy and market decisions – applicability and impact assessment**, Integrated Environmental Assessment and Management, Volume 11, Issue 3, Pages 417–424,
DOI: 10.1002/ieam.1658

iii. M. Henßler, V. Bach, M. Berger, M. Finkbeiner, K. Ruhland (2016): **Resource Efficiency Assessment—Comparing a Plug-In Hybrid with a Conventional Combustion Engine**, Resources, 5(1), 5
DOI: 10.3390/resources5010005

iv. L. Schneider, V. Bach, M. Finkbeiner (2016): **LCA Perspectives for Resource Efficiency Assessment**, in: LCA Compendium - The Complete World of Life Cycle Assessment –Special Types of Life Cycle Assessment, M. Finkbeiner (ed.), Springer: Dodrecht, The Netherlands, pp 179-218, ISBN 978-94-017-7608-0
DOI: 10.1007/978-94-017-7610-3_5

v. V. Bach, M. Berger, T. Helbig, M. Finkbeiner (2015): **Measuring a product's resource efficiency – a case study of smartphones**, Proceedings of CILCA 2015 – VI International Conference on Life Cycle Assessment, March 13-16, Lima, Peru, ISBN 978-9972-674-11-2

vi. V. Bach, M. Henßler, M. Berger, K. Ruhland, L. Schneider, M. Finkbeiner (2017): **Integrated method to assess resource use in the context of sustainable development (ESSENZ+)**, Proceedings of LCM 2017

2.3 Research targets and connection of papers

In this chapter, the four research questions introduced in chapter 2.1 are further detailed by specific research targets. To better guide trough the research targets, the research questions are presented in this chapter again. Overall four to five research targets (numbered consecutively with a) to d) or e)) specify the research questions. Further, in Table 1 the publications listed in chapter 2.2 are related to the research questions and targets to demonstrate their contribution to this thesis.

1) **How can the assessment of abiotic resource and raw material use on product level be enhanced towards a more holistic sustainability assessment?**

   a) Identify methodological gaps of existing approaches and determine relevant dimensions and categories for the assessment of abiotic resource and raw material use on product level

   b) Identify existing indicators to quantify these relevant dimensions and categories

   c) Develop indicators for quantification of missing dimensions and categories

   d) Join existing and newly developed indicators to establish an integrated methodology to assess abiotic resource and raw material use

   e) Test developed methodology in case studies

2) **How can the availability assessment of terrestrial biotic resource and raw material use on product level be enhanced?**

   a) Identify methodological gaps of existing approaches and determine relevant dimensions and categories for the assessment of terrestrial biotic resource and raw material use on product level

b) Identify existing indicators to quantify these relevant dimensions and categories

c) Develop indicators for quantification of missing dimensions and categories

d) Join existing and newly developed indicators to establish an integrated methodology to assess abiotic resource and raw material use

e) Test developed methodology in case study

3) **How can a consistent assessment of abiotic and biotic resource and raw material use be enhanced?**

a) Identify shortcomings of existing approaches

b) Develop approach to combine existing methodologies for abiotic and biotic resource and raw material use in a consistent way

c) Combine developed methodologies for abiotic and biotic resource and raw material use assessment on product level to establish a combined methodology

d) Test developed methodology in case study

4) **How can the assessment of abiotic resource and raw material use on regional level be improved?**

a) Identify methodological gaps of existing approaches and determine relevant dimensions and categories for the assessment of abiotic resource and raw material use on regional level

b) Identify existing indicators to quantify these relevant dimensions and categories

c) Develop indicators for the quantification of missing dimensions and categories

d) Join existing and newly developed indicators to establish an integrated methodology to assess abiotic resource and raw material use on regional level

e) Test developed methodology in case study

Within **Table 1** the research questions and targets are set in relation to the publications of this thesis (which are introduced in chapter 2.2). It can be seen that the publications addressing the ESSENZ methodology (Bach et al. 2016a; Bach et al. 2016b; Bach et al. 2017c) primarily answers the first research question and associated targets (for more details see chapter 3.1). The publication addressing the BIRD methodology gives answers to the second research question (for more details see chapter 3.2). The third research question is answered by the publication addressing the approach for evaluating different resource types (for more details see chapter 3.3) and the fourth research question is answered by the publication addressing the SCARCE method (for more details see chapter 3.4). The complementary publication relate to several research targets, but address mostly the first research question and related targets (for more details see chapter 3.5).

Table 1: Overview of publications of this thesis (main papers as well as additional publications) related to the established research questions and targets

	Research questions and targets	**1**					**2**					**3**				**4**				
		a	b	c	d	e	a	b	c	d	e	a	b	c	d	a	b	c	d	e
Core publications	ESSENZ paper	✓	✓	✓	✓	✓	✓		✓							✓				
	ESSENZ book	✓	✓	✓	✓	✓														
	BIRD						✓	✓	✓	✓	✓									
	Approach for combining assessment methods for different resource use types											✓	✓	✓	✓					
	SCARCE															✓	✓	✓	✓	✓
Additional publications	PEF Impact assessment methods		✓					✓												
	Maturity of impact assessment methods		✓					✓												
	LCA perspective for resource use efficiency assessment						✓	✓				✓								
	Case study on smartphone					✓														
	Case study on Plug-In Hybrid vs. conventional combustion engine					✓														
	ESSENZ +			✓																

2.4 Structure of thesis

This thesis comprises of six chapters (see **Figure 4**). An introduction to the field (motivation), and the presentation of existing methods as well as gaps and challenges are given in this first chapter. In chapter 2 (Goals and research targets) the research questions and targets are presented and set in relation to the core and complementary publications the thesis consists of. These publications are introduced in more detail and presented in chapter 3 (Results). Further, it is demonstrated to which of the identified shortcomings the publications propose solutions. The results are discussed and an outlook with regard to remaining challenges is given in chapter 4 (Discussion & outlook). In chapter 0 conclusions are drawn. In the last chapter (chapter 6 – References) the sources referred to throughout this thesis are given. Sources referred to within the publications are not part of this chapter, but can be found within the references chapters of the individual publications. The thesis further provides a glossary and an appendix, where the complementary publications as well as updated data of the introduced methodologies are presented.

Chapter 1: Introduction

Importance of topic is introduced

Existing methods and gaps are presented

Chapter 2: Goal and research targets

Goals and research targets are presented

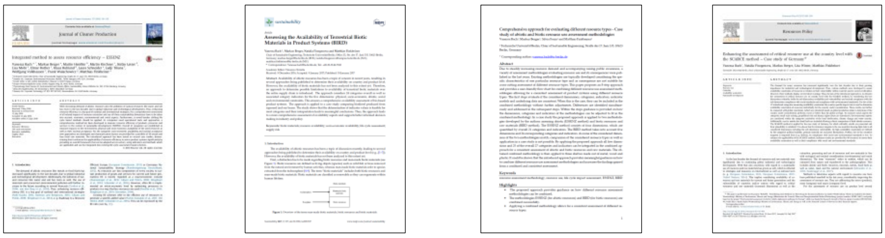
Connection to publications is shown

Chapter 3: Results

Publication of thesis are composed

Connection to research targets are shown

3.1 3.2 3.3 3.4



Chapter 4: Discussion and outlook

Remaining and newly identified challenges are discussed

Future work needed in this field is identified

Chapter 5: Conclusion

Conclusion of thesis are presented

Figure 4: Structure of thesis

In the following chapter, the publications are introduced and their contribution to answer the research questions and targets is described further.

3. Results

In this chapter, the publications of this thesis are presented and their contribution in answering the research questions and targets is explained.

3.1 Assessment of abiotic resource and raw material use on product level

This chapter contains the following publications:

V. Bach, M. Berger, M. Henßler, M. Kirchner, S. Leiser, L. Mohr, E. Rother, K. Ruhland, L. Schneider, L. Tikana, W. Volkhausen, F. Walachowicz, M. Finkbeiner (2016): **Integrated method to assess resource use efficiency – ESSENZ**, Journal of Cleaner Production, 137 (2016), 118–130

DOI: 10.1016/j.jclepro.2016.07.077

V. Bach, M. Berger, M. Henßler, M. Kirchner, S. Leiser, L. Mohr, E. Rother, K. Ruhland, L. Schneider, L. Tikana, W. Volkhausen, F. Walachowicz, M. Finkbeiner (2016): **Messung von Ressourceneffizienz mit der ESSENZ-Methode - Integrierte Methode zur ganzheitlichen Bewertung**, Springer, Berlin/Heidelberg, ISBN 978-3-662-49263-5, The final publication is available at Springer via www.springer.com/de/book/9783662492635, but is not displayed in this thesis due to its size

DOI: 10.1007/978-3-662-49264-2

The publications introduce an integrated method to assess resource use efficiency (ESSENZ) for abiotic resource and raw materials use on product level along the supply chain in the context of sustainable development. First, within an iterative procedure the methodological gaps of existing methods and relevant dimensions and categories for the assessment of resource use in general as well as for abiotic resource and raw material use on product level are determined (contribution to research target *1a;* as well as to research target *2a* and *4a*). Overall the following four dimensions with 21 associated categories are identified:

- availability (with the sub dimensions physical and socio-economic availability)
- societal acceptance with the sub dimensions
    - compliance with social standards
    - compliance with environmental standards
- environmental impacts

The identified categories are quantified by existing (contribution to research target *1b*; and to research target *2b*) as well as newly established approaches and indicators (contribution to research target *1c;* as well as to research target *2c*). For the categories socio-economic availability and societal acceptance, new approaches are developed and CFs are provided for a portfolio of 40 raw materials. Further, existing and newly developed methods are joined and establish the integrated methodology ESSENZ (contribution to research target *1d*). In a last step, the methodology is applied to a case study of two cables (contribution to research target *1e*).

The CFs and indicator values were updated since the publications were issued (see appendix 10.1).

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Integrated method to assess resource efficiency – ESSENZ

Vanessa Bach [a,*], Markus Berger [a], Martin Henßler [b], Martin Kirchner [c], Stefan Leiser [d], Lisa Mohr [e], Elmar Rother [c], Klaus Ruhland [b], Laura Schneider [a], Ladji Tikana [f], Wolfgang Volkhausen [e], Frank Walachowicz [g], Matthias Finkbeiner [a]

[a] Technische Universität Berlin, Chair of Sustainable Engineering, Straße des 17. Juni 135, 10623 Berlin, Germany
[b] Daimler AG, Corporate Environmental Protection, RD/RSE, 70546 Stuttgart, HPC G211, Germany
[c] Evonik Techn. & Infr. GmbH, Rodenbacher Chaussee 4, 63457 Hanau, Germany
[d] Knauer Wissenschaftliche Geräte GmbH, Hegauer Weg 38, 14163 Berlin, Germany
[e] Thyssenkrupp Steel Europe AG, Environmental and Climate Affairs, Sustainability, Kaiser-Wilhelm-Str. 100, 47166 Duisburg, Germany
[f] Deutsches Kupferinstitut, Am Bonneshof 5, 40474 Düsseldorf, Germany
[g] Siemens AG, Corporate Technology (CT RTC PET SEP-DE), Siemensdamm 50, 13629 Berlin, Germany

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ABSTRACT

With increasing demand of abiotic resources also the pollution of natural resources like water and soil has risen in the last decades due to global industrial and technological development. Thus, enhancing resource efficiency is a key goal of national and international strategies. For a comprehensive assessment of all related impacts of resource extraction and use all three sustainability dimensions have to be taken into account: economic, environmental and social aspects. Furthermore, to avoid burden shifting life cycle based methods should be applied. As companies need operational tools and approaches, a comprehensive method has been developed to measure resource efficiency of products, processes and services in the context of sustainable development (ESSENZ). Overall 21 categories are established to measure impacts on the environment, physical and socio-economic availability of the used resources as well as their societal acceptance. For the categories socio-economic availability and societal acceptance new approaches are developed and characterization factors are provided for a portfolio of 36 metals and four fossil raw materials. The introduced approach has been tested on several case studies, demonstrating that it enhances the applicability of resource efficiency to assess product systems significantly by providing an overall framework that can be adopted across sectors, using indicators and methods which are applicable and can be integrated into existing life cycle assessment based schemes.

1. Introduction

The demand of abiotic resources like metals or fossil fuels has increased significantly in the last decades due to global industrial and technological development. Additionally, the pollution of natural resources like water and soil has risen as well. The use of materials and concurrent environmental pollution will further increase in the future according to several forecasts (Gordon et al., 2006; van den Berg et al., 2016). Thus, enhancing resource efficiency (RE) is a key goal of national and international strategies (Aoki-Suzuki, 2016; Bontoux and Bengtsson, 2016; Giljum and Polzin, 2009; Klinglmair et al., 2014), e. g. Roadmap to a Resource Efficient Europe (European Commission, 2011) or Germanys National Sustainability Strategy (Bundesregierung Deutschland, 2012). As resources are key components of every society to sustain production of goods and services for current and future generations RE is mostly regarded as a macroeconomic concept (Eisenmenger et al., 2016; Giljum and Polzin, 2009; Klinglmair et al., 2014; Schneider et al., 2016). However, often RE is implemented on micro-economic level by optimizing processes or products in a way that less resources are used (Henßler et al., 2016; Klinglmair et al., 2014; Schneider, 2014).

Existing RE schemes refer to the efficient use of resources to generate a specific added value (Fischer-Kowalski et al., 2011; ISO 14044, 2006; Schneider et al., 2016). This can be expressed by the RE ratio (see Eq. (1)).

* Corresponding author.
E-mail address: vanessa.bach@tu-berlin.de (V. Bach).

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$$\text{Resource } \mathit{efficiency} = \frac{\text{added value}}{\text{resources}} \qquad (1)$$

Added value often refers to economic values (e. g. Gross Domestic Product (Scoreboard, 2013)) but can also include physical values as proposed by ISO 14045 (2012) depending on the overall goal of the evaluation (for more information regarding the added value see supplementary material – section 5.1).

Contrary to life cycle impact assessment practice existing methods determining the denominator *resources* in resource efficiency assessments so far typically only consider the mass of used metals, minerals and fossil energy carriers (Klinglmair et al., 2014). For an economy-wide perspective mostly material flow indicators like the Domestic Material Input are used (European Commission, 2001). On product level Material Input Per Service Unit (Ritthoff et al., 2002) is applied (e. g. Hinterberger et al. (1997), (Welfens et al., 2016) and von Geibler et al. (2016)). Even though the basic idea of using fewer resources per added value is good, by only measuring the mass of the used resources other relevant aspects associated with the extraction and use of resources (e. g. environmental pollution) are not taken into account. Thus, by applying only mass based indicators and no additional indicators measuring for example environmental impacts little information for a comprehensive RE assessment is provided (Bach et al., 2014; Behrens et al., 2007; Eisenmenger et al., 2016; Schneider et al., 2016; Steen, 1999).

The European Commission already expanded their definition of resources in the year 2005, when the protection of environmental compartments was included in the Strategy on the Sustainable Use of Natural Resources (European Commission, 2005). Furthermore, the environmental dimension was included in the resource efficiency scoreboard, which is a scheme to assess the resource efficiency of Europe and its member states (Scoreboard, 2013). The scoreboard also considers a few social impacts (e. g. condition of infrastructure). Furthermore, the academic community agrees that other aspects besides the mass of a used material have to be considered when determining the resource efficiency of products and/or companies (e. g. BIO Intelligence Service (2012), Fischer-Kowalski et al. (2011), Geldermann et al. (2016), Horton et al. (2016), Schneider et al. (2016) and University of the West of England (2012)). Thus, for a comprehensive assessment of all related impacts of resource extraction and use the existing framework for RE has to be expanded to be integrated into existing sustainability frameworks (Horton et al., 2016; Robèrt et al., 2002; Sonnemann et al., 2015; United Nations, 2016). Therefore, all three sustainability dimensions have to be taken into account: economic, environmental and social dimension (see Fig. 1). As the availability of resources is a precondition for economic development (Eisenmenger et al., 2016; UNEP, 2010), the economic dimension can be expressed through security of resource supply. Restrictions to resource availability can limit the productivity of companies which rely on certain resources to be available anytime to produce goods and services. Thus, they might be forced to discontinue their production if resources they rely on become scarce. This would not only damage the company itself, but also the country/region where the company produces, pays taxes, provides jobs and healthcare to people etc. and therefore ultimately the whole society (BIO Intelligence Service, 2012; Eisenmenger et al., 2016; Gemechu et al., 2016; Rosenau-Tornow et al., 2009; Schneider, 2014). Further, a differentiation between long-term (also called physical availability) and medium-term (also called socio-economic availability) has to be made. Long-term availability refers to the resources in the earth crusts as well as anthropogenic stocks (e. g. electric components consisting of various metals like printed circuit boards in dump sites or buildings). Both have direct influence on availability: if a resource cannot be extracted from either of these sources, it is not available for industrial processes. As it will be very unlikely that this situation occurs in the next years it is referred to as a long-term availability (Schneider, 2014; Schneider et al., 2016).

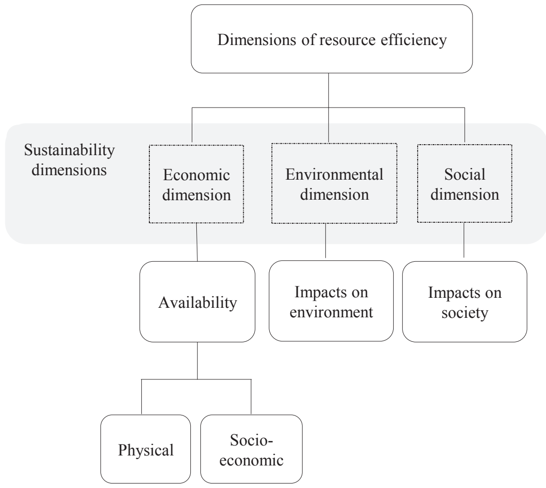

Fig. 1. Considered dimensions for resource efficiency assessment in the context of sustainable development.

Medium-term availability is influenced by socio-economic aspects (e. g. political stability) inhibiting the supply security of resources and leading to a restriction in availability. For example political instabilities of countries due to corruption can disrupt the capacity to effectively implement robust policies including ones related to resource extraction, export, etc. Thus, the availability of a specific resource produced in such a country could be limited. This aspect as well as other socio-economic factors can lead to restrictions of resource availability at different supply chain stages. Availability and criticality of resource supply on macro (country), meso (company) and micro (product) level has been a topic of discussion in various working groups recently (Buchert et al., 2012, 2009; Eggert et al., 2007; European Commission, 2014; Gemechu et al., 2016; Graedel et al., 2012; Klinglmair et al., 2014; Rosenau-Tornow et al., 2009; Schneider et al., 2013; Sonnemann et al., 2015). However, existing approaches are often only applicable for assessing the risk of limited availability on country level (e.g. Eggert et al. (2007), Erdmann et al. (2011), and European Commission (2014)) or are not easily integrated into existing approaches already applied by companies like Life Cycle Assessment (LCA) (e. g. Graedel et al. (2012) and Schneider et al. (2013)).

The LCA method according to ISO 14040 (2006) and ISO 14044 (2006) has been used to assess environmental impacts over the entire life cycle of products for several years (Finkbeiner et al., 2006). Environmental impacts refer to pollution of the environmental compartments land, water and soil. Many industries apply LCA and use existing Life Cycle Impact Assessment (LCIA) methods and indicators to assess the environmental performance of their products (Guinée et al., 2002). To avoid shifting impacts and to capture all potential effects associated with resource use life cycle based approaches should be used as a basis for evaluation. By considering the life cycle of the product system important aspects regarding resource efficiency such as recycling and reuse of resources (Ardente and Mathieux, 2014) in the different supply chain stages are measured as well.

Last, in the context of sustainable development also the social dimension has to be considered in RE assessment to assure that impacts on society (e. g. social inequality, meaning that some people do not have access to products or services they need like food or health care) due to production and use of resources are managed. The guide for Social Life Cycle Assessment (SLCA) published by the UNEP/SETAC Life Cycle Initiative (United Nations Environment Programme, 2009) is applied to assess social impacts over the life cycle of products (Benoît et al., 2010). However, SLCA is less established than LCA (Jørgensen, 2013; Jørgensen et al., 2013) due to limited data inventory and challenges in data collection (Martínez-Blanco et al., 2014). As social conditions highly depend on the geographic location (Benoit-Norris et al., 2012; United Nations Environment Programme, 2009) data has to be collected for every production site individually (Dreyer et al., 2010). Different to environmental data, where emissions can be estimated based on the state-of-the-art of the plant, social conditions can vary depending on the companies involved, the region where the plant is operating etc. Furthermore, applicable and valid impact assessment methods to determine social impacts are missing (Lehmann et al., 2013; Neugebauer et al., 2014). However, some data and indicators exist, which can be used as a starting point (Benoit-Norris et al., 2012; Dreyer et al., 2006; Martínez-Blanco et al., 2014). Currently the most comprehensive database is the Social Hotspot Data Base (SHDB) (Norris et al., 2013), which provides data for several sectors and countries on social conditions and can be used to identify social hotspots of product systems (Benoit-Norris et al., 2012; Martínez-Blanco et al., 2014).

Even though several sectors have guidelines how to measure RE (e.g. Geraghty (2011), Clean Technology Centre (2012), García et al. (2013), Manara and Zabaniotou (2014), Heinemann (2016), Geldermann et al. (2016), and Wiedemann et al. (2016)), general guidance is missing. Existing approaches mostly focus on sector specific aspects but do not take general aspects which are valid for different sectors into account and therefore do not comprehensively evaluate the RE.

To assess RE in the context of sustainable development companies need operational tools and approaches. Thus, a comprehensive method has been developed to measure and assess RE of products in the context of sustainable development (ESSENZ method), which will be explained in more detail in the next sections.

2. ESSENZ method

Following the ESSENZ method (further referred to as ESSENZ) is introduced (Bach et al., 2016). As many companies already use LCA for assessing their environmental impacts, ESSENZ is established to be integrated into LCA (ISO 14040, 2006). All three sustainability dimensions are considered within ESSENZ (see Fig. 2). The environmental impacts are measured by using existing LCIA methods and indicators (see section 2.1). The economic dimension is considered by assessing the physical (long-term) and socio-economic (medium-term) availability of resources (see section 2.2). Two screening indicators are developed to be applied in ESSENZ to measure social impacts (see section 2.3).

The ESSENZ method was developed by Technische Universität Berlin (TUB) in cooperation with the six European companies Daimler, Evonik, Knauer, ThyssenKrupp, German Copper Institute (Deutsches Kupferinstitut) and Siemens during a three year project funded by the German Federal Ministry of Education and Research. In Fig. 3 it is shown how the applied indicators and methods of ESSENZ were determined in the project. First in a bottom-up approach existing methods and indicator for evaluating RE or one of the individual dimensions were identified (e. g. approaches by Graedel et al. (2012) etc. to assess socio-economic availability) including company internal approaches as well as the work done by TUB (e.g. Schneider et al. (2011)). They were analyzed by means of meta criteria and correlation analysis (for more information please see supplementary material – section 1). Based on these results a preselection of indicators was made. In a top-down approach aspects with regard to RE were determined (e.g. compliance with social standards) and their relevancy was discussed within project group meetings. Not all of the identified aspects were seen as relevant enough to be included in the approach, e. g. differences of metals mined in underground or surface mines regarding time frame of extraction and development of mines were determined as minor. For the relevant aspects new indicators or methods were developed (see section 2.1 and 2.2). Data availability had to be checked to guarantee that values can be determined for a variety of materials (thus, to ensure that the overall method is applicable in practice). Then, the newly developed indicators were calculated for as many materials as possible (it became apparent that data availability was a limiting factor). Furthermore, several correlation analyses were carried out to determine if the number of the overall indicators could be reduced. The preselected indicators were tested on several case studies (e. g. Henßler et al. (2016)) to verify the applicability of the indicators and methods as well as to test if the results are reasonable. Based on the results the established indicators were reduced to a set of reliable and applicable indicators (e. g. as toxicity results are not mature enough at this point especially for metals (Joint Research Centre, 2011; Potting et al., 1999; Westh et al., 2015) the category was not included despite its relevance). An iterative approach was chosen to finalize the newly developed indicators and methods: they were applied in several case studies, improved, applied in case studies again, improved again, etc. This way the adequacy of results could be ensured. This led to the final selection of indicators and methods.

The ESSENZ concept is developed with focus on abiotic resources metals and fossil raw materials (this includes but is not limited to fossil energy carriers). Thus, except the environmental dimension, where the chosen indicators are also valid for other abiotic resources (e. g. minerals), the indicators of the dimensions' availability and societal acceptance are more specific for these materials.

2.1. Availability

As mentioned in section 1 the availability of resources can be divided into physical (long-term) as well as socio-economic (medium-term) availability.

2.1.1. Physical availability

The physical availability is composed of the availability of geological and anthropogenic stocks. To measure the physical availability the Abiotic Depletion Potential (ADP) indicator (baseline approach – based on ultimate reserves) (Guinée et al., 2002, 1993; Oers et al., 2002) can be used. Several options to calculate the ADP indicator are available (e. g. based on economic resources or ultimate reserves) (Guinée et al., 2002). However, as shown by Schneider et al. (2015), Drielsma et al. (2016a, b) for assessing the availability of resources the baseline approach “ultimate reserves in the earth crusts” should be applied. The baseline approach is also the approach, which has been used in LCA case studies for many years (Lehmann et al., 2015; Schneider et al., 2015). The assessment of metals ($ADP_{elemental}$) and fossil raw materials (ADP_{fossil}) is carried out separately.

To account for the availability of anthropogenic stocks the anthropogenic stock extended abiotic depletion potential (AADP) is applied (Schneider et al., 2015, 2011). However, as the name of the indicator already suggests it does not only measure the depletion

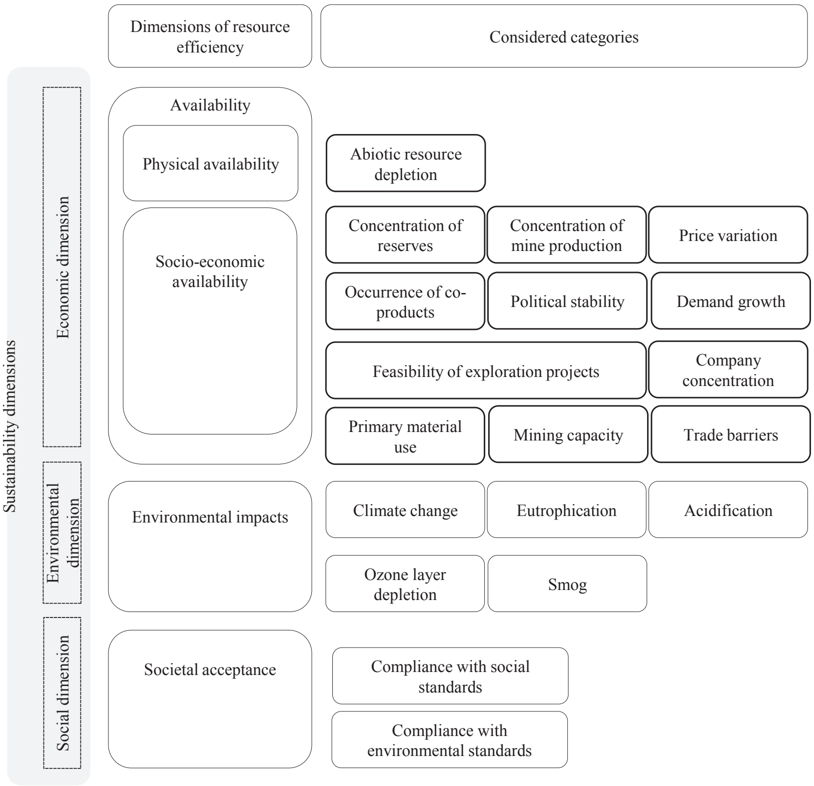

Fig. 2. Overview of considered dimensions and categories in ESSENZ.

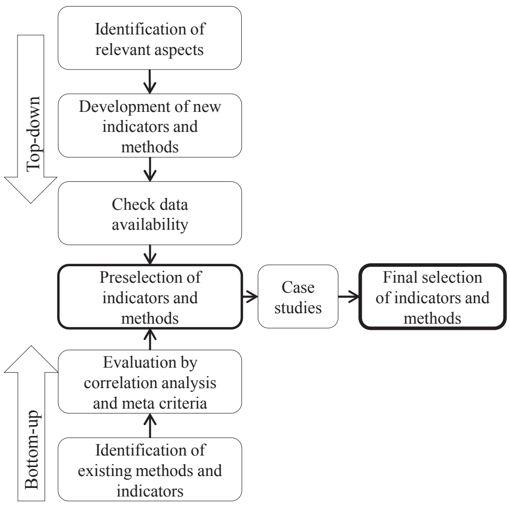

Fig. 3. Combine top-down and bottom-up approach to determine the final indicators and methods used in ESSENZ.

potential of anthropogenic stocks but also considers abiotic (geologic) *ultimately extractable reserves* (resource for which economic extraction is currently or potentially feasible (Schneider et al., 2015, 2011; USGS, 2015)). Therefore, by applying AADP and ADP together the geological resources are overrepresented. However, as only few values for AADP are available, it can only be used for product systems, where these few materials occur. To cover a wider range of materials the ADP is applied in addition (for more information see supplementary materials section 2).

2.1.2. Socio-economic availability

Additional to physical also socio-economic aspects can influence the availability of resources significantly and have to be evaluated for a comprehensive RE assessment. Based on this existing work (with focus on Graedel et al. (2012) and Schneider et al. (2013)) eleven potential economic constraints leading to supply shortages along the product's value chain are quantified in ESSENZ (see Table 1). In Fig. 4 the stages of the supply chain considered in ESSENZ and related socio-economic aspects restricting availability are shown. Overall the supply chain is divided into four stages: ore stocks, mining of ores, raw materials and (intermediate) product. In all four stages constrains to availability can occur. Besides the physical availability, the concentration of resources is the most important restriction influencing the availability of ore stocks. In the extraction stage concentration of production, company concentration, mining capacity, feasibility of exploration projects, occurrence as co-product and political stability of ore extracting countries can impact the availability. Trade barriers, price fluctuations, demand growth and primary material use can affect the availability in the raw material stage. For (intermediate) products various socio-economic constrains occur, which can lead to a

Table 1
Overview of the eleven considered categories reflecting socio-economic availability including a description and related category indicators.

Category	Description	Category indicator
Company concentration	Company concentration for producing and trading companies	Herfindahl-Hirschman-Index (HHI) is calculated by squaring the market share of each company or country with regard to the production or reserves (Rhoades, 1993)
Concentration of reserves	Reserve concentration based on reserves in countries	
Concentration of production	Concentration of mine production based on production in countries	
Mining capacity	Overall mining time of a material considering current production	Reserve-to-annual-production ratio (based on data from (USGS, 2015) and BGS (Brown et al., 2014))
Feasibility of exploration projects	Political and societal factors influencing opening of mines	Policy Potential Index (Cervantes et al., 2013)
Occurrence as co-product	Companion metals within host metal ore bodies	Percentage of production as companion metal (Angerer et al., 2009)
Trade barriers	Materials underlying trade barriers	Enabling Trade Index (Hanouz et al., 2014)
Political stability	Governance stability of raw material producing countries	World Governance Indicators (World Bank Group, 2013)
Demand growth	Increase of demand over the last five years	Percentage of annual growth based on past developments (based on data from BGS (Brown et al., 2014))
Primary material use	Recycled content of a material	Percentage of new material content (Graedel, 2011)
Price fluctuation	Unexpected price fluctuations	Volatility (Federal Institute for Geosciences and Natural Resources, 2014)

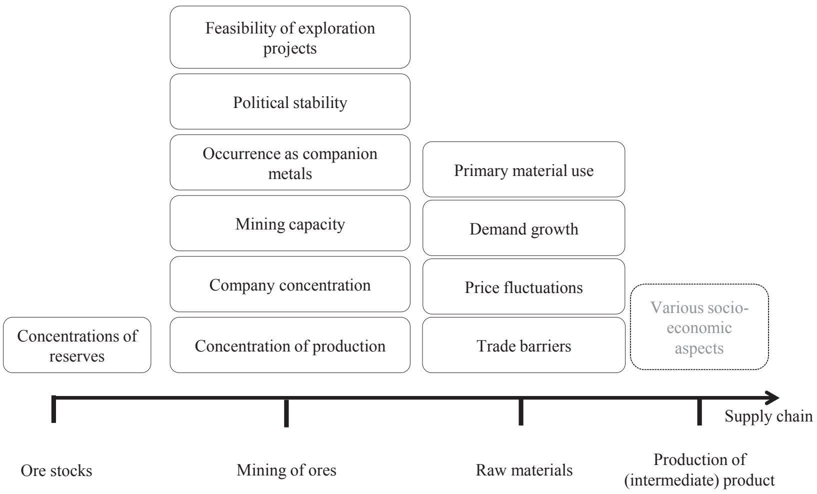

Fig. 4. Supply chain stages and related socio-economic aspects restricting availability of resources for companies.

limited availability for companies. However, as these restrictions are mostly product dependent they have to be determined individually for each (intermediate) product. Thus, characterization factors (CFs) are not provided for the supply chain stage (intermediate) product. As the identified aspects of the other supply chain stages can also influence the availability of (intermediate) products, the approach can be transferred to measure restrictions for (intermediate) products.

Following the eleven categories are described and the approach for the determination of the CFs is explained. Unless otherwise mentioned United States Geological Survey (USGS – United States Geological Survey, 2015) and British Geological Survey (BGS – Brown et al., 2014) data are used to calculate the indicator results. All indicators are calculated according to the same principle: the higher the determined value, the greater possible supply restrictions. If necessary the indicator values were reversed to follow this principle.

- Concentration: A high concentration of one activity (e.g. mining) refers to the extent to which a relatively small number of companies or countries account for a large share of this activity (e.g. Rosenau-Tornow et al. (2009) and Graedel et al. (2012)). High concentrations increase the risk of limited accessibility of a resource. Within ESSENZ the concentration of reserves, concentration of production and company concentration is quantified by means of the Herfindahl-Hirschmann-Index (HHI) (Rhoades, 1993) as done by Erdmann et al. (2011), Graedel et al. (2012) and Schneider et al. (2013) (see supplementary material – section 3.1 for more details).
- Mining capacity: The category reflects how long a *reserve* can be extracted considering the current conditions (e. g. amount of recoverable ores with regard to technological and economic feasibility) before all mines are exhausted. Thus, the calculated number reflects the time (in years) until new mines have to be developed[1]; no statement can be made with regard to the

[1] Development includes the discovery process of the reserves as well as the mining of the discovered ores and returning the land to its natural state after extraction is finished.

Table 2
Qualitative information about occurrence as main and companion metals by Angerer et al. (2009) and transferred quantitative data used in ESSENZ.

Qualitative criteria as reported by Angerer et al. (2009)	Quantitative criteria used in ESSENZ
Only mined as main product	0
Mostly mined as main product	0.33
Mostly mined as companion product	0.67
Only mined as companion product	1

physical availability of the resource. As the development of new mines typically takes around 10–15 years restrictions to availability due to supply bottlenecks might occur when the capacity of existing mines lasts for only a few more years and new mines are not under development yet. The capacity of existing mines is determined by the indicator static lifetime (see Eq. (2)).

$$\text{Static lifetime}_i = \frac{\text{Reserves}_i}{\text{Annual production}_i} \tag{2}$$

To determine the lifetime of a raw material *i* its *reserves* are set in relation to the annual production.

- Feasibility of exploration projects: Laws and regulations, societal conditions (e. g. civil movement) and other framework conditions (e. g. infrastructure) can support as well as restrict development of new mines. When these aspects complicate the development of a new mine the time until the mine is operating can be prolonged for several years or even worse in some cases the mine is not able to be opened at all. Thus, the amount of extracted raw materials decreases, which can lead to restrictions to availability. The feasibility of exploration projects (FEP) is determined by multiplying the raw materials' *i* share of global production (sgp) per country *x* with the Policy Potential Index (PPI) (Cervantes et al., 2013) (see Eq. (3)). The PPI assesses the current regulatory situation within a country regrading mining activities (e. g. explorations of new mines) by considering the countries policies e. g. on taxation, environmental regulations, administration of regulations, or infrastructure (Cervantes et al., 2013).

$$FEP_i = \sum \left(sgp_{x,i} \times PPI_x \right) \tag{3}$$

- Occurrence as co-product: Main reason to put a mine in operation is typically the existence of one main product that shall be extracted. Additionally, other metals are present and are extracted as well. These are called companion metals (co-products) as they are mined next to the main metal(s). The guarantee that these co-products are mined is low as feasibility of mining is only evaluated based on the market value of the main metal(s). If the economic importance of the main metal(s) is decreasing and the mining activities are diminished or discontinued, the co-product is not extracted further as well. To determine if a metal occurs as a main or companion product, qualitative values by Angerer et al. (2009) were transformed into quantitative values according to Table 2. Occurrence as co-product can influence the availability over the whole supply chain. In ESSENZ the category is considered for the step mining of ores (see Fig 4).

The quantitative values are assigned by dividing one (which is set as the highest value) by three (as numbers for three other criteria have to be assigned). The criteria only mined as main product is set to zero as restrictions to availability are not to be expected in this case.

- Trade barriers: Availability of raw materials can be restricted by barriers of trade regarding export (e. g. export duty) of these materials. If an ore producing country limits the export of its produced raw materials to few individual countries the availability to companies in certain countries can be impacted. Trade barriers (TB) are measured according to the same principle as FEP: by multiplying the raw materials' *i* share of global production (sgp) per country *x* with the Enabling Trade Index (ETI)[2] (Hanouz et al., 2014) (see Eq. (4)).

$$TB_i = \sum \left(sgp_{x,i} \times ETI_x \right) \tag{4}$$

Trade barriers can occur over the whole supply chain. In ESSENZ restrictions to trade are considered for the step raw materials (see Fig 4).

- Political stability: In unstable countries, where political systems and legal procedures are not reliable, the risk of limited availability of raw materials rises as potential revolutions or riots but also corruption or financial crises may interrupt production. Political stability (PS) of raw material producing countries is determined according to the same principle as for FEP: by multiplying the raw materials' *i* share of global production (sgp) per country *x* with the Worldwide Governance Indicators (Kaufmann et al., 2011; World Bank Group, 2013) (see Eq. (5)). Overall six key aspects of governance for over 210 countries are established: voice and accountability, political stability and absence of violence, government effectiveness, regulatory quality, rule of law and control of corruption. As all six world governance indicators reflect parts of an unstable system, in ESSENZ they are all combined as an aggregated evenly weighted index ($WGII_x$).

$$PS_i = \sum \left(sgp_{x,i} \times WGII_x \right) \tag{5}$$

Politically unstable systems can influence the availability of raw materials or products over the whole supply chain. In ESSENZ only the effect on raw materials is considered.

- Demand growth: Demand describes the need for goods as raw materials. Increasing demand is referred to as demand growth. When the demand growth is higher than the actual production possible restrains to availability can occur. Demand growth (DG) of raw materials is determined by calculating their production increase (or decrease) over the last five years (see Eq. (6)).

[2] The ETI – established by the World Economic Forum – ranks countries regarding their policy for trading goods (Hanouz et al., 2014).

$$DG_i = \frac{\sum_1^5 \left(\frac{global\ production\ of\ year\ n+1}{global\ production\ of\ year\ n} - 1\right)}{4} \tag{6}$$

- Primary material use: During production primary as well as secondary materials might be used. If more secondary materials are utilized less primary materials have to be produced. As a result the demand for this primary material is reduced and its overall availability increases. To determine the effects of primary material use, the recycled content[3] of the raw material is determined based on data published by Graedel (2011). To determine the primary material use (PMU) the recycled content (given in percentage) is subtracted from 100% (see Eq. (7)). Is the recycled contend low, more primary material is used. The bigger the recycled content the less primary material has to be used. Thus, the higher the PMU value the higher are possible restrictions to availability as more primary materials have to be produced.

$$PMU_i = 100\% - recycled\ content_i \tag{7}$$

- Price fluctuation: Prices of raw materials always fluctuate depending on current market situations. For predictable fluctuations compensation can be expected, as purchasers consider them in their calculations. However, when unexpected fluctuations of raw material prices occur and compensation is not possible, availability of raw materials can be restricted. These fluctuations can be quantified by the volatility indicator applied by Federal Institute for Geosciences and Natural Resources (2014).

For determining the CFs for the dimension socio-economic availability the developed 4-step approach is explained as follows:

Step 1) Determination of indicator values of the eleven categories as introduced in section 2.1 for the individual raw materials. In ESSENZ a portfolio of 36 metals and four fossil raw materials is considered as data for these materials is available.

Step 2a) Targets for all eleven categories c are determined (see supplementary material –section 3.2). These targets were established based on expert judgment and a stakeholder survey (not published). However, these targets are default values, which can be adapted by practitioners and stakeholders according to their preferences.

Step 2b) The indicator values are then set in relation to the target (see Eq. (8)) to determine the Distance-to-Target (DtT) value based on the ecological scarcity approach (Frischknecht et al., 2009; Müller-Wenk et al., 1990).

$$DtT - value_{i,c} = \left(\frac{indicator\ value_{i,c}}{target_c}\right)^2 \tag{8}$$

Is the DtT value lower than 1, no constraints on availability are expected and the DtT value is set to zero. When the DtT value is 1 or greater than 1 a possible limitation to availability occurs. The larger the number the higher the probability of possible limitations. The chosen target values have a significant influence on the overall result as they determine whether a material is assigned a possible restriction to availability or not. When the targets are set too low materials with no risk would be classified as risky and an overestimation of the limitations in the product system occurs. Whereas possible limitations cannot be identified when the target is set too high, which could lead to an underestimation of the overall restrictions. As setting the target values is a scientifically informed value choice, but not a scientific result as such, sensitivity analyses are recommended to address the associated uncertainties.

Step 3) Normalization of the DtT values is carried out (see Eq. (9)) to determine the normalized DtT (nDtT) value.

$$nDtT\ value_{i,c} = \frac{DtT\ value_{i,c}}{normalization\ value_i} \tag{9}$$

Contrary to the ecological scarcity approach, where the normalization factor is equivalent to the critical flow (e. g. amount of greenhouse gas emissions released during a year) in the considered area (e. g. Switzerland) (Frischknecht et al., 2009; Müller-Wenk et al., 1990), in ESSENZ the critical flows are based on global production data. As ESSENZ is applicable for products with global supply chains (which is the case for almost all industrial products (Berger et al., 2015, 2012)) a global normalization factor was chosen. The normalization values were determined based on USGS (2015) and BGS (Brown et al., 2014) data. By normalizing with global production data the overall amount of the resource currently produced is taken into consideration. For raw materials with small amounts of production, e. g. gallium, the above mentioned effects (quantified in the eleven categories) can be even worse for their availability than for raw materials, where the overall annually produced amount is high.

Step 4) By dividing the DtT values by global production the CFs are expressed in small numbers. Common production systems however use large amounts of materials (e.g. to produce a car (Henßler et al., 2016)). Thus, if the nDtT values are multiplied with the raw material flows, predominantly the amount of the raw material but not the potential raw material specific risk of restriction to availability determines the result. Therefore, for ESSENZ to be able to assess different product systems also ones with large amounts of materials, the nDtT values are scaled up to 1.7 × 1013 (this number was chosen as it presents the highest global production value of the raw material portfolio considered). According to Eq. (10) the final CFs are calculated.

$$CFs_i = \begin{cases} nDtT\ value_{c,i,max} \gg 1.7 \times 10^{13} \\ other\ values\ of\ category\ are\ calculated \gg \dfrac{1.7 \times 10^{13}}{nDtT\ vlaue_{c,i,max}} \times nDtT\ value_{c,i} \end{cases} \tag{10}$$

The highest value of each category ($nDtT\ value_{c,i,max}$) is set to 1.7 × 1013. The CFs of the other raw materials are calculated by applying the rule of three[4] (Swetz et al., 2001).

[3] The recycled content refers to the annual amount of material scrap consumed divided by the amount of material produced (Schneider et al., 2013).

[4] The rule of three (also referred to as the Golden Rule) is a method, which supports solving basic linear equations with four terms where three of the terms are known (Swetz et al., 2001).

The calculated CFs can be found in the supplementary materials – section 6.1. They range from zero to 1.7 × 1013: A value of zero means that the material has no potential restriction to available in this category (e. g. for aluminum no potential limitations to availability occur due to demand growth). The higher the value of the CF the higher are the potential possible restriction to availability. For example, for the category trade barriers the CF of aluminum is lower as the CF of antimony. Thus, the possible restrictions to availability due to trade barriers are lower for 1 kg aluminum as for 1 kg antimony (all CFs refer to 1 kg material). However, with regard to assessing these materials within a product system also their amount has to be taken into account. In the example presented here possible limitations to availability will be higher for aluminum than for antimony when the amount of the used aluminum is much higher than the amount of antimony.

As the developed approach is in line with existing LCIA methods for determining environmental impacts it can be applied as well as interpreted accordingly. To demonstrate the relation between existing LCIA methods and the developed approach the scheme of ISO 14040 for impact categories is applied for an established method as well as for a category of the developed approach (see supplementary material – section 3.3).

2.2. Societal acceptance

To assess RE in the context of sustainable development social aspects have to be considered. As assessing social impacts is challenging (Lehmann et al., 2013; Neugebauer et al., 2014) the screening indicator *compliance with social standards* was developed for ESSENZ based on the approach by Schneider (2014). This approach considers on the one hand (some of the) social impacts related to the product system (Missimer et al., 2016) and adds on the other hand additional motivation for a company to apply the indicator as consumers are more and more interested in compliance with social standards by companies (e. g Tsurukawa and Manhart (2011), Kannan (2014), The Guardian (2015) and Osburg et al. (2016)). Thus, societal acceptance can be an additional limitation for companies with regard to purchasing materials. In the worst case a certain material cannot be used by a company because of its low societal acceptance, even though it is available from a physical and socio-economic perspective.

The developed indicator is based on SHDB data and provides information for the supply stage extraction. Impacts in other stages are not covered. To quantify the compliance with social standards the aspects child labor (CL), high conflict zones (CZ) and forced labor (FL) are considered. The indicator were chosen according to Schneider (2014) based on relevance for the mining and minerals sector, high public interest and low societal acceptance. The SHDB (Norris et al., 2013) provides a social hotspot index (Benoit-Norris et al., 2012; Norris et al., 2013) for the mining sector (to assess metals) as well as oil, coal and gas sector (to assess fossil raw materials) on country level for these three aspects (Pelletier et al., 2016). The category indicator result for a material *i* quantifying the compliance with social standards (SC – social compliance) is determined by multiplying the three social hotspot index (ranging from 0 to 10) with the global production shares (sgp) of different countries *x* and summing them up (see Eq. (11)). To be methodological consistent with the approach of the socio-economic availability the result is squared. Therefore, it will be spread so that small values become smaller and big values become bigger. Thus, differences between low and high impacts are more significant.

$$SC_i = \sum [sgp_x \times (CL_x + CZ_x + FL_x)]^2 \tag{11}$$

Finally, according to the same principle as in Eq. (10) the results are scaled to a range of 0–100.

Next to social standards also compliance with environmental standards is gaining in importance for consumers (Balanay and Halog, 2016; Evgeny et al., 2016; Kirchner, 2012). Contrary to global impacts like climate change consumers expect that local impacts like eutrophication and eco toxicity are prevented by using proper technology. Thus, for ESSENZ a screening indicator for compliance with environmental standards is established.

To quantify the compliance of a metal or fossil raw material with regard to environmental standards (EC – environmental compliance) the Environmental Performance Index (EPI) (Yale Center for Environmental Law and Policy, 2014) is applied. The EPI consists of overall 16 sub indicators to measure the performance of countries with regard to their environmental protection efforts. For determining the environmental compliance of countries the sub indicators Critical Habitat Protection (CHP), Marine Protected Areas (MPA) and Terrestrial Protected Areas (TPA) are chosen. The assumption is made that the way a country takes care of their protected areas is similar to their overall compliance with environmental standards during extraction of raw materials. The global production shares of different countries (sgp_x) are multiplied with the EPI indicators, squared and summed up (see Eq. (12)).

$$EC_i = \sum [sgp_x \times (CHP_x + MPA_x + TPA_x)]^2 \tag{12}$$

Then, the result is scaled to 0–100 according to the same principle as in Eq. (10) to have the same dimension as the category indicator assessing social compliance.

The calculated CFs can be found in the supplementary materials – section 6.2. They range from 0.07 to 100 for social compliance and from 0.18 to 100 for environmental compliance. The higher the value of the CF the higher is the potential of being not compliant with social or environmental standards. For example, the potential of antimony producing countries not be compliant with social standards is higher as for beryllium producing countries. On the other hand, these antimony producing countries are more likely to comply with environmental standards as beryllium producing countries.

2.3. Environmental impacts

As the environmental impacts are evaluated over the entire life cycle consequently the whole life cycle has to be modelled. For several impact categories e.g. climate change, eutrophication, etc. LCIA models and methods are available, which have been applied in LCA case studies for many years. Thus, no new indicators and methods were developed, but rather existing methods were chosen to be integrated into ESSENZ. Based on recent publications determining the maturity of LCIA models and methods (e. g Bach and Finkbeiner (2016), Joint Research Centre (2010) and Lehmann et al. (2015)) the CML-IA method (Guinée et al., 2002) for the categories climate change, eutrophication, acidification, ozone depletion and formation of photochemical oxidants (smog) is chosen to be applied in ESSENZ (for more information see supplementary materials – section 1.2).

3. Case study

Following the results of a hypothetical case study are displayed. This case study is simple on purpose as its main goal is to demonstrate the applicability of the introduced ESSENZ approach. Thus, several simplifications were made with regard to the system boundaries. The goal of this case study is to demonstrate, how the ESSENZ method can be applied and how results can be obtained.

However, the approach was also tested for applications to more complex products like cars (e.g. Henßler et al. (2016)). ESSENZ can be used to compare two (or more) options or to analyze one specific product system. For the case study two cables (a silver and an aluminum cable) with the same function (transmission of electricity) used in sound systems are compared. As the electrical conductivity of the silver cable is higher than for aluminum less material has to be used for the same function. Thus, for one cable of silver 0.24 kg silver and for one cable of aluminum 0.44 kg aluminum are used. The plastic coating and other components are not considered for simplicity. Furthermore, utilized fossil fuels in the upstream processes are only considered for the environmental impacts but not in the assessment of availability. The results for the considered dimensions and categories are shown in Fig. 5 (results for the environmental dimension are shown in the supplementary information – section 4). On the top left the result for the physical availability is demonstrated (only $ADP_{elemental}$ is displayed as no fossil raw materials were considered in the example and the AADP value of silver is missing) showing that silver has a much higher risk of restricted (geological) availability than aluminum, meaning that less extractable silver stocks exist in comparison to aluminum. On the top right the (first) result for the socio-economic availability is shown: here the overall results of both materials are set to 100%. This way it can be demonstrated which category influences the overall supply risk of each metal the most. The socio-economic availability of aluminum is most likely to be restricted by trade barriers, (low) feasibility of exploration projects and (low) political stability. Silvers supply risk is most probable to be influenced by price fluctuations, trade barriers and (low) political stability. Thus, for both materials trade barriers as well as political stability might lead to low availability. Center left the overall (second) results for the dimension socio-economic availability are shown. Overall the supply risk is much higher for silver than for aluminum. However, when considering the societal acceptance (see center right for compliance with social standards and bottom left for compliance with environmental standards) aluminum performs worse than silver. Thus, the possibility of societal outrage due to non-compliance with standards is higher for aluminum than it is for silver. Last the result of all dimensions (including environmental impacts) is displayed (bottom right) by setting the highest value of each metal in every category to 100% and determining the percentage of the other metal accordingly. It can be seen that silver performs worse in most of the categories with exception of societal acceptance, and the socio-economic categories concentration of reserves and concentration of production. However, as seen in the results of the socio-economic availability both categories do not contribute much to aluminum's overall supply risk. When comparing both options to decide which material should be used in the sound system the choice would most likely fall on aluminum as the geological availability, the socio-economic supply risks as well as the environmental impacts are lower than for silver. Regarding the societal acceptance however the company should have a more detailed analysis from which countries and/or companies their aluminum is coming from and if compliance with standards can be ensured.

4. Results and discussion

The ESSENZ approach was developed to determine the resource efficiency of product systems. The numerator of the RE formula (see Eq. (1)) is referred to as added value. So far there is no common agreement on how to determine this added value. The main measure for added value in ESSENZ is the benefit of the product system quantified by the functional unit as done in LCA (ISO 14044, 2006). To calculate the RE of product systems Eq. (1) is applied: dividing the added value (functional unit) by the results of the three dimensions (availability, environmental impacts and societal acceptance). Results with a higher number reflect high RE, whereas low numbers reflects low RE. This step is shown in the supplementary material (section 5.1), where also a detailed description about the added value and the determination of the RE is provided.

Company specific monetary values, e. g. investments, or value added to the resources used (Neugebauer et al., 2016), are heterogeneous and also depend on which actor along the supply chain the assessment is focused on (e.g. selling a product for a high price is good for the business case of the seller, but results in high costs for the buyer). These monetary values cannot be determined in a generic way and may even distort the overall RE result (as demonstrated in an example in the supplementary materials – section 5.1). They can only be used as additional factors, but do not replace the RE assessment by ESSENZ, which focusses on the generic socio-economic resource efficiency risks of materials. This information can be used by companies to decide their specific purchasing strategies, which then determine their pecuniary costs and benefits.

The developed approach measures RE in the context of sustainable development. Several dimensions influencing RE are combined to achieve a comprehensive evaluation enabling meaningful decision making processes. CFs are available in literature as well as provided for the dimensions socio-economic availability and societal acceptance for a portfolio of 36 metals and four fossil raw materials (see supplementary materials – section 6). The developed ESSENZ approach has several uncertainties, which have to be considered when interpreting the results. These are described in detail in the supplementary material (section 5.3).

One very prominent challenge is described here as it should be especially considered when interpreting the results. For the assessment of the socio-economic availability ESSENZ currently only considers primary materials. The socio-economic availability of secondary materials is not taken into account. This might not be a significant limitation for natural gas, oil and coal as they are mostly burned and therefore not feed back into the system as a recycled product. Metals on the other hand are often recycled with only low restrictions to quality. Most product systems do not use only primary but often secondary metals as well. A first approach to determine socio-economic availability of secondary materials was established by Finkbeiner and Schneider (2012). However, due to lack of data the socio-economic availability of secondary materials could not be determined. Thus, in ESSENZ limitations to the socio-economic availability are determined based on the restrictions on primary metals only.

5. Conclusion

The introduced approach enhances the applicability of RE to assess product systems significantly by providing an overall framework that can be adopted across sectors. As already 21 categories are included in the approach ESSENZ serves as a starting point to carry out a comprehensive assessment of RE. However, as every sector has its individual characteristics sector specific aspects should be added.

Furthermore, ESSENZ considers RE in the context of sustainable development by considering existing sustainability goals (United Nations, 2016). For decision making support on a product level and in the context of sustainable development a comprehensive assessment of sustainability is needed (Schneider et al., 2016). ESSENZ contributes to this aim by considering all three sustainability dimensions and providing indicators for quantifying environmental, economic and social implications of material use.

For determining the socio-economic availability a new approach

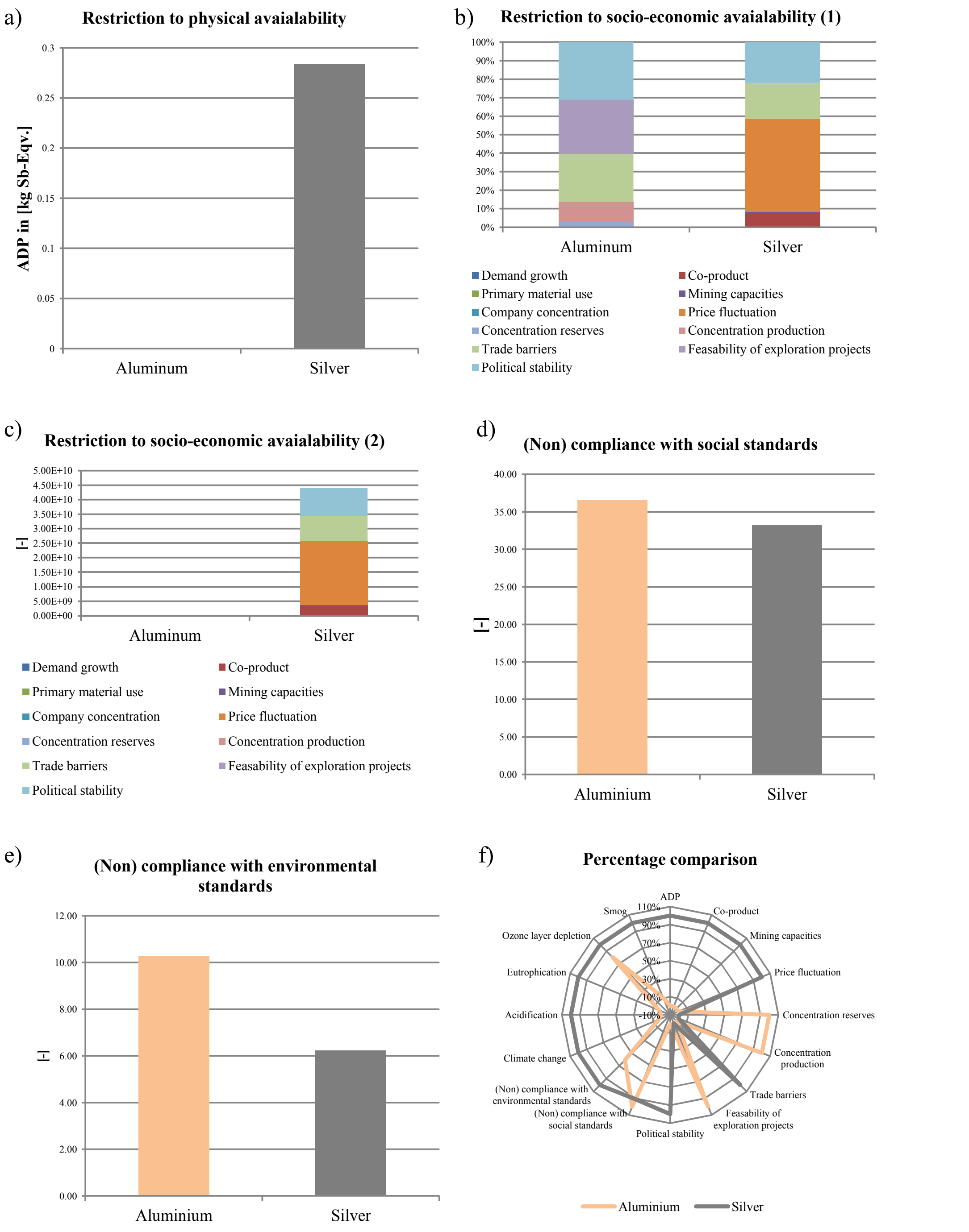

Fig. 5. Results of the case study a) for the dimensions physical (top left) b) for the dimension socio-economic availability in percentage (top right) c) overall result for the dimension socio-economic availability (center left) d) for societal acceptance – (non) compliance with environmental standards (center right) e) for societal acceptance – (non) compliance with social standards (bottom left) f) for all dimensions in percentage (bottom right).

is developed, which complements existing approaches as it can be integrated in existing life cycle assessment based schemes. Thus, companies which already use LCA for determining their environmental impacts can adapt their framework and integrate the assessment of additional aspects more easily. As the developed approach has the same framework condition interpretation of the results is also straightforward. CFs for 36 metals and four fossil raw materials are provided. This enhances the applicability tremendously.

To make the developed approach even more practical a reduction of the current indicator set should be pursued. As 21 indicators are applied the communication of the results can be challenging, especially with regard to stakeholders with less experience in the field of LCA and sustainability. Thus, identifying key indicators which represent the individual dimensions could be one option to reduce the indicator set. So far there is no experience on how to determine such key indicators. Another option could be to aggregate the indicators into a single score. However, numerous challenges accompany aggregation of indicators. These are explained in more detail in the supplementary material (section 5.2).

Even though several case studies were carried out during the development of the approach (e.g. Bach et al. (2015) and Henßler et al. (2016)) further case studies also from other sectors so far not included should be performed to continuing the testing of the developed indicators and related results. Furthermore, as ESSENZ is so far focused on metals and fossil raw materials only, but various product systems also include biotic raw materials (or a comparison on abiotic vs. abiotic based products is performed), the ESSENZ approach should be adapted to be applicable for biotic raw materials as well. Preliminary studies have shown that the ESSENZ framework can be applied to other materials (e.g. biotic materials like wood or organic substances) as well.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2016.07.077.

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Supplementary material

Integrated method to assess resource efficiency – ESSENZ

Vanessa Bach[1*], Markus Berger[1], Martin Henßler[2], Martin Kirchner[3], Stefan Leiser[4], Lisa Mohr[5], Elmar Rother[3], Klaus Ruhland[2], Laura Schneider[1], Ladji Tikana[6], Wolfgang Volkhausen[5], Frank Walachowicz[7], Matthias Finkbeiner[1]

[1] Technische Universität Berlin, Chair of Sustainable Engineering, Straße des 17. Juni 135, 10623 Berlin

[2] Daimler AG, Corporate Environmental Protection, RD/RSE, 70546 Stuttgart, HPC G211

[3] Evonik Techn. & Infr. GmbH, Rodenbacher Chaussee 4, 63457 Hanau

[4] Knauer Wissenschaftliche Geräte GmbH, Hegauer Weg 38, 14163 Berlin

[5]Thyssenkrupp Steel Europe AG, Environmental and Climate Affairs, Sustainability, Kaiser-Wilhelm-Str. 100, 47166 Duisburg

[6] Deutsches Kupferinstitut, Am Bonneshof 5, 40474 Düsseldorf

[7] Siemens AG, Corporate Technology (CT RTC PET SEP-DE), Siemensdamm 50, 13629 Berlin

* Corresponding author: vanessa.bach@tu-berlin.de

Table of content:

1. Indicator selection

In this section it is described how the selection of the indicators for the dimensions socio-economic availability and environmental impacts was carried out.

1.1 Indicator selection for socio-economic availability

Following indicators and methods were identified as options for the assessment of the socio-economic availability:

- Economic Resource Scarcity Potential (Schneider et al., 2013)
- Methodology of metal criticality determination (Graedel et al., 2012)
- DERA Rohstoffinformationen (raw material information) (Buchholz et al., 2012)
- Kritische Rohstoffe für Deutschland (Critical raw materials for Germany) (Erdmann et al., 2011)
- Minerals, Critical Minerals, and the U.S. Economy (Eggert et al., 2007)
- Report on critical raw materials for the EU (European Commission, 2014a)
- VDI 4800 Blatt 2 Bewertung des Rohstoffaufwands - Bilanzierungsgrundsätze und Rohstoffkritikalität (Assessment of raw material use - accounting principles and raw material criticality) (Association of German Engineers - Verein Deutscher Ingenieure e.V., 2016)

There were analyzed using the criteria shown in Table 1. The evaluation of these methods follows the assessment approach shown in the publication by Lehmann et al. (2015).

Table 1: Criteria used to evaluate existing approaches for the assessment of the socio-economic availability

Main criteria	**Sub criteria**
Stakeholder acceptance	Is the method currently applied?
	Are the principles of the methods easy to understand?
	Is the method able to reflect the goals of German and European politics?
Documentation & review	Is the method reviewed?
	Is the method (sufficiently) documented?
Relevance	Is the method able to reflect the impacts on the socio-economic availability adequately?
Applicability	Is the method applicable worldwide?
	Are elementary flows available in common LCA database?
	Can the method assess product systems?
	Is enough data available to determine characterization factors?

Due to the complexity of the analysis not all criteria are discussed in detail. However, the result for the overall evaluation shows how well the assessed method can contribute to the development of an approach, which is applicable for determining the socio-economic availability of materials used in product systems. For the classification an easy-to-understand traffic light rating system was chosen. As shown in Figure 1 only one method could not be used for developing ESSENZ as the assumptions made and indicators used were not transparently documented.

Figure 1: Results for the evaluation of methods determining socio-economic availability

Method	Schneider et al. (2013)	Graedel et al. (2012)	Buchholz et al. (2012)	Erdmann et al. (2011)	Eggert et al. (2007)	European Commission (2014a)	VDI Verein Deutscher Ingenieure e.V. (2013)
Overall assessment							

The method by Schneider et al. (2013) performed best and was used as a basis for ESSENZ. The others methods can contribute partly to the development of ESSENZ. For example, every method uses the HHI-Index (Rhoades, 1993) to measure the concentration of production, reserves or companies. This indicator is also implemented in ESSENZ.

1.2 Indicator selection for environmental impacts

Following indicators and methods were identified as options for the assessment of the environmental dimension:

- ReCiPe method (Goedkoop et al., 2009)
- CML-IA (Guinée et al., 2002)
- Additional methods and indicators identified for application by Product Environmental Footprint (PEF) (European Commission, 2014b), e. g. USEtox method to assess toxicity impacts (Rosenbaum et al., 2008)

Even though more impact assessment methods (e. g. TRACI (Bare, 2002)) are available the focus was on methods used and/or discussed for application in Europe. The identified indicators and methods were assessed by means of meta criteria as shown in the publication by Lehmann et al. (2015). The publication shows that several of the proposed methods and indicators by PEF are not reliable enough to be used in LCA for decision making. This is supported by the analysis of the Joint Research Center (JRC), which came to a very similar conclusion (JRC, 2011). Thus, only for the categories climate change, eutrophication, acidification, ozone layer depletion and photochemical ozone formation (smog) reliable and applicable methods are available. These categories can be measured by ReCiPe as well as by CML-IA method. However, as the ReCiPe method only considers a fraction of the elementary flows covered by CML-IA (Bach and Finkbeiner, 2016), the project consortiums choice fell on CML-IA. Correlations and mutual dependencies of these indicators are examined by correlation analysis as presented by Berger and Finkbeiner (2011). However, correlations could not be identified. Thus, the amount of categories analyzed in ESSENZ could not be decreased. Applicable and reliable methods for land use are currently not available as shown by the analysis of the Joint Research Center (JRC, 2011). Furthermore, endpoint methods assessing impacts on biodiversity are not mature enough to be applied in decision making processes (JRC, 2011; JRC, 2010). Thus, both categories are not included in ESSENZ.

2. Physical availability

Following the Abiotic Depletion Potential (ADP) indicator and the Anthropogenic Stock Extended Abiotic Depletion Potential (AADP) indicator used to determine the physical availability are described in more detail.

ADP is used to determine the physical availability. The indicator assesses the depletion of abiotic resources considering stocks and extraction rates of resource (see Eq. 1). By using ADP the potential depletion of a metal or a fossil raw material used in a product system is assessed. Thus, as more metal stocks are depleted, the lower the physical (geological) availability is.

CFs are calculated by dividing the yearly extraction rate of a resource by the reserve squared. They are determined for each element and are then set in relation with the depletion of 'antimony' (Sb) as a reference (see Eq. 1) (Guinée et al., 2002, 1993; Oers et al., 2002).

$$\mathrm{ADP}_{\mathrm{i,\,ultimate\ reserves}} = \frac{\text{extraction rate}_{\mathrm{i}}}{(\text{ultimate reserves}_{\mathrm{i}})^2} \times \frac{(\text{ultimate reserves}_{\mathrm{Sb}})^2}{\text{extraction rate}_{\mathrm{Sb}}} \qquad \text{Equation 1}$$

CFs are provided by the method developers and can be found in the publication by Oers et al., (2002).

AADP is used to additionally determine the anthropogenic availability. The occurrence of anthrophonic stocks in the technosphere can influence the physical availability significantly. CFs are determined by dividing the extraction rate of a resource by the ultimately extractable reserves and anthropogenic stock of this resource. This value is then set in relation with the reference substance antimony (see Eq. 2) (Schneider et al., 2015, 2011).

$$\mathrm{AADP}_{\mathrm{i,\,Ressources}} = \frac{\text{extraction rate}_{\mathrm{i}}}{(\text{ultimately extractable reserves}+\text{anthropogenic stocks}_{\mathrm{i}})^2} \times \frac{(\text{ultimately extractable reserves}_{\mathrm{sb}}+\text{anthropogenic stock}_{\mathrm{sb}})^2}{\text{extraction rate}_{\mathrm{sb}}}$$

Equation 2

CFs are provided by the method developers and can be downloaded here: http://www.see.tu-berlin.de/menue/forschung/ergebnisse/modellierung_der_verfuegbarkeit_abiotischer_ressourcen/parameter/de/

3. Socio-economic availability

In this section the three indicators measuring concentration of reserves, of production and of companies are introduced. Furthermore, the targets used for calculating the characterization factors for socio-economic availability are introduced. The comparison according to ISO 14040 scheme for impact categories of the IPCC (2007) method for climate change with the developed approach for political stability is introduced.

3.1 Concentration of reserves, production and companies

Following the three categories related to concentration are explained in more detail:

- Concentration of reserves: By determining the concentration of *reserves*, the number of countries where the considered reserve can be extracted as well as their share of the global stocks is reflected. When only few countries have almost all of the *reserves* the concentration as well as the possibility of limited availability is high as future mining explorations can only focus on few countries.
- Concentration of production: The category expresses the number of countries extracting ores as well as their share of the global mining activities. High concentrations mean that only few countries produce most of the ores. This can lead to limited availability as most of the mining activities occur in only few countries.
- Company concentration: When only few companies market most of the produced raw materials, a high company concentration occurs, which can reduce the availability of resources. High company concentrations can influence the whole supply chain as the different supply chain stages are managed by different sectors and companies. Within ESSENZ though only the concentration of ore marketing companies is considered. SNL[1] (SNL, 2015) data are used to calculate the HHI indicator.

3.2 Targets used for calculating the characterization factors for socio-economic availability

In Table 2 the target values including their sources are shown. These were used to calculate the characterization factors for the socio-economic availability. The targets are default values and can be changed by the company or stakeholder according to their individual preference.

Table 2: Targets of the eleven categories including their sources

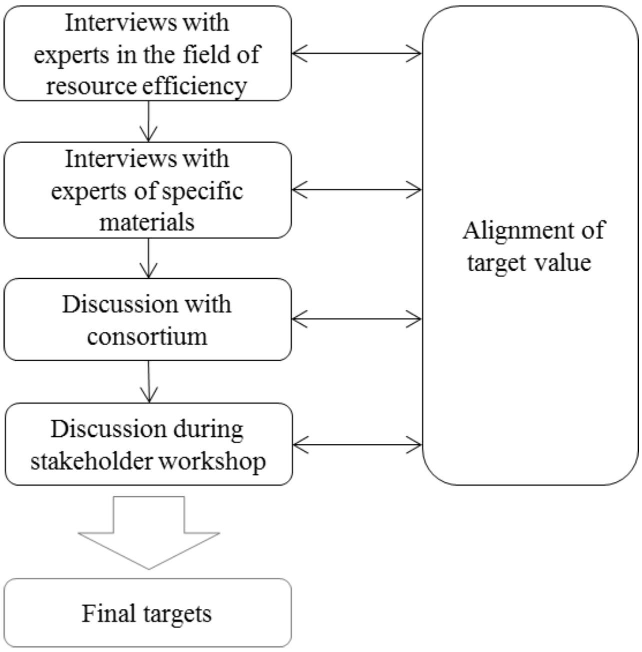
Category	**Target**	**Source**
Concentration	0.15 [-]	(Rhoades, 1993)
Mining capacity	50 years	Stakeholder survey
Feasibility of exploration projects	55 [-]	Expert judgment
Companion metal	0.35 [-]	Expert judgment
Trade barriers	3.15 [-]	Expert judgment
Political stability	1.9 [-]	Expert judgment

[1] SNL - SNL Metals & Mining

Demand growth	5 %	Stakeholder survey
Primary material use	75%	Stakeholder survey
Price fluctuations	20%	Expert judgment

Following it is explained in more detail how the targets based on expert judgment were determined. As shown in Figure 2 an iterative process was chosen to align the target after every consultation step. First interviews with experts in the field of resource efficiency were carried out. Together with the experts target values were determined. These values were discussed with experts with specific knowledge of certain materials. Together with the experts the targets were adjusted if necessary. These values where then discussed within the project consortium, which consists of experts in the field of resource efficiency as well as experts with knowledge of specific materials. As a last step the target values were debated in a stakeholder workshop attended by around 70 experts in the field of resources, which approved the predefined target values.

Figure 2: Approach to determine target values for ESSENZ method



3.3 Comparing the IPCC (2007) method for climate change with the developed approach for political stability according to ISO 14040 scheme for impact categories

Following the relation between existing Life Cycle Impact Assessment (LCIA) methods and the developed approach is demonstrated by applying the scheme of ISO 14040 for impact categories for the established method by Intergovernmental Panel on Climate Change (IPCC 2007) for the category climate change and for the category political stability of the developed approach (see Table 3).

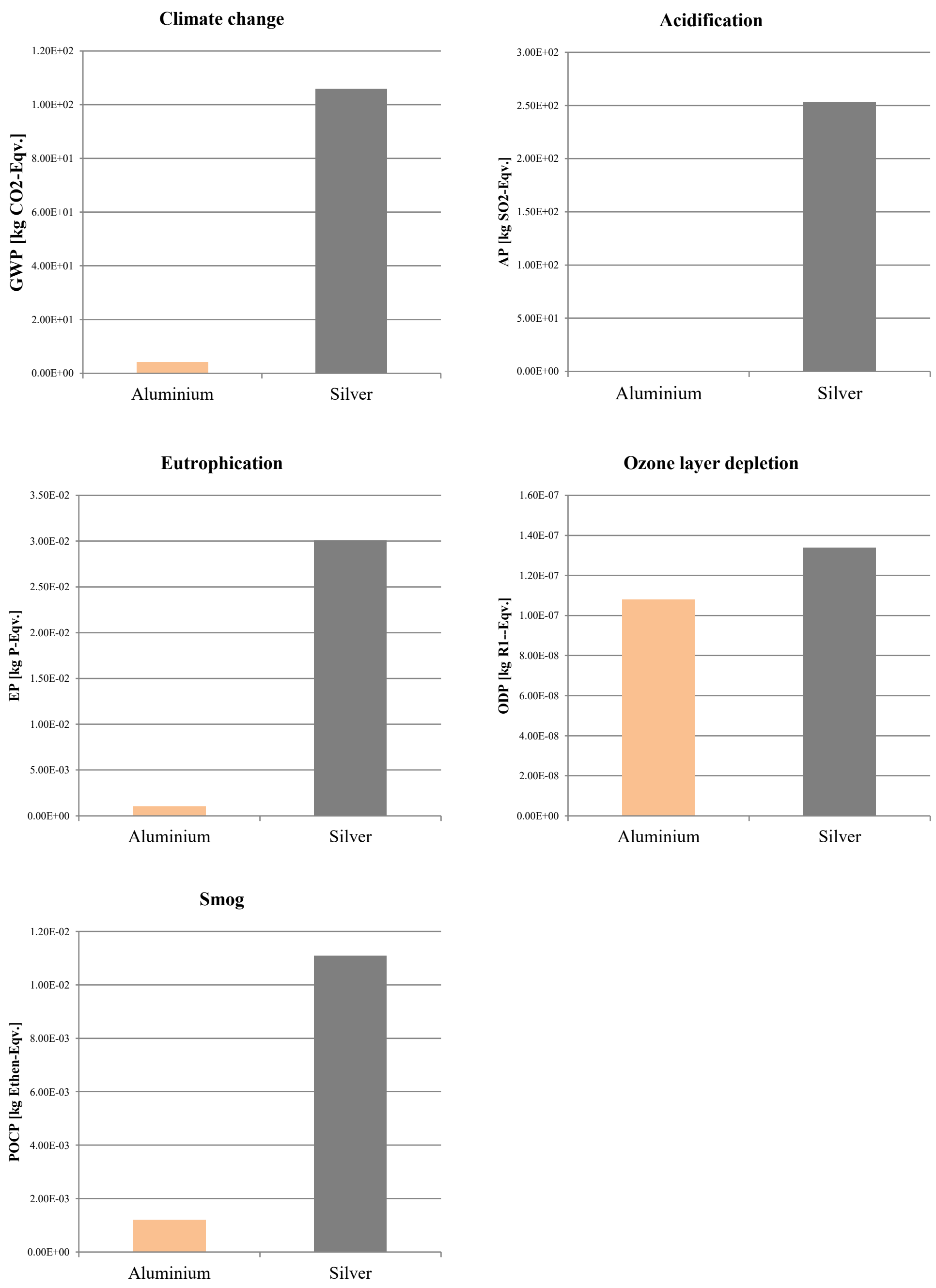
Table 3: Comparison of IPCC (2007) method for climate change with developed approach for political stability according to ISO 14040 scheme for impact categories

ISO terminology	**IPCC method for climate change as demonstrated in ISO 14040**	**Developed approach for political stability**
Impact category	Climate change	Political stability
LCI results	Amount of greenhouse gases over the life cycle per functional unit	Amount of raw materials per functional unit
Characterization model	IPCC model (100 years)	ESSENZ model
Category indicator	Infrared radiative forcing	World Governance Index based on World Governance Indicators (Kaufmann et al., 2011; World Bank Group, 2013)
Characterization factor	Global warming potential for every greenhouse gas (kg CO_2-eq per functional unit)	Potential restriction to availability due to political (un)stability for every raw material (1/kg per functional unit)
Determining category indicator result	LCI result multiplied with category indicator	LCI result multiplied with category indicator
Category indicator result	Kilograms of CO_2-eq per functional unit	Dimensionless, value per functional unit
Category endpoints	Coral reefs, forests, crops	Limited availability for products and companies
Relevance	Infrared radiative forcing is a proxy for potential effects of climate change on the environment.	World Governance Index is a proxy for potential effects of political (un)stability on availability of raw materials.

4. Results for the environmental dimension

The results for the environmental dimension are shown in Figure 3. They were determined using the GaBi software and database (Thinkstep, 2016). It can be seen that silver performs worse in all categories compared to aluminum. The least differences occur in the category ozone layer depletion, where the variance in the impacts of both materials is around 20%.

Figure 3: Results for the environmental dimension a) for climate change (top left), b) for acidification (top right), c) for eutrophication (center left), d) for ozone layer depletion (center right) e) for smog (bottom left)



5. Discussion

In this section the topics 'added value', 'determination of resource efficiency', 'aggregation' of indicators into a single score and 'uncertainties' associated with the ESSENZ method are discussed in more detail.

5.1 Added value and determination of resource efficiency

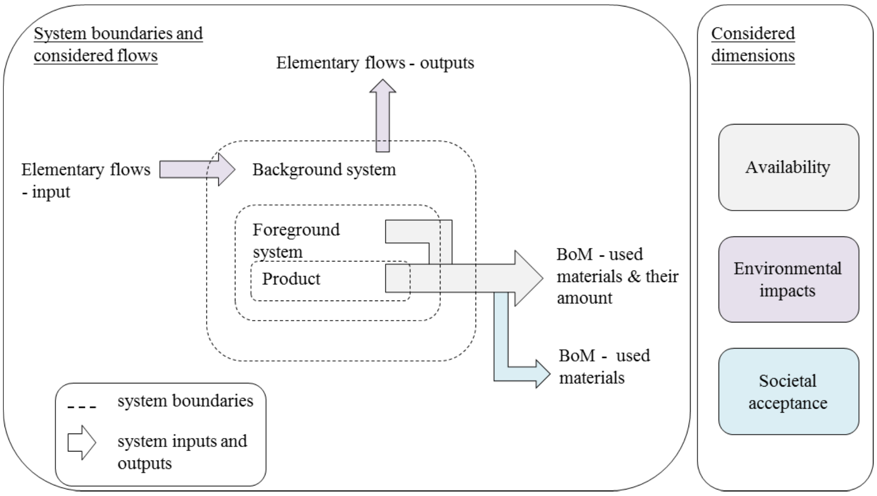
Following, the calculations of the RE results are explained. Even though ESSENZ is developed to be integrated into LCA due to current inventory data structure is not possible to consider the whole life cycle in all dimensions. Assessing environmental impacts over the life cycle is possible as LCA databases provide life cycle inventory data for many processes and products. The ADP and AADP indicators are original integrated into LCA in the same way as environmental impacts are. However, as in ESSENZ the indicators are used to determine the physical availability of metals and fossil raw material often rather than the overall depletion potential, current LCA databases cannot be used, which make the assessment over the life cycle of a product system challenging. Currently the databases apply economic allocation to assign metal contents from mixed ores to metal datasets. Thus, the amount of metals given in the inventory results does not reflect the physically present metal content. Over- or underestimations in the inventory occur depending on the economic value of the considered metals. Thus, the physical availability can so far only be determined for the bill of material (BoM - materials incorporated in the product under investigation). The aim should also be to include data of the foreground system as these should be easier to gather as data for background systems. This is especially important when alternatives are compared which use different auxiliary materials. For the assessment of the socio-economic availability the actual amount of the used materials is needed as well. Thus, the same principle applies.

For evaluating the societal acceptance material data about amounts are not needed as the acceptance of a product by society (consumers) is independent of the amount of the material. It only matters, if a certain material occurs at all and not if 1 g or 1 kg is used. Thus, the consideration of the entire life cycle is easily possible. However, as consumers mostly care about the materials included in the purchased product rather than all materials used over the life cycle and the interpretation of the indicators is relatively new, it is advised to analyze the BoM first before considering the whole life cycle.

Summarizing (see Figure 4), to determine environmental impacts the elementary flows of the entire system (foreground as well as background system – considering the entire life cycle of the product system) have to be collected. For determining the availability (physical as well as socio-economic) the BoM (amount of used materials occurring in product) and possibly materials used in the foreground system have to be identified. Same applies for determining the societal acceptance without taken the amounts of the resources into account.

The numerator of the RE formula is referred to as added value (see Eq. 1 in paper). In ESSENZ the main measure for added value is the benefit of the product system quantified by the functional unit as done in LCA (ISO 14044, 2006). The European Commission, which measures the RE of European states and Europe as a whole, uses the gross domestic product (GDP) (European Commission, 2015a). For product systems an economic value is often applied as well. However, using economic measures for the added value can lead to inadequate results. As applying economic values might be reasonable in some cases (e.g. to evaluate investment decisions) ESSENZ allows to use economic measures only as an add-on.

Figure 4: Overview of considered system boundaries and flows to determine the results of the dimensions availability, environmental impacts as well as societal acceptance



Following a small example is introduced to explain the challenges of using monetary values in more detail. Two product systems with the same benefit are compared: painting a 12 m^2 wall with white color (functional unit). When using product A four liter paint are needed, for the same result five liter paint from product B have to be applied as the quality (e. g. coverage) is lower. The product system A´s global warming potential adds up to 117 kg CO_2-equivalents. For product system B 190 kg CO_2-equivalents are emitted over the life cycle - more as for product system A as more paint has to be used. Thus, as shown in Eq. 3 and 4 the calculated RE is higher (and therefore better) for product A. Results with a higher number reflect high RE, whereas low numbers reflects low RE.

$$\mathrm{RE_A} = \frac{\text{added value}}{\text{global warming potential}} = \frac{\text{(painting a) 12 m}^2\text{ (wall with white color)}}{\text{117kg CO}_2\text{-eqv}} = 0.1\,\frac{m^2\text{ painted white wall}}{\text{kg CO}_2\text{-eqv}} \quad \text{(Equation 3)}$$

$$\mathrm{RE_B} = \frac{\text{added value}}{\text{global warming potential}} = \frac{\text{(painting a) 12 m}^2\text{ (wall with white color)}}{\text{190 kg CO}_2\text{-eqv}} = 0.06\,\frac{m^2\text{ painted white wall}}{\text{kg CO}_2\text{-eqv}} \quad \text{(Equation 4)}$$

When economic measures are applied for the added value, inadequate results might occur. Following an example is introduced to highlight this challenge. Again the products A and B for painting a wall are compared. Based on the results for climate change one would assume that the product system with the lower CO_2-equivalent emissions is more resource efficient (product system A). However, when considering for example the price of the two products and product B costs twice as much as product A (e. g. 10 € versus 5 €), the RE of product system B is higher. The price of each product is divided by its released CO_2-equivalent emissions: thus for product system B, where the price is twice as high as for A, but the emissions are not double, the calculated value is higher as for product system A (see Eq. 5 and 6).

$$\mathrm{RE_A} = \frac{\text{added value}}{\text{global warming potential}} = \frac{5€}{\text{117 kg CO}_2\text{-eqv.}} = 0.043\,\frac{€}{\text{kg CO}_2\text{-eqv.}} \quad \text{(Equation 5)}$$

$$\mathrm{RE_B} = \frac{\text{added value}}{\text{global warming potential}} = \frac{10€}{\text{190 kg CO}_2\text{-eqv.}} = 0.053\,\frac{€}{\text{kg CO}_2\text{-eqv.}} \quad \text{(Equation 6)}$$

As applying economic values might be reasonable in some cases (e.g. to evaluate investment decisions) ESSENZ allows economic measures only as an additional measure.

5.2 Aggregation of indicators

The aggregation of the overall 21 indicators included in ESSENZ is not science based (Finkbeiner et al., 2014). Each dimension has to be summed up individually first before they can be combined into a single score. This means that the five indicators of the environmental dimension, the 11 indicators of the socio-economic availability dimension, the three indicators of the physical availability dimension and the two indicators of the societal acceptance dimension have to be aggregated into one value respectively. Then, these values have to be aggregated into the final single score. However, aggregation of the individual dimensions is challenging and thus not recommended within ESSENZ. To aggregate the environmental categories the normalization and weighing approach could be applied, which first normalizes the results to global (or regional) values and then weights the categories with a defined weighting scheme (European Commission, 2015b; ISO 14044, 2006). Even though this approach is supported by some authors (e. g. European Commission (2015) and Goedkoop et al. (2009)) it has many challenges e. g. missing elementary flows to determine the normalization factors and leads to distorted results (Lehmann et al., 2016). Therefore, it is not recommended for ESSENZ. The eleven indicators determining the socio-economic availability could be aggregated when applying a weighting scheme. However, as combining socio-economic aspects are individual for every company, defining a weighting scheme might be challenging. The three indicators for the physical availability ($ADP_{elemental}$, ADP_{fossil} and AADP) cannot be aggregated as they are calculated based on different resource stocks (AADP is based on *ultimately extractable reserves*whereas $ADP_{elemental}$ is based on ultimate reserves and ADP_{fossil} is based on ultimate fossil resources). Both indicators for determining the societal acceptance can be aggregated individually by adding up the results of the metals and fossil raw materials. They can also be aggregated together as there are both scaled to 100. However, a weighting scheme has to be applied, where so far no experience is available. Overall the aggregation of the individual dimensions, even though theoretically possible, is challenging and not recommended in practice.

5.3 Uncertainties

General uncertainties: All indicators used in ESSENZ – established ones as well as newly developed ones – face the challenge of the underlying data quality. If the data quality is poor, the indicator will have greater uncertainties as if the underlying data is good. However, established indicators, which have been used more frequently, tend to have lower uncertainties as they were improved over time.

Only the environmental impacts can be determined over the entire life cycle. Due to the current database structure the availability can only be determined for the bill of material. Thus, materials in the downstream and upstream processes are not taken into account. This could lead to over- and underestimations of impacts as well as disregard of trade-offs between alternatives.

The determination of the considered (environmental) impacts and risks (to availability) is based on models trying to reflect the reality as best as possible. However, several aspects are no considered in these models e. g. time course or regional specifications (Finkbeiner et al., 2006; ISO 14044, 2006). These restrictions have to be considered during interpretation of the results.

Environmental impacts: To determine the environmental impacts of a product system over its life cycle corresponding data has to be identified (e.g. using LCA databases like Ecoinvent (2016), or Thinkstep (2016)). The quality of life cycle based data especially for the background system can be poor. Thus, a sensitivity analysis should always be carried out and data of environmental hotspots should be reviewed carefully to avoid over- and underestimations of environmental impacts. Even though the categories biodiversity and land use are of upmost importance when assessing extraction and use of resources, they cannot be included at this point as no mature models and methods exist (Curran et al., 2016, 2011; Finkbeiner et al., 2014; Joint Research Centre, 2011, 2010; Koellner et al., 2013)

Physical availability: To determine the physical availability the ADP (Guinée et al., 2002) and AADP (Schneider et al., 2011) indicators are used for which CFs are provided by the authors. However, due to lack of data only few AADP values are available (Schneider et al., 2015). Hence, for product systems containing metals for which CFs are not available a comprehensive assessment of anthropogenic stocks is not possible. A wide variety of metals and fossil raw materials are covered by the ADP indicator. However, the indicator is developed for abiotic resources only. Thus, when considering other materials like biotic resources alternative indicators e.g. the biotic resource depletion (BDP) (Lindeijer et al., 2002) have to be applied additionally.

Socio-economic-availability: ESSENZ provides overall 40 CFs. However, not all metals are included due to lack of data. Thus, if a product (or its foreground system) consists of metals not included in the set, the availability assessment is impaired. Even though the calculation of additional CFs is possible, it is challenging due to lack of data. The concept of assessing the socio-economic availability is for the most part transferable to other materials (case studies including chemical products as well as biotic raw materials were carried out during the development of the approach, but are not published yet). Furthermore, currently only primary materials are considered (see paper – section 4).

Societal acceptance: The indicator for determining the compliance with social standards is based on the Social Hotspot Database (Norris et al., 2013), which provides data on sector and country level. Thus, no statement in regard to the status of specific mines can be made. For the indicator determining the compliance with environmental standards data by Yale Center for Environmental Law & Policy (2014) are used, which reflect the quality and quantity of conservation areas of countries, but not the standards of mining specifically. As there are no indicators available evaluating the compliance with standards of mines, the introduced approaches seem valid for application. However, uncertainties have to be kept in mind when interpreting the results.

6. Characterization factors of socio-economic availability and societal acceptance

Following the calculated characterization factors for the dimensions socio-economic availability as well as societal acceptance are provided (status: June 2016). The material set indicators will be updated and the most recent data can be found here: http://www.see.tu-berlin.de/menue/forschung/ergebnisse/essenz/parameter/en/

6.1 Characterization factors of socio-economic availability

The displayed CFs are calculated based on the approach shown in section 2 of the paper.

Category/ raw material	Demand growth $\frac{1}{kg}$	Occurrence as Co-product $\frac{1}{kg}$	Primary material use $\frac{1}{kg}$	Mining capacities $\frac{1}{kg}$	Company conc. $\frac{1}{kg}$	Price fluctuation $\frac{1}{kg}$	Conc. of reserves $\frac{1}{kg}$	Conc. of production $\frac{1}{kg}$	Trade barriers $\frac{1}{kg}$	Feasibility of exploration projects $\frac{1}{kg}$	Political stability $\frac{1}{kg}$
Aluminium	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	1.91E+06	7.86E+06	1.87E+07	2.10E+07	2.24E+07
Antimony	9.32E+08	5.82E+08	5.20E+09	5.05E+08	4.23E+09	9.49E+09	3.01E+09	1.55E+10	6.03E+09	1.12E+10	9.80E+09
Beryllium	1.14E+12	3.49E+11	2.94E+12	1.70E+13	3.40E+12	6.88E+12	6.32E+12	1.70E+13	0.00E+00	0.00E+00	0.00E+00
Bismuth	0.00E+00	2.87E+11	3.87E+11	0.00E+00	2.36E+11	6.68E+11	4.49E+11	2.86E+11	2.85E+11	4.98E+11	3.90E+11
Chromium	0.00E+00	0.00E+00	2.56E+07	1.31E+06	0.00E+00	0.00E+00	7.76E+06	1.74E+07	3.65E+07	0.00E+00	3.89E+07
Cobalt	3.99E+09	2.99E+09	0.00E+00	0.00E+00	0.00E+00	2.15E+10	2.16E+09	8.04E+09	1.07E+10	8.86E+09	1.73E+10
Copper	0.00E+00	5.02E+06	0.00E+00	3.88E+05	0.00E+00	6.84E+07	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Crude oil	0.00E+00	0.00E+00	2.67E+05	0.00E+00	8.96E+03	3.30E+05	0.00E+00	0.00E+00	0.00E+00	3.09E+05	2.23E+05
Gallium	1.70E+13	1.89E+12	0.00E+00	0.00E+00	7.40E+11	3.74E+12	2.05E+11	0.00E+00	0.00E+00	2.25E+12	0.00E+00
Germanium	0.00E+00	5.28E+12	0.00E+00	2.29E+11	4.16E+12	8.47E+12	6.77E+11	7.05E+12	0.00E+00	0.00E+00	0.00E+00

Category/ raw material	Demand growth $\frac{1}{kg}$	Occurrence as Co-product $\frac{1}{kg}$	Primary material use $\frac{1}{kg}$	Mining capacities $\frac{1}{kg}$	Company conc. $\frac{1}{kg}$	Price fluctuation $\frac{1}{kg}$	Conc. of reserves $\frac{1}{kg}$	Conc. of production $\frac{1}{kg}$	Trade barriers $\frac{1}{kg}$	Feasibility of exploration projects $\frac{1}{kg}$	Political stability $\frac{1}{kg}$
Gold	0.00E+00	0.00E+00	0.00E+00	9.68E+09	0.00E+00	0.00E+00	0.00E+00	0.00E+00	3.46E+11	0.00E+00	3.85E+11
Hard coal	0.00E+00	0.00E+00	1.63E+05	0.00E+00	5.45E+03	3.53E+05	1.63E+04	8.14E+04	1.29E+05	0.00E+00	1.78E+05
Indium	2.86E+11	1.04E+12	0.00E+00	0.00E+00	5.11E+11	2.44E+12	4.54E+11	7.13E+11	0.00E+00	0.00E+00	0.00E+00
Iron	7.48E+04	0.00E+00	0.00E+00	4.38E+03	0.00E+00	8.39E+05	2.86E+04	1.45E+05	2.87E+05	0.00E+00	3.25E+05
Lead	5.10E+07	1.62E+07	0.00E+00	7.76E+06	0.00E+00	3.46E+08	4.17E+07	1.15E+08	1.57E+08	2.25E+08	2.08E+08
Lignite Coal	0.00E+00	0.00E+00	1.06E+07	0.00E+00	3.56E+05	2.31E+07	9.67E+05	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Lithium	1.01E+10	0.00E+00	3.30E+10	0.00E+00	0.00E+00	0.00E+00	1.70E+10	1.44E+10	0.00E+00	0.00E+00	0.00E+00
Magnesium	3.10E+07	1.94E+06	0.00E+00	0.00E+00	1.11E+06	0.00E+00	3.15E+06	5.52E+07	1.97E+07	3.09E+07	3.27E+07
Molybdenum	1.04E+07	0.00E+00	0.00E+00	6.47E+05	0.00E+00	0.00E+00	6.33E+06	7.64E+06	4.99E+07	5.11E+07	4.69E+07
Natural Gas	0.00E+00	0.00E+00	0.00E+00	2.34E+07	0.00E+00	1.07E+10	8.26E+08	1.47E+09	0.00E+00	3.80E+09	3.56E+09
Nickel	0.00E+00	0.00E+00	4.55E+05	0.00E+00	1.52E+04	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Niobium	2.22E+08	3.56E+07	0.00E+00	4.07E+06	0.00E+00	9.58E+08	0.00E+00	6.94E+07	3.76E+08	0.00E+00	3.93E+08
Palladium	0.00E+00	1.53E+09	1.15E+10	0.00E+00	1.30E+10	3.37E+10	5.25E+10	7.17E+10	1.84E+10	2.55E+10	1.56E+10
Platinum	0.00E+00	1.84E+12	0.00E+00	0.00E+00	2.99E+11	7.03E+12	1.54E+13	3.35E+12	5.08E+12	5.40E+12	5.34E+12
Raw earth	0.00E+00	4.96E+11	0.00E+00	0.00E+00	2.10E+11	0.00E+00	1.70E+13	1.22E+13	5.68E+12	5.99E+12	5.40E+12
Rhenium	0.00E+00	3.40E+09	1.02E+10	0.00E+00	7.20E+09	7.49E+10	3.25E+09	3.34E+10	8.12E+09	0.00E+00	1.41E+10

9

Category/ raw material	**Demand growth $\frac{1}{kg}$**	**Occurrence as Co-product $\frac{1}{kg}$**	**Primary material use $\frac{1}{kg}$**	**Mining capacities $\frac{1}{kg}$**	**Company conc. $\frac{1}{kg}$**	**Price fluctuation $\frac{1}{kg}$**	**Conc. of reserves $\frac{1}{kg}$**	**Conc. of production $\frac{1}{kg}$**	**Trade barriers $\frac{1}{kg}$**	**Feasibility of exploration projects $\frac{1}{kg}$**	**Political stability $\frac{1}{kg}$**
Selenium	0.00E+00	1.70E+13	1.56E+13	0.00E+00	1.70E+13	0.00E+00	7.82E+12	1.39E+13	0.00E+00	0.00E+00	0.00E+00
Silicon	1.48E+07	0.00E+00	0.00E+00	0.00E+00	2.70E+07	0.00E+00	0.00E+00	1.70E+08	1.07E+08	2.14E+08	1.59E+08
Strontium	0.00E+00	1.43E+10	0.00E+00	9.75E+08	0.00E+00	9.22E+10	0.00E+00	0.00E+00	3.56E+10	0.00E+00	4.03E+10
Tantalum	0.00E+00	0.00E+00	0.00E+00	0.00E+00	1.81E+09	8.12E+09	1.72E+09	2.99E+09	0.00E+00	5.06E+09	2.71E+09
Tellurium	1.70E+11	7.76E+10	6.30E+11	0.00E+00	0.00E+00	1.45E+12	1.07E+12	5.53E+11	1.02E+12	0.00E+00	1.39E+12
Tin	0.00E+00	5.05E+12	0.00E+00	0.00E+00	7.59E+11	1.70E+13	5.45E+12	2.38E+12	0.00E+00	0.00E+00	0.00E+00
Titanium	0.00E+00	0.00E+00	1.99E+09	1.56E+08	0.00E+00	5.43E+09	3.18E+08	1.47E+09	2.88E+09	5.09E+09	4.63E+09
Tungsten	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	4.81E+08	1.43E+07	0.00E+00	1.12E+08	0.00E+00	0.00E+00
Uranium	0.00E+00	0.00E+00	0.00E+00	7.44E+07	4.22E+09	0.00E+00	6.37E+09	3.21E+10	1.22E+10	1.62E+10	2.02E+10
Vanadium	3.00E+12	0.00E+00	1.70E+13	4.26E+11	0.00E+00	0.00E+00	0.00E+00	4.27E+12	1.70E+13	1.70E+13	1.70E+13
Zinc	0.00E+00	1.21E+10	0.00E+00	0.00E+00	3.02E+09	3.75E+10	4.75E+09	1.68E+10	1.51E+10	2.33E+10	2.30E+10
Zirconium	0.00E+00	6.72E+06	5.00E+07	2.68E+06	0.00E+00	1.14E+08	7.93E+06	1.55E+07	6.63E+07	0.00E+00	7.82E+07

10

6.2 Characterization factors of societal acceptance

The displayed CFs are calculated based on the approach shown in section 2 of the paper.

Categorie/ raw material	(Non) compliance with social standards [-]	(Non) compliance with environmental standards [-]
Aluminium	36.57	10.27
Antimony	45.51	8.26
Beryllium	0.83	9.83
Bismuth	23.84	6.62
Chromium	35.73	6.96
Cobalt	25.85	20.81
Copper	1.64	3.02
Crude oil	23.37	8.92
Gallium	0.07	0.18
Germanium	30.28	5.64
Gold	2.87	1.41
Hard Coal	20.23	3.87
Indium	20.22	5.03
Iron	56.34	32.48
Lead	8.36	4.19
Lignite Coal	2.02	2.77
Lithium	3.15	2.29
Magnesium	100.00	18.00
Manganese	21.73	11.14
Molybdenum	12.96	3.40
Natural Gas	0.64	2.41
Nickel	18.19	5.55
Niobium	21.34	100.00
Palladium	25.25	15.89
Platinum	62.35	42.20
Raw earth	36.57	10.27

Categorie/ raw material	(Non) compliance with social standards [-]	(Non) compliance with environmental standards [-]
Selenium	3.32	6.45
Silicon	0.87	7.04
Silver	56.14	9.87
Strontium	5.03	2.68
Tantalum	33.28	6.23
Tellurium	14.21	9.96
Tin	13.82	24.90
Titanium	2.72	4.19
Tungsten	5.94	4.30
Uranium	4.59	19.37
Vanadium	32.20	8.72
Zinc	46.00	8.42
Zirconium	11.24	2.39

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The ESSENZ book is not reproduced here entirely due to its length (161 pages). However, key aspects are addressed within the journal publication, which is included in this thesis. Additionally to these key aspects, the book provides further details regarding the ESSENZ methodology.

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Vanessa Bach • Markus Berger • Martin Henßler
Martin Kirchner • Stefan Leiser • Lisa Mohr
Elmar Rother • Klaus Ruhland • Laura Schneider
Ladji Tikana • Wolfgang Volkhausen
Frank Walachowicz • Matthias Finkbeiner

Messung von Ressourceneffizienz mit der ESSENZ-Methode

Integrierte Methode
zur ganzheitlichen Bewertung

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Inhaltsverzeichnis

3.2 Assessment of terrestrial biotic resource and raw material use on product level

This chapter contains the following publication:

V. Bach, M. Berger, N. Finogenova, M. Finkbeiner (2017): **Assessing the Availability of Terrestrial Biotic Materials in Product Systems** (**BIRD**), Sustainability, 9(1), 137

DOI:10.3390/su9010137

The publication introduces an approach to determine the availability of terrestrial biotic resource and raw material use in product systems (BIRD) along the supply chain. First, applying an iterative approach methodological gaps of existing methods and relevant dimensions and categories for the assessment of terrestrial biotic resource and raw material use on product level are determined (contribution to research target *2a*). Overall the following five dimensions with 24 associated categories are identified:

- physical constraints
- socio-economic constraints
- abiotic constraints
- social constraints and
- environmental constraints

The dimensions and categories are quantified by existing (contribution to research target *2b*) as well as newly established indicators (contribution to research target *2c*). For the dimensions physical constraints, socio-economic constraints, abiotic constraints and social constraints new indicators (partly based on ESSENZ) are developed. Further, existing and newly developed methods are joined for the integrated methodology BIRD (contribution to research target *2d*). In a last step, the methodology is applied to a case study of biodiesel produced from rapeseed and soy beans (contribution to research target *2e*).

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Assessing the Availability of Terrestrial Biotic Materials in Product Systems (BIRD)

Vanessa Bach *, Markus Berger, Natalia Finogenova and Matthias Finkbeiner

Chair of Sustainable Engineering, Technische Universität Berlin, Office Z1, Str. des 17. Juni 135, 10623 Berlin, Germany; markus.berger@tu-berlin.de (M.B.); natalia.finogenova@campus.tu-berlin.de (N.F.); matthias.finkbeiner@tu-berlin.de (M.F.)

* Correspondence: Vanessa.bach@tu-berlin.de; Tel.: +49-30-3142-7941

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Abstract: Availability of abiotic resources has been a topic of concern in recent years, resulting in several approaches being published to determine their availability on country and product level. However, the availability of biotic materials has not been analyzed to this extent yet. Therefore, an approach to determine possible limitations to availability of terrestrial biotic materials over the entire supply chain is introduced. The approach considers 24 categories overall as well as associated category indicators for the five dimensions: physical, socio-economic, abiotic, social and environmental constraints. This ensures a comprehensive availability assessment of bio-based product systems. The approach is applied to a case study comparing biodiesel produced from rapeseed and soy beans. The study shows that the determination of indicator values is feasible for most categories and their interpretation leads to meaningful conclusions. Thus, the approach leads to a more comprehensive assessment of availability aspects and supports better informed decision making in industry and policy.

Keywords: biotic materials; resource availability; socio-economic availability; life cycle assessment; supply risk

1. Introduction

The availability of abiotic resources has been a topic of discussion recently, leading to several approaches being published to determine their availability on country and product level (e.g., [1–7]). However, the availability of biotic materials has not been analyzed to this extent yet.

First, a distinction has to be made regarding biotic resources and man-made biotic materials (see Figure 1). Biotic resources are defined as living objects (species) such as wild fish or trees removed from the natural environment by human activities, whereas man-made biotic materials refer to species extracted from the technosphere [8,9]. The term "biotic materials" includes both biotic resources and man-made biotic materials. Biotic materials are classified as renewable as they can regenerate within human lifetime.

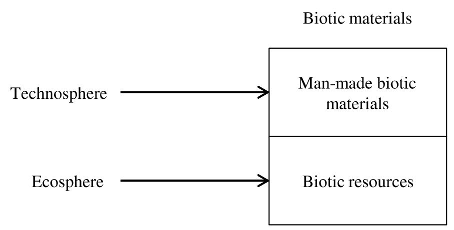


Figure 1. Overview of the terms man-made biotic materials, biotic resources and biotic materials.

Having materials available at any time is a precondition for economic development as companies rely on certain materials to produce goods and services [10–12]. Limitations to availability can restrict productivity and (in the worst case) might lead to production stops. Such a scenario would result in severe damage to the company as well as to the affected regions and countries. In addition to job losses further aspects like healthcare system for the employees might be affected. Therefore, ultimately the whole society is impacted when a company stops production due to availability restrictions of materials [13–16].

Availability of biotic resources has been a topic of concern for several years, especially in relation to such topics as overfishing, elephants being killed for ivory as well as deforestation of rain forest [17]. Fish consumption plays a vital role for the livelihood of many people as over two billion people rely on fish as an important part of their daily diet [18,19] and the rainforest is one of the biggest hotspots for biodiversity and billions of people depend on the services it supplies (e.g., food and shelter). The hunt for ivory has led to the decrease of African elephant population to the point where they are almost extinct [20,21]. Considering recent rises in fish yields and increasing shares of the rainforest being transformed into agricultural areas the pressure on these resources is steadily intensifying. For the assessment of biotic resource use of products, the Life Cycle Assessment (LCA) methodology according to ISO 14040/44 is commonly used. Several Life Cycle Impact Assessment (LCIA) methods exist to determine depletion of biotic resource (e.g., of fish [22–24] or loss of biodiversity in relation to deforestation of rain forest areas e.g., [11,25,26]).

Man-made biotic materials are restricted in their availability for industrial processes. However, these restrictions somewhat differ compared to the constraints of abiotic resources. So far no method exists for the assessment of man-made biotic materials extracted from the technosphere, e.g., agricultural products such as maize, rapeseed or timber from cultivated forests (silviculture). The availability of these materials has so far not been considered in LCA, even though their accessibility can be restricted as well. For example, predicted high demand of cellulosic fibers (from cotton) in the coming years [27] might lead to the restriction of the overall availability of cotton as a consequence.

Biotic materials are subject to various constraints which can influence their availability (as addressed by [1,28–32]). Most of these studies consider physical and socio-economic constraints of abiotic resources only, but do not include biotic materials in their approaches ([1,28,32]). Carrying out a bottom-up analysis, the compatibility of these categories and indicators for biotic materials was analyzed. With regard to socio-economic availability of biotic materials the approach considering the widest range of categories and indicators is the one of Fraunhofer (2013) [31], which takes the following categories into account: substitution, recycling, concentration of producing countries and poor governance as well as environmental performance of producing countries. Besides substitution all categories are also considered within the introduced (BIRD) approach as follows: For the category recycling, another indicator is applied than that proposed by Fraunhofer (2013) [31] because the recycled content was evaluated to be more adequate for the assessment of primary material availability. Indicators for the categories concentration and governance of producing countries are identical. Substitution is not included within the introduced approach as it is typically an aspect considered within vulnerability [3,5,6,32,33] and is challenging to determine on a material level. Additionally, the introduced approach provides indicators for seven more socio-economic constraints not considered by Fraunhofer (2013).

Furthermore, using a top-down procedure existing case studies of species used as biotic materials as well as bio-based products were reviewed, which address individual aspects related to the availability of agricultural and silvi-cultural products (e.g., land and phosphorus use [29,30]). Thus, based on the applied Top-down-Bottom-up procedure the following aspects are identified as being relevant for the availability of biotic materials:

- physical constraints
- socio-economic constraints
- abiotic constraints

- social constraints and
- environmental constraints

The introduced approach to determine the availability of terrestrial biotic materials in product systems (BIRD) proposes several categories and indicators to quantify these aspects and, therefore, represents the first assessment framework to comprehensively evaluate the availability of biotic materials. Its aim is to provide a methodology to adequately assess potential restrictions to availability of biotic materials for product systems.

2. BIRD Method

BIRD focuses on terrestrial biotic materials because, firstly, most biotic materials for human consumption except fish are produced from terrestrial materials and, secondly, the availability of aquatic materials is influenced by other aspects (e.g., ocean acidification [34]) and thus should be assessed separately [35]. The aim of the introduced approach is to evaluate possible restrictions to availability of terrestrial biotic materials (in the following terrestrial biotic materials are referred to as biotic materials) along the supply chain. Based on recent publications regarding the availability of abiotic materials (e.g., [1,28,32]) as well as additional aspects of biotic material availability (e.g., Food First Principle [36]) a Top-down-Bottom-up approach (established and already applied for the assessment of abiotic resource availability by Bach et al. (2016) [28]) is applied. Dimensions and categories influencing the availability of biotic materials are identified with regard to supply chain stages where these limitations occur (see Figure 2). Overall the five dimensions physical, socio-economic, abiotic, social and environmental constraints are considered. Physical constraints refer to limited availability of species used as biotic materials and are quantified for the categories biotic resource depletion, replenishment rate and anthropogenic availability. Socio economic constraints decrease the access to biotic materials. Following categories with regard to socio-economic constraints are considered within BIRD: concentration of resources, of harvesting and company concentration, demand growth, political instability, trade barriers, price fluctuations, occurrence as co-product, storage complexity as well as recycling. Phosphorus, land and water availability as well as natural disasters can reduce the occurrence of species used as biotic materials and are assessed within the dimension abiotic constraints. Social constraints refer to limited availability of biotic materials due to challenges regarding compliance with social and environmental standards as well as food security. Possible limitations in availability can occur due to the environmental constraints climate change, acidification, eutrophication, ozone depletion and smog.

The supply chain of products produced from biotic materials can be divided into the following stages: nature, cultivation and harvesting of terrestrial species used as resources/materials, processing of materials (where an intermediate product is the output) and production of several (additional intermediate) products depending on the considered product system (e.g., the final product rapeseed oil has less supply chain stages than the final product biofuel, which is made out of vegetable oils like rapeseed oil). Whereas some categories are only valid for one specific supply chain stage (e.g., replenishment rate), other apply for several stages (e.g., demand growth). Some categories are predominately valid for one supply chain stage, but can—under special circumstances—also influence other supply chains stages, e.g., water availability (these are marked with a dotted grey line in Figure 2).

Most of the categories are valid for biotic resources as well as man-made biotic materials. Exceptions exist for the categories biotic resource depletion and concentration of resources, which are only valid for biotic resources and the categories phosphorus availability and food security, which are only valid for man-made biotic materials.

Furthermore, indicators for quantifying the categories are proposed. All indicators are constructed globally. Thus, possible restrictions to availability are determined as average limitation and do not consider individual regions.

The assessment of product systems is often carried out by LCA [37–39]. Thus, the approach introduced is designed to be implemented into LCA in the future, e.g., the defined categories of the introduced approach are similar to the categories in LCA and the proposed indicators can function as category indicators. In the following the identified dimensions and categories as well as related indicators are introduced in more detail. The approach is further tested in a case study (see Section 3) to analyze applicability and discuss robustness of results.

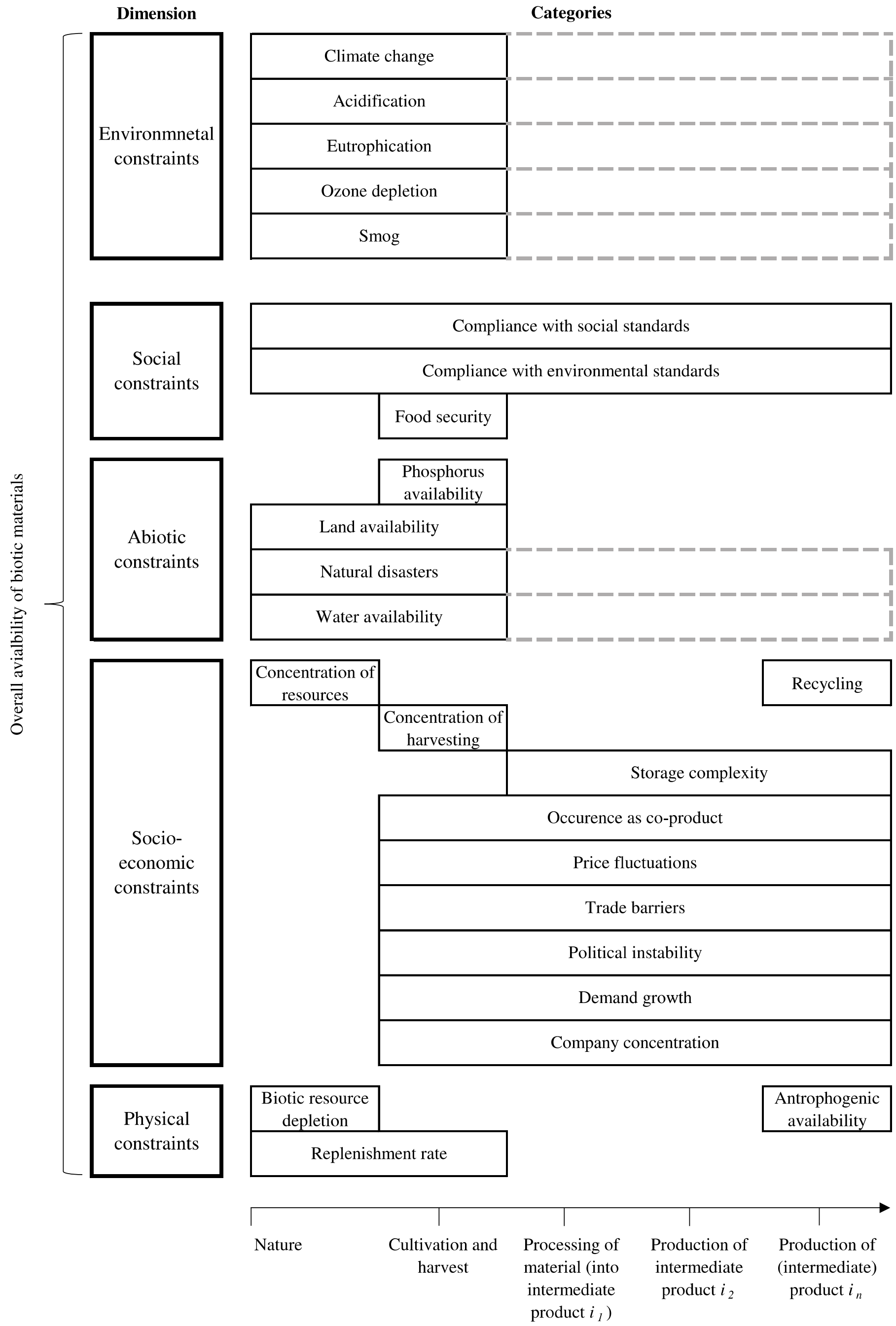


Figure 2. Overview of identified dimensions and categories influencing the availability of biotic materials and (intermediate) products as well as the related stages of the supply chain.

2.1. Physical Constraints

Physical constraints refer to availability restrictions due to limited existence of biotic resources and materials in the ecosphere (environment) and/or technosphere. These restrictions are influenced by existing stocks, extraction rate, replenishment rate and anthropogenic availability (see Figure 3).

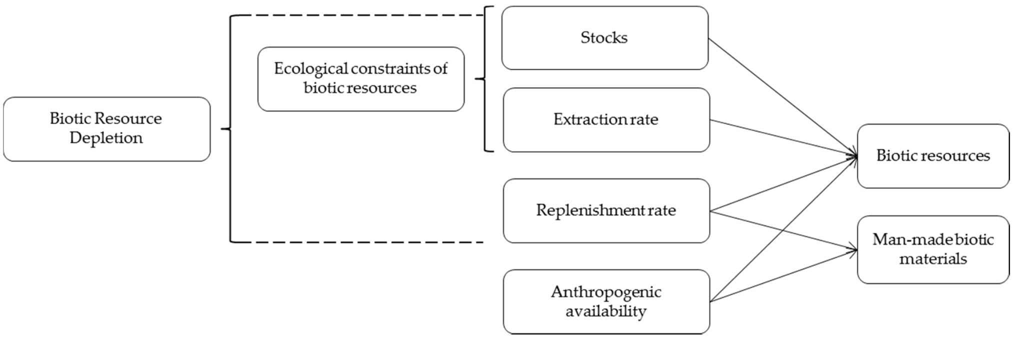


Figure 3. Overview of physical constraints influencing biotic resources and man-made biotic materials.

2.1.1. Biotic Resources

Availability of biotic resources decreases when the amount of resources extracted from the environment exceeds the replenishment rate and therefore decreases the resource stock (biotic resource depletion) [8]. Some basic concepts to measure biotic resource depletion exist modelled in line with the Abiotic Resource Depletion indicator [40,41], by Heijungs et al. (1992) [42] and Sas (1997) [43]. However, these frameworks have never reached a mature level to be applicable in case studies because biotic resource depletion has seldom been considered in LCA. Only very specific products (e.g., exotic animal leather, ivory, rare timber and medical plants) consist of biotic resources.

Based on these studies the Biotic Resource Availability (BRA) indicator according to Equation (1) is proposed as the first approach to determine resource depletion within the BIRD method.

$$\mathrm{BRA_i} = \frac{\mathrm{BRAI_i}}{\mathrm{BRAI_{reference}}} = \frac{\left[\frac{\text{extraction rate}_i - \text{replenishment rate}_i}{(\text{resource stocks}_i)^2}\right] \times \mathrm{TSI_i}}{\mathrm{BRAI_{reference}}} \quad (1)$$

The Biotic Resource Availability Indicator (BRAI) of a species i is set in relation to the BRAI of a reference species. The BRAI is determined by subtracting the replenishment rate from the extraction rate and dividing it by the squared resource stock. The resource stock is squared as the BRA method is based on the Abiotic Resource Depletion Potential approach [40,41]. The higher the extraction rate is the more species are extracted. If the extraction rate is higher than the replenishment rate, existing stocks are depleted. However, as the calculated value is only a snapshot of the current situation and does not reflect the depletion in the last years, the Threatened Species Index (TSI) is considered in addition. Depletion of endangered species is worse as depletion of less or one endangered species. The TSI is based on the evaluation of the rating system of the International Union for Conservation of Nature and Natural Resources (IUCN) Red List of Threatened Species, which is divided into six classes [20,44]. These qualitative classes are translated into quantitative values according to Table 1. For a species of least concern the quantitative value is set to 1, thus, the TSI does not influence the BRAI result. For a critically endangered species the TSI value is set to 100. This way, the BRAI is influenced by the TSI, but not exclusively. To determine the values for the other classes there are divided into 3 categories with the same range (25).

Table 1. Classes of the International Union for Conservation of Nature and Natural Resources (IUCN) (2016) [20] Red List of Threatened Species and translated quantitative Threatened Species Index (TSI) values.

Classes of IUCN Red List of Threatened Species	TSI Value
Least concern	1
Near threatened	25
Vulnerable	50
Endangered	75
Critically endangered	100

To compare the availability of different biotic resources the calculated value is set in relation to the reference species African elephant (this is the same approach as for the Abiotic Resource Depletion indicator [40], which uses antimony as a reference). The African elephant, which is hunted for its ivory, is classified as vulnerable. It is chosen as the reference species because data are easily available (calculations of the BRAI for African elephant are shown in the supplementary material—Section 1). A high BRA refers to high possible restrictions to availability of the considered resource, whereas a smaller BRA relates to lower possible restrictions to availability.

2.1.2. Man-Made Biotic Materials

For man-made biotic materials existing stocks as well as extraction rates are not limiting factors. These materials do not occur naturally and therefore do not have a stock. Furthermore, they are harvested to be cultivated (extraction rate is predefined). Not extracting man-made biotic materials would not automatically lead to an accumulation of these materials as they will be deteriorating in a certain time frame. The replenishment rate (growth period and/or the amount of yield), however, influences the availability of man-made biotic materials as it determines how often and to what extend a species used as a material is replenished. Species with high growth rates can replenish within a short time frame, for example maize, which can be harvested after half a year. Thus, the availability is higher than for species with average (e.g., bamboo, which must grow three years before being harvested) or low growth rates (e.g., shea or walnut trees with an initial growth time of 10 years before the first harvest).

For the assessment of the influence of the replenishment rate on availability within the BIRD method the following approach is introduced (see Figure 4).

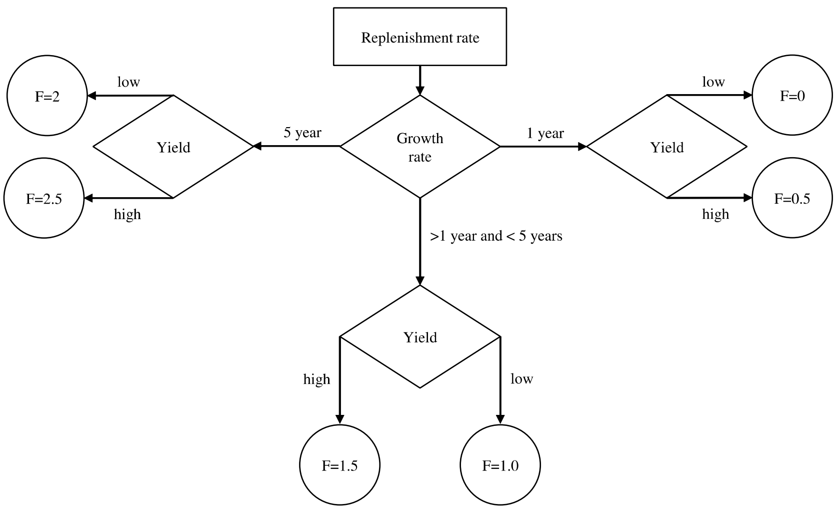


Figure 4. Decision tree to determine the replenishment rate of man-made biotic materials.

The growth rate is divided into three groups: growth rate less than one year, growth rate beyond 1 year but less than 5 years and growth rate beyond 5 years. These groups are determined on the basis that a growth rate below one year is not critical because the species used as a biotic material can be regenerated within a short time frame. A growth rate beyond 5 years is critical as during this time frame (or even longer) no species can be harvested for human purposes. Furthermore, the overall yield of a species influences the availability of a biotic material. In case of a low yield only a small amount of this specific species used as a biotic material is available. Is the yield high, the availability of a certain species is high as well. Thereby, only the average yield of the species is considered, no distinctions regarding different regions are made. However, such a distinction could be made when regionalized data is available. Determining which amount of yield can be considered as low or high is challenging and will not be fully answered within this article (an example on how to classify materials can be found in the case study—Section 3). Low indicator values refer to a high replenishment rate and thus fewer restrictions to availability whereas high indicator values refer to a low replenishment rate and therefore high possible restrictions to availability. The scenario with no (or the lowest) restrictions to availability is assigned the number zero, whereas the other scenarios are assigned a higher number depending on the increasing importance for restricting the availability. The numbers are chosen based on experiences from former work of the authors (e.g., ESSENZ [28], ESP [1]).

The annual yield is not considered to determine the replenishment rate (but for the socio-economic availability) as it does not allow conclusions regarding the overall replenishment capability. If a species is quantitative available is important for the current supply situations (thus considered within the socio-economic availability), but can change when the market structure changes. The replenishment rate of a species is independent of the market structure and only depends on the characteristics of the species.

2.1.3. Biotic Materials

Biotic resources as well as man-made biotic materials are transferred into the technosphere, where they can accumulate and thus, are available to be used further. Currently, there are no existing methods to measure the anthropogenic availability of biotic materials. In the BIRD method, the use of biotic materials is applied as a basis to determine the influence of anthropogenic stocks on the availability of biotic materials (see Table 2). Whether a biotic material enriches the anthropogenic stock depends on its original use. Materials, which are consumed (e.g., as food, feed or fuel) during their first use phase cannot be reused. Materials used for or in products stay in the technosphere and thus have the potential to be reused. However, their reuse depends further on the product design. If the product is almost completely made out of a biotic material like paper or wooden furniture, it is likely to be recycled. Products where the fiber of the biotic material is used together with several other materials like in bio polymers are harder to recycle and thus, are often incinerated after one use phase. However, because they are accumulated in the technosphere, they have the potential to enrich the anthropogenic stock when improved recycling technologies are available. The values for quantification were chosen based on the experiences of the authors obtained from former method development. However, as other indicator values within this approach rank from 0 to 1, a similar scale was preferred. To evaluate to what extent a biotic material contributes to the anthropogenic stock (ASR—anthropogenic stock restraint) the global production shares (sgp) of a material i are multiplied with the quantitative factor *F* and then summed up (see Equation (2)).

$$\mathrm{ASR_i} = \left(\mathrm{sgp_{i,F}} \times \mathrm{F_F}\right) + \left(\mathrm{sgp_{i,P1}} \times \mathrm{F_{P1}}\right) + \left(\mathrm{sgp_{i,P2}} \times \mathrm{F_{P2}}\right) \quad (2)$$

A high ASR refers to a low contribution to the anthropogenic stock, whereas a low ARA refers to a high contribution. This approach is used as no data is available on anthropogenic bio-based products.

Table 2. Classes related to use of the biotic material and translated quantifiable factors.

Classes Related to Use of Biotic Material	Quantitative Factor F
F: Food, feed and fuel	1
P1: Product, made from several materials	0.5
P2: Product, primarily made from biotic materials	0

2.2. Socio-Economic Constraints

The socio-economic availability of materials is influenced by structural conditions of the market as well as societal structures inhibiting the supply security. For example, the political instabilities of a country can lead to restraints in availability as e.g., corruption or revolutions disrupt the ability to effectively implement robust policies including ones related to material export, etc. So far several methods to determine the socio-economic availability of abiotic materials exist (e.g., [15,28,32]), which are not adapted for the application to biotic materials but can be used as a basis to determine socio-economic aspects influencing the availability of biotic materials. Within the method of the Association of German Engineers (Verein Deutscher Ingenieure—VDI) biotic materials are considered, but often only evaluated through expert judgment [45]. The study of (Fraunhofer 2013) [31] provides a first assessment methodology regarding the availability of biotic materials. Considered aspects are substitution, recycling capability, concentration of producing countries as well as political stability. Thus, the method by Fraunhofer (2013) [31] and (VDI 2013) [45] are used as a basis for the development of a comprehensive approach for the socio-economic availability of biotic materials. Furthermore, the ESSENZ method [28] developed by the authors is taken into account as it is a methodology to assess the resource efficiency including the socio-economic availability of abiotic resources for product systems.

In the following, the categories as well as associated indicators for quantifying these categories are introduced. Overall 10 potential economic constraints leading to possible supply shortages along the product's value chain are identified. For all categories high values are referring to high restrictions to availability and low values relate to low restrictions.

2.2.1. Concentration of Resources, Harvesting and Company Concentration

A high concentration of an activity (e.g., trading biotic materials) refers to the extent to which a relatively small number of companies or countries account for a large share of this activity (e.g., [32,46]). High concentrations increase potential restrictions to availability. In the introduced approach the concentration of resources, company concentration and concentration of harvesting are considered. The concentration can be measured by the Herfindahl-Hirschmann-Index (HHI) [47], which is calculated as the sum of the squared market shares (global production share (sgp)) (see Equation (3)) and ranks from 0 to 1.

$$HHI_i = \sum\left(sgp_{i,x}\right)^2 \tag{3}$$

By determining the concentration of resources, the number of countries where the considered species (resource) is available and can be extracted as well as their share in the global stocks are reflected. In case all species occur in only few countries, the concentration and as consequence risk of limited availability is high. This category only applies to species used as biotic resources and not to man-made biotic materials because only species used as biotic resources occur in nature.

Concentration of harvesting refers to the number of countries harvesting species used as biotic materials and the share of the globally produced material. Limited availability can occur when most of the harvesting activities occur in only few countries. This category applies to biotic resources as well as man-made biotic materials because for both species are harvested.

Company concentration reflects the number of companies trading and their share of the globally produced material. When only few companies market most of the materials, a high company concentration occurs, which can reduce the availability of resources. High company concentration can have an influence in every supply chain stage. This category is important for biotic resources as well as man-made biotic materials because the availability of both is influenced by companies trading them.

2.2.2. Political Instability

The risk of limited availability of biotic materials is higher for unstable countries, where political systems and legal procedures are not reliable. For example, potential uprisings and corruption might interrupt the cultivation and harvest of species used as biotic materials. Politically unstable countries can influence the availability of biotic materials over the whole supply chain. Next to cultivation and harvesting, also processing and production of (intermediate) products can take place in unstable countries. Political instability of countries can be a limiting factor for the availability of biotic resources as well as man-made biotic materials because both might be processed in instable countries. The quantification of the political instability (PIS) is based on the Worldwide Governance Indicators [48,49]. The indicators consider the key aspects voice and accountability, political stability and absence of violence, government effectiveness, regulatory quality, rule of law and control of corruption for over 210 countries. As all six indicators reflect fragments of an unbalanced system, they are combined to an aggregated evenly weighted index ($WGII_x$). To determine the political instability in relation to a biotic material i the material's global production (or consumption) share (sgp) per country x is multiplied with the $WGII_x$ and summed up (see Equation (4)). The global production shares are used as a basis to determine the country distribution of species used as biotic materials for the supply chain stages nature as well as cultivation and harvest. The global consumption shares are applied as a basis to determine the country distribution regarding the production of biotic materials (made out of species) for the supply chain stages processing of material (into intermediate product), production of intermediate product(s) and production of the final product. If the specific countries in which production occurs are known, these shares should be used instead of the generalized country distribution.

$$PIS_i = \sum (sgp_{x,i} \times WGII_x) \quad (4)$$

2.2.3. Demand Growth

Demand describes the need for biotic materials. Demand growth occurs when the demand is increasing. When the demand is higher than the amount of materials currently obtained, possible restraints to availability can occur. Demand growth can occur in all supply chain stages because not only the demand of harvested materials but also of (intermediate) products can increase. It occurs for biotic resources as well as man-made biotic materials as both are used. If the demand for one specific (intermediate) product increases, the demand of harvested materials increases as well. Demand growth (DG) of raw materials is determined by calculating their production (or consumption) increase (or decrease) over the last five years (see Equation (5)).

$$DG_i = \frac{\sum_1^5 \left(\frac{\text{global production of year n + 1}}{\text{global production of year n}} - 1 \right)}{4} \quad (5)$$

2.2.4. Trade Barriers

Barriers to trade regarding the export (e.g., export duty) of biotic materials can limit their availability. This might occur when biotic material producing countries reduce or terminate the export of specific materials or (intermediate) products. These trade barriers can occur in all stages of the supply chain since both harvested and processed materials or (intermediate) products can be subject to trade. Trade barriers apply to biotic resources as well as man-made biotic materials. To quantify the trade barriers (TB) the Enabling Trade Index (ETI) [50] is used. The ETI is established by the World Economic Forum and ranks countries regarding their policy for trading goods. To determine existing trade barriers in relation to a biotic material i the global production (or consumption) share (sgp) of the material i per country x is multiplied with the ETI and summed up (see Equation (6)).

$$TB_i = \sum \left(sgp_{x,i} \times ETI_x \right) \tag{6}$$

2.2.5. Price Fluctuation

Prices of biotic materials fluctuate depending on current market situations. Companies consider predictable price fluctuations in their raw material planning processes. However, when unexpected price fluctuations occur, compensation might not be possible and the availability of materials is restricted. Price fluctuation can occur in every supply chain stage for biotic materials as well as (intermediate) products because they are sold at the world market. Fluctuations can be quantified by the volatility indicator, e.g., [51].

However, often the necessary market data is not available to calculate the volatility of a biotic material. Thus, other indicators have to be applied, e.g., commodity price index by Barrientos and Soria (2016) [52].

2.2.6. Occurrence as Co-Product

Production processes are established to produce a specific main product (e.g., production of oil from rapeseed). Next to the main product co-products can be produced alongside (e.g., rapeseed cake). If the economic importance of the main product is decreasing and production is declining, the co-product is not produced further as well. This leads to limited availability of the co-product [53–55]. Occurrence as co-product can influence the availability over the whole supply chain and affects biotic resources as well as man-made biotic materials since both can occur as main and co-products. Qualitative information regarding occurrence of co-products can be transformed into quantitative values according to the scheme presented in Table 3. The values are assigned based on the ESSENZ approach [28]. One is set as the highest value and is divided into 3 same-range categories (as numbers for three other classes have to be assigned). The class only mined as main product is set to zero as restrictions to availability are not to be expected.

Table 3. Qualitative information about main and co-products transferred into quantitative data.

Information Regarding Production as Main or Co-Product	Quantitative Value
Only main product	0
Mostly main product	0.33
Mostly co-product	0.67
Only co-product	1

2.2.7. Storage Complexity

The produced materials and (intermediate) products might have to be stored before they are used. Depending on the characteristics of the material it can be difficult to provide adequate storage conditions. For example, some agricultural products need to be refrigerated and/or turned over for air circulation. The complexity of storage can lead to possible restrictions to availability as the biotic materials can decompose and therefore cannot be used further. It is relevant for all supply chain stages. Storage has to be provided for biotic resources as well as man-made biotic materials because both are stored during their life time. The moisture content of the material and (intermediate) product can be related to most of the storage challenges, e.g., insect infestation, fungal growth and turn-over of the goods [56–58], so it is used as an indicator to describe storage complexity. When the moisture content is low (and therefore there is less water in the product) the complexity of storage decreases. The moisture content can differ depending on the country where the biotic material is cultivated and stored as the humidity in different countries varies. Hence, if data is available, the moisture content of a biotic material should be determined for every storing country individually. Since the successful storage also highly depends on the countries development [59], the economic vulnerability of the country where the material or (intermediate) product is being stored is also taken into account. Economic vulnerability refers to the weakness of a country to absorb and overcome severe shocks while supporting strong economic growth [60]. It is assumed that a country with a low vulnerability is also able to handle complex storage of materials and (intermediate) products. Thus, the Economic Vulnerability Indicator (EVI) by Organization for Economic Cooperation and Development—OECD (2016) [60] is used as the basis for quantification. The storage complexity is determined by multiplying the global production share (sgp) with the EVI. Furthermore, the value is added to the moisture content (mc) (see Equation (7)). Both components can range from 0 to 100 and thus equally influence the result.

$$SC_i = mc_{i,x} + \sum\left(sgp_{x,i} \times EVI_x\right) \tag{7}$$

2.2.8. Recycling

During production of (intermediate) products primary as well as secondary materials might be used. If more secondary materials are utilized less primary materials have to be produced. Thus, the demand for primary materials is reduced and the overall availability increases. The higher the recycled content of a product is, the less primary material has to be harvested. To determine the primary material use (PMU) the recycled content of a product (given in percentage) is subtracted from 100% (see Equation (8)).

$$PMU_i = 100\% - \text{recycled content}_i \tag{8}$$

2.2.9. Determination of Characterization Factors

Based on the method by Bach et al. (2016) [28] for abiotic resources the determined indicator values are transferred into characterization factors (CFs) using the Distance-to-Target (DtT) approach. In the following this 4-step DtT approach is introduced:

(Step 1) For all categories indicator values have to be determined (as shown in Sections 2.2.1 and 2.2.8) for the materials of the study.

(Step 2a) Targets for all categories have to be determined. For nine of the ten categories targets are already established in the approach for abiotic materials. These targets can be used for the assessment of biotic materials. The approach how the target values were established can be found in the publication by Bach et al. (2016) [28]. These values are introduced as default values. Thus, they can be adapted by practitioners and stakeholders according to their preferences. There is no target for the category storage complexity available because the category was not considered in the approach for abiotic materials. Thus, a target value is set to 60 for the category. This value is not established based on stakeholder survey or expert judgment, but only by the authors to show the applicability of the DtT approach for BIRD. The value of 60 was considered reasonable as it is assumed that a moisture content of 30% as well as a vulnerability of 30 (based on the global production of a material and the corresponding EVI) can be seen as potentially limiting the availability of biotic materials significantly.

(Step 2b) Based on the ecological scarcity approach by Müller-Wenk et al. (1990) [61] and Frischknecht et al. (2009) [62] the indicator values are set in relation to the target to determine the DtT value for each material i in each category c (see Equation (9)). Is the DtT value lower than 1, no constraints on availability can be expected. Thus, the DtT value is set to zero. If the DtT value is equal 1 or greater than 1 possible availability restrictions might occur. The ratio of the current to the critical flow is squared to weigh the exceeding of the target above proportional [62].

$$DtT - value_{i,c} = \left(\frac{\text{indicator value}_{i,c}}{\text{target}_c}\right)^2 \tag{9}$$

(Step 3) To take the overall amount of the material currently produced into account, the DtT values are normalized (nDtT) (see Equation (10)). For raw materials with small amounts of production, e.g., cotton, the restrictions quantified in the ten categories can be even more significant compared to materials for which the overall annually produced amount is higher, e.g., wheat.

$$\text{nDtT value}_{\text{i,c}} = \frac{\text{DtT value}_{\text{i,c}}}{\text{normalization value}_{\text{i}}} \tag{10}$$

(Step 4) The nDtT values are expressed in small numbers which are challenging for the application within the LCA framework (in LCA the characterization factor is multiplied by the amount of the material in the product system). Thus, they are scaled to 6.3×10^{15} (this number was chosen as it presents the highest global production value of total grains in the year 2015 according to United States Department of Agriculture (USDA) [63]. According to Equation (11) the final CFs are calculated. The highest value of each category ($\text{nDtT value}_{\text{c,i,max}}$) is set to 6.3×10^{15}. The CFs of the other materials are calculated by applying the rule of three [64]. In the case study, the application of this approach is demonstrated.

$$\text{CFs}_{\text{i}} = \begin{cases} \text{nDtT value}_{\text{c, i, max}} \rightarrow 6.3 \times 10^{15} \\ \text{other values of category are calculated} \rightarrow \frac{6.3 \times 10^{15}}{\text{nDtT vlaue}_{\text{c, i, max}}} \times \text{nDtT value}_{\text{c, i}} \end{cases} \tag{11}$$

2.3. Abiotic Constraints

Abiotic constraints refer to abiotic factors influencing the ecosystem at the location where the species used as biotic material grows. Several abiotic constraints influence the environment where species used as biotic materials grow (see Figure 5). They are divided into 'constraints on the ecosystem' and 'limiting factors: resources needed for cultivation'. Some of these constraints impact the availability of species used as biotic materials locally, others more globally. They are equal for biotic resources and man-made biotic materials.

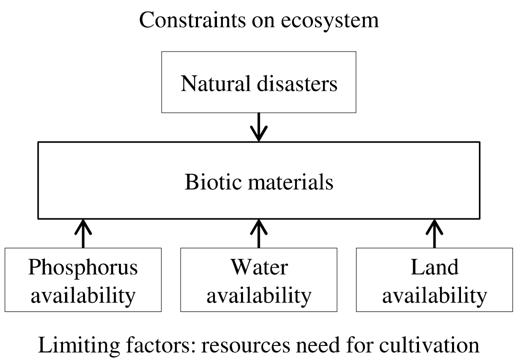


Figure 5. Overview of abiotic aspects impacting biotic materials.

Water availability is one very dominant aspect since over 70 percent of all water resources worldwide are used for agriculture purposes [65]. Thus, the cultivation and harvest of species used as biotic materials highly depend on the local water scarcity, which is influenced by local precipitation, run-offs evapotranspiration and consumption [66,67]. When a species used as a material predominantly grows in water scare regions, the possibility of restricted availability is higher than for species grown in water rich regions [68]. Methods to assess water scarcity of bio-based products systems exist and have been tested in several case studies, e.g., [69,70]. To determine possible restrictions to availability due to water scarcity (WS), the global production share (gps) of a biotic material i is multiplied with the water depletion index (WDI) by Berger et al. (2014) [71] and summed up according to Equation (12).

$$WS_i = \sum(gps_i \times WDI_x) \quad (12)$$

The higher the WS_i, the higher are the potential restrictions to availability for a biotic material. Water availability is especially important for the cultivation stage as species used as biotic materials need water to grow. However, lack of available water can also impact other supply chain stages, e.g., when industrial processes require large amounts of water.

Furthermore, land and phosphorus availability can be constraints for accessibility of species used as biotic material. Currently enough land and phosphorus are available for cultivation of species used as biotic materials. However, several studies predict (e.g., [71–78]) that with increasing use of biotic materials (e.g., for biofuels) in the next decades not enough land and phosphorus will be available to meet all human needs (e.g., for food, feed and industrial processes). However, both challenges are rather universal and are less related to one specific species used as a biotic material [79,80]. Only phosphorus and no other soil nutrients are considered, because phosphorus is a limited resource itself and thus can limit the production of agricultural products significantly. Other nutrients like potassium, calcium, sulfur, magnesium and nitrogen (which can be easily extracted from the air e.g., via Haber-Bosch process [81]) are not considered being scarce as they are available in great quantities within nature or can be easily recycled [82–84]. Thus, limitations to their availability are not expected in the near future and, therefore, not included within this approach.

Phosphorus amounts used in the cultivation stage are reported frequently in case studies focusing on agricultural systems (e.g., [85–87]). To assess land use and accompanied environmental impacts, several methods exist (e.g., [26,88,89]). Often, the overall area in hectare per year (ha/year is also reported. However, these methods have so far only been used in the context of assessing the impacts of a product system, but not for the evaluation of possible restrictions to the availability of species used as biotic materials.

The use of land and phosphorus to produce species used as biotic materials is considered within the BIRD method by reporting the amount of the land used (in h/year) and phosphorus applied (in kg) in a specific product system. Thus, by comparison of two or several product systems a statement is possible regarding the land and phosphorus use. The amount of land use for the cultivation of a specific species used as a material (with regard to the functional unit) can be determined either by measurements provided by the practitioner (e.g., farmer) or average values based on literature research or within common LCA databases like GaBi [90] and ecoinvent [91]. For the determination of phosphor use either measured values provided by the practitioner (e.g., farmer) or average values based on literature research can be used [92].

Another abiotic constraint is the occurrence of natural disasters. Natural disasters can affect regions by floods (FL), droughts (DR), hurricanes, earthquakes, volcanic eruptions, forest fires, landslide, pests and diseases [93]. In the last years natural disasters have impacted the agriculture dramatically: around 30% of all agricultural products were destroyed by natural disasters between 2003 and 2013 [94]. Especially droughts and floods play an important role with regard to agricultural product loss, whereas volcanic eruptions, earthquakes, hurricanes, forest fires, pests and diseases—even though have dramatic outcomes when they occur [95]—do proportionally not destroy as much agricultural products [93,94,96,97]. Natural disasters are monitored and reported by several organizations (e.g., [98,99]). However, these factors are so far only being applied for monitoring purposes, but not in the context of assessing possible limitations to the availability of biotic materials.

For the quantification of the influence of natural disasters on the availability of biotic materials within the BIRD method the natural disaster risk (NDR) indicator is determined on country level based on data by (United Nations Office for Disaster Risk Reduction 2013) [100]. The impacts within a specific region x are summed up and multiplied with the global production share of the material considered (see Equation (13)). The higher the NDR_i the more likely are possible restrictions to availability of biotic materials. Natural disasters can affect biotic resources as well as man-made biotic materials. For the supply chain stage, cultivation and harvesting natural disasters are predominantly important. However, they might also affect other supply chain stages, when, e.g., producing plants are destroyed by floods.

$$NDR_i = \sum gps_{i,x} \times (FL_x + DR_x) \quad (13)$$

2.4. Social Constraints

Social constraints refer to societal aspects which limit the availability of resources and man-made biotic materials. For once the "Food first" principle has to be considered when assessing availability of biotic materials. The main function of biotic materials as an agricultural product is to provide food for human consumption. The use in industrial processes can only be an option, when food security in all countries is guaranteed [100]. As of today several studies exist discussing food security in the global context. Most studies agree that currently no food crises is initiated due to biotic material use in the industrial sectors, but rather socio-economic aspects in the countries are responsible for food scarcity [101]. However, when demand of biotic materials significantly increases, food security could be impaired. So far there are no methods to estimate if biotic materials violate the "Food first" principle. To comply with the "Food first" principle it has to be ensured that the used biotic material is not traded by a country where food crises occur. Thus, for the assessment within the BIRD method an indicator is introduced to measure the risk of a material to be exported by a country, which cannot ensure food security. This indicator is determined according to Equation (14), where the global production share (pgs) of a material i is multiplied by the food security index (FSI) [102] of the related country x and then summed up (see Equation (14)). The FSI is created based on the food security indicators (e.g., depth of the food deficit, cereal import dependency ratio, etc.) by FAO (2016) [102] (see supplementary material—Section 2 on how the FSI is constructed). The food first principle is important for man-made biotic material as they are used for food and feed. Thus, the supply chain stages nature, cultivation and harvesting are impacted.

$$FSI_i = \sum (gps_i \times FSI_x) \quad (14)$$

Furthermore, social constraints can also occur due to lacking societal acceptance with regard to a company's compliance of social standards. The consumers' perception of the company has been influencing the decision to buy products more and more in the recent years [103,104], e.g., consumers boycotted blueberries due to poor working conditions of farm workers [105]. In the worst case, a certain material cannot be used by a company because of its low societal acceptance, even though it is available from a physical and socio-economic perspective. Furthermore, societal dismissal can also occur with regard to compliance with environmental standards, e.g., consumers are boycotting palm oil as it is seen as one major contributor in destroying orangutan habitats [106]. Bach et al. (2016) [28] developed an approach to measure the compliance with social standards based on data by Social Hotspot Data Base (SHDB) [107,108] and with environmental standards based on the Environmental Performance Indicators (EPI) by [109] for abiotic resources.

To determine the compliance with social standards aspects with low societal acceptance are identified as child labor (CL), forced labor (FL) and high conflict zones (CZ). For these aspects data from the SHDB [107,108] is identified. The SHDB provides data on country and sector level. Several sectors are available for the evaluation of biotic materials (including crops, food products, oil seeds, plant based fibers, sugar cane, vegetables and wheat). Based on the product system under investigation the appropriate sector/s have to be identified by the practitioner. The screening indicator for a material i is determined by multiplying the three social hotspot indexes with the global production shares (sgp) x and summing them up (see Equation (15)). Compliance with social standards is important for biotic resources as well as man-made biotic materials and play a role in all supply chain stages as within every stage child labor, forced labor and high conflict zones may occur.

$$SC_i = \sum[sgp_x \times (CL_x + CZ_x + FL_x)] \quad (15)$$

For the quantification of compliance with environmental standards the EPI [109] is applied. The EPI provides 16 sub indicators measuring the performance of countries regarding their environmental protection efforts including protection of biodiversity. Therefore, the sub indicators Critical Habitat Protection (CHP), Marine Protected Areas (MPA) and Terrestrial Protected Areas (TPA) are used to determine the compliance with environmental standards. It is assumed, that the manner of a country taking care of its biodiversity is similar to the overall compliance with environmental standards. The compliance with environmental standards (EC—environmental compliance) is determined by multiplying the global production shares of the countries (gps_x) with the EPI indicators and summing it up (see Equation (16)).

$$EC_i = \sum[sgp_x \times (CHP_x + MPA_x + TPA_x)] \quad (16)$$

Compliance with environmental standards is important for biotic resources as well as man-made biotic materials. For both the extraction and cultivation can occur with high loss of biodiversity, which is of more concern to consumers as gradual pollution of the environment over time and therefore indirect loss of biodiversity. Thus, species used as biotic materials causing direct loss of biodiversity have a higher potential restriction as species causing indirect loss of biodiversity. Environmental compliance plays a role in all supply chain stages as within every stage environmental pollution can occur. However, as the direct loss of biodiversity (which mostly occurs due to harvesting of agricultural plants) is seen as more severe by consumers the supply chain stage harvesting is of most concern.

2.5. Environmental Constraints

Environmental impacts of species cultivation, extraction and use can lead to various impacts, which can change the ecosystem significantly up to the point where the cultivation of species used as biotic materials and thus the availability of these materials is jeopardized [11,110]. Emissions during cultivation, extraction and use (as well as the end of life) can lead to direct and indirect pollution of the environment (e.g., acidification, eutrophication, etc.) including degradation of soils, contamination of freshwater, etc. which are extremely important for the successful and efficient cultivation of species used as biotic materials (see Figure 6) [111].

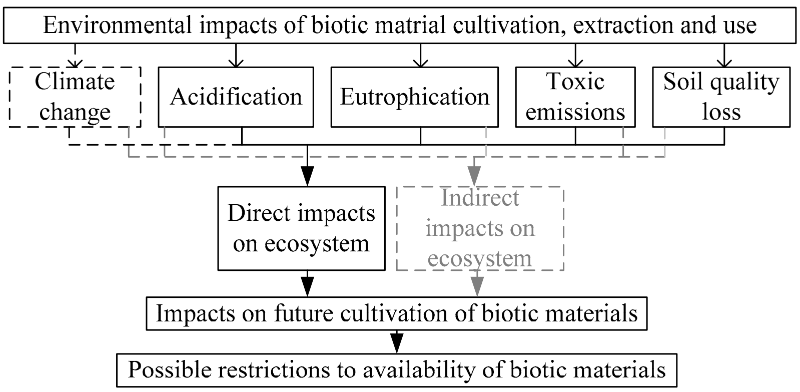


Figure 6. Overview of cause-effect-chain with regard to environmental impacts and restrictions to availability of biotic materials.

Thus, environmental impacts can constraint the availability of biotic materials. The LCA method has been used for decades to assess environmental impacts over the entire life cycle of products [38]. Several methods exist within the LCA framework assessing impacts to water, soil and air, e.g., [37]. For the assessment of biotic materials, following aspects are relevant and should be considered: impacts into soil, water and air (like eutrophication, acidification and toxicity) as well as resulting impacts like soil quality loss and biodiversity loss. Furthermore, impacts due to land use and land use change have to be taken into account. Climate change is taken into account as one of the most important prospective factor regarding changes in ecosystems. These changes can lead to a reduced availability. Several studies confirm that climate change leads to, e.g., extreme weather events influencing the availability of agricultural products (e.g., [112–114]).

However, not for all these aspects mature methods are available. As shown by [115–117] several methods have high uncertainties and thus, have to be applied with caution. However, in the current work of the UNEP/SETAC Life Cycle Initiative some more mature methods are developed [118]. Currently several impact assessment methods are tested in the Product Environmental Footprint (PEF) initiative [119]. In case these methods are evaluated to be mature enough for implementation, they should be added to the ones proposed here. For the BIRD method it is recommended to only use mature impact assessment methods to ensure adequate decision making. These mature methods are CML-IA [37] for acidification, eutrophication and photochemical ozone formation, the Intergovernmental Panel on Climate Change (IPCC) method for climate change [120] and the World Meteorological Organization (WMO) method for ozone depletion [121]. Environmental constraints are important for species used as biotic resources as well as man-made biotic materials as for both of them the underlying ecosystem can be affected. Even though these environmental impacts can occur over the whole supply chain, there are mostly affecting the supply chain stages 'nature' and 'cultivation and harvest'. However, the other supply chain stages can be affected also indirectly; e.g., when due to water pollution (e.g., acidification, eutrophication) not enough clean water is available for industrial processes. Even though the state of the environment is directly related to the availability of species used as materials, it is so far not determined to which extend. Emitting twice the greenhouse gases will not lead to double the restrictions to availability. Thus, the LCIA results cannot be directly related to the potential restrictions to availability. However, as a first approach the principle "less impacts, less constraints" is used.

3. Case Study

In the following the BIRD method is applied in a case study. For simplicity only two materials are considered. For the case study 1 L biofuel made from rapeseed or soy beans is analyzed. For both plants the cultivation and harvest of the agricultural products as well as all processing steps are taken into account.

3.1. Physical Constraints

The considered biotic materials can be classified as man-made (and are therefore not a biotic resource). These plants might be also available within nature, but are not being harvested to produce biofuels in this case. Thus, resource depletion is not considered (an example how to calculate the BRA for a biotic resource is included in the Supplementary Materials—Section 3.1.1). Physical limitations on man-made biotic materials include anthropogenic constraints as well as restrictions due to replenishment rate. The replenishment rate is determined according to Figure 4. Both rapeseed and soy beans have a growth rate less than one year [122]. Furthermore, both have a high yield, which is one reason they are cultivated for producing biofuels [123]. Thus, the corresponding indicator value is zero, which means that for both plants limitations to availability due to replenishment are not a limiting factor.

To determine the anthropogenic restraints of the biotic materials the concept presented in Section 2 is applied. Data is lacking with regard to the amount of globally produced rapeseed and soy beans used for biofuel. Thus, the shares of the largest consumers of rapeseed and soy beans are applied instead based on the data by Barrientos and Soria (2016) [52]. The five biggest consumers of rapeseed are EU, Canada, China, USA and India: for soy beans USA, Brazil, Argentina, China and EU can be identified [52]. In the considered countries rapeseeds and soy beans are used for food, feed and fuel production [124,125]. No data was found stating if soy beans and rapeseeds are used within any products or for biofuels only. However, if they are used for products the amount is most likely very small and would not change the overall result significantly. As the factor for both materials in all considered countries is 1, the overall anthropogenic constraints result in 1 as well (for more details see Supplementary Materials—Section 3.1.2). Both materials are predominantly used for biofuels, food or feed and thus are consumed and cannot be used again (see Figure 7). Hence, they do not contribute to the anthropogenic stock.

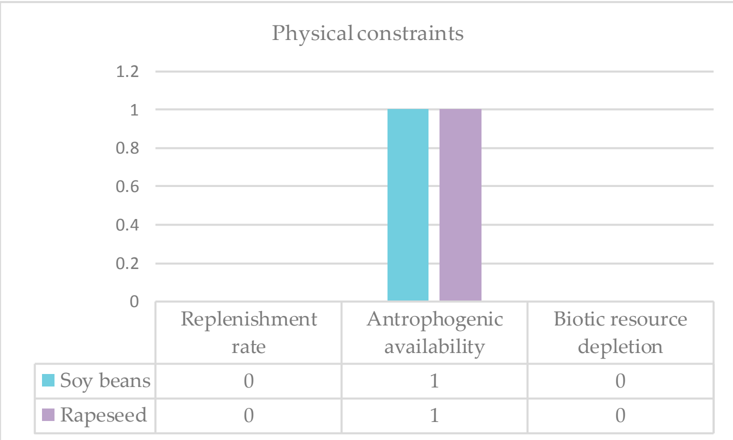


Figure 7. Results for the dimension physical availability of rapeseed and soy beans.

3.2. Socio-Economic Constraints

In the following, the results for the category indicators for the dimension socio-economic constraints are shown (detailed calculations can be found in the Supplementary Materials—Section 3.2):

Concentration of resources: This category is not considered as only man-made biotic materials are considered in the case study.

Company concentration: This category (more precise categories as company concentration plays a role in all supply chain stages) could not be determined because there is no global data available with regard to companies trading rapeseed and soy beans.

Concentration of harvesting: This category is determined for the harvesting step of soy beans and rapeseed by applying the HHI according to Equation (6) based on data by Barrientos and Soria (2016) [52]. To determine the HHI the global production shares are squared and summed up. For soy beans the HHI is 0.25 and for rapeseed it is 0.13. Only three countries (USA, Brazil and Argentina) produce around 82% of all soy beans worldwide [52]. For rapeseed the three biggest producers (Canada, China and EU) worldwide add up to around 78% [52]. However, as within the EU overall 26 countries produce rapeseed [52], the HHI is lower as for soy beans. Thus, potential limitations to availability due to concentration of harvesting are higher for soy beans than for rapeseed. However, considering the target value of the category (0.15) potential restrictions occur only for soy beans.

Political instability: Political instability determined according to Equation (3) can occur during the cultivation as well as during processing of the materials and production of biofuel. It is determined by multiplying the global production or consumption share [52] with the WGI [49]. For the cultivation and harvesting step global production data [52] are used whereas global consumption data [52] are applied for the processing and the final product step. For the production step the political instability is

1.9 for soybeans producing countries and 1.4 for rapeseed producing countries. For soybeans the three countries with the highest production contribute most to the result: Brazil, Argentina and USA [52]. However, especially the contribution of Brazil and Argentina is significant since their WGI values are high [49]. For rapeseed, China influences the result the most, even though it is only the third biggest producing country (next to Canada and the EU). Thus, possible limitations to availability due to political unstable countries are higher for soy beans than for rapeseed. When considering the target value (1.9) potential restrictions occur only for soy bean. It is assumed that soybeans and rapeseed are pressed into meal and oil, which are further processed into biofuel within the same country. This assumption is made as global data regarding rapeseed and soy bean oil (and meal) production is not available. Thus, the political instability can be determined once for both steps using the global consumption share based on data by Barrientos and Soria (2016) [52]. For the processing and product step limitations due to political instability for rapeseed add up to 1.3, whereas for soy beans the limitations are lower with 2.1. The biggest consuming country of soy beans is China [52], which also has a high WGI [49]. For rapeseed the biggest consuming country is also China [52]. Considering the target value (1.9) only for soy beans potential restrictions to availability occur. However, compared to the cultivation and harvesting step, the possible limitations are higher within the processing and product step.

Demand growth: The demand growth for the cultivation stage is calculated according to Equation (4) based on USDA (2016) [126] providing annual production data of soy beans and rapeseed. The demand growth is 7.6% for soy beans and 2.6% for rapeseed. Annual production data are used to determine the demand growth in the processing step based on USDA (2015) [127] for all countries producing biodiesel out of soy beans and rapeseed. The demand growth for soy beans is 8.9%. For rapeseed the demand growth adds up to 1.6%. Therefore, the possible restrictions to availability due to demand growth are much higher for soy beans than for rapeseed. Furthermore, for soy beans the demand growth of the processing step where biodiesel is produced is higher than for the soy bean production itself. Since the demand growth of both materials in the considered supply chain stages is above the target (5%), potential restrictions to availability occur for both materials.

Trade barriers: Trade barriers are determined by multiplying the global production shares [52] with the ETI [50] and aggregating the values (see Equation (5)). Trade barriers can occur when soy beans and rapeseed are harvested and exported for further production as well as when biodiesel is produced. Trade barriers are sum up to 3.4 for soy beans and 2.4 for rapeseed. For rapeseed the country influencing the result the most is China, as it is one of the biggest producers [52] and has several trade restrictions [50]. For soy beans the country with the biggest influence is Brazil. Considering the target value (3.15) potential restrictions to availability occur only for soy beans. For the processing step trade barriers are determined using country consumption data [52] and summed up to 3.2 for soy beans and 2.2 for rapeseed. Here China has the highest influence both for rapeseed and for soy beans. Considering the target value (3.15) potential restrictions to availability occur only for soy beans. Thus, the potential restrictions to availability due to trade barriers are higher for soy beans than for rapeseed within all stages. Furthermore, restrictions are higher for countries cultivating the species used as biotic materials than for countries producing biodiesel.

Price fluctuations: As sufficient data regarding the monthly prices over the last five years are not available for soy beans and rapeseed, the commodity prices index published by Barrientos and Soria (2016) [52] is used to assess price fluctuations for rapeseed oil and soy bean oil (processing step). The commodity price index is an average of selected commodity prices based on monthly or daily prices over the period of several months [52,128]. Prices for soy bean oil vary much more (7.1%) than prices for rapeseed oil (2.7%). Thus, possible restrictions to availability due to price fluctuations are higher for soy bean oil than for rapeseed oil. Data regarding the price fluctuations of rapeseed and soy beans as well as biodiesel are not accessible. Since both materials are below the target (20%), potential restrictions to availability do not occur for both materials.

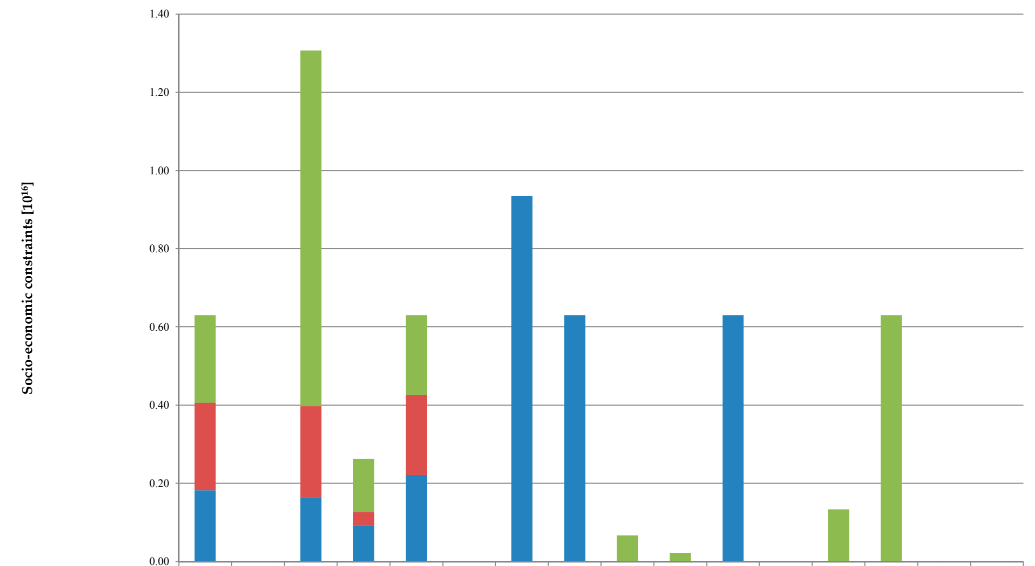
Occurrence as co-product: When considering the supply chain of soy beans and rapeseed, co-product occur within every stage. After harvesting straw is left, which often remains on the field, but can also be used to produce second-generation biofuels [129]. During processing soy beans and rapeseed are pressed into oil and cake [130]. The oil is being processed further into biofuel by e.g., refinement of oils, transesterification, etc. In all these process steps co-products occur [130]. To determine if possible restriction to availability, all products need to be classified as shown in Table 3 for every supply chain stage. Both materials can be classified as 'only main product' for the harvesting step. Even though the straw can be used for production of second-generation biofuels, the main products are still the soy beans and rapeseed. Thus, both materials get assigned the value of zero. For the next step—material pressing—the co-product cake is an important product for animal feed and thus, soy bean and rapeseed oil are classified as 'mostly main products' (value of 0.33). Even though several co-products are generated during the biodiesel production, the main product is still biodiesel and thus it can be classified as 'only main product' (value of zero). There is no difference for soy beans and rapeseed as both are processed into biodiesel in a similar way. Since both materials are below the target, no potential restrictions to availability occur.

Storage complexity: This category has to be determined for the storage of soy beans and rapeseed as well as for the storage of the produced oils and biodiesel. However, as the products oils and biodiesel can be stored for a long time and both have a high economic value [131], it is assumed that no potential restrictions due to storage complexity occur. To determine possible restrictions for soy beans and rapeseed Equation (7) is applied by multiplying the global production shares [52] with the EVI [60] and aggregating the results. Furthermore, the moisture content of the material in storage is taken into account. For soy beans and rapeseed the average moisture content during storage is 11%–15% [132,133]. Thus, the average of 13 is used for the calculation. The results for the storage complexity for soy beans is 20.8 and for rapeseed 44.0. As the countries storing soy beans have a low economic vulnerability the influence of the moisture content is more significant. For rapeseed the countries storage ability has a higher influence than the moisture content, with China being most influential. Thus, the potential restrictions to availability due to high storage complexity are higher for rapeseed than for soy beans. However, since both materials are below the target (60), no potential restrictions to availability occur.

Recycling: This category is applied for the final product. As biodiesel is burned and thus cannot be recycled [134], the recycled content is zero for both biotic materials and thus the PMU is 100% (see Equation (9)). Since both materials are above the target, for both, potential restriction in terms of availability occurs.

Next to the calculation of the indicator values, the DtT approach is also applied for better interpretation of the results. To implement the Distance-to-Target (DtT) approach the determined indicator values for the case study are set into relation to the target to determine the DtT value according to Equation (8). Next, the DtT values are normalized according to Equation (9). Finally, the results have to be rescaled to the 6.3×10^{15} according to Equation (10) (for detailed calculations please see Supplementary Materials—Section 3.2.6).

In Figure 8, the results for all supply chain stages and both evaluated materials are shown. It can be seen that demand growth and price fluctuation are the biggest restrictions for soy bean availability. For the availability of rapeseed price fluctuations and recycling have the most influence. The cultivation and harvesting step is mostly impacted by price fluctuations, concentration of harvesting and trade barriers. Demand growth and recycling mostly influence the availability of the final product biodiesel. Limitations in the processing step (oil production) are comparably low.



	Political instability		Demand growth		Trade barriers		Price fluctuations		Occurence as co-product		Concentration of harvesting		Recycling		Storage complexity	
	Soy beans	Rapeseeds	Soy beans	Rapeseeds	Soy beans	Rapeseeds	Soy beans	Rapeseeds	Soy beans	Rapeseeds	Soy beans	Rapeseeds	Soy beans	Rapeseeds	Soy beans	Rapeseeds
(intermediate) product	0.22	0.00	0.91	0.14	0.20	0.00	0.00	0.00	0.07	0.02	0.00	0.00	0.13	0.63	0.00	0.00
Process step	0.22	0.00	0.23	0.03	0.20	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cultivation and harvesting step	0.18	0.00	0.16	0.09	0.22	0.00	0.94	0.63	0.00	0.00	0.63	0.00	0.00	0.00	0.00	0.00

Figure 8. Results for the dimension socio-economic constraints for all supply chain stages.

3.3. Abiotic Constraints

In the following, abiotic constraints to the product systems are determined. For limitations due to water availability Equation (12) is applied for the supply chain stage cultivation by multiplying the global production shares [52] with the WDI [69] and aggregating the results (for more information see Supplementary Materials—Section 3.3.1). The water availability adds up to 0.33 for both materials. Thus, possible limitations to availability due to water scarcity are the same.

Furthermore, restrictions due to natural disasters are determined for the supply chain stage cultivation) by multiplying the global production shares [52] with the NDI [98] and aggregating the results (for detailed information see Supplementary Materials—Section 3.3.2). For soy beans, the natural disaster risk adds up to 16.8. For rapeseed, the overall risk is 10.8. Thus, potential restrictions to availability are higher for soy beans than for rapeseed.

Land use and phosphorous input can be determined carrying out an LCA. As this would go beyond the scope of this article, existing studies are used. Based on Zulka et al. (2012) [86] land used for cultivation of rapeseed is 7.7 m^2/year per liter biofuel. For soy beans, the land use is 7.6 m^2/year per liter biofuel based on Pradhan et al. (2012) [85]. As the results are not from the same study, they should not be compared (e.g., due to different system boundaries, etc.). However, to show how the results would be interpreted we still use them here for an exemplary comparison. Based on the results, possible limitations to availability are higher for soy beans than for rapeseed. With regard to phosphorus use the phosphor input for rapeseed cultivation is 4250 g/L biofuel [86] whereas it is only 9.64 g/L for soy beans [85]. Thus, the possible restrictions for rapeseed are much higher than for soy beans. However, it has to be considered again, that the results of two different studies were taken into account. Especially for soy beans, amounts of phosphorus inputs differ widely depending on the study [85,135,136].

As the categories within this dimension have different units they cannot be compared directly (e.g., the amount of phosphorus of soy beans cannot be compared with the water availability of rapeseed).

Thus, for better visualization within a diagram the comparison is based on shares (see Figure 9), where the higher result within a category is set to 1 and the other result is calculated accordingly.

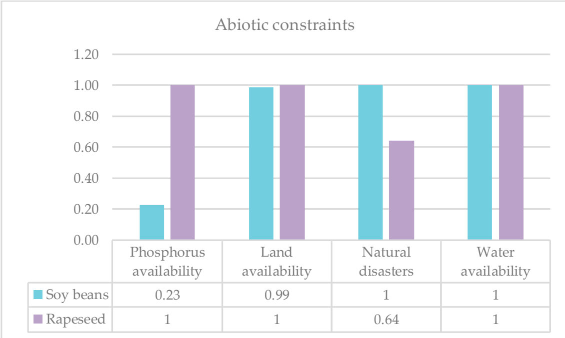


Figure 9. Results for the dimension abiotic constraints of rapeseed and soy beans.

3.4. Social Constraints

Next, social constraints for the product systems under assessment are evaluated. To determine if both systems comply with the "Food First" principle, the global production share [52] are multiplied with the FSI [102] according to Equation (14) for the supply chain stage cultivation (for detailed information see Supplementary Materials—Section 3.4.1). The food security adds up to 25.4 for soy beans and to 19.1 for rapeseed. Thus, potential restrictions to availability due to countries not complying with the food first principle are higher for soy beans than for rapeseed. To determine the compliance with social standards during cultivation and harvest data regarding child labor, forced labor and conflict zones from SHDB [108] is identified and applied according to Equation (15) based on data from Barrientos and Soria (2016) [52] and Norris et al. (2013) [107] (for detailed information see Supplementary Materials—Section 3.4.2). The social acceptance adds up to 9.0 for soy beans and to 7.0 for rapeseed. Thus, potential restrictions to availability due to non-compliance with social standards are higher for soy beans. To determine the compliance with environmental standards (cultivation and harvesting) EPI data [109] are identified on country level. The environmental compliance is determined for the supply chain stage cultivation and harvest according to Equation (16) based on data from Barrientos and Soria (2016) [52] and Yale Center for Environmental Law and Policy (2014) [109] (for detailed information see Supplementary Materials—Section 3.4.2). The non-compliance with environmental standards adds up to 11.0 for soy beans and to 11.2 for rapeseed. Thus, potential restrictions to availability due to non-compliance with environmental standards are similar for both.

Similar to the results of the dimension abiotic constraints, the results of the dimension social constraints cannot be compared directly. Thus, a comparison based on shares is performed (see Figure 10).

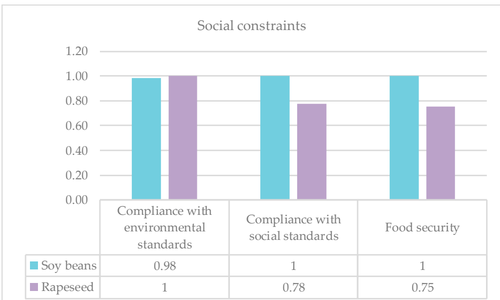


Figure 10. Results for the dimension social constraints of rapeseed and soy beans.

3.5. Environmental Constraints

To determine environmental constraints an LCA case study has to be carried out. Again, existing case studies are used as performing an own case study is beyond the scope of this article. As the results for environmental impacts of soy beans and rapeseed are not taken from the same study, they are difficult to compare (e.g., due to different system boundaries etc.). However, as stated before, we still use them here for an exemplary comparison to show how the results can be interpreted. Considering the study of Panichelli et al. (2009) [135] for soybean-based biodiesel and by González-García et al. (2013) [137] for rapeseed-derived biodiesel results for the three impact categories climate change, eutrophication and acidification can be obtained. Other case studies had to be used as for identifying the amount of phosphorus and land use since these studies did not contain results for the desired impact assessment categories. Based on this data, the production of biodiesel made from rapeseed leads to less environmental impacts than the biofuel production made from soybeans. However, it has to be considered, that results of two different studies are taken into account. Thus, the results should be validated and are only used here to demonstrate the introduced approach. Even though impacts are determined over the whole life cycle, they are mostly affecting the cultivation stage since the environment pollution leads to ecosystems changing and thus possible restrictions in the availability of species used as biotic materials. However, as stated before, no direct correlation between environmental impacts and limited availability can be made, as the amount of emitted substances (or the impact assessment results) cannot be related to a certain restriction in availability. Thus, the "less impacts, less constraints" principle is applied for now.

Summarizing, for most categories the possible restrictions to availability are higher for soy beans than for rapeseed within all considered supply chains stages (see Table 4). Exceptions are the category recycling for the stages processing of oil and final biodiesel production as well as phosphorus use and land availability for rapeseed. Data for the supply chain stage cultivation and harvest was easier to collect than for the other supply chain stages. However, it was demonstrated that the introduced approach can be applied and leads to plausible results which can be interpreted.

Table 4. Results for physical, socio-economic and abiotic constraints of soybeans (SB) and rapeseed (RS) for the considered categories in the corresponding supply chain stages.

Dimension	Category	Supply Chain Stage			
		Nature	Cultivation and Harvest of Soy Beans and Rapeseed	Processing (and Corresponding Steps) of Soy Beans and Rapeseed	Production Steps to Produce Biodiesel
Environmental constraints	Acidification Eutrophication Climate change		SB > RS SB > RS SB > RS		
Social constraints	Compliance with environmental standards		SB = 11.0 RS = 11.2		
	Compliance with social standards		SB = 9.0 RS = 7.0		
	Food security		SB = 25.4 RS = 19.1		
Abiotic constraints	Phosphorus availability		SB = 9.64 RS = 42.5		
	Land availability		SB = 7.6 RS = 7.7		
	Natural disasters		SB = 16.8 RS = 10.8		
	Water availability		SB = RS = 0.33		

Table 4. *Cont.*

Dimension	Category	Supply Chain Stage			
		Nature	Cultivation and Harvest of Soy Beans and Rapeseed	Processing (and Corresponding Steps) of Soy Beans and Rapeseed	Production Steps to Produce Biodiesel
Socio-economic constraints	Recycling				SB = 1.3 × 10^{15} RS = 6.3 × 10^{15}
	Storage complexity		SB = 0 RS = 0		
	Concentration of harvesting		SB = 9.4 × 10^{15} RS = 0		
	Occurrence as co-products		SB = 0 RS = 0	SB = 6.7 × 10^{14} RS = 2.2 × 10^{14}	
	Price fluctuations			SB = 9.4 × 10^{15} RS = 6.3 × 10^{14}	
	Trade barriers		SB = 2.2 × 10^{15} RS = 0	SB = 2.0 × 10^{15} RS = 0	
	Demand growth		SB = 1.6 × 10^{15} RS = 9.2 × 10^{14}	SB = 2.3 × 10^{15} RS = 3.5 × 10^{14}	SB = 9.1 × 10^{15} RS = 1.4 × 10^{15}
	Political instability		SB = 1.8 × 10^{15} RS = 0	SB = 2.2 × 10^{15} RS = 0	
	Concentration of resources	SB = 0 RS = 0			
Physical constraints	Replenishment rate		SB = 0 RS = 0		
	Anthropogenic availability				SB = 1 RS = 1
	Biotic resource depletion	SB = 0 RS = 0			

4. Discussion

The aim of the developed approach is to assess the availability of biotic materials within 24 categories with related category indicators. The indicator values for a broad range of materials are currently not available, which limits the applicability of the method. However, as shown in the case study, the determination of indicator values is feasible.

The BIRD approach can be applied to different taxonomy levels, e.g., order, family, species etc. For the introduced example of rapeseed this would mean that besides determining possible restrictions to rapeseed in general, also possible restrictions to the availability of different rapeseed species (e.g., annual rape and summer rape, swede rape) could have be identified. Furthermore, the availability could have also been determined for a higher taxonomic level, e.g., for all brassicaceae (cabbage family). Based on current data availability the determination of possible restrictions to availability for different rapeseed species as well as for higher taxonomic level like family is challenging as data is mostly available for a group of species (e.g., rapeseed) or different plants grouped together based on their function (e.g., oilseeds) and not for their taxonomic classification.

The developed approach has several uncertainties, which have to be considered when interpreting the results. These are described in detail in the following.

4.1. Physical Constraints

The required data for the introduced BRA indicator is missing. Thus, it is hard to determine indicator values. For future measurement of biotic resources availability, the BRA should be developed further specifically for different resource groups. For example, there are a lot of methods available and being further developed for fish consumption [138,139]. Furthermore, the calculation of the BRA for two species (African elephant and Great Indian Bustard [140,141]) has shown that the influence of the TS is not as significant for the result as it was anticipated. The interpretation of the results is challenging as well because negative values can occur demonstrating stock replenishment. To quantify the influence of the replenishment rate the introduced approach needs to be refined. Currently, the growth rate is divided into three classes, which have to be validated regarding their meaningfulness; e.g., perhaps a division into more or less classes would be more precise. Furthermore, the classification in high and low yields is currently carried out based on the practitioner's judgment. Thus, more precise rules regarding the way to determine high and low yields are needed. The global increase of production and thus the globally increasing replenishment rate was not considered in the introduced approach. However, as global production amounts influence the availability this aspect should be considered in the future.

To assess whether biotic materials contribute to the anthropogenic stock, an approach based on the use of biotic materials is introduced. However, this approach should be refined. Classifying food, feed and biofuel in one category could be misleading as, e.g., in some countries oil used for frying food is afterwards used for fuel and thus contributes to the anthropogenic stock. Furthermore, it has to be factored in how often a material contributes to the anthropogenic stock. Reuse of biotic materials often goes hand in hand with a down cycling of the material quality, e.g., furniture made out of wood will most likely not be used for another furniture but will be shredded and used for particle board. Thus, biotic materials cannot be recycled without quality loss.

4.2. Socio-Economic Constraints

Some category indicators are based on existing indicators, e.g., WGI, ETI, etc., for which data is not available for every country and thus had to be determined based on existing correlations with other indicators. These calculated values are more uncertain than the provided values. For some categories, e.g., occurrence of co-product qualitative data is transformed into quantitative ones. The transformation of quantitative data can be challenging when the classification is not conclusive, e.g., when data is obtained from different sources. For the DtT approach the target value for storage capacity was defined by the authors to carry out the calculations. This target has to be redefined by expert participation.

4.3. Abiotic Constraints

For the water availability only scarcity impacts, but not socio-economic aspects are taken into account. Even though water scarcity of a region or country is important to determine water availability, socio-economic factors also play a significant role regarding accessibility of water resources [142]. With regard to natural disasters only floods and drought are currently considered. For a more comprehensive assessment other disasters should be taken into account as well. For the assessment of phosphorus and land use only the amounts are reported. This allows a comparison of two product systems. However, it is not possible to make a statement when only one system is analyzed.

4.4. Societal Constraints

The food security index is based on five indicators since sufficient data is available for these indicators. However, additional analysis is needed to check if these indicators are sufficient to realistically reflect the situation in countries. The indicator for determining compliance with social standards is based on the three aspects child labor, forced labor and high conflict zones. These indicators were chosen based on the approach for abiotic materials. Even though these aspects are important for every sector, it should be verified if the indicators are sufficient for the assessment of biotic materials as well. Furthermore, the SHDB data are only available for broad sectors but not for small sectors. Thus, societal acceptance of biofuel production could not be determined. For the compliance with environmental standards, some of the EPI are chosen to represent the compliance of a country with environmental standards. It is necessary to check if these chosen indicators realistically present a country's compliance with environmental standards.

4.5. Environmental Constraints

Environmental constrains are determined based on the results of an LCA study. First, mature methods for several important environmental aspects are missing, e.g., biodiversity loss. Second, the reported values do not allow an overall statement with regard to possible limitation to availability. It can only be determined whether the impacts to the environment are lower when two or more options are compared. Even though it can be argued that lower environmental impacts are better for the affected ecosystems, it is not possible to determine how much and even if a certain impact actually leads to restriction in availability. Determining a target value with regard to the amount of impacts which can be seen as 'not critical' is challenging as presented in recent publications related to planetary boundaries [143–145]. Furthermore, as shown by Milà i Canals et al. (2011) [146] as well as within the ongoing PEF Pilot phase [147,148], challenges with regard to data availability and quality exist for bio-based products and thus for biotic materials. Furthermore, occurrence of invasive species can also lead to reduction of species used as biotic materials [149]. However, as it is challenging to assess such effects without detailed regionalized data, it is not considered within the BIRD approach.

4.6. General

For most indicators country based values are used to determine the overall potential restrictions for a material. These restrictions however are determined globally and do not allow conclusions regarding any regional aspects. Thus, the proposed indicators are applied as screening indicators to determine hotspots. Based on the hotspots a deeper analysis with regard to the regional conditions for the specific product system should be carried out.

Established indicators as well as newly developed ones face the challenge of underlying data quality. If the data quality is poor, the indicator will have greater uncertainties as if the underlying data is good. However, established indicators, which have been used more frequently, tend to have lower uncertainties as they were improved over time. For all applied indicators, the maturity level and meaningfulness has to be validated.

All introduced indicators can be used to assess a product and its associated life cycle. However, so far the implementation of the introduced approach into LCA is not possible (except for the dimension socio-economic availability and environmental constraints), because the indicator values are not established to be multiplied with the mass of a material.

5. Conclusions and Outlook

The introduced approach significantly enhances the availability assessment of biotic materials by providing a framework, which considers a broad range of aspects in relation to availability restrictions. Overall, five dimensions and 24 categories with corresponding category indicators are introduced. A comprehensive assessment of availability aspects as well as more meaningful decision making processes are therefore possible.

The next steps should include a comprehensive analysis of the proposed category indicators (as well as underlying indicators and data). Furthermore, indicator values for several biotic materials should be calculated to enhance applicability of the approach. These indicator values should then be applied in several case studies to test and refine them. Additional steps would be, for example, determination of the missing target value and refinement of the dimension physical constraints.

As the communication of the overall 24 indicators will be challenging (especially regarding stakeholders with less experience in the field of LCA, supply risk assessment and sustainability) future efforts should include the aggregation of the indicators within the individual dimensions. Currently, the aggregation of indicators is too challenging to be achieved within this work. Despite the benefits in communication, aggregation of indicators also has several disadvantages. By aggregating the indicator values within one dimension, transparency of the results is decreased significantly, which lowers the informative value of the communicated results. Further, weighting implies that aspects can be balanced against each other, e.g., physical constraints can be compensated with fewer environmental constraints. This kind of offsetting is a purely subjective decision, for which no commonly agreed on weighting scheme exists.

The assessment of availability is often seen as a part of the resource efficiency evaluation (e.g., [12,150]). Thus, the introduced approach can be seen as a relevant step with regard to a comprehensive resource efficiency assessment of biotic materials. Since the introduced approach is partly based on the ESSENZ method (a method to determine the resource efficiency of abiotic materials) [28] it could be implemented into ESSENZ in the future.

Supplementary Materials: The following are available online at www.mdpi.com/2071-1050/9/1/137/s1, Table S1: Overview of countries consuming rape seed and soy beans, the share of the categories as determined in Table S2 (in the main part of the article) and the corresponding factor, Table S2: Data to determine the political instability of soy bean producing countries: countries, global production share and Worldwide Governance Index, Table S3: Data to determine the political instability of rapeseed producing countries: countries, global production share and Worldwide Governance Index, Table S4: Data to determine the political instability of soy bean consuming countries: countries, global consumption share and Worldwide Governance Index; Table S5: Data to determine the political instability of rapeseed consuming countries: countries, global consumption share and Worldwide Governance Index, Table S6: Data for calculating the yearly change in demand growth: year and global production, Table S7: Data to determine the trade barriers of soy bean producing countries: countries, global production share and Enabling Trade Indicator, Table S8: Data to determine trade barriers of rapeseed producing countries: countries, global production share and Enabling Trade Indicator, Table S9: Data to determine trade barriers of soy bean consuming countries: countries, global consumption share and Worldwide Governance Index, Table S10: Data to determine trade barriers of rapeseed consuming countries: countries, global consumption share and Enabling Trade Indicator, Table S11: Production data to determine Herfindahl-Hirschmann-Index for soy beans, Table S12: Production data to determine Herfindahl-Hirschmann-Index for rapeseed, Table S13: Data to determine the storage complexity of soy bean producing countries: countries, global production share and Economic Vulnerability Indicator, Table S14: Data to determine storage complexity of rapeseed producing countries: countries, global production share and Economic Vulnerability Indicator, Table S15: Indicator results of case study for considered categories and supply chain stages, Table S16: Calculation and results of DtT-value for considered categories for soy beans, Table S17: Calculation and results of normalized DtT-value for considered categories, Table S18: Overall results for the biotic materials soy bean and rapeseed for considered categories,

Table S19: Calculation and results of scaled values for considered categories, Table S20: Data for determination of the water availability of soy bean producing countries: countries, global production share and Water Depletion Index, Table S21: Data for determination of water availability of rapeseed producing countries: countries, global production share and Water Depletion Index. Table S22: Data for determination of the natural disaster risk of soy bean producing countries: countries, global production share and natural disaster index, Table S23: Data for determination of the natural disaster risk of rapeseed producing countries: countries, global production share and natural disaster index, Table S24: Data for determination of the food security of soy bean producing countries: countries, global production share and Food Security Index, Table S25: Data for determination of food security of rapeseed producing countries: countries, global production share and Food security Index, Table S26: Data to determine the societal acceptance of soy bean producing countries: countries, global production share and indicator for compliance with social standards, Table S27: Data to determine the societal acceptance of rapeseed producing countries: countries, global production share and indicator for compliance with social standards, Table S28: Data for determination of the compliance with environmental standards of soy bean producing countries: countries, global production share and Environmental Performance Indicators (EPI), Table S29: Data for determination of environmental compliance of rapeseed producing countries: countries, global production share and Environmental Performance Indicators (EPI).

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Conflicts of Interest: The authors declare no conflict of interest.

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Supplementary Materials: Assessing the Availability of Terrestrial Biotic Raw Materials in Product Systems (BIRD)

Vanessa Bach, Markus Berger, Natalia Finogenova and Matthias Finkbeiner

1. Calculation of the Resource Availability for African Elephants

In the following the determination of the reference for the Biotic Resource Availability Indicator (BRAI) is explained. As shown in Equation (1) (in the main part of the article) the BRAI of a species is calculated by subtracting the regeneration rate from the extraction rate and setting the value into relation to the resource stock.

The extraction rate (based on data by World Wildlife Fund (2016)), regeneration rate (based on data by African Elephant Specialist Group (2013)), resource stock (based on data by World Wildlife Fund (2016)) and TSI (based on data by IUCN (2016)) are determined. As the African elephant is classified as vulnerable, the TSI value is set to 50. Thus, the BRAI of the African elephant (AI) can be calculated (see Equation (1)).

$$\mathrm{BRAI_{AI}} = \left[\frac{\text{extraction rate}_{\mathrm{AI}} \text{ - regeneration rate}_{\mathrm{AI}}}{(\text{resource stocks}_{\mathrm{AI}})^2}\right] \times \mathrm{TSI_{AI}} = \left[\frac{30{,}000 - 23{,}500}{470{,}000^2}\right] \times 50 = 1.47 \times 10^{-6}\,\frac{1}{\text{individuals}} \quad (1)$$

The BRA of GIB is set to 1 as African elephants are also the reference species.

2. Determining the Food Security Index

In the following it is described how the Food Security Index (FSI) is established based on several Food Security Indicators by Food and Agriculture Organization of the United Nations (FAO) (2016). Based on data availability and relevance the following five indicators were chosen:

- Average dietary energy supply adequacy (DES)
- Depth of the food deficit (FD)
- Prevalence of food inadequacy (FI)
- Cereal import dependency ratio (CID)
- Prevalence of undernourishment (PU)

These indicators have to be converted to the same scale before they can be aggregated to the food security index (FSI).

The scaling occurs in three steps:

(1) Indicators with negative values are recalculated so that all values are positive. Therefore, the minimum value is subtracted from the original value.
(2) All indicators are scaled to the range from 0 to 100 by dividing the original value by the maximal value and multiplying it with 100.
(3) All indicators need to have the same direction, e.g., high numbers refer to big impacts. Thus the direction has to be changed if necessary. This can be achieved by subtracting the original value from 100.

The final FSI is calculated by determining the average of all five indicators according to Equation (2).

$$\mathrm{FSI_i} = \frac{\mathrm{DES_i} + \mathrm{FD_i} + \mathrm{FI_i} + \mathrm{CID_i} + \mathrm{PU_i}}{5} \quad (2)$$

3. Case Study

In the following additional calculations for the introduced case study are presented.

3.1. Physical Constraints

Next the calculations for the dimension physical constraints are shown.

3.1.1. Ecological Constraints

The introduced case study does not consider biotic resources. However, to show how the final BRA is calculated an example is introduced, for which the detailed calculation steps are shown. The biotic resource considered is the Great Indian Bustard, which is hunted for sport as well as for food. As shown in Equation (1) (in the main part of the article) the BRAI of a species is calculated by subtracting the regeneration rate from the extraction rate and setting the value into relation to the stock. Furthermore, the Threatened Species Index (TSI) of the Great Indian Bustard (GIB) is taken into account (see Equation (3)). The Great Indian Bustard is a bird, which is classified as critically endangered.

$$\mathrm{BRAI}_{\mathrm{GIB}} = \left[\frac{\text{extraction rate}_{\mathrm{GIB}}\text{- regeneration rate}_{\mathrm{GIB}}}{(\text{resource stocks}_{\mathrm{GIB}})^2}\right] \times \mathrm{TSI}_{\mathrm{GIB}} = \left[\frac{24-50}{300^2}\right] \times 100 = -2.89 \times 10^{-2}\,\frac{1}{\text{individuals}} \quad (3)$$

The extraction rate is determined by considering the population rate of the year 1969 (before depletion occurred) and comparing it with the population of the year 2009 (see Equation (4)) based on data by Dutta et al. (2011). The current resource stock consists of 300 individuals (current population).

$$\text{Extraction rate}_{\mathrm{GIB}} = \frac{1260-300}{40} = 24 \text{ individuals} \quad (4)$$

The regenerated rate can be determined according to Equation (5) based on data published by Mohammed and Indira (2015).

$$\begin{aligned}&\text{Replenishment rate}\\&\quad = \text{amount of mature female individuals}\\&\quad \times \text{ amount of descendants per individual} \times \text{mortality rate in first year}\\&\quad = 100 \times 1 \times 0.5 = 50 \text{ individuals}\end{aligned} \quad (5)$$

The TSI is defined according to Table S1 (in the main part of the article) and according to data by International Union for Conservation of Nature (2016). As the GIB is classified as critically endangered, the TSI value is set to 100.

Table S1. Overview of countries consuming rape seed and soy beans, the share of the categories as determined in Table S2 (in the main part of the article) and the corresponding factor.

Country	Biotic Material	Share of Categories	Factor(s)
EU	Rapeseed	100% category F	1
	Soybeans	100% category F	1
USA	Rapeseed	100% category F	1
	Soybeans	100% category F	1
Brazil	Soybeans	100% category F	1
Argentina	Soybeans	100% category F	1
Canada	Rapeseed	100% category F	1
India	Rapeseed	100% category F	1

Thus, the overall BRAI of the GIB sums up to $-2.89 \times 10^{-2} \times \frac{1}{\text{individuals}}$. As the current regeneration rate per year is higher as the extraction rate, no resource stocks are depleted. The stock is rather growing. Thus, the calculated value is negative.

To compare BRAs of different biotic resources the BRAIs have to be set in relation to the reference resource African elephants. Thus, for the GIB the resource depletion adds up to -2.1×10^5. The negative value refers to the replenishment of the resource stock compared to African elephants, which are still hunted in larger amounts as they can replenish.

3.1.2. Anthropogenic Stock

In the following the calculation of the factor determining the anthropogenic stock for soy bean and rapeseed is demonstrated. As there is no data available on the amount of the globally produced rapeseed and soy bean used for biofuel, the shares of the largest consumers of rapeseed and soy beans are applied instead based on the data by Barrientos and Soria (2016) [1]. As shown in Table S1 rapeseeds and soy beans in the considered countries are used for food, feed and fuel production. No data was found stating if soy beans and rapeseeds are used within any products. However, if they are used for products the amount is most likely very small and would not change the overall result significantly.

As the factor for both materials in all considered countries is 1 the overall anthropogenic constraints results in 1 as well.

3.2. Socio-Economic Availability

Next the calculations for the categories of the dimension socio-economic constraints are introduced.

3.2.1. Political Instability

In the following the calculations to determine the political instability of the product system are shown. Possible limitations due to political instable countries can occur during the cultivation as well as during processing of the materials and production of biofuel. It is assumed that the production of soy bean and rapeseed oil occurs in the same country as the production of biodiesel. Thus, two values are determined based on Equation (3) (in the main part of the article). The political instability for soy bean producing countries is determined as shown in Table S2.

Table S2. Data to determine the political instability of soy bean producing countries: countries, global production share and Worldwide Governance Index.

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Worldwide Governance Index (WGII) Based on World Bank Group (2013)	Global Production Share × WGII
USA	33.5%	1.34	4.50×10^{-1}
Brazil	31.3%	2.14	6.71×10^{-1}
Argentina	17.9%	2.48	4.43×10^{-1}
China	3.8%	2.66	1.00×10^{-1}
Paraguay	2.8%	2.76	7.61×10^{-2}
India	2.5%	2.46	6.18×10^{-2}
Canada	2.0%	0.52	1.01×10^{-2}
Ukraine	1.2%	1.49	1.77×10^{-2}
Uruguay	1.0%	0.90	8.82×10^{-3}
Bolivia	1.0%	2.66	2.59×10^{-2}
Russia	0.9%	2.83	2.53×10^{-2}
South Africa	0.3%	1.89	5.26×10^{-3}
Nigeria	0.2%	3.26	6.65×10^{-3}
Indonesia	0.2%	2.47	4.79×10^{-3}
Serbia	0.2%	2.22	3.68×10^{-3}
Mexico	0.1%	2.27	2.57×10^{-3}
Japan	0.1%	0.81	5.58×10^{-4}
Zambia	0.1%	2.34	1.57×10^{-3}
Myanmar	0.1%	3.44	2.16×10^{-3}
Iran	0.1%	3.24	1.98×10^{-3}

Table S2. *Cont.*

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Worldwide Governance Index (WGII) Based on World Bank Group (2013)	Global Production Share × WGII
Uganda	0.1%	2.72	1.62×10^{-3}
Vietnam	0.1%	2.63	1.44×10^{-3}
Korea, Dem. Rep.	4.7×10^{-4}%	3.75	1.76×10^{-3}
Korea, Rep.	3.8×10^{-4}%	1.36	5.13×10^{-4}
Turkey	2.8×10^{-4}%	2.19	6.19×10^{-4}
Zimbabwe	2.8×10^{-4}%	3.44	9.70×10^{-4}
Venezuela	2.4×10^{-4}%	3.43	8.06×10^{-4}
Colombia	2.3×10^{-4}%	2.42	5.54×10^{-4}
Ecuador	2.2×10^{-4}%	2.69	5.91×10^{-4}
Australia	1.9×10^{-4}%	0.55	1.04×10^{-4}
Thailand	1.6×10^{-4}%	2.41	3.78×10^{-4}
Guatemala	1.1×10^{-4}%	2.73	3.09×10^{-4}
Egypt	6.3×10^{-5}%	3.02	1.90×10^{-4}
Bosnia	2.2×10^{-5}%	2.36	5.17×10^{-5}
Nicaragua	2.2×10^{-5}%	2.65	5.81×10^{-5}
Taiwan	1.9×10^{-5}%	3.31	6.23×10^{-5}
Peru	9.4×10^{-6}%	2.36	2.22×10^{-5}
Switzerland	9.4×10^{-6}%	0.38	3.62×10^{-6}
Pakistan	6.3×10^{-6}%	3.24	2.03×10^{-5}
Syrian Arab.Rep.	6.3×10^{-6}%	3.80	2.38×10^{-5}
Philippines	3.1×10^{-6}%	2.43	7.63×10^{-6}
$\Sigma = 1.93$			

The political instability for soy bean producing countries adds up to 1.93. For rapeseed the political stability with regard to production is shown in Table S3 and sums up to 1.43.

Table S3. Data to determine the political instability of rapeseed producing countries: countries, global production share and Worldwide Governance Index.

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Worldwide Governance Index (WGII) Based on World Bank Group (2013)	Global Production Share × WGII
Canada	32%	1.31×10^{-1}	0.41
China	25%	5.54×10^{-1}	2.18
India	21%	2.18×10^{-1}	1.05
Australia	9%	7.73×10^{-2}	0.96
France	8.07%	4.62×10^{-2}	0.65
Germany	7.12%	2.52×10^{-2}	0.28
Ukraine	5%	1.07×10^{-1}	2.83
UK	3.79%	3.51×10^{-2}	1.28
USA	3%	3.73×10^{-2}	0.82
Poland	2.74%	2.59×10^{-2}	1.03
Russia	2%	4.70×10^{-2}	2.44
Belarus	2%	2.05×10^{-2}	1.25
Czech Rep.	1.64%	1.21×10^{-2}	1.29
Lithuania	0.93%	2.39×10^{-3}	0.33
Denmark	0.72%	8.68×10^{-3}	1.47
Hungary	0.59%	1.49×10^{-3}	0.31
Sweden	0.48%	6.33×10^{-3}	1.41
Latvia	0.45%	1.31×10^{-2}	0.79
Bulgaria	0.40%	7.97×10^{-3}	1.99
Slovakia	0.31%	1.03×10^{-2}	2.33

Table S3. *Cont.*

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Worldwide Governance Index (WGII) Based on World Bank Group (2013)	Global Production Share × WGII
Estonia	0.24%	9.29×10^{-3}	2.74
Romania	0.23%	4.43×10^{-3}	1.41
Austria	0.22%	2.78×10^{-3}	0.84
Finland	0.11%	2.48×10^{-3}	1.03
Spain	0.08%	7.66×10^{-3}	2.59
Belgium	0.07%	4.63×10^{-3}	1.97
Italy	0.04%	1.28×10^{-3}	0.58
Ireland	0.04%	3.58×10^{-3}	1.51
Slovenia	0.03%	2.90×10^{-4}	0.27
Luxembourg	0.02%	3.92×10^{-4}	0.33
Netherlands	0.01%	1.94×10^{-3}	1.91
Greece	0.01%	1.00×10^{-3}	1.30
Bangladesh	4×10^{-3}%	5.30×10^{-4}	0.75
Kazakhstan	3×10^{-3}%	1.87×10^{-3}	2.12
Chile	3×10^{-3}%	2.89×10^{-4}	0.70
Pakistan	3×10^{-3}%	5.97×10^{-4}	1.62
Ethiopia	2×10^{-3}%	3.17×10^{-4}	1.26
Switzerland	1×10^{-3}%	9.11×10^{-5}	0.41
Turkey	1×10^{-3}%	4.95×10^{-5}	0.07
Paraguay	9×10^{-4}%	4.48×10^{-5}	0.43
Norway	7×10^{-4}%	1.33×10^{-4}	1.81
Japan	2×10^{-4}%	2.39×10^{-5}	0.15
Korea, Rep.	3×10^{-5}%	2.01×10^{-5}	0.68
Morocco	1×10^{-5}%	3.63×10^{-5}	2.46
$\Sigma = 1.43$			

To determine the political instability with regard to the processing step (oil production) and the production of biofuel the countries global consumption share of rapeseed and soy bean is multiplied with the corresponding WGII. The calculation for rapeseed is shown in Table S4 and the calculation for soy bean in Table S5. For rapeseed the political instability adds up to 1.32 and for soy bean to 2.14.

Table S4. Data to determine the political instability of soy bean consuming countries: countries, global consumption share and Worldwide Governance Index.

Soy Bean Consuming Countries	Global Consumption Share (gps) Based on Barrientos and Soria (2016)	Worldwide Governance Index (WGII) Based on World Bank Group (2013)	Global Consumption Share × WGII
China	30.29%	1.34	2.37×10^{-1}
USA	17.69%	2.14	2.96×10^{-1}
Argentina	15.41%	2.48	3.82×10^{-1}
Brazil	13.83%	2.66	8.06×10^{-1}
India	2.60%	2.76	3.73×10^{-2}
Russia	1.46%	2.46	6.41×10^{-2}
Mexico	1.41%	0.52	4.04×10^{-3}
Paraguay	1.35%	1.49	8.02×10^{-3}
Japan	1.00%	0.90	8.00×10^{-4}
Indonesia	0.94%	2.66	2.36×10^{-2}
Bolivia	0.89%	2.83	4.14×10^{-2}
Taiwan	0.81%	1.89	6.77×10^{-3}
Turkey	0.79%	3.26	8.40×10^{-3}
Thailand	0.79%	2.47	2.31×10^{-2}

Table S4. *Cont.*

Soy Bean Consuming Countries	Global Consumption Share (gps) Based on Barrientos and Soria (2016)	Worldwide Governance Index (WGII) Based on World Bank Group (2013)	Global Consumption Share × WGII
Canada	0.78%	2.22	2.71×10^{-3}
Egypt	0.67%	2.27	3.21×10^{-2}
Vietnam	0.63%	0.81	8.06×10^{-3}
Ukraine	0.54%	2.34	1.61×10^{-3}
Iran	0.49%	3.44	2.21×10^{-3}
Pakistan	0.48%	3.24	1.57×10^{-2}
Korea, Rep.	0.45%	2.72	1.62×10^{-3}
South Africa	0.36%	2.63	1.66×10^{-2}
Nigeria	0.26%	3.75	1.98×10^{-3}
Colombia	0.21%	1.36	6.16×10^{-3}
Serbia	0.12%	2.19	1.73×10^{-2}
Peru	0.11%	3.44	9.96×10^{-4}
Uruguay	0.09%	3.43	3.05×10^{-3}
Venezuela	0.09%	2.42	5.06×10^{-3}
Zambia	0.07%	2.69	6.07×10^{-4}
Myanmar	0.06%	0.55	1.05×10^{-4}
Uganda	0.06%	2.41	1.90×10^{-2}
Korea, Dem. Rep.	0.05%	2.73	5.72×10^{-4}
Philippines	0.04%	3.02	2.03×10^{-2}
Zimbabwe	0.03%	2.36	2.96×10^{-4}
Ecuador	0.02%	2.65	5.96×10^{-5}
Australia	0.02%	3.31	2.70×10^{-2}
Guatemala	0.02%	2.36	2.67×10^{-3}
Syrian Arab. Rep.	0.02%	0.38	2.23×10^{-5}
Switzerland	0.01%	3.24	1.57×10^{-2}
Bosnia	0.01%	3.80	8.81×10^{-4}
Nicaragua	0.00%	2.43	1.03×10^{-3}
	$\Sigma = 2.14$		

Table S5. Data to determine the political instability of rapeseed consuming countries: countries, global consumption share and Worldwide Governance Index.

Rapeseed Consuming Countries	Global Consumption Share (gcs) Based on Barrientos and Soria (2016)	Worldwide Governance Index (WGII) Based on World Bank Group (2013)	Global Consumption Share × WGII
EU-27	36.91%	1.14	4.20×10^{-1}
China	27.83%	2.18	6.07×10^{-1}
Canada	13.06%	0.41	5.32×10^{-2}
India	9.00%	1.05	9.44×10^{-2}
Japan	3.65%	0.15	5.38×10^{-3}
USA	2.48%	1.03	2.55×10^{-2}
Russia	1.53%	2.44	3.72×10^{-2}
Australia	1.40%	0.28	3.99×10^{-3}
Pakistan	1.19%	2.59	3.09×10^{-2}
Turkey	0.74%	1.91	1.41×10^{-2}
Ukraine	0.48%	0.82	3.88×10^{-3}
Belarus	0.45%	0.79	3.54×10^{-3}
Bangladesh	0.45%	2.33	1.04×10^{-2}
Chile	0.28%	0.84	2.35×10^{-3}
Kazakhstan	0.25%	2.74	6.88×10^{-3}
Ethiopia	0.12%	1.51	1.80×10^{-3}
Switzerland	0.11%	0.33	3.75×10^{-4}
Paraguay	0.04%	2.12	9.45×10^{-4}
Norway	0.03%	0.07	2.17×10^{-5}
	$\Sigma = 1.32$		

3.2.2. Demand Growth

In the following it is shown how the demand growth for soy bean for the cultivation (and harvesting) step is calculated (see Equation (6)) based on the data by U.S. Department of Agriculture (USDA) (2016). In Table S6 it is shown how to calculate the yearly growth (numerator of Equation (5)) for soy beans.

$$DG_{soybeans,cultivation} = \frac{\sum_1^5\left(\frac{\text{global cultivation of year n + 1}}{\text{global cultivation of year n}} - 1\right)}{4} = \frac{\sum(11.78+5.26+12.67+0.51)}{4} = \frac{30.22}{4} = 7.55\% \quad (6)$$

Table S6. Data for calculating the yearly change in demand growth: year and global production.

Year	Global Production in Million Metric Tons Based on USDA (2016)	Calculation	Yearly Change
2015	320.21	$\frac{320.21}{318.57} - 1$	0.51%
2014	318.57	$\frac{318.57}{282.75} - 1$	12.67%
2013	282.75	$\frac{282.75}{268.63} - 1$	5.26%
2012	268.63	$\frac{268.63}{240.32} - 1$	11.78%
2011	240.32		

To determine the demand growth in the processing step the share of yearly consumption of all produced oil used for biodiesel has to be identified. Thus, yearly production data of all countries producing biodiesel out of soy beans have to be determined. As these data is not available for all countries, only the three biggest producers are taken into account: USA, Brazil and China. The data for China is not available, that's why the fourth biggest producing country Argentina is considered. For these three countries the yearly growth of soybean derived biodiesel is calculated based on data by USDA (2015) to determine the overall demand growth for soy bean in the processing stage (see Equation (7)).

$$DG_{soybeans,processing} = \frac{\sum_1^5\left(\frac{\text{global processing of year n + 1}}{\text{global processing of year n}} - 1\right)}{4} = \frac{\sum(2.42 + 5.28 + 14.72 + 13.15)}{4} = \frac{35.56}{4} = 8.89\% \quad (7)$$

Thus, the demand growth for soy beans is 7.55% in the cultivation stage and 8.89% in the processing stage.

For rapeseed the demand growth is calculated as shown in Equations (8) and (9) using the data from USDA (2015) and USDA (2016).

$$DG_{rapeseed,cultivation} = \frac{\sum_1^5\left(\frac{\text{global cultivation of year n + 1}}{\text{global cultivation of year n}} - 1\right)}{4} = \frac{\sum(3.54 + 13.05 + 0.19 + (-6.27))}{4} = \frac{10.51}{4} = 2.63\% \quad (8)$$

$$DG_{rapeseed,processing} = \frac{\sum_1^5\left(\frac{\text{global processing of year n + 1}}{\text{global processing of year n}} - 1\right)}{4} = \frac{\sum(0.18 + 7.31 + 7.31 + (-8.14))}{4} = \frac{6.3}{4} = 1.58\% \quad (9)$$

For rapeseed the demand growth sums up to 2.63% in the cultivation stage and 1.58% in the processing stage.

3.2.3. Trade Barriers

The trade barriers are determined by multiplying the global production share with the Enabling Trade Indicator [2] and summing it up for the production as well as for the consumption of soy beans and rapeseed (see Tables S7–S10). The trade barriers for producing countries add up to 3.36 for soy bean and to 2.89 for rapeseed. For the processing step the trade barriers sum up to 3.23 for soy beans and to 2.42 for rapeseed.

Table S7. Data to determine the trade barriers of soy bean producing countries: countries, global production share and Enabling Trade Indicator.

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Enabling Trade Indicator (ETI) Based on Hanouz et al. (2014)	Global Production Share × EPI
USA	33.5%	2.60	8.72×10^{-1}
Brazil	31.3%	3.80	1.19
Argentina	17.9%	3.90	6.97×10^{-1}
China	3.8%	3.30	1.24×10^{-1}
Paraguay	2.8%	4.10	1.13×10^{-1}
India	2.5%	4.00	1.00×10^{-1}
Canada	2.0%	2.60	5.08×10^{-2}
Ukraine	1.2%	3.80	4.53×10^{-2}
Uruguay	1.0%	3.40	3.31×10^{-2}
Bolivia	1.0%	3.90	3.79×10^{-2}
Russia	0.9%	4.10	3.66×10^{-2}
South Africa	0.3%	3.40	9.49×10^{-3}
Nigeria	0.2%	4.50	9.17×10^{-3}
Indonesia	0.2%	3.40	6.61×10^{-3}
Serbia	0.2%	3.90	6.48×10^{-3}
Mexico	0.1%	3.50	3.95×10^{-3}
Japan	0.1%	2.50	1.72×10^{-3}
Zambia	0.1%	3.90	2.62×10^{-3}
Myanmar	0.1%	4.40	2.76×10^{-3}
Iran	0.1%	4.60	2.81×10^{-3}
Uganda	0.1%	4.00	2.38×10^{-3}
Vietnam	0.1%	3.60	1.97×10^{-3}
Korea, Dem. Rep.	4.7×10^{-4}%	3.70	1.74×10^{-3}
Korea, Rep.	3.8×10^{-4}%	2.90	1.09×10^{-3}
Turkey	2.8×10^{-4}%	3.30	9.31×10^{-4}
Zimbabwe	2.8×10^{-4}%	4.70	1.33×10^{-3}
Venezuela	2.4×10^{-4}%	4.37	1.03×10^{-3}
Colombia	2.3×10^{-4}%	3.60	8.24×10^{-4}
Ecuador	2.2×10^{-4}%	3.50	7.68×10^{-4}
Australia	1.9×10^{-4}%	2.70	5.08×10^{-4}
Thailand	1.6×10^{-4}%	3.40	5.33×10^{-4}
Guatemala	1.1×10^{-4}%	3.50	3.95×10^{-4}
Egypt	6.3×10^{-5}%	4.00	2.51×10^{-4}
Bosnia	2.2×10^{-5}%	3.70	8.12×10^{-5}
Nicaragua	2.2×10^{-5}%	3.60	7.90×10^{-5}
Taiwan	1.9×10^{-5}%	2.70	5.08×10^{-5}
Peru	9.4×10^{-6}%	3.30	3.10×10^{-5}
Switzerland	9.4×10^{-6}%	2.40	2.26×10^{-5}
Pakistan	6.3×10^{-6}%	4.10	2.57×10^{-5}
Syrian Arab. Rep.	6.3×10^{-6}%	4.10	2.57×10^{-5}
Philippines	3.1×10^{-6}%	3.50	1.10×10^{-5}
	$\Sigma = 3.36$		

Table S8. Data to determine trade barriers of rapeseed producing countries: countries, global production share and Enabling Trade Indicator.

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Enabling Trade Indicator (ETI) Based on Hanouz et al. (2014)	Global Production Share × ETI
Canada	32%	2.05	6.60×10^{-1}
China	25%	2.71	6.87×10^{-1}
India	21%	1.70	3.54×10^{-1}
Australia	9%	0.76	2.18×10^{-1}
France	8.07%	0.71	1.78×10^{-1}
Germany	7.12%	1.40	1.24×10^{-1}
Ukraine	5%	0.68	9.09×10^{-2}
UK	3.79%	0.93	9.05×10^{-2}
USA	3%	2.08	9.54×10^{-2}
Poland	2.74%	2.00	5.01×10^{-2}
Russia	2%	3.53	6.81×10^{-2}
Belarus	2%	0.91	5.24×10^{-2}
Czech Rep.	1.64%	0.91	2.99×10^{-2}
Lithuania	0.93%	0.74	1.86×10^{-2}
Denmark	0.72%	0.93	1.95×10^{-2}
Hungary	0.59%	0.71	1.19×10^{-2}
Sweden	0.48%	0.91	1.44×10^{-2}
Latvia	0.45%	0.99	1.64×10^{-2}
Bulgaria	0.40%	1.02	1.44×10^{-2}
Slovakia	0.31%	3.22	1.43×10^{-2}
Estonia	0.24%	3.82	1.30×10^{-2}
Romania	0.23%	0.93	1.04×10^{-2}
Austria	0.22%	2.22	7.38×10^{-3}
Finland	0.11%	0.79	6.74×10^{-3}
Spain	0.08%	3.28	9.69×10^{-3}
Belgium	0.07%	1.05	8.69×10^{-3}
Italy	0.04%	0.74	5.72×10^{-3}
Ireland	0.04%	2.20	5.20×10^{-3}
Slovenia	0.03%	0.68	2.59×10^{-3}
Luxembourg	0.02%	2.07	2.45×10^{-3}
Netherlands	0.01%	2.87	2.92×10^{-3}
Greece	0.01%	0.79	2.15×10^{-3}
Bangladesh	4×10^{-3}%	0.76	1.91×10^{-3}
Kazakhstan	3×10^{-3}%	3.14	2.79×10^{-3}
Chile	3×10^{-3}%	0.79	1.16×10^{-3}
Pakistan	3×10^{-3}%	0.93	1.22×10^{-3}
Ethiopia	2×10^{-3}%	0.91	8.03×10^{-4}
Switzerland	1×10^{-3}%	0.71	5.54×10^{-4}
Turkey	1×10^{-3}%	0.60	4.06×10^{-4}
Paraguay	9×10^{-4}%	0.65	2.38×10^{-4}
Norway	7×10^{-4}%	1.02	2.66×10^{-4}
Japan	2×10^{-4}%	0.45	7.38×10^{-5}
Korea, Rep.	3×10^{-5}%	1.45	4.28×10^{-5}
Morocco	1×10^{-5}%	3.20	4.73×10^{-5}
		$\Sigma = 2.89$	

Table S9. Data to determine trade barriers of soy bean consuming countries: countries, global consumption share and Worldwide Governance Index.

Soy Bean Consuming Countries	Global Consumption Share (gps) Based on Barrientos and Soria (2016)	Enabling Trade Indicator (ETI) Based on Hanouz et al. (2014)	Global Consumption Share × ETI
China	30.29%	2.60	4.60×10^{-1}
USA	17.69%	3.80	5.26×10^{-1}
Argentina	15.41%	3.90	6.01×10^{-1}
Brazil	13.83%	3.30	9.99×10^{-1}
India	2.60%	4.10	5.54×10^{-2}
Russia	1.46%	4.00	1.04×10^{-1}
Mexico	1.41%	2.60	2.03×10^{-2}
Paraguay	1.35%	3.80	2.05×10^{-2}
Japan	1.00%	3.40	3.01×10^{-3}
Indonesia	0.94%	3.90	3.46×10^{-2}
Bolivia	0.89%	4.10	6.00×10^{-2}
Taiwan	0.81%	3.40	1.22×10^{-2}
Turkey	0.79%	4.50	1.16×10^{-2}
Thailand	0.79%	3.40	3.18×10^{-2}
Canada	0.78%	3.90	4.77×10^{-3}
Egypt	0.67%	3.50	4.94×10^{-2}
Vietnam	0.63%	2.50	2.49×10^{-2}
Ukraine	0.54%	3.90	2.68×10^{-3}
Iran	0.49%	4.40	2.83×10^{-3}
Pakistan	0.48%	4.60	2.23×10^{-2}
Korea, Rep.	0.45%	4.00	2.38×10^{-3}
South Africa	0.36%	3.60	2.27×10^{-2}
Nigeria	0.26%	3.70	1.95×10^{-3}
Colombia	0.21%	2.90	1.31×10^{-2}
Serbia	0.12%	3.30	2.60×10^{-2}
Peru	0.11%	4.70	1.36×10^{-3}
Uruguay	0.09%	4.37	3.89×10^{-3}
Venezuela	0.09%	3.60	7.53×10^{-3}
Zambia	0.07%	3.50	7.88×10^{-4}
Myanmar	0.06%	2.70	5.12×10^{-4}
Uganda	0.06%	3.40	2.68×10^{-2}
Korea, Dem. Rep.	0.05%	3.50	7.32×10^{-4}
Philippines	0.04%	4.00	2.69×10^{-2}
Zimbabwe	0.03%	3.70	4.64×10^{-4}
Ecuador	0.02%	3.60	8.11×10^{-5}
Australia	0.02%	2.70	2.20×10^{-2}
Guatemala	0.02%	3.30	3.74×10^{-3}
Syrian Arab. Rep.	0.02%	2.40	1.39×10^{-4}
Switzerland	0.01%	4.10	1.98×10^{-2}
Bosnia	0.01%	4.10	9.50×10^{-4}
Nicaragua	0.00%	3.50	1.49×10^{-3}
	$\Sigma = 3.23$		

Table S10. Data to determine trade barriers of rapeseed consuming countries: countries, global consumption share and Enabling Trade Indicator.

Rapeseed Consuming Countries	Global Consumption Share (gcs) Based on Barrientos and Soria (2016)	Enabling Trade Indicator (ETI) Based on Hanouz et al. (2014)	Global Consumption Share × ETI
EU-27	36.91%	0.83	3.07×10^{-1}
China	27.83%	2.71	6.87×10^{-1}
Canada	13.06%	2.05	6.60×10^{-1}
India	9.00%	1.70	3.54×10^{-1}
Japan	3.65%	0.45	7.38×10^{-5}
USA	2.48%	2.00	5.01×10^{-2}
Russia	1.53%	3.53	6.81×10^{-2}
Australia	1.40%	1.40	1.24×10^{-1}
Pakistan	1.19%	3.28	9.69×10^{-3}
Turkey	0.74%	2.87	2.92×10^{-3}
Ukraine	0.48%	2.08	9.54×10^{-2}
Belarus	0.45%	0.99	1.64×10^{-2}
Bangladesh	0.45%	3.22	1.43×10^{-2}
Chile	0.28%	2.22	7.38×10^{-3}
Kazakhstan	0.25%	3.82	1.30×10^{-2}
Ethiopia	0.12%	2.20	5.20×10^{-3}
Switzerland	0.11%	2.07	2.45×10^{-3}
Paraguay	0.04%	3.14	2.79×10^{-3}
Norway	0.03%	0.60	4.06×10^{-4}
	$\Sigma = 2.42$		

3.2.4. Concentration of Harvesting

In the following it is explained how the Herfindahl-Hirschmann-Index (HHI) is calculated for the harvesting step for soy beans and rapeseeds. To determine the HHI the global production shares are squared and summed up (see Tables S11 and S12). For soy bean the HHI is 0.25 and for rapeseed it is 0.13.

Table S11. Production data to determine Herfindahl-Hirschmann-Index for soy beans.

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Squared Global Production Shares
USA	33.5%	1.12×10^{-1}
Brazil	31.3%	9.83×10^{-2}
Argentina	17.9%	3.19×10^{-2}
China	3.8%	1.42×10^{-3}
Paraguay	2.8%	7.61×10^{-4}
India	2.5%	6.29×10^{-4}
Canada	2.0%	3.82×10^{-4}
Ukraine	1.2%	1.42×10^{-4}
Uruguay	1.0%	9.50×10^{-5}
Bolivia	1.0%	9.44×10^{-5}
Russia	0.9%	7.98×10^{-5}
South Africa	0.3%	7.78×10^{-6}
Nigeria	0.2%	4.15×10^{-6}
Indonesia	0.2%	3.78×10^{-6}
Serbia	0.2%	2.76×10^{-6}

Table S11. *Cont.*

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Squared Global Production Shares
Mexico	0.1%	1.27×10^{-6}
Japan	0.1%	4.76×10^{-7}
Zambia	0.1%	4.50×10^{-7}
Myanmar	0.1%	3.93×10^{-7}
Iran	0.1%	3.74×10^{-7}
Uganda	0.1%	3.55×10^{-7}
Vietnam	0.1%	3.01×10^{-7}
Korea, Dem. Rep.	4.7×10^{-4}%	2.21×10^{-7}
Korea, Rep.	3.8×10^{-4}%	1.42×10^{-7}
Turkey	2.8×10^{-4}%	7.96×10^{-8}
Zimbabwe	2.8×10^{-4}%	7.96×10^{-8}
Venezuela	2.4×10^{-4}%	5.53×10^{-8}
Colombia	2.3×10^{-4}%	5.24×10^{-8}
Ecuador	2.2×10^{-4}%	4.81×10^{-8}
Australia	1.9×10^{-4}%	3.54×10^{-8}
Thailand	1.6×10^{-4}%	2.46×10^{-8}
Guatemala	1.1×10^{-4}%	1.27×10^{-8}
Egypt	6.3×10^{-5}%	3.93×10^{-9}
Bosnia	2.2×10^{-5}%	4.81×10^{-10}
Nicaragua	2.2×10^{-5}%	4.81×10^{-10}
Taiwan	1.9×10^{-5}%	3.54×10^{-10}
Peru	9.4×10^{-6}%	8.84×10^{-11}
Switzerland	9.4×10^{-6}%	8.84×10^{-11}
Pakistan	6.3×10^{-6}%	3.93×10^{-11}
Syrian Arab.Rep.	6.3×10^{-6}%	3.93×10^{-11}
Philippines	3.1×10^{-6}%	9.83×10^{-12}
	$\Sigma = 0.25$	

Table S12. Production data to determine Herfindahl-Hirschmann-Index for rapeseed.

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Squared Global Production Shares
Canada	32%	6.45×10^{-2}
China	25%	4.34×10^{-2}
India	21%	7.85×10^{-3}
Australia	9%	2.10×10^{-3}
France	8.07%	6.51×10^{-3}
Germany	7.12%	5.07×10^{-3}
Ukraine	5%	6.30×10^{-4}
UK	3.79%	1.43×10^{-3}
USA	3%	3.71×10^{-4}
Poland	2.74%	7.52×10^{-4}
Russia	2%	2.76×10^{-4}
Belarus	2%	1.96×10^{-5}
Czech Rep.	1.64%	2.68×10^{-4}
Lithuania	0.93%	8.74×10^{-5}
Denmark	0.72%	5.13×10^{-5}
Hungary	0.59%	3.51×10^{-5}

Table S12. *Cont.*

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Squared Global Production Shares
Sweden	0.48%	2.28×10^{-5}
Latvia	0.45%	2.02×10^{-5}
Bulgaria	0.40%	1.60×10^{-5}
Slovakia	0.31%	9.89×10^{-6}
Estonia	0.24%	5.79×10^{-6}
Romania	0.23%	5.51×10^{-6}
Austria	0.22%	4.84×10^{-6}
Finland	0.11%	1.16×10^{-6}
Spain	0.08%	5.90×10^{-7}
Belgium	0.07%	5.02×10^{-7}
Italy	0.04%	1.36×10^{-7}
Ireland	0.04%	1.71×10^{-7}
Slovenia	0.03%	6.30×10^{-8}
Luxembourg	0.02%	4.91×10^{-8}
Netherlands	0.01%	1.07×10^{-8}
Greece	0.01%	5.45×10^{-9}
Bangladesh	4×10^{-3}%	1.15×10^{-5}
Kazakhstan	3×10^{-3}%	1.10×10^{-5}
Chile	3×10^{-3}%	8.72×10^{-6}
Pakistan	3×10^{-3}%	5.58×10^{-6}
Ethiopia	2×10^{-3}%	1.40×10^{-6}
Switzerland	1×10^{-3}%	1.04×10^{-6}
Turkey	1×10^{-3}%	7.85×10^{-7}
Paraguay	9×10^{-4}%	4.61×10^{-7}
Norway	7×10^{-4}%	2.64×10^{-8}
Japan	2×10^{-4}%	8.72×10^{-10}
	$\Sigma = 0.13$	

3.2.5. Storage Complexity

In the following it is shown how the category storage complexity is calculated for soy beans and rapeseeds (see Tables S13 and S14). For soy beans the results add up to 7.78 and for rapeseed to 31.04. Furthermore according to Equation (7) (in the main part of the article) the moisture content of the material in storage has to be considered as well. For soy beans and rapeseed the average moisture content during storage is 11%–15% [3,4]. Thus, to both values 13 (average of 11 and 15) is added. The values for the storage complexity for soy beans is 20.78 and for rapeseed 44.04.

Table S13. Data to determine the storage complexity of soy bean producing countries: countries, global production share and Economic Vulnerability Indicator.

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Economic Vulnerability Indicator (EVI) Based on Organisation for Economic Co-Operation and Development (2016)	Global Production Share × EVI
USA	33.5%	5.01	1.68
Brazil	31.3%	0.11	3.41×10^{-2}
Argentina	17.9%	8.39	1.50
China	3.8%	0.00	0
Paraguay	2.8%	24.72	6.82×10^{-1}
India	2.5%	16.77	4.21×10^{-1}
Canada	2.0%	86.00	1.68
Ukraine	1.2%	58.25	6.94×10^{-1}
Uruguay	1.0%	24.07	2.35×10^{-1}
Bolivia	1.0%	24.94	2.42×10^{-1}
Russia	0.9%	20.04	1.79×10^{-1}
South Africa	0.3%	12.31	3.43×10^{-2}
Nigeria	0.2%	56.42	1.15×10^{-1}
Indonesia	0.2%	14.49	2.82×10^{-2}
Serbia	0.2%	44.68	7.42×10^{-2}
Mexico	0.1%	3.81	4.30×10^{-3}
Japan	0.1%	8.82	6.08×10^{-3}
Zambia	0.1%	42.33	2.84×10^{-2}
Iran	0.1%	42.37	2.59×10^{-2}
Uganda	0.1%	49.78	2.96×10^{-2}
Vietnam	0.1%	36.97	2.03×10^{-2}
Korea, Rep.	3.8×10^{-4}%	60.55	2.28×10^{-2}
Turkey	2.8×10^{-4}%	15.25	4.30×10^{-3}
Zimbabwe	2.8×10^{-4}%	21.90	6.18×10^{-3}
Venezuela	2.4×10^{-4}%	38.77	9.12×10^{-3}
Colombia	2.3×10^{-4}%	21.13	4.84×10^{-3}
Ecuador	2.2×10^{-4}%	35.78	7.85×10^{-3}
Australia	1.9×10^{-4}%	15.36	2.89×10^{-3}
Thailand	1.6×10^{-4}%	30.28	4.75×10^{-3}
Guatemala	1.1×10^{-4}%	35.03	3.95×10^{-3}
Egypt	6.3×10^{-5}%	54.89	3.44×10^{-3}
Bosnia	2.2×10^{-5}%	42.05	9.23×10^{-4}
Nicaragua	2.2×10^{-5}%	48.14	1.06×10^{-3}
Peru	9.4×10^{-6}%	20.26	1.91×10^{-4}
Switzerland	9.4×10^{-6}%	14.81	1.39×10^{-4}
Pakistan	6.3×10^{-6}%	29.08	1.82×10^{-4}
Syrian Arab.Rep.	6.3×10^{-6}%	15.15	9.50×10^{-5}
Philippines	3.1×10^{-6}%	40.41	1.27×10^{-4}
	$\Sigma = 7.78$		

Table S14. Data to determine storage complexity of rapeseed producing countries: countries, global production share and Economic Vulnerability Indicator.

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Economic Vulnerability Indicator (EVI) Based on Organisation for Economic Co-Operation and Development (2016)	Global Production Share × EVI
Canada	25.40%	67.85	2.18×10^{1}
China	20.82%	0.00	0.00
India	8.86%	7.14	1.49
France	8.07%	10.78	8.70×10^{-1}
Germany	7.12%	8.28	5.89×10^{-1}
Australia	4.58%	7.93	7.03×10^{-1}
UK	3.79%	8.82	3.34×10^{-1}
Poland	2.74%	14.59	4.00×10^{-1}
Ukraine	2.51%	31.94	1.46
USA	1.93%	3.85	9.66×10^{-2}
Russia	1.66%	17.28	3.33×10^{-1}
Czech Republic	1.64%	25.70	4.21×10^{-1}
Lithuania	0.93%	38.88	3.63×10^{-1}
Denmark	0.72%	33.87	2.43×10^{-1}
Hungary	0.59%	24.51	1.45×10^{-1}
Sweden	0.48%	17.32	8.26×10^{-2}
Latvia	0.45%	59.90	2.69×10^{-1}
Belarus	0.44%	8.27	1.37×10^{-1}
Bulgaria	0.40%	48.83	1.95×10^{-1}
Bangladesh	0.34%	20.04	8.88×10^{-2}
Kazakhstan	0.33%	33.12	1.12×10^{-1}
Slovakia	0.31%	29.73	9.35×10^{-2}
Chile	0.30%	88.89	2.95×10^{-1}
Estonia	0.24%	75.70	1.82×10^{-1}
Pakistan	0.24%	23.26	6.87×10^{-2}
Austria	0.22%	18.08	3.98×10^{-2}
Ethiopia	0.12%	14.79	3.50×10^{-2}
Finland	0.11%	23.85	2.57×10^{-2}
Switzerland	0.10%	12.78	1.51×10^{-2}
Turkey	0.09%	13.26	1.35×10^{-2}
Spain	0.08%	20.91	1.61×10^{-2}
Belgium	0.07%	32.02	2.27×10^{-2}
Paraguay	0.07%	18.96	1.68×10^{-2}
Ireland	0.04%	30.93	1.28×10^{-2}
Italy	0.04%	6.75	2.49×10^{-3}
Slovenia	0.03%	25.60	6.43×10^{-3}
Norway	0.02%	10.83	7.36×10^{-3}
Netherlands	0.01%	30.39	3.14×10^{-3}
Greece	0.01%	54.57	4.03×10^{-3}
Japan	0.00%	1.60	2.61×10^{-4}
	$\Sigma = 31.04$		

3.2.6. Application of Distance-to-Target Approach

In the following the in Section 2 (in the main part of the article) introduced Distance-to-Target (DtT) approach is applied. For more information regarding this approach in relation to the assessment of socio-economic availability see publications by Bach et al. (2016).

(Step 1) Determination of indicator values is carried out for soy beans and rapeseeds (see Table S15). The category storage complexity has been excluded as no targets are available for this category.

Table S15. Indicator results of case study for considered categories and supply chain stages.

Categories	Biotic Materials	Cultivation and Harvesting Step	Process Step	(Intermediate) Product
Political instability	Soy beans	2.66	2.24	2.24
	Rapeseeds	2.04	2.95	2.95
Demand growth	Soy beans	7.55	8.99	8.99
	Rapeseeds	2.6	1.6	1.6
Trade barriers	Soy beans	3.65	3.23	3.23
	Rapeseeds	2.89	2.73	2.73
Price fluctuations	Soy beans	7.14	n.a.	n.a.
	Rapeseeds	2.7	n.a.	n.a.
Occurence as co-product	Soy beans	0	0.5	0
	Rapeseeds	0	0.5	0
Concentration of harvesting	Soy beans	0.25	0	0
	Rapeseeds	0.13	0	0
Storage capacity	Soy beans	20.78	0	0
	Rapeseeds	44.04	0	0
Recycling	Soy beans	0	0	100
	Rapeseeds	0	0	100

(Step 2a) Determination of targets: Targets were identified in the previous project by Bach et al. (2016a). These are shown in Table S4 (in the main part of the article).

(Step 2b) Calculation of DtT-value (see Equation (9) in the main part of the article). Calculations and results are presented here exemplary for soy beans in the supply chain stage cultivation and harvest in Table S16.

Table S16. Calculation and results of DtT-value for considered categories for soy beans.

Categories	Indicator Results	Targets	DtT Calculation and Results
Political instability	2.66	1.9	$\left(\frac{2.66}{1.9}\right)^2 = 1.96$
Demand growth	7.55	5	$\left(\frac{7.55}{5}\right)^2 = 22801$
Trade barriers	3.65	3.15	$\left(\frac{3.65}{3.15}\right)^2 = 1.34$
Price fluctuations	7.14	20	$\left(\frac{7.14}{20}\right)^2 = 1275$
Occurence as co-product	0	0.5	$\left(\frac{0}{0.5}\right)^2 = 0$
Concentration of harvesting	0.25	0.15	$\left(\frac{0.25}{0.15}\right)^2 = 2.78$
Storage complexity	20.78	60	$\left(\frac{20.78}{60}\right)^2 = 0.12$ >>set to zero
Recycling	0	0.75	$\left(\frac{0}{0.75}\right)^2 = 0$

(Step 3) Normalization with the global production amount: results are presented exemplarily for soy beans and the supply chain stage cultivation and harvest (see Table S17). The global production for soy beans is 319 million tones [1].

Table S17. Calculation and results of normalized DtT-value for considered categories.

Categories	DtT values	Calculation and Results of Normalized DtT Values
Political instability	1.96	$\frac{1.96}{319\,008\,000} = 6.14 \times 10^{-9}$
Demand growth	22801	$\frac{22801}{319\,008\,000} = 7.15 \times 10^{-5}$
Trade barriers	1.34	$\frac{1.34}{319\,008\,000} = 4.21 \times 10^{-9}$
Price fluctuations	1275	$\frac{1275}{319\,008\,000} = 4.0 \times 10^{-6}$
Occurence as co-product	0	0
Concentration of harvesting	2.78	$\frac{2.78}{319\,008\,000} = 8.71 \times 10^{-9}$
Recycling	0	0
Storage complexity	0	0

(Step 4) Scaling of results: the determined results are rescaled to the 6.30×10^{15} (overall amount of cereals being produced in a year [5]). Furthermore, the largest value for each category has to be determined. Therefore, the results for all supply chain stages for soy bean and rapeseed are summed up (see Table S18). For some categories the overall value for soy beans is higher (demand growth), for some categories the overall value for rapeseeds is higher (political instability, trade barriers, price fluctuations, occurrence as co-product and recycling).

Table S18. Overall results for the biotic materials soy bean and rapeseed for considered categories.

Category/Biotic Material	Political Instability	Demand Growth	Trade Barriers	Price Fluc-Tuations	Occurence as co-Product	Concentration of Harvesting	Recycling
Soy beans	1.49×10^{-8}	2.74×10^{-4}	1.08×10^{-8}	4.00×10^{-6}	3.92×10^{-8}	8.71×10^{-9}	5.57×10^{-5}
Rapeseeds	8.82×10^{-8}	7.02×10^{-5}	1.24×10^{-8}	2.69×10^{-6}	1.85×10^{-7}	0	2.63×10^{-4}

The calculations are shown exemplary for soy beans and the supply chain stage cultivation and harvest (see Table S19) based on Equation (11) (in the main part of the article).

Table S19. Calculation and results of scaled values for considered categories.

Categories	Normalized DtT Values	Calculation and Results of Scaled Values
Political instability	6.14×10^{-9}	$\frac{6.3 \times 10^{15} \times 6.14 \times 10^{-9}}{8.82 \times 10^{-8}} = 4.39 \times 10^{14}$
Demand growth	7.15×10^{-5}	$\frac{6.3 \times 10^{15} \times 7.15 \times 10^{-5}}{2.74 \times 10^{-4}} = 1.64 \times 10^{15}$
Trade barriers	4.21×10^{-9}	$\frac{6.3 \times 10^{15} \times 4.21 \times 10^{-9}}{1.23 \times 10^{-8}} = 2.13 \times 10^{15}$
Price fluctuations	4.0×10^{-6}	$\frac{6.3 \times 10^{15} \times 4.0 \times 10^{-6}}{2.69 \times 10^{-6}} = 9.35 \times 10^{15}$
Occurence as co-product	0	0
Concentration of harvesting	8.71×10^{-9}	$\frac{6.3 \times 10^{15} \times 6.14 \times 10^{-9}}{8.82 \times 10^{-8}} = 4.39 \times 10^{14}$
Recycling	0	0

3.3. Abiotic Constraints

In the following it is shown how the abiotic constraints water availability and natural disasters are calculated.

3.3.1. Water Availability

Next it is explained how the water availability is determined for soy beans and rapeseeds according to Equation (12) (in the main part of the article) (see Tables S20 and S21). The water availability sums up to 0.33 for both biotic materials.

Table S20. Data for determination of the water availability of soy bean producing countries: countries, global production share and Water Depletion Index.

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Water Depletion Index (WDI) Based on Berger et al. (2014)	Global Production Share × WDI
USA	33.5%	0.55	1.83×10^{-1}
Brazil	31.3%	0.08	2.48×10^{-2}
Argentina	17.9%	0.38	6.82×10^{-2}
China	3.8%	0.58	2.17×10^{-2}
Paraguay	2.8%	0.01	2.76×10^{-4}
India	2.5%	0.74	1.86×10^{-2}
Canada	2.0%	0.07	1.39×10^{-3}
Ukraine	1.2%	0.46	5.52×10^{-3}
Uruguay	1.0%	0.01	1.37×10^{-4}
Bolivia	1.0%	0.22	2.09×10^{-3}
Russia	0.9%	0.08	7.37×10^{-4}
South Africa	0.3%	0.83	2.33×10^{-3}
Nigeria	0.2%	0.28	5.68×10^{-4}
Indonesia	0.2%	0.17	3.25×10^{-4}
Serbia	0.2%	0.18	2.92×10^{-4}
Mexico	0.1%	0.78	8.76×10^{-4}
Japan	0.1%	0.48	3.33×10^{-4}
Myanmar	0.1%	0.02	1.07×10^{-5}
Iran	0.1%	0.95	5.80×10^{-4}
Uganda	0.1%	0.01	6.92×10^{-6}
Vietnam	0.1%	0.28	1.55×10^{-4}
Korea, Dem. Rep.	4.7×10^{-4}%	0.50	2.34×10^{-4}
Korea, Rep.	3.8×10^{-4}%	0.42	1.59×10^{-4}
Turkey	2.8×10^{-4}%	0.72	2.02×10^{-4}
Venezuela	2.4×10^{-4}%	0.39	9.27×10^{-5}
Colombia	2.3×10^{-4}%	0.02	3.78×10^{-6}
Ecuador	2.2×10^{-4}%	0.19	4.13×10^{-5}
Australia	1.9×10^{-4}%	0.91	1.70×10^{-4}
Thailand	1.6×10^{-4}%	0.05	8.24×10^{-6}
Guatemala	1.1×10^{-4}%	0.01	1.13×10^{-6}
Egypt	6.3×10^{-5}%	1.00	6.27×10^{-5}
Bosnia	2.2×10^{-5}%	0.18	3.88×10^{-6}
Nicaragua	2.2×10^{-5}%	0.01	2.20×10^{-7}
Peru	9.4×10^{-6}%	0.73	6.89×10^{-6}
Switzerland	9.4×10^{-6}%	0.36	3.35×10^{-6}
Pakistan	6.3×10^{-6}%	0.97	6.07×10^{-6}
Syrian Arab.Rep.	6.3×10^{-6}%	0.94	5.88×10^{-6}
Philippines	3.1×10^{-6}%	0.02	7.21×10^{-8}
$\Sigma = 0.33$			

Table S21. Data for determination of water availability of rapeseed producing countries: countries, global production share and Water Depletion Index.

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Water Depletion Index (WDI) Based on Berger et al. (2014)	Global Production Share × WDI
Canada	32%	0.06	1.80×10^{-2}
China	25%	0.47	1.20×10^{-1}
India	21%	0.32	6.57×10^{-2}
Australia	9%	0.47	4.14×10^{-2}
France	8.07%	0.22	1.75×10^{-2}
Germany	7.12%	0.17	1.24×10^{-2}
Ukraine	5%	0.25	1.16×10^{-2}
UK	3.79%	0.28	1.06×10^{-2}
USA	3%	0.42	1.05×10^{-2}
Poland	2.74%	0.04	1.03×10^{-3}
Russia	2%	0.07	1.37×10^{-3}
Belarus	2%	0.03	4.66×10^{-4}
Czech Rep.	1.64%	0.14	2.37×10^{-3}
Lithuania	0.93%	0.02	1.78×10^{-4}
Denmark	0.72%	0.28	2.02×10^{-3}
Hungary	0.59%	0.18	1.07×10^{-3}
Sweden	0.48%	0.15	7.37×10^{-4}
Latvia	0.45%	0.02	8.54×10^{-5}
Bulgaria	0.40%	0.56	2.26×10^{-3}
Slovakia	0.31%	0.18	5.57×10^{-4}
Estonia	0.24%	0.02	3.66×10^{-5}
Romania	0.24%	0.21	4.85×10^{-4}
Austria	0.22%	0.18	3.86×10^{-4}
Finland	0.11%	0.20	2.11×10^{-4}
Spain	0.08%	0.72	5.53×10^{-4}
Belgium	0.07%	0.86	6.12×10^{-4}
Italy	0.04%	0.64	2.36×10^{-4}
Ireland	0.04%	0.10	4.10×10^{-5}
Slovenia	0.03%	0.17	4.36×10^{-5}
Luxembourg	0.02%	0.25	5.47×10^{-5}
Netherlands	0.01%	0.74	7.63×10^{-5}
Greece	0.01%	0.69	5.09×10^{-5}
Bangladesh	4×10^{-3}%	0.15	6.50×10^{-4}
Kazakhstan	3×10^{-3}%	0.93	3.15×10^{-3}
Chile	3×10^{-3}%	0.61	2.03×10^{-3}
Pakistan	3×10^{-3}%	0.77	2.29×10^{-3}
Ethiopia	2×10^{-3}%	0.26	6.09×10^{-4}
Switzerland	1×10^{-3}%	0.31	3.63×10^{-4}
Turkey	1×10^{-3}%	0.62	6.34×10^{-4}
Paraguay	9×10^{-4}%	0.01	6.79×10^{-6}
Norway	7×10^{-4}%	0.06	3.80×10^{-5}
Japan	2×10^{-4}%	0.09	1.43×10^{-5}
Korea, Rep.	3×10^{-5}%	0.21	6.25×10^{-6}
Morocco	1×10^{-5}%	0.99	1.46×10^{-5}
	$\Sigma = 0.33$		

3.3.2. Natural Disasters

In the following it is explained how the results for the category natural disasters are obtained. Based on Equation (13) (in the main part of the article) the global production shares are multiplied with the Natural Disaster Index (NDI) per country and are summed up (see Tables S22 and S23). For soy beans the natural disaster risk adds up to 16.75. For rapeseed the overall risk is 10.84.

Table S22. Data for determination of the natural disaster risk of soy bean producing countries: countries, global production share and natural disaster index.

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Natural Disaster Index (NDI) Based on United Nations Office for Disaster Risk Reduction (2013)	Global Production Share × NDI
USA	33.5%	31.37	1.05×10^{1}
Brazil	31.3%	10.88	3.41
Argentina	17.9%	5.79	1.03
China	3.8%	25.37	9.54×10^{-1}
Paraguay	2.8%	1.26	3.46×10^{-2}
India	2.5%	15.21	3.81×10^{-1}
Canada	2.0%	7.10	1.39×10^{-1}
Ukraine	1.2%	1.38	1.64×10^{-2}
Uruguay	1.0%	0.95	9.24×10^{-3}
Bolivia	1.0%	2.63	2.55×10^{-2}
Russia	0.9%	14.33	1.28×10^{-1}
South Africa	0.3%	5.16	1.44×10^{-2}
Nigeria	0.2%	2.74	5.59×10^{-3}
Indonesia	0.2%	7.14	1.39×10^{-2}
Serbia	0.2%	0.43	7.08×10^{-4}
Mexico	0.1%	8.09	9.13×10^{-3}
Japan	0.1%	2.55	1.76×10^{-3}
Myanmar	0.1%	3.51	2.20×10^{-3}
Iran	0.1%	5.80	3.55×10^{-3}
Uganda	0.1%	1.19	7.11×10^{-4}
Vietnam	0.1%	2.96	1.63×10^{-3}
Korea, Dem. Rep.	4.7×10^{-4}%	1.10	5.17×10^{-4}
Korea, Rep.	3.8×10^{-4}%	1.72	6.48×10^{-4}
Turkey	2.8×10^{-4}%	2.67	7.52×10^{-4}
Venezuela	2.4×10^{-4}%	2.50	5.88×10^{-4}
Colombia	2.3×10^{-4}%	3.06	7.00×10^{-4}
Ecuador	2.2×10^{-4}%	0.79	1.74×10^{-4}
Australia	1.9×10^{-4}%	23.77	4.47×10^{-3}
Thailand	1.6×10^{-4}%	3.88	6.08×10^{-4}
Guatemala	1.1×10^{-4}%	0.47	5.25×10^{-5}
Egypt	6.3×10^{-5}%	1.07	6.70×10^{-5}
Bosnia	2.2×10^{-5}%	0.37	8.09×10^{-6}
Nicaragua	2.2×10^{-5}%	0.63	1.39×10^{-5}
Taiwan	1.9×10^{-5}%	0.81	1.52×10^{-5}
Peru	9.4×10^{-6}%	3.63	3.42×10^{-5}
Switzerland	9.4×10^{-6}%	0.35	3.25×10^{-6}
Pakistan	6.3×10^{-6}%	4.73	2.96×10^{-5}
Syrian Arab.Rep.	6.3×10^{-6}%	0.66	4.14×10^{-6}
Philippines	3.1×10^{-6}%	7.16	2.24×10^{-5}
$\Sigma = 16.75$			

Table S23. Data for determination of the natural disaster risk of rapeseed producing countries: countries, global production share and natural disaster index.

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Natural Disaster Index (NDI) Based on United Nations Office for Disaster Risk Reduction (2013)	Global Production Share × NDI
Canada	32%	7.10	1.80
China	25%	25.37	5.28
India	21%	15.21	1.35
Australia	9%	23.77	1.09
France	8.07%	2.59	2.09×10^{-1}
Germany	7.12%	0.90	6.39×10^{-2}
Ukraine	5%	1.38	3.47×10^{-2}
UK	3.79%	1.64	6.19×10^{-2}
USA	3%	31.37	6.05×10^{-1}
Poland	2.74%	0.74	2.02×10^{-2}
Russia	2%	14.33	2.38×10^{-1}
Czech Rep.	1.64%	0.52	8.59×10^{-3}
Lithuania	0.93%	0.06	5.66×10^{-4}
Denmark	0.72%	0.34	2.44×10^{-3}
Hungary	0.59%	0.30	1.75×10^{-3}
Sweden	0.48%	0.47	2.25×10^{-3}
Latvia	0.45%	0.26	1.16×10^{-3}
Bulgaria	0.40%	0.79	3.14×10^{-3}
Slovakia	0.31%	0.24	7.56×10^{-4}
Estonia	0.24%	0.03	7.38×10^{-5}
Romania	0.24%	1.74	4.08×10^{-3}
Austria	0.22%	0.37	8.07×10^{-4}
Finland	0.11%	0.56	5.99×10^{-4}
Spain	0.08%	3.07	2.36×10^{-3}
Belgium	0.07%	0.07	5.13×10^{-5}
Italy	0.04%	2.87	1.06×10^{-3}
Ireland	0.04%	0.03	1.32×10^{-5}
Slovenia	0.03%	0.44	1.11×10^{-4}
Luxembourg	0.02%	0.00	0.00
Greece	0.01%	2.66	1.96×10^{-4}
Bangladesh	4×10^{-3}%	5.43	1.84×10^{-2}
Kazakhstan	3×10^{-3}%	3.24	1.08×10^{-2}
Chile	3×10^{-3}%	2.17	6.40×10^{-3}
Pakistan	3×10^{-3}%	4.73	1.12×10^{-2}
Ethiopia	2×10^{-3}%	3.16	3.74×10^{-3}
Switzerland	1×10^{-3}%	0.35	3.53×10^{-4}
Turkey	1×10^{-3}%	2.67	2.36×10^{-3}
Paraguay	9×10^{-4}%	1.26	8.53×10^{-4}
Norway	7×10^{-4}%	1.22	1.99×10^{-4}
Japan	2×10^{-4}%	2.55	7.52×10^{-5}
Korea, Rep.	3×10^{-5}%	1.72	2.55×10^{-5}
Morocco	1×10^{-5}%	1.93	2.85×10^{-5}
	$\Sigma = 10.84$		

3.4. Social Constraints

Next it is described how social constraints are determined.

3.4.1. Food Security

In the following it is explained how the food security is calculated for soy beans and rapeseeds according to Equation (14) (in the main part of the article) (see Tables S24 and S25). The food security sums up to 25.39 for soy beans and to 19.13 for rapeseed.

Table S24. Data for determination of the food security of soy bean producing countries: countries, global production share and Food Security Index.

Soy bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Food Security Index (FSI) Based on FAO (2016)	Global Production Share × FSI
USA	33.5%	15.29	5.13
Brazil	31.3%	39.50	1.24×10^{1}
Argentina	17.9%	20.64	3.69
China	3.8%	28.64	1.08
Paraguay	2.8%	28.86	7.96×10^{-1}
India	2.5%	32.55	8.16×10^{-1}
Canada	2.0%	11.27	2.20×10^{-1}
Ukraine	1.2%	19.66	2.34×10^{-1}
Uruguay	1.0%	28.77	2.80×10^{-1}
Bolivia	1.0%	4.85	4.71×10^{-2}
Russia	0.9%	13.85	1.24×10^{-1}
South Africa	0.3%	71.91	2.01×10^{-1}
Nigeria	0.2%	27.89	5.68×10^{-2}
Indonesia	0.2%	34.91	6.78×10^{-2}
Serbia	0.2%	32.55	5.41×10^{-2}
Mexico	0.1%	35.42	4.00×10^{-2}
Japan	0.1%	17.14	1.18×10^{-2}
Myanmar	0.1%	26.40	1.65×10^{-2}
Iran	0.1%	36.65	2.24×10^{-2}
Uganda	0.1%	34.50	2.05×10^{-2}
Vietnam	0.1%	48.46	2.66×10^{-2}
Korea, Dem. Rep.	4.7×10^{-4}%	28.15	1.32×10^{-2}
Korea, Rep.	3.8×10^{-4}%	45.20	1.70×10^{-2}
Turkey	2.8×10^{-4}%	35.77	1.01×10^{-2}
Venezuela	2.4×10^{-4}%	14.04	3.30×10^{-3}
Colombia	2.3×10^{-4}%	24.69	5.65×10^{-3}
Ecuador	2.2×10^{-4}%	32.82	7.20×10^{-3}
Australia	1.9×10^{-4}%	32.55	6.12×10^{-3}
Thailand	1.6×10^{-4}%	32.55	5.10×10^{-3}
Guatemala	1.1×10^{-4}%	36.13	4.08×10^{-3}
Egypt	6.3×10^{-5}%	24.29	1.52×10^{-3}
Bosnia	2.2×10^{-5}%	32.55	7.14×10^{-4}
Nicaragua	2.2×10^{-5}%	24.48	5.37×10^{-4}
Peru	9.4×10^{-6}%	20.08	1.89×10^{-4}
Switzerland	9.4×10^{-6}%	15.78	1.48×10^{-4}
Pakistan	6.3×10^{-6}%	27.11	1.70×10^{-4}
Syrian Arab.Rep.	6.3×10^{-6}%	25.96	1.63×10^{-4}
Philippines	3.1×10^{-6}%	35.18	1.10×10^{-4}
		$\Sigma = 25.39$	

Table S25. Data for determination of food security of rapeseed producing countries: countries, global production share and Food security Index.

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Food Security Index Based on FAO (2016)	Global Production Share × FSI
Canada	32%	8.89	2.86
China	25%	23.48	5.96
India	21%	13.85	2.88
Australia	9%	16.82	1.49
France	8.07%	14.69	1.19
Germany	7.12%	8.30	5.91×10^{-1}
Ukraine	5%	10.78	4.94×10^{-1}
UK	3.79%	12.41	4.70×10^{-1}
USA	3%	11.74	2.95×10^{-1}
Poland	2.74%	18.40	5.05×10^{-1}
Russia	2%	11.94	2.30×10^{-1}
Belarus	2%	3.80	6.32×10^{-2}
Czech Rep.	1.64%	26.29	4.31×10^{-1}
Lithuania	0.93%	32.55	3.04×10^{-1}
Denmark	0.72%	13.00	9.31×10^{-2}
Hungary	0.59%	32.55	1.93×10^{-1}
Sweden	0.48%	19.69	9.39×10^{-2}
Latvia	0.45%	24.57	1.10×10^{-1}
Bulgaria	0.40%	19.37	7.75×10^{-2}
Slovakia	0.31%	32.55	1.02×10^{-1}
Estonia	0.24%	15.13	3.64×10^{-2}
Romania	0.24%	17.21	4.04×10^{-2}
Austria	0.22%	5.24	1.15×10^{-2}
Finland	0.11%	32.55	3.51×10^{-2}
Spain	0.08%	19.77	1.52×10^{-2}
Belgium	0.07%	16.26	1.15×10^{-2}
Italy	0.04%	19.50	7.20×10^{-3}
Ireland	0.04%	16.28	6.73×10^{-3}
Slovenia	0.03%	16.34	4.10×10^{-3}
Luxembourg	0.02%	9.10	2.02×10^{-3}
Netherlands	0.01%	20.93	2.16×10^{-3}
Greece	0.01%	32.55	2.40×10^{-3}
Bangladesh	4×10^{-3}%	38.74	1.72×10^{-1}
Kazakhstan	3×10^{-3}%	32.39	1.10×10^{-1}
Chile	3×10^{-3}%	17.29	5.75×10^{-2}
Pakistan	3×10^{-3}%	21.69	6.41×10^{-2}
Ethiopia	2×10^{-3}%	18.16	4.29×10^{-2}
Switzerland	1×10^{-3}%	13.61	1.61×10^{-2}
Turkey	1×10^{-3}%	31.10	3.17×10^{-2}
Paraguay	9×10^{-4}%	22.13	1.96×10^{-2}
Norway	7×10^{-4}%	4.49	3.05×10^{-3}
Japan	2×10^{-4}%	3.12	5.06×10^{-4}
Korea, Rep.	3×10^{-5}%	22.60	6.68×10^{-4}
Morocco	1×10^{-5}%	39.40	5.82×10^{-4}
	$\Sigma = 19.13$		

3.4.2. Societal Acceptance

In the following the calculations for the categories compliance with social and environmental standards is shown.

Compliance with Social Standards

In the following it is shown how the results for the category compliance with social standards are determined according to Equation (15) (in the main part of the article) (see Tables S26 and S27). The compliance with social standards sums up to 9.03 for soy beans and to 7.04 for rapeseed.

Table S26. Data to determine the societal acceptance of soy bean producing countries: countries, global production share and indicator for compliance with social standards.

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Indicator for Compliance with Social Standards (ICS) Based on Norris et al. (2013)	Global Production Share × ICS
USA	33.5%	8.3	2.78
Brazil	31.3%	10.9	3.42
Argentina	17.9%	3.67	6.56×10^{-1}
China	3.8%	18.3	6.88×10^{-1}
Paraguay	2.8%	10.5	2.90×10^{-1}
India	2.5%	20.9	5.24×10^{-1}
Ukraine	1.2%	13.1	1.56×10^{-1}
Uruguay	1.0%	4	3.90×10^{-2}
Bolivia	1.0%	14	1.36×10^{-1}
Russia	0.9%	15.8	1.41×10^{-1}
South Africa	0.3%	17.7	4.94×10^{-2}
Nigeria	0.2%	22.6	4.60×10^{-2}
Indonesia	0.2%	14.8	2.88×10^{-2}
Serbia	0.2%	9.6	1.59×10^{-2}
Mexico	0.1%	9.7	1.09×10^{-2}
Japan	0.1%	1.5	1.03×10^{-3}
Myanmar	0.1%	20.3	1.27×10^{-2}
Iran	0.1%	14.1	8.62×10^{-3}
Uganda	0.1%	20.6	1.23×10^{-2}
Korea, Rep.	3.8×10^{-4}%	5.9	2.22×10^{-3}
Turkey	2.8×10^{-4}%	13.9	3.92×10^{-3}
Venezuela	2.4×10^{-4}%	7.2	1.69×10^{-3}
Colombia	2.3×10^{-4}%	12.9	2.95×10^{-3}
Ecuador	2.2×10^{-4}%	8.3	1.82×10^{-3}
Australia	1.9×10^{-4}%	3.5	6.58×10^{-4}
Thailand	1.6×10^{-4}%	10.3	1.61×10^{-3}
Guatemala	1.1×10^{-4}%	10.9	1.23×10^{-3}
Egypt	6.3×10^{-5}%	8	5.02×10^{-4}
Bosnia	2.2×10^{-5}%	10.8	2.37×10^{-4}
Nicaragua	2.2×10^{-5}%	10.1	2.22×10^{-4}
Taiwan	1.9×10^{-5}%	2.7	5.08×10^{-5}
Peru	9.4×10^{-6}%	14	1.32×10^{-4}
Pakistan	6.3×10^{-6}%	19.8	1.24×10^{-4}
Syrian Arab. Rep.	6.3×10^{-6}%	12.3	7.71×10^{-5}
Philippines	3.1×10^{-6}%	18.2	5.71×10^{-5}
$\Sigma = 9.03$			

Table S27. Data to determine the societal acceptance of rapeseed producing countries: countries, global production share and indicator for compliance with social standards.

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	Indicator for Compliance with Social Standards (ICS) Based on Norris et al. (2013)	Global Production Share × ICS
China	25%	15.00	3.81
India	21%	8.89	1.85
Australia	9%	1.81	1.60×10^{-1}
France	8.07%	1.00	8.07×10^{-2}
Germany	7.12%	0.26	1.85×10^{-2}
Ukraine	5%	7.18	3.29×10^{-1}
USA	3%	6.37	1.60×10^{-1}
Poland	2.74%	1.00	2.74×10^{-2}
Russia	2%	13.62	2.62×10^{-1}
Belarus	2%	2.64	4.39×10^{-2}
Czech Rep.	1.64%	0.70	1.15×10^{-2}
Lithuania	0.93%	1.70	1.59×10^{-2}
Hungary	0.59%	4.10	2.43×10^{-2}
Latvia	0.45%	1.80	8.08×10^{-3}
Bulgaria	0.40%	6.00	2.40×10^{-2}
Slovakia	0.31%	0.67	2.11×10^{-3}
Estonia	0.24%	6.00	1.41×10^{-2}
Austria	0.22%	1.50	3.30×10^{-3}
Greece	0.01%	2.40	1.77×10^{-4}
Bangladesh	4×10^{-3}%	11.04	4.89×10^{-2}
Kazakhstan	3×10^{-3}%	8.71	2.96×10^{-2}
Chile	3×10^{-3}%	6.22	2.07×10^{-2}
Pakistan	3×10^{-3}%	15.84	4.68×10^{-2}
Ethiopia	2×10^{-3}%	9.70	2.29×10^{-2}
Switzerland	1×10^{-3}%	12.09	1.23×10^{-2}
Turkey	1×10^{-3}%	8.05	7.13×10^{-3}
Paraguay	9×10^{-4}%	0.27	4.43×10^{-5}
Norway	7×10^{-4}%	2.95	8.71×10^{-5}
Japan	2×10^{-4}%	14.00	2.07×10^{-4}
		$\Sigma = 7.04$	

Compliance with Environmental Standards

In the following it is shown how the results for the category compliance with environmental standards are determined according to Equation (16) (in the main part of the article) (see Tables S28 and S29). The compliance with environmental standards sums up to 11.04 for soy beans and to 11.17 for rapeseed.

Sustainability **2017**, *9*, 137; doi:10.3390/su9010137

Table S28. Data for determination of the compliance with environmental standards of soy bean producing countries: countries, global production share and Environmental Performance Indicators (EPI).

Soy Bean Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	EPI Indicators Based on Yale Center for Environmental Law & Policy (2014)	Global Production Share × EPI
USA	33.5%	8.80	2.95
Brazil	31.3%	9.77	3.06
Argentina	17.9%	13.96	2.49
China	3.8%	9.25	3.48×10^{-1}
Paraguay	2.8%	25.53	7.04×10^{-1}
India	2.5%	16.19	4.06×10^{-1}
Canada	2.0%	10.67	2.09×10^{-1}
Ukraine	1.2%	18.36	2.19×10^{-1}
Uruguay	1.0%	21.55	2.10×10^{-1}
Bolivia	1.0%	15.00	1.46×10^{-1}
Russia	0.9%	13.58	1.21×10^{-1}
South Africa	0.3%	7.12	1.99×10^{-2}
Nigeria	0.2%	12.36	2.52×10^{-2}
Indonesia	0.2%	5.90	1.15×10^{-2}
Serbia	0.2%	18.72	3.11×10^{-2}
Mexico	0.1%	10.21	1.15×10^{-2}
Japan	0.1%	8.18	5.64×10^{-3}
Myanmar	0.1%	20.31	1.27×10^{-2}
Iran	0.1%	17.58	1.07×10^{-2}
Uganda	0.1%	9.82	5.85×10^{-3}
Vietnam	0.1%	14.19	7.79×10^{-3}
Korea, Dem. Rep.	4.7×10^{-4}%	28.67	1.35×10^{-2}
Korea, Rep.	3.8×10^{-4}%	17.18	6.46×10^{-3}
Turkey	2.8×10^{-4}%	17.02	4.80×10^{-3}
Venezuela	2.4×10^{-4}%	1.50	3.53×10^{-4}
Colombia	2.3×10^{-4}%	4.83	1.11×10^{-3}
Ecuador	2.2×10^{-4}%	1.85	4.06×10^{-4}
Australia	1.9×10^{-4}%	2.59	4.86×10^{-4}
Thailand	1.6×10^{-4}%	14.00	2.19×10^{-3}
Guatemala	1.1×10^{-4}%	12.34	1.39×10^{-3}
Egypt	6.3×10^{-5}%	14.28	8.95×10^{-4}
Bosnia	2.2×10^{-5}%	28.42	6.24×10^{-4}
Nicaragua	2.2×10^{-5}%	10.32	2.26×10^{-4}
Taiwan	1.9×10^{-5}%	14.02	2.64×10^{-4}
Peru	9.4×10^{-6}%	8.11	7.63×10^{-5}
Switzerland	9.4×10^{-6}%	18.67	1.76×10^{-4}
Pakistan	6.3×10^{-6}%	17.57	1.10×10^{-4}
Syrian Arab.Rep.	6.3×10^{-6}%	21.99	1.38×10^{-4}
Philippines	3.1×10^{-6}%	9.80	3.07×10^{-5}
	$\Sigma = 11.04$		

Table S29. Data for determination of environmental compliance of rapeseed producing countries: countries, global production share and Environmental Performance Indicators (EPI).

Rapeseed Producing Countries	Global Production Share (gps) Based on Barrientos and Soria (2016)	EPI Based on Yale Center for Environmental Law & Policy (2014)	Global Production Share × EPI
Canada	32%	8.42	2.71
China	25%	7.59	1.93
India	21%	6.89	1.43
Australia	9%	1.34	1.18×10^{-1}
France	8.07%	15.26	1.23
Germany	7.12%	8.67	6.17×10^{-1}
Ukraine	5%	10.07	4.61×10^{-1}
UK	3.79%	10.62	4.02×10^{-1}
USA	3%	6.75	1.70×10^{-1}
Poland	2.74%	11.14	3.05×10^{-1}
Russia	2%	11.70	2.26×10^{-1}
Belarus	2%	6.54	1.09×10^{-1}
Czech Rep.	1.64%	18.83	3.08×10^{-1}
Lithuania	0.93%	9.89	9.25×10^{-2}
Denmark	0.72%	15.82	1.13×10^{-1}
Hungary	0.59%	25.71	1.52×10^{-1}
Sweden	0.48%	14.83	7.08×10^{-2}
Latvia	0.45%	28.41	1.28×10^{-1}
Bulgaria	0.40%	18.01	7.21×10^{-2}
Slovakia	0.31%	19.89	6.26×10^{-2}
Estonia	0.24%	8.67	2.09×10^{-2}
Romania	0.24%	14.18	3.33×10^{-2}
Austria	0.22%	19.64	4.32×10^{-2}
Finland	0.11%	15.14	1.63×10^{-2}
Spain	0.08%	16.52	1.27×10^{-2}
Belgium	0.07%	20.03	1.42×10^{-2}
Italy	0.04%	3.05	1.13×10^{-3}
Ireland	0.04%	24.08	9.96×10^{-3}
Slovenia	0.03%	18.67	4.69×10^{-3}
Luxembourg	0.02%	18.67	4.13×10^{-3}
Netherlands	0.01%	28.67	2.96×10^{-3}
Greece	0.01%	11.66	8.61×10^{-4}
Bangladesh	4×10^{-3}%	14.45	6.40×10^{-2}
Kazakhstan	3×10^{-3}%	26.40	8.97×10^{-2}
Chile	3×10^{-3}%	8.89	2.95×10^{-2}
Pakistan	3×10^{-3}%	14.06	4.15×10^{-2}
Ethiopia	2×10^{-3}%	6.38	1.51×10^{-2}
Switzerland	1×10^{-3}%	16.10	1.90×10^{-2}
Turkey	1×10^{-3}%	14.80	1.51×10^{-2}
Paraguay	9×10^{-4}%	19.58	1.73×10^{-2}
Norway	7×10^{-4}%	3.56	2.42×10^{-3}
Japan	2×10^{-4}%	1.49	2.42×10^{-4}
Korea, Rep.	3×10^{-5}%	8.59	2.54×10^{-4}
Morocco	1×10^{-5}%	20.93	3.09×10^{-4}
$\Sigma = 11.17$			

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3.3 Consistent assessment of abiotic and biotic resources and raw materials

This chapter contains the following publication:

V. Bach, M. Berger, S. Forin, M. Finkbeiner (2017): **Comprehensive approach for evaluating different resource types –** Case study of abiotic and biotic resource use assessment methodologies**,** Ecological Indicators 87C (2018) pp. 314-322 (postprint)

DOI: 10.1016/j.ecolind.2017.12.049

The publication introduces an 8-step approach presented as a user-friendly flow chart to combine methodologies addressing different resource types. First, shortcomings of existing methods are introduced (contribution to research target *3a*). Next, the proposed approach is outlined and related steps are explained in detail (contribution to research target *3b*). Further, the proposed approach is applied to combine ESSENZ and BIRD establishing a combined methodology to assess abiotic and biotic resource and raw material use on product level in a consistent way (contribution to research target *3c*). Finally, the combined methodology is applied to a case study of shelves (contribution to research target *3d*; as well as *1d and 2d*).

Comprehensive approach for evaluating different resource types - Case study of abiotic and biotic resource use assessment methodologies

Vanessa Bach[1], Markus Berger[1], Silvia Forin[1] and Matthias Finkbeiner[1]

[1] Technische Universität Berlin, Chair of Sustainable Engineering, Straße des 17. Juni 135, 10623 Berlin, Germany

* Corresponding author: vanessa.bach@tu-berlin.de

Abstract

Due to steadily increasing resource demand and accompanying raising public awareness, a variety of assessment methodologies evaluating resource use and its consequences were published in the last years. Existing methodologies are typically developed considering the specific characteristics of one particular resource type and as consequence are not suitable for cross-cutting assessment of different resource types. This paper proposes an 3-step approach for combining different resource use assessment methodologies allowing for a consistent assessment of product systems using different resource types. The first steps evaluate if the considered dimensions, categories, indicators, indicator models and underlying data are consistent. When this is the case, they can be included in the combined methodology without further adjustments. Differences are identified simultaneously and addressed in the subsequent steps. Within the steps guidance is provided on how the dimensions, categories and indicators of the methodologies can be adjusted to fit in the combined methodology. In a case study the proposed approach is applied to two methodologies developed by the authors assessing abiotic (ESSENZ method) and biotic resources and raw materials (BIRD method). The ESSENZ method consists of four dimensions, which are quantified by overall 21 categories and indicators. The BIRD method takes into account five dimensions and 24 corresponding categories and indicators. As none of the considered dimension of the two methodologies match, comparison of the considered resource types as well as application in a case study is not possible. By applying the proposed approach all five dimensions and 25 of the overall 27 categories and indicators can be integrated in the combined approach for a consistent assessment of abiotic and biotic resources and raw materials. The obtained combined methodology is then applied to three shelves made out of metal, wood and plastic. It could be shown that the introduced approach provides meaningful guidance on how to combine different resource use assessment methodologies and increases the findings gained from a combined and consistent assessment.

Keywords

resource assessment methodology, resource use, life cycle impact assessment, ESSENZ, BIRD

Highlights

- The proposed approach provides guidance on how different resource assessment methodologies can be combined.
- The methodologies ESSENZ (for abiotic resources) and BIRD (for biotic resources) are combined successfully.
- Applying a combined methodology allows for a consistent assessment of different resource types.

1. Introduction

Continuing global industrial and technological development has steadily increased the demand for resources. Their use has therefore been a topic of discussion throughout the last decades with regard to competition on resources (availability of resources and raw materials as well as related vulnerabilities of companies and countries) and corresponding environmental (e. g. climate change) as well as social aspects (e. g. working conditions). With that also the need to assess resource use and its related impacts has been growing. This led to the publication of a variety of assessment methodologies for evaluating resource use (mostly for abiotic resources and raw materials (e. g. Schneider et al. (2016), van Oers and Guinée (2016) and Berger and Sonderegger (2017)) as well as water (e. g. Pfister et al. (2009), Berger et al. (2014) and Núñez et al. (2016)), but also for biotic resources and raw materials (e. g. Oakdene Hollins(2014) and Bach et al. (2017)) as well as land (e. g. Beck et al. (2010) and Koellner et al. (2013)). So far, almost all of these methodologies are developed explicitly for one type of resource only.

The term resource refers to entities, which can be extracted from nature and transferred to the anthroposphere. This includes abiotic and biotic resources, abiotic and biotic raw materials as well as water, land, and the natural environment (European Commission 2005; Schneider et al. 2016; Sonderegger et al. 2017). Based on the specific characteristics of the considered resource or raw material, relevant aspects (and corresponding indicators) are defined. For instance, availability constraints are mostly associated with abiotic resources and raw materials (Dewulf et al. 2016), renewability rates with biotic resources (Crenna and Sala 2017), scarcity for water (Pfister et al. 2017), etc. The development of methodologies for specific resource types allows accounting for relevant aspects in a consistent way. However, these methods do not allow for the assessment of any other resource types besides the one they are developed for. Some of the specific dimensions, categories and indicators applied cannot be transferred to other resource types and/or are not valid for these resources. This makes it challenging to assess different resource types in a consistent way. Following challenges occur in current assessments of several resource types:

• Dimension(s) and corresponding categories and indicators considered represent only the intersecting set (and therefore often only a small amount of dimensions, categories and indicators) and thus do not comprehensively reflect all aspects of resource use and its related implications (e. g. as shown by Ritthoff et al. (2002); Zabalza Bribián et al. (2011), Alvarenga et al. (2013) and Klinglmair et al. (2014)). For example, even though biofuels are made from renewable resources, which cannot be consumed in the same way as fossil fuels can, their use is limited by land and phosphorus availability (Hein and Leemans 2012; Rulli et al. 2016; March et al. 2016). However, currently phosphorus and land use are often not addressed in the assessment of biotic materials (Rack et al. 2013; Finkbeiner et al. 2014; Mousavi-Avval et al. 2017). An adequate assessment of different resource types together can therefore not be achieved.

• Inadequate comparison of dimensions, categories and indicators occurs, when different aspects of the various resource types are addressed. For example: Acidification can be determined with different indicators (using different models and underlying data). Thus, even though results are provided for the category acidification within two methodologies, these results cannot be compared due to the differences in the applied indicators. If there are compared anyways, the comparison is inadequate.

This methodological gap is addressed in this paper, which has the aim of providing guidance for combining different resource use assessment methodologies and thus achieving a more comprehensive assessment and adequate comparison of different resource types.

In the next section, the proposed approach including a user-friendly flow chart for easy application is introduced (section 2). Next, the approach is applied to a case study (section 3), where two methodologies developed by the authors (one for the assessment of abiotic resources and raw materials and one for biotic ones) are combined according to the proposed approach. The combined methodology is then applied for an exemplary product system considering three sorts of shelves (made out of wood, plastic and metal). Further, challenges of the proposed approach are discussed (section 4) and conclusions are drawn (section 5).

2. Method

In this section the proposed approach to combine methodologies evaluating different resource types is introduced. To apply the proposed approach, it is assumed that the practitioner is familiar with the methodologies and aware of their shortcomings. The proposed approach can be applied for methodologies assessing resource use, independently from the number of aspects and indicators considered. However, as the goal of the approach is to combine multi indicator methodologies, the focus is on methodologies taking into account several categories and indicators (e. g. Graedel et al. (2012), European Commission (2014) and Bach et al. (2016a)) instead of methodologies considering only one or few indicators (e. g. Oers et al. (2002), Valero et al. (2014) and Finnveden et al. (2016)). As indicators are designed differently depending on the level considered (micro (product), meso (company) or macro (company) level), methodologies can only be combined when they address the same level. The approach can be applied to combine two or more methodologies. For the sake of simplicity, this paper describes the combination of two methodologies.

The introduced approach consists of three steps, which guide the user to identify equal and different dimensions, categories and indicators applied within the considered methodologies. Step 1) and 2) have three possible outcomes: i) comparison of dimensions, categories and/or indicators is possible; ii) comparison is possible, because dimensions, categories and/or indicators can be rearranged, renamed and/or (re) calculated; iii) comparison is not possible, because rearranging, renaming and/or (re) calculating is not feasible and the user is guided to a subsequent step 3). Within this step, dimensions, categories and indicators are addressed, which could not be matched within step 1) and 2).

For non-experts in the field of life cycle assessment, life cycle impact assessment and sustainability assessment, a more detailed procedure is provided in the supplementary materials, including a user-friendly flow chart. Within the detailed approach the same principles are considered as within the 3-step approach, but are broken down to a more detailed level (section 4 Supplementary Materials).

In the following, the 3-step approach is introduced and described in detail:

1) Check if the considered dimensions address the same aspects and categories

a. Yes, comparison is possible without adaptation

b. No, dimensions do not consider the same aspects and categories, but can be rearranged and/or renamed

c. No, dimensions do not consider the same aspects and categories and cannot be rearranged and/or renamed → they need to be further analyzed in step 3

2) Check if indicators, their models and underlying data are comparable

a. Yes, comparison is possible without adaptation

b. No, indicators, their models and underlying data are not comparable, but can be renamed and/or (re) calculated

c. No, indicators, their models and underlying data are not comparable and cannot be rearranged and/or (re) calculated → they need to be further analyzed in step 3

3) Check if missing categories and corresponding indicators are relevant for the evaluated resource and raw material

a. No → set indicator value to zero

b. Yes, categories are relevant and findings of other studies can be used to determine results of missing categories

c. Yes, but applying results of other studies is not possible → categories have to be excluded from the combined approach

In step 1) the user determines if the considered dimensions of the methodologies are equivalent by comparing their naming as well as addressed aspects and categories. Possible dimensions could be the classical sustainability dimension: environmental, economic and social (Giddings et al. 2002) as well newly developed dimensions like criticality (Sonnemann et al. 2015). If the same dimensions and categories are addressed, they can be considered in the combined approach without adaptation.

However, when the methodologies do not consider the same dimensions and categories, it has to be analyzed if they can be rearranged and/or renamed. For example: human health impacts are often considered as part of the environmental dimension, because impacts are determined as part of an Life Cycle Assessment (LCA) case study. Sometimes though these categories are seen as part of human well-being and are placed in the social dimension. These dimensions can be rearranged and renamed to be combined in a consistent way by shifting the assessment of human health impacts from the environmental dimension to the social dimension. Before dimensions and categories are renamed and rearranged it should be decided which of the considered methodologies is selected as the standard framework. When different categories address the same aspect, they can be renamed and/or rearranged accordingly as well. For example: categories named company concentration and producer diversity address the same aspect (concentration of raw material producing companies) (Achzet and Helbig 2013) and thus could be renamed by introducing a new terminology or by renaming one of the categories accordingly. If it is not possible to rearrange and/or rename the categories and dimensions to achieve comparison, they need to be further analyzed in step 3).

In step 2) the user checks if indicators, their models and underlying data are equivalent, which is only the case when the indicators as such as well as the applied models and underlying data match. Thus, the applied indicators also have to be compared across categories (and dimensions) in case the same indicator is used within different categories (and dimensions). If the same indicators, models and underlying data are applied, they can be considered in the combined approach without adaptation. However, methodologies often apply different indicators, models and underlying data for the same categories (and therefore for the corresponding dimensions). Several options exist with regard to differences of indicators, models and data:

i) Indicators are different. For example: To assess acidification the CML-IA (Impact Assessment methodology by Institute of Environmental Sciences – Universiteit Leiden) (Guinée et al. 2002) method applies the category indicator hydrogen ion release , whereas the method developed by Seppälä et al. (2006) uses cumulative exceedance as the category indicator. In this case, the indicator results cannot be compared directly, because the quantitative results have different units and ranges.

ii) Indicators are equal, but are based on different models. For example: the category indicator hydrogen ion release is applied to assess the category acidification by the methodologies TRACI (Tool for the Reduction and Assessment of Chemical and other environmental Impacts) (Bare 2002) as well as by CML-IA (Guinée et al. 2002). However, the underlying models are different. TRACI considers emission release and distribution within the US, whereas CML-IA applies European release and distribution pathways. In this case, the indicator results cannot be compared, because the reference frameworks differ.

iii) Indicators as well as models are equal, but underlying data varies. For example: To determine the category political stability (dimension socio-economic availability) of different resource types, different global production data has to be applied. For abiotic resources and raw materials data provided by United States Geological Survey (USGS) (2015) is used, whereas for biotic raw resources and raw materials data provided by Food and Agriculture Organization of the United Nations (FAO) (2015) is applied. When underlying data is resource and raw materials specific, indicator results cannot be compared. Resource and raw material specific data refers to underlying data, which differs because different resource types (e.g. abiotic and biotic resources) are considered. Is the data not resource and raw material specific, a comparison is possible. Underlying data is not specific for a resource type when it is not influenced by the resource type (e.g. Worldwide Governance Indicators (Kaufmann et al. 2011; World Bank Group 2013) to determine political stability). When underlying data is specific for the considered resources and raw materials of both methods, application of different data sources still facilitates adequate comparison and the indicators can be included in the combined methodology. Otherwise, comparison is not possible.

iv) When applying different methodologies some of the addressed categories are not considered in one of the methodologies. Thus, no indicator is provided for quantification. In this case, it has to be decided whether the indicator provided by the other methodology can be applied.

When indicators, models and data differ and comparison is not possible, it has to be determined if the applied indicators, models and data can (re)calculated. (Re)calculation is possible, when the applied indicator, model and data are not resource specific. If model and indicator are resource and raw material specific, (re) calculation is not possible. For example: abiotic resource depletion cannot be determined by the indicator measuring biotic resource depletion and vice versa, because the considered parameter (e.g. dissipation, regeneration, etc.) are not applicable. If models and indicators are not resource and raw material specific, the indicator results can be re (calculated). For example: for the category acidification: resource use specific acidification impacts are determined by considering specific inventory data, but the underlying model does not take into account which resource type is evaluated. If rearrangement and (re)calculation are not possible, indicators, models and data have to be further analyzed in step 3.

The following paragraphs provide guidance on which indicators, models and underlying data to choose, when indicator results can be (re) calculated. If both methodologies provide approaches for a category, the user has to decide which of the approaches should be used in the combined methodology. Approaches which have been applied for some years (which is often the case for methods applied in LCA case studies), are assessed in several publications with regard to their maturity (e.g. (European Commission-Joint Research Centre 2011; Rack et al. 2013; Klinglmair et al. 2014; Lehmann et al. 2015)). For determining the maturity of newly developed and established methods, it is referred to the publication by Bach and Finkbeiner (2016). Further, approaches addressing categories and dimensions, which so far have not been included in assessment methodologies, are most likely not evaluated so far (e.g. this is the case for approaches assessing socio-economic availability). Thus, an evaluation of the approaches has to be carried out by the user. The evaluation schemes provided in the above mentioned publications can be used as a basis to assess the newly developed approaches.

Within step 3), dimensions, categories and indicators, which could not be rearranged, renamed or (re) calculated in steps 1) and 2) are further analyzed. First, it is determined if the category is relevant. If the category is not relevant, it can be set to zero. For example: the regeneration rate is a relevant aspect for biotic resources but not for abiotic ones, because abiotic resources are formed over very long geological periods. Therefore, the indicator regeneration rate is set to zero for abiotic resources.

There is no straightforward approach for determine if a category is relevant or not. However, following criteria should be taken into account: i) if the aspect (which the category represents) was/is addressed in past and ongoing discussions within society and politics; ii) if the topic is addressed in scientific publications, iii) if it is based on expert judgment. Is the topic not addressed within societal and political discussions or within scientific publications, experts should be consulted to make sure that the decision to set a category as not relevant is thoroughly reviewed. Mistakes can lead to an inadequate comparison of resource types.

If the category is identified as relevant, findings of other studies can be used to determine results of missing categories. Practitioners should keep in mind that findings from other studies might not consider the same system boundaries, functions, etc. Due to simplifications and assumptions, results might have higher uncertainties, which have to be taken into account. Thus, a thorough analysis is important before using the findings for comparing resources and raw materials. When findings are applicable the results can be included in the overall results. If the category is determined as relevant, but findings of other studies are not available or cannot be applied, the category has to be excluded from the combined methodology. This option however should be avoided and significant effort should be made to include the broadest possible range of categories.

3. Case study

In this section the proposed approach is applied to the case study of two methodologies developed by the authors: the integrated methodology to assess resource efficiency (referred to as ESSENZ) for abiotic resources and raw materials (Bach et al. 2016) and the methodology assessing the availability of terrestrial biotic materials in product systems (referred to as BIRD) (Bach et al. 2017).

Both methodologies are shortly introduced in the following. These sections can be skipped by readers familiar with these methods. Further, the implementation of the approach is described in detail for better illustration. The combined methodology is then applied for the case study of three shelves made out of metal, wood and plastic.

3.1 ESSENZ

ESSENZ has been developed in cooperation with key players of the European industry (Daimler, Evonik, Knauer, ThyssenKrupp, German Copper Institute (Deutsches Kupferinstitut) and Siemens) to comprehensively measure resource efficiency of products made out of abiotic resources and raw materials (Bach et al. 2016a; Bach et al. 2016b). The aim of the methodology is to enhance the applicability of resource efficiency as well as assess resources in the context of sustainable development. Therefore all three sustainability dimensions (economic, environmental and social) are considered. As the availability of resources and raw materials is a precondition for economic development, the economic dimension is expressed through security of resource supply. ESSENZ takes into account physical as well as socio-economic constraints. The physical availability refers to resources in the earth crusts as well as anthropogenic stocks (e. g. copper cables in dump sites or buildings), whereas the socio-economic availability is influenced by aspects inhibiting the supply security of resources throughout the supply chain (e. g. trade barriers can prevent the import of necessary materials). To determine the sub dimension physical availability the two indicators abiotic resource depletion (Guinée et al. 1993; Oers et al. 2002) and anthropogenic stock extended abiotic depletion potential (Schneider et al. 2011; Schneider et al. 2015) are applied. The socio-economic availability is addressed by eleven categories and corresponding indicators: Concentration of reserves and production as well as company concentration (Rhoades 1993), demand growth and mining capacity established based on data from (British Geological Survey (BGS) 2014; USGS 2015), feasibility of exploration projects (Cervantes et al. 2013), occurrence as co-product (Angerer et al. 2009), trade barriers (Hanouz et al. 2014), political stability (World Bank Group 2013), primary material use (Graedel 2011) and price fluctuation (Federal Institute for Geosciences and Natural Resources 2014). To determine the characterization factors the distance-to-target approach (Müller-Wenk et al. 1990; Frischknecht et al. 2009) is applied, which sets indicator values in relation to a target value to determine whether the considered materials have a potential supply restriction (which is the case when the indicator value exceeds the target value). Further, environmental impacts related to the extraction and processing of resources and raw materials as well as for the entire product life cycle are assessed applying well established impact assessment methodologies for the categories climate change (Intergovernmental Panel on Climate Change (IPCC) 2007), acidification (Hauschild and Wenzel 1998; Huijbregts 1999), eutrophication (Heijungs et al. 1992), smog (Guinée et al. 2002) and ozone depletion (World Meteorological Organization (WMO) 2010). The social dimension is taken into account by determining aspects considered as not acceptable by society. Thus, the corresponding dimension is referred to as societal acceptance. Within ESSENZ social aspects not accepted by society are identified as child labor, forced labor and materials originating in high conflict zones (Norris et al. 2013) and are expressed in the category compliance with social standards. As not only the violation of social but also environmental standards can lead to consumer's boycotting products and companies, the category compliance with environmental standards is added to the dimension as well. Within this category country specific laws and regulations for conservation areas are considered (Yale Center for Environmental Law & Policy 2014), assuming that a country with better regulations also has fewer violations of environmental standards. Indicators applied to quantify the categories and dimensions considered in ESSENZ are shown in the supplementary material – section 1. Characterization factors for the overall 21 indicators are provided for 40 metals and fossil raw materials by Bach et al. (2016a).

3.2 BIRD

BIRD assesses potential restrictions to availability of terrestrial biotic materials and resources for product systems along the supply chain (Bach et al. 2017). Overall the five dimensions physical, socio-economic, abiotic, social and environmental constraints are taken into account. Physical constraints refer to biotic materials extracted from the natural environment (this category is only established for resources) based on Heijungs et al. (1992) and Sas (1997), extraction from the man-made environment as well as the anthroposphere (Bach et al. 2017). The characterization factors of the socio-economic constraints are established based on the distance-to-target approach (Müller-Wenk et al. 1990; Frischknecht et al. 2009), quantifying the following ten categories: concentration of resources and of harvesting as well as company concentration (Rhoades 1993), political instability (World Bank Group 2013), trade barriers (Hanouz et al. 2014), price fluctuations (Barrientos and Soria 2016), storage complexity (Organisation for Economic Cooperation and Development 2016), occurrence as co-product as well as recycling and demand growth according to the same calculation principle as applied in ESSENZ (Bach et al. 2016a). Further, phosphorus, land and water as well as natural disasters (United Nations Office for Disaster Risk Reduction 2013; Berger et al. 2014) reduce the occurrence of species used as biotic materials and are considered within the dimension abiotic constraints. The availability of phosphorus and land is only indirectly determined by taking into account the amount of phosphorus and the square meters of land used per functional unit. Limited availability of resources and raw materials can also be caused by social constraints, i.e. compliance with social standards (Norris et al. 2013) and environmental standards (Yale Center for Environmental Law & Policy 2014) as well as food security (FAO 2016). Further, environmental aspects can reduce the availability of species used as biotic materials. Within BIRD the categories climate change (IPCC 2007), acidification (Hauschild and Wenzel 1998; Huijbregts 1999), eutrophication (Heijungs et al. 1992a), ozone depletion (WMO 2010) and smog (Guinée et al. 2002) are considered. Indicators applied to quantify the categories and dimensions are shown in the supplementary material – section 2.

3.3 Case study: combining ESSENZ and BIRD

Following it is described in detail how the three steps of the proposed approach are applied for the combination of ESSENZ and BIRD. The combined methodology is shown in Figure 1. Dimensions and categories are visualized with black frames for ESSENZ and with grey frames for BIRD.

The proposed approach starts with step 1), where dimensions and their corresponding aspects and categories of both methodologies are compared to identify which of them match. The considered dimensions do not match regarding terminology (all considered dimensions within ESSENZ and BIRD are referred to differently). Thus, a comparison of abiotic and biotic resources and raw materials applying ESSENZ and BIRD is not possible without further adapting both methodologies. However, as they partly address the same aspects, some of the dimensions and categories can be aligned through renaming and rearranging and be integrated into the combined methodology. Both methodologies consider the dimensions physical and socio-economic constraints/availability as well as environmental impacts/constraints. The dimensions are named slightly differently - constraints/availability - but refer to the same overall challenges and thus can be renamed. The structure of ESSENZ is used as the basis to establish a framework to combine both methodologies. ESSENZ has been chosen as it has been applied in several case studies already and can therefore be seen as more accepted as well as more mature. Within the dimension environmental impacts the same categories are taken into account. As the two categories compliance with social and environmental standards are named the same in both methodologies, the dimension social impacts (in BIRD) is renamed to societal acceptance (as in ESSENZ). BIRD additionally considers the category food security as part of the dimension social impacts, which is therefore also shifted to the dimension societal acceptance. Further, within the dimension socio-economic availability the three categories company concentration, demand growth and trade barriers match. Thus, overall eight categories can be included in the combined approach without adaptation. Several categories within the dimension socio-economic availability address the same aspect and can be renamed. This includes the categories price variation and price fluctuation (both address rapidly changing prices – renamed to price fluctuations), political instability and stability (renamed to political (in)stability), primary material use and recycling (both address the decreasing pressure on primary resources, when secondary resources are used – renamed to limitations due to recycling capacity), concentration of resources and reserves (both consider the concentration of the resource within nature; the term resources is applied for biotic resources, whereas the occurrence of abiotic resources in nature is referred to as reserves – renamed to concentration of resources/reserves), occurrence of co-products and occurrence as co-product (both refer to possible availability constraints due to being a co-product – renamed to occurrence as co-products) as well as concentration of mine production and concentration of harvesting. The last two categories refer to different activities (mining and harvesting), but both address the concentration of production sites. Thus, a new name is defined (concentration of production), because none of the existing names is adequate to reflect both aspects. After the renaming of the categories, another six categories (additionally to the eight identified before) can be included in the combined approach. The dimension abiotic constraints considered in BIRD does not match any dimension addressed in ESSENZ and cannot be included in the combined approach at this point. It is further analyzed in step 3).

Within step 2) the indicators, their models and underlying data are compared. For the following categories the same indicators, models and underlying data are applied: all categories of the dimension environmental impacts as well as for the categories company concentration, demand growth, trade barriers, price fluctuation, political (in)stability, limitations due to recycling capacity, concentration of resources/reserves, occurrence as co-products and concentration of production within the dimension socio-economic availability. For the indicators of the dimension socio-economic availability some of the used data are equal (underlying indicators, e.g. Worldwide Governance Indicators (Kaufmann et al. 2011; World Bank Group 2013)) and some are different (source of the global production shares). For the calculation of the indicator results for the dimension socio-economic availability, data sources are resource and raw material specific. Production shares of abiotic materials are taken from USGS (2015) and BGS (2014), whereas for biotic raw materials various data sources addressing biotic materials e.g. FAO (2015) are used. Indicators for the categories compliance with social and environmental standards can be recalculated by adapting the model provided in BIRD according to the model applied in ESSENZ. For the category storage complexity as well as the categories natural disasters, land and water availability, indicators can be calculated for abiotic resources according to BIRD.

Within Step 3) the relevance of the categories that could not be included in the combined methodology in step 1) and 2) is determined. Both methodologies address categories, which are not covered by the other methodology. Categories addressed in ESSENZ but not in BIRD are mining capacity and feasibility of exploration projects. Both can be set to zero:

O Mining capacity: As biotic resources and raw materials can be replenished, their availability is not limited by the extraction and reserve rate used to determine mining capacity.

O Feasibility of exploration projects: Cultivation of agricultural and forestry products is easier to establish than mining activities.

Categories addressed by BIRD but not by ESSENZ are replenishment rate of the dimension physical availability, storage complexity of the dimension socio-economic availability, all categories of the dimension abiotic constraints as well as the category food security of the dimension societal acceptance (originally social constraints). Following categories are not relevant for the assessment of abiotic resources and can be set to zero:

O Replenishment rate: Abiotic resources are formed over very long geological periods and are therefore not renewable.

O Food security: Abiotic resources cannot be used as food and are therefore not in direct competition of use. Indirect influences such as land use or impacts on the environment are considered in the other dimension.

O Phosphorus availability: Phosphorus itself is an abiotic resource, but other abiotic resources and raw materials are not influenced by its availability.

Next, it is analyzed if categories considered relevant, can be determined considering findings of other studies. This applies to the categories physical and anthropogenic availability. Here not the methodologies as such, but the case study for which the combined methodology shall be applied for, is the basis for completing the step. The case study chosen here (comparison of shelves made out of steel, polypropylene (PP) and oak wood) is introduced later, but results are addressed here for the sake of completeness. Thus, research is carried out with regard to existing studies addressing the physical and anthropogenic availability of the materials steel, PP and oak wood. Such studies do not exist. Thus, these categories and indicators can be included in the combined approach.

The advantage of applying the proposed approach is displayed in Figure 2, where the number of dimensions, categories and indicators considered in ESSENZ and BIRD individually as well as within the ideal combination of ESSENZ and BIRD, the combination without the proposed approach and within the combined methodology applying the proposed approach are shown. ESSENZ consists of four dimensions and 21 categories and related indicators. BIRD considers five dimensions and 25 categories and corresponding indicators. In the ideal case, where all dimensions, categories and indicators considered in ESSENZ and BIRD can be combined, results for overall five dimensions with 27 categories and related indicators could be determined.

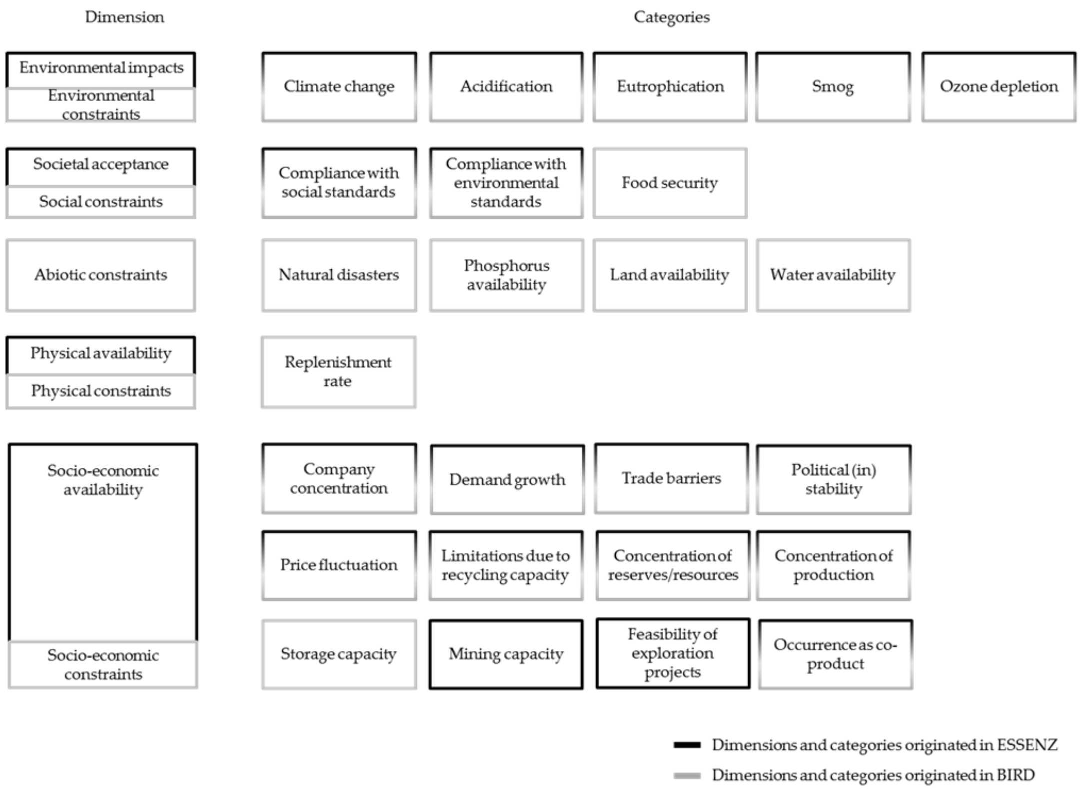

Figure 1: Combined methodology applying the proposed approach: dimensions and categories are highlighted by black frames when considered in ESSENZ and by grey frames for BIRD

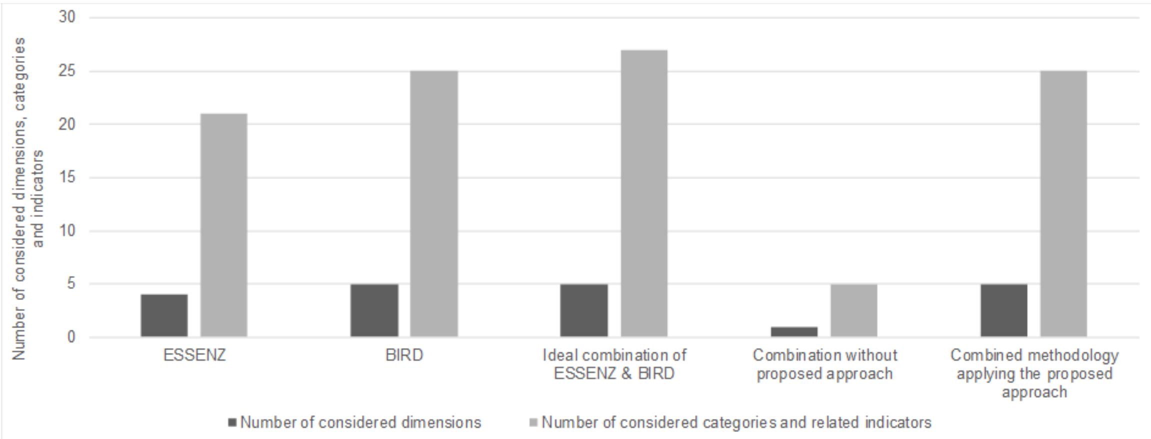

Figure 2: Overview of dimensions, categories and indicators considered in ESSENZ and BIRD as well as within the ideal combination of ESSENZ and BIRD, the combination without the proposed approach and within the combined methodology applying the combined approach

Is the proposed approach not applied only one dimension with five categories and corresponding indicators can be identified as equal – the environmental impacts. By applying the proposed approach to join ESSENZ and BIRD the combined methodology consists of overall five dimensions composed of 25 categories and indicators. Only the two categories resource depletion and anthropogenic availability with their corresponding indicators cannot be included due to different and not transferrable indicator models. Thus, applying the proposed approach allows determining results for a significantly higher number of dimensions, categories and indicators.

3.4 Application of combined approach

Next, the combined methodology is applied for the case study of three shelves. Shelves with the same function (storage of undefined items with a carrying capacity of 30kg) made out of the three different materials steel (10.7 kg), PP (5.8 kg) and oak (38 kg) are considered. The masses of the shelves are assumed based on data provided by Inter IKEA Systems B.V. (2017). For all three shelves only the production phase is taken into account (excluding use and end of life phase), which were modeled in GaBi (Thinkstep 2016), using GaBi and ecoinvent data (Ecoinvent 2016). Further, several additional simplifications were made with regard to the system boundaries as the purpose of this case study is to demonstrate the applicability of the combined approach only and not to provide reliable results.

The combined methodology consists of five dimensions with overall 25 categories and indicators. A direct comparison of the results can only be carried out individually for every category. As the categories have different units, comparison across categories is not possible (results for each category are displayed in the supplementary materials – section 3). Thus, a proportional assessment is used to compare the results of all categories and dimensions for the three shelves (see Figure 3). The proportional comparison is determined by setting the highest result within each category to 100% and determining the percentage share of the other results of the categories.

As mentioned before the purpose of the case study is to demonstrate the applicability of the combined approach. Thus, when reflecting upon the results it should be kept in mind that several assumptions were made. Further, as no weighting factors are applied, neither the importance of the individual categories can be assessed, nor a single score result can be determined. An imbalance of the results also occurs as more categories for the dimension socio-economic availability are taken into account than for other dimensions. For a better understanding of the individual categories, please see the publications of ESSENZ (Bach et al. 2016) and BIRD (Bach et al. 2017).

Further, the categories resource depletion and anthropogenic availability could not be displayed in Figure 3, because the indicators of the categories are not transferable (as shown in the results derived by applying the approach). However, as especially these two categories are of importance for plastic and its fossil raw materials, the overall comparison of the tree shelves is limited. Due to the importance of the categories resource depletion and anthropogenic availability the results are still displayed in Table 1 for all three shelves. However, it is pointed out that the results cannot be compared, because the indicators are not equal.

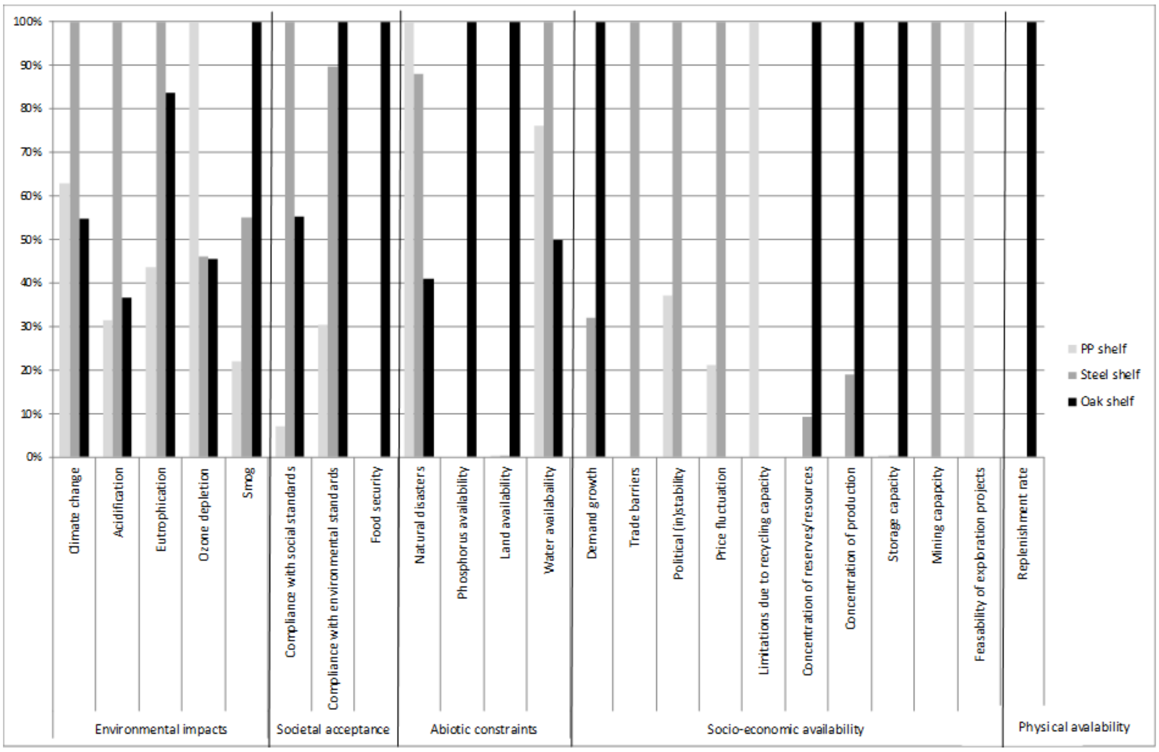

Figure 3: Proportional comparison of results of case study - shelves made out of polypropylene (PP), steel and oak

Dimension	Category	PP shelf	Steel shelf	Oak shelf
Physical availability	Resource depletion – metals [kg Sb – eqv.]	n.t.	5.61E-07	n.t.
	Resource depletion – biotic [kg]	n.t.	n.t.	1.03E-12
	resource depletion – fossils [MJ]	245.51	n.t.	n.t.
	Anthropogenic availability [kg Sb-eqv.]	n.t.	8.51E-04	n.t.

4. Discussion

The aim of the proposed approach is to provide guidance on how different methodologies can be combined to complement each other to accomplish an adequate evaluation of different resource types. Related challenges are addressed in the following.

For experts in the field of life cycle assessment, life cycle impact assessment or sustainability assessment, some of the phases of the proposed approach might seem trivial. However, for users with less experience in this field, more detailed guidance can be useful. If no guidance is provided on how to combine two methodologies correctly (e.g. making sure that indicator models and underlying data match), results of non-comparable indicators might be equated and wrong conclusions are derived. Further, if users only compare dimensions and categories with the same name the intersecting set of indicators might be very small. Challenges can arise when the methodology is not documented sufficiently. In fact, many methodologies provide accurate information with regard to the indicator itself but the underlying data cannot be accessed (e. g. the EUTREND model used for calculating characterization factors for the category acidification in ReCiPe (Huijbregts et al. 2017)), and therefore not all details are available.

Further, lack of understanding can lead to wrong decisions in the individual steps of the approach. This is especially significant when the category is classified as not relevant for the considered resource type. If the practitioner is not familiar with all aspects of a specific resource type, faulty evaluations in this step will lead to an inadequate combined methodology. Findings of other studies can be used to determine category results. However, applying these results correctly can be challenging, because parameters of the studies have to be thoroughly reviewed and interpreted appropriately.

The proposed approach is developed for the combination of methodologies assessing resource use of different resource types. The maturity of indicators has to be determined when two methodologies considering the same resource type are combined. However, establishing such an assessment method for indicator maturity is out of the scope of this paper, which only provides useful literature and first ideas on how to tackle such an assessment.

To guarantee adequate comparability, categories have to be excluded, when quantification is not possible. However, exclusion of categories always means that relevant aspects are not taken into account. As shown in the case study the exclusion of categories can be minimized when the other steps are thoroughly carried out.

The resources phosphorus, water and land can be evaluated as own resources, but are also considered within assessment methodologies for the evaluation of abiotic and biotic resources. Within the methodologies of the case study land as well as water availability are considered applying one indicator. In reality both resource types are more complex and can be assessed applying several indicators as shown for example in the publication by Sonderegger et al. (2015). The same applies to phosphorus, which is considered as one aspect for biotic resources, but is a resource itself with its own availability constraints and environmental implications.

Combining multi-indicator methodologies supports a more comprehensive assessment of resource use. However, the practitioner should be aware that this does not mean that all aspects relevant for the resource type are taken into account. Especially when only one indicator per methodology is established the practitioner has to be aware that several aspects relevant for the considered resource types are not addressed. However, depending on the goal of the case studies addressing only few indicators might be feasible.

5. Conclusion

The proposed approach provides guidance on how different methodologies can be combined to complement each other to accomplish the assessment of different resource types. It leads the user through three steps to adequately combine assessment methodologies for different resource types. Dimensions, categories, indicators, underlying models and data can be included in the combined approach when they fulfill the requirements of the individual steps.

A case study of two methodologies developed by the authors demonstrates the applicability of the approach. The individual methodologies cannot be used for the consistent assessment of different resources types, because none of the dimensions, categories and indicators of ESSENZ and BIRD match. After the proposed approach has been applied all five dimensions and 25 of the 27 considered categories and indicators can be integrated into the combined approach. The combined methodology can be applied to determine implications of resource use of different resource types. Further, the combined methodology is tested for a case study of three shelves (made out of oak, PP and steel). The results show an increased level of knowledge gained from the case study compared to the isolated application of the methods. Thus, the approach presented in this paper allows for a more comprehensive and meaningful assessment of resource use in life cycle assessment.

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Supplementary Materials

Comprehensive approach for evaluating different resource types - Case study of abiotic and biotic resource use assessment methodologies

Vanessa Bach[1], Markus Berger[1], Silvia Forin[1] and Matthias Finkbeiner[1]

[1] Technische Universität Berlin, Chair of Sustainable Engineering, Straße des 17. Juni 135, 10623 Berlin, Germany

* Corresponding author: vanessa.bach@tu-berlin.de

Table of contents

1. Dimensions, categories and indicators of ESSENZ

The dimensions, categories and indicators applied in the integrated method to assess resource efficiency (referred to as ESSENZ) (Bach et al. 2016) are shown in Table S.1.

Table S.1: Overview of dimensions, categories and indicators considered in ESSENZ

Dimension	Category	Category indicator
Physical availability	Abiotic resource depletion	Abiotic resource depletion indicator (Guinée et al. 1993; Oers et al. 2002) or anthropogenic stock extended abiotic depletion potential (Schneider et al. 2011; Schneider et al. 2015)
Socio economic availability	Company concentration	Herfindahl-Hirschman-Index (HHI) is calculated by squaring the market share of each company or country with regard to the production or reserves (Rhoades 1993)
	Concentration of reserves	
	Concentration of production	
	Mining capacity	Reserve-to-annual-production ratio (based on data from (United States Geological Survey 2015) and BGS (British Geological Survey 2014))
	Feasibility of exploration projects	Policy Potential Index (Cervantes et al. 2013)
	Occurrence as co-product	Percentage of production as companion metal (Angerer et al. 2009)
	Trade barriers	Enabling Trade Index (Hanouz et al. 2014)
	Political stability	World Governance Indicators (World Bank Group 2013)
	Demand growth	Percentage of annual growth based on past developments (based on data from BGS (British Geological Survey 2014))
	Primary material use	Percentage of new material content (Graedel 2011)
	Price fluctuation	Volatility (Federal Institute for Geosciences and Natural Resources 2014)
Societal acceptance	Compliance with social standards	Child labor, forces labor and high conflict zones (based on Norris et al. 2013)
	Compliance with environmental standards	Sub indicators of Environmental Performance Index (Yale Center for Environmental Law & Policy 2014)
Environmental impacts	Climate change	Increasing radiative forcing (Intergovernmental Panel on Climate Change 2007)
	Acidification	Proton release (Hauschild and Wenzel 1998; Huijbregts 1999)
	Eutrophication	Increased supply of nutrients (Heijungs et al. 1992)
	Ozone layer depletion	Release of chloride and bromide ions (World Meteorological Organization 2010)
	Smog	Tropospheric ozone formation (Guinée et al. 2002)

2. Dimensions, categories and indicators of BIRD

The dimensions, categories and indicators applied in the method assessing the availability of terrestrial biotic materials in product systems (referred to as BIRD) (Bach et al. 2017) are shown in Table S.2.

Table S.2: Overview of dimensions, categories and indicators considered in BIRD

Dimension	Category	Category indicator
Physical constraints	Biotic resource depletion	Biotic resource depletion (Heijungs et al. 1992; Sas 1997)
	Replenishment rate	Approach developed by (Bach et al. 2017)
	Anthropogenic availability	Approach developed by (Bach et al. 2017)
Socio economic constraints	Company concentration	Herfindahl-Hirschman-Index (HHI) is calculated by squaring the market share of each company or country with regard to the production or reserves (Rhoades 1993)
	Concentration of reserves	
	Concentration of harvesting	
	Political instability	World Governance Indicators (World Bank Group 2013)
	Trade barriers	Enabling Trade Index (Hanouz et al. 2014)
	Price fluctuations	Commodity price index by (Barrientos and Soria 2016)
	Storage complexity	Economic Vulnerability Indicator (Organisation for Economic Cooperation and Development 2016)
	Primary material use	According to the same principle as applied in ESSENZ (Bach et al. 2016)
	Occurrence as co-product	
	Demand growth	
Abiotic constraints	Phosphorus availability	Amount of phosphorus used
	Land availability	Square meters of land used
	Water availability	Water Depletion Index (Berger et al. 2014)
	Natural disasters	Droughts and floods (United Nations Office for Disaster Risk Reduction 2013)
Societal acceptance	Compliance with social standards	Child labor, forces labor and high conflict zones based on (Norris et al. 2013)
	Compliance with environmental standards	Sub indicators of Environmental Performance Index (Yale Center for Environmental Law & Policy 2014)
	Food security	Food security index (Food and Agriculture Organization of the United Nations 2016)
Environmental impacts	Climate change	Increasing radiative forcing (Intergovernmental Panel on Climate Change 2007)
	Acidification	Proton release (Hauschild and Wenzel 1998; Huijbregts 1999)
	Eutrophication	Increased supply of nutrients (Heijungs et al. 1992)
	Ozone layer depletion	Release of chloride and bromide ions (World Meteorological Organization 2010)
	Smog	Tropospheric ozone formation (Guinée et al. 2002)

3. Results of case study

The results for the three shelves analyzed in section 3 of the manuscript are shown in Table S.3.

Table S.3: Results of the three considered shelves (polypropylene, steel and oak)

Dimension	Category	Polypropylene shelf	Steel shelf	Oak shelf
Environmental dimension	Climate change [kg CO_2-eqv.]	1.75E+01	2.79E+01	1.53E+01
	Acidification [kg SO_2-eqv.]	2.97E-02	9.41E-02	3.45E-02
	Eutrophication [kg P-eqv.]	3.71E-03	8.50E-03	7.11E-03
	Ozone depletion [kg R11-eqv.]	1.32E-09	6.1E-10	6.02E-10
	Smog [kg Ethene-eqv.]	4.85E-03	1.21E-02	2.20E-02
Societal acceptance	Compliance with social standards [-]	1.64E+00	2.34E+01	1.29E+01
	Compliance with environmental standards [-]	3.02E+00	8.92E+00	9.94E+00
	Food security [-]	0.00E+00	0.00E+00	2.29E+01
Abiotic constraints	Natural disasters [-]	1.82E+02	1.60E+02	7.43E+01
	Phosphorus availability [kg]	0.00E+00	0.00E+00	5.30E+01
	Land availability [ha/year]	1.15E+00	9.22E-01	2.48E+02
	Water availability [$m^3_{depleted}/m^3_{consumed}$]	7.75E+01	1.02E+02	5.08E+01
Socio-economic availability	Company concentration [-]	5.20E+04	0.00E+00	n.a.
	Demand growth [-]	0.00E+00	8.00E+05	2.51E+06
	Trade barriers [-]	0.00E+00	3.08E+06	0.00E+00
	Political (in) stability [-]	1.29E+06	3.48E+06	0.00E+00
	Occurrence as/of co-product [-]	0.00E+00	0.00E+00	0.00E+00
	Price fluctuation (Variation) [-]	1.92E+06	8.98E+06	0.00E+00
	Limitations due to recycling capacity [-]	1.55E+06	0.00E+00	0.00E+00
	Concentration of reserves/resources [-]	0.00E+00	3.06E+05	3.26E+06
	Concentration of production [-]	0.00E+00	1.55E+06	8.09E+06
	Storage capacity [-]	2.38E+00	5.71E+00	6.46E+04
	Mining capacity [-]	0.00E+00	4.68E+04	0.00E+00
	Feasibility of exploration projects [-]	1.79E+06	0.00E+00	0.00E+00
Physical availability	Resource depletion – metals [kg Sb-eqv.]	n.t.	5.61E-07	n.t.
	Resource depletion – biotic [kg]	n.t.	n.t.	1.03E-12
	resource depletion – fossils [MJ]	2.46E+02	n.t.	n.t.
	Anthropogenic availability [kg Sb-eqv.]	5.80E-06	8.51E-04	n.t.
	Replenishment rate [-]	0.00E+00	0.00E+00	5.00E-01
n.t. – indicators are not transferable				

4. Detailed 8-step approach for non-experts

Following, additional to the 3-step approach provided within the paper, a more detailed 8-step approach is provided for non-experts in the field of life cycle assessment, life cycle impact assessment and sustainability assessment.

The approach is presented as a flow chart to enhance applicability (see Figure S1). Every step is phrased as a question, which can only be answered with “yes” or “no”. Depending on the answer, the user is guided to the subsequent step. If the question is answered with “yes”, the requirements formulated in this step are fulfilled (e. g. the same dimensions are taken into account). If the question is answered with “no”, the requirements are not met. The approach has to be carried out until one of the four ending points for all considered dimensions, categories and indicators is reached.

Following, the individual steps of the proposed approach are described in detail. First, the procedure is described for steps fulfilling the formulated requirements (questions are all answered with “yes”). In this case, the considered methodologies take into account equal dimensions, categories and indicators as well as underlying models and data. Next, the procedure is described when the formulated requirements are not fulfilled and the questions are therefore answered with “no”.

In step 1a the question “Are the same dimensions taken into account?” is addressed. When the same dimension are included (the question of step 1a can be answered with “yes”) the approach continues with step 2a, where the question “Are the same categories considered within equal dimensions?” is addressed. When the same categories are addressed, they are further analyzed in step 3a, where the question “Are the same indicators applied within equal categories?” is addressed. Are the same indicators used, it is evaluated next if the underlying models (step 4a: Are the same models applied for each indicator?) and data (step 5a: Are the same data applied for each indicator?) are consistent. For indicators that apply the same models and data and are further addressing the same categories and dimensions, one of the four ending points (“Dimensions, categories and indicators are included in the combined methodology”) is reached: these indicators can be used within the combined methodology without further adjustments.

Differences identified in the first step (1a) refer to discrepancies of the established dimensions (e. g. one methodology only considers availability of resources, whereas the other methodology also takes into account social implications). In this case, the approach continues with step 1b, where the question “Do different dimensions address the same aspects?” has to be answered. Is the answer “yes”, the dimensions and associated categories should be renamed and/or rearranged accordingly.

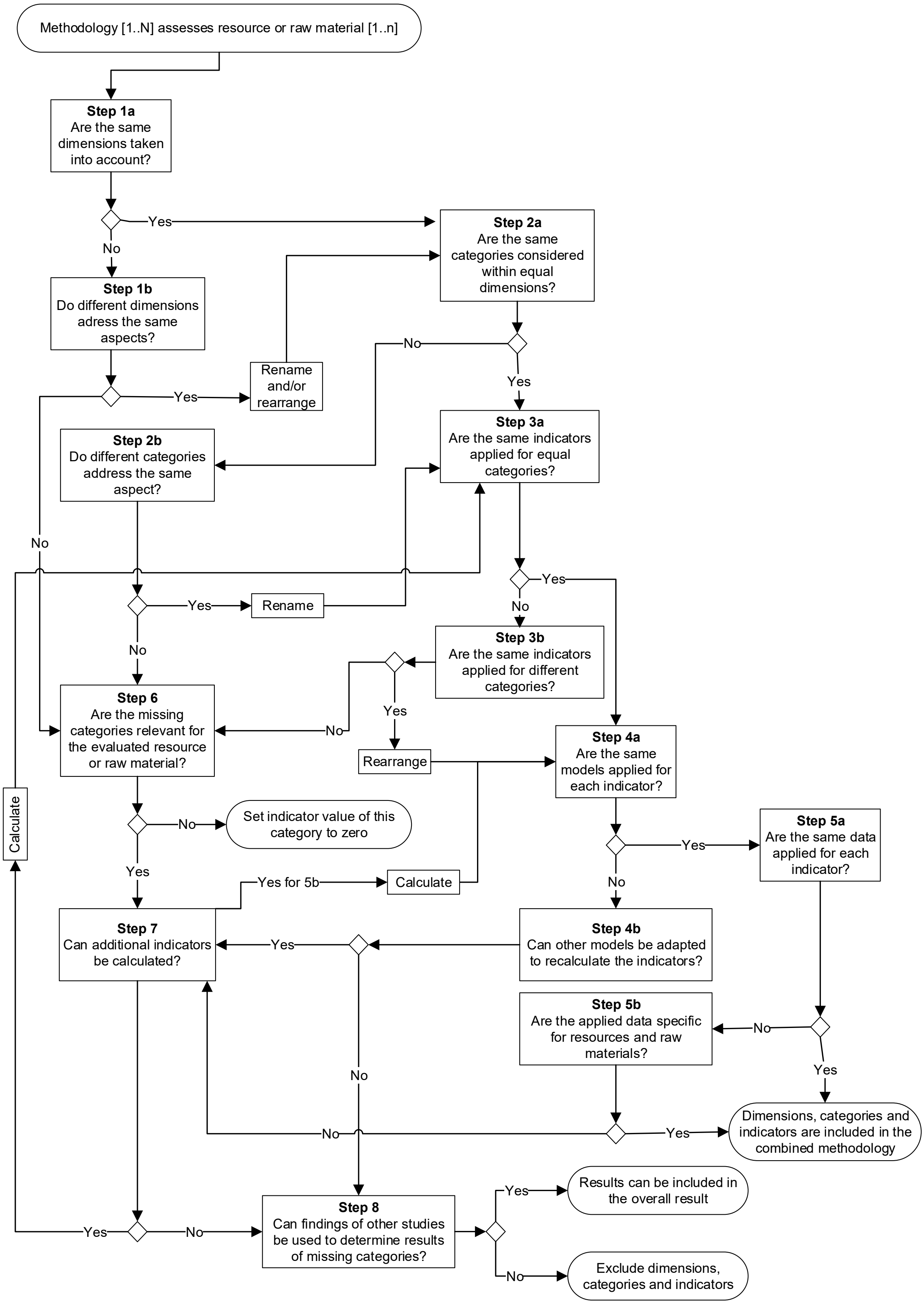

Figure S1: Proposed approach to combine methodologies evaluating different resource types

Before dimensions and categories are renamed and rearranged it should be decided, which of the considered methodologies is selected as the standard framework. For the dimensions which can be renamed and/or rearranged the approach continues with step 2a, the categories of other dimensions are further analyzed in step 6 (which will be explained later on).

In step 2b the question "Do different categories address the same aspect?" is addressed. When different categories address the same aspect, they can be renamed accordingly. For categories which can be renamed, the approach continues with step 3a, otherwise categories are analyzed further in step 6.

Within Step 3b the question "Are the same indicators applied for different categories?" is addressed when step 3a is answered with "no". Therefore, the applied indicators are compared across categories (and dimensions). Are indicators identical, they can be rearranged accordingly and are analyzed in step 4a. Different indicators are further addressed in step 6.

In step 4a the indicators are analyzed with regard to their underlying models. If the same models are applied, the approach continues with step 5a. Different models are further analyzed regarding possible adaptations in step 4b. To determine if a model can be adopted, it is identified whether the model is specific for a resources type. If it is specific for a resources type, model adjustments are difficult and often not possible. If adaptation of the models is possible, the approach continues with step 7, otherwise step 8 is addressed (both steps will be explained later).

Besides the models, also the used data has to be consistent (evaluated in step 5a). When different data is used, the question of step 5b "Are the data specific for resources and raw materials?" is addressed. When underlying data is specific for resources and raw materials, application of different data sources still facilitates adequate comparison and the indicators can be included in the combined methodology (same ending point as before with step 5a). If the underlying data is not resource and raw material specific, the same data sources have to be used and the approach continues with step 7.

Following, step 6 addresses the question "Are the missing categories relevant for the evaluated resource or raw material?". This step considers categories addressing different dimensions (from step 1b), different aspects (from step 2b) as well as different indicators (from step 3b). Is the category determined as not relevant, the indicator value can be set to zero and another ending point of the approach is reached ("Set indicator value of this category to zero"). The decision about relevance of a category should be thoroughly reviewed, because mistakes can lead to an inadequate comparison of resource types. Thus, it is recommended to make this decision based on the judgment of multiple experts. Is the category considered relevant, the approach continues with step 7.

In step 7 the question "Can additional indicators be calculated?" is addressed. This step considers indicators which apply different models (from step 4b), different data sources (from step 5b) as well as categories for which so far no indicator results are determined (from step 2b and step 3b). If so far no indicator has been established to quantify a certain category, it is decided whether an indicator of the other methodology can be applied. This is only possible if the underlying data is independent from the resource type (as already determined in step 5b). Thus, there are no specific challenges involved in determining them, besides the additional time that needs to be invested for recalculation. If additional indicators can be calculated the approach continues with step 3a to guarantee that the newly calculated indicators are consistent. If it is not possible to calculate additional indicators, the approach continues with step 8.

The steps so far are established to compare dimensions, categories and indicators of specific assessment methodologies and combine them to one methodology. Step 8 addresses the question "Can findings of other studies be used to determine results of missing categories?" and thus no longer addresses the considered methodologies, but takes into account results of existing case studies and whether they can be used to determine results of missing dimensions, categories and indicators. Thus, step 8 can only be carried out after the system to be examined was defined. Practitioners should keep in mind that findings from other studies might not consider the same system boundaries, functions, etc. Due to simplifications and assumptions, results might have higher uncertainties, which have to be taken into account. Thus, a thorough analysis is important before using the findings for comparing resources and raw materials. When findings are applicable the results can be included in the overall results and another ending point of the approach is reached ("Results can be included in the overall results"). Otherwise categories (and dimensions), which cannot be assessed are not included in the combined methodology. Here, the fourth ending point is reached ("Exclude dimensions, categories and indicators"). This option however should be avoided and significant effort should be made to include the broadest possible range of categories.

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3.4 Assessment of abiotic resource and raw material use on regional level

This chapter contains the following publication:

V. Bach, N. Finogenova, M. Berger, L. Winter and M. Finkbeiner (2017**): Enhancing the assessment of critical resource use use at the country level with the SCARCE method – case study of Germany,** Resources Policy - The International Journal of Minerals Policy and Economics

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The publication introduces the SCARCE method to assess criticality of abiotic resource and raw material use at regional level. First, methodological gaps of existing methods and relevant dimensions and categories for the assessment of abiotic resource and raw material use on region level are determined in an iterative first step (contribution to research target *4a*). Overall the following three dimensions with 25 associated categories are identified:

- availability
- vulnerability
- societal acceptance with the sub dimensions
    - compliance with social standards
    - compliance with environmental standards

The dimensions and categories are quantified by existing (contribution to research target *4b*) as well as newly established indicators (contribution to research target *4c*). For the dimensions availability and societal acceptance new indicators (based on ESSENZ) are developed. Further, existing and newly developed methods are joined for the integrated methodology SCARCE (contribution to research target *4d*). In a last step, the methodology is applied to the case study of Germany (contribution to research target *4e*).

The updated indicator values are shown in appendix 10.3.

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Enhancing the assessment of critical resource use at the country level with the SCARCE method – Case study of Germany☆

Vanessa Bach*, Natalia Finogenova, Markus Berger, Lisa Winter, Matthias Finkbeiner

Technische Universität Berlin, Chair of Sustainable Engineering, Straße des 17. Juni 135, 10623 Berlin, Germany

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ABSTRACT

The demand for many resources has increased significantly over the last decades due to their growing importance for industrial and technological development. Thus, various methods were developed to assess availability constraints of resources in relation to their vulnerability within countries and/or sectors (criticality). However, these methods display several short-comings. Thus, the aim of the introduced approach is, to enhance the assessment of critical resource use on country level with the SCARCE method, by considering the two dimensions criticality (with the sub dimensions availability and vulnerability) and societal acceptance (with the sub dimensions compliance with social standards and compliance with environmental standards). For five of the 12 introduced categories measuring availability constraints the country specific import mix is used to determine availability constraints of resources individually for the country under consideration. These results can further be compared with global constraints (which are calculated based on global production data) to determine if the country under consideration performs worse or better than the global average. To measure social aspects the categories small scale mining, geopolitical risk and human rights abuse are introduced. Environmental aspects are considered within the categories sensitivity of the local biodiversity, climate change and water scarcity. Additionally, next to metals also fossil fuels are included allowing a direct comparison of both abiotic resources. The SCARCE method is applied for the case study of Germany for which criticality results are presented and their plausibility is validated. It is shown that for Germany tungsten is the raw material showing high risks in all considered dimensions excluding the sub dimension vulnerability. Its high availability constraints are defined by the categories political stability, primary material use and price fluctuations. Further, due to the countries tungsten is imported from (e.g. Bolivia), its compliance with social and environmental standards is low. To enhance the applicability of the SCARCE method, indicator results are provided for 40 resources to assess their availability constraints as well as their compliance with social and environmental standards.

1. Introduction

In the last decades the demand of resources and raw materials rose significantly due to continuing global industrial and technological development. With that also awareness with regard to a sustainable use of resources and raw materials has grown as well, which is reflected in strategies and measures on international as well as national level (e. g. European Commission, 2011, European Commission, 2015; United Nations, 2016). This implies considering availability of resources and raw materials for current and future generations and the vulnerability of countries and/or sectors with regard to critical resources and raw materials (economic dimension) as well as the extraction, processing and use of resources and raw materials in line with ecological and societal considerations (environmental and social dimension). The term “resources” refers to entities, which can be extracted from nature and transferred to the anthroposphere. This includes abiotic and biotic resources, minerals, metals, fossil fuels as well as water, land, and the natural environment (Schneider et al., 2016; Sonderegger et al., 2017).

Methods to determine aspects with regard to resource use have been published manifold in the last years, considerably improving the assessment of resource use. They are addressing the micro (product), meso (company) and macro (company) level.

For the assessment of resource use on product level several

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* Corresponding author.
E-mail address: vanessa.bach@tu-berlin.de (V. Bach).

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approaches exist (e. g. Guinée et al., 1993, Graedel et al., 2012, VDI e.V. (2013), Schneider et al., 2013, Schneider et al., 2015, Dewulf et al., 2015, Bach et al., 2016 and Gemechu et al., 2016). Most of them complement the existing Life Cycle Assessment (LCA) methodology according to ISO 14040/44 (Finkbeiner et al., 2006). These approaches range from considering single aspects (e.g. depletion of abiotic resource (Guinée et al., 1993)) over multiple aspects (e.g. several socio-economic availability constraints (Schneider et al., 2013)) to first approaches with regard to sustainability assessments (e.g. Bach et al., 2016). So far the focus has been on metals and minerals, with only few methodologies also considering biotic resources and raw materials (Oakdene Hollins, 2014; Bach et al., 2017).

For the assessment on company level so far only few approaches exist (e. g. (Duclos et al., 2010; Graedel et al., 2012; VDI Verein Deutscher Ingenieure e.V e.V, 2013; Bensch et al., 2015)), which often consider the same socio-economic limitations to availability as on product level. Additionally to availability, the vulnerability of the considered companies with regard to these materials is taken into account. Assessing the availability of materials within the context of a company's vulnerabilities is referred to as criticality. So far existing methodologies focus on abiotic resources only.

For the assessment of resource use on the country level several methodologies and studies exist (e.g. Eggert et al., 2007, Morley and Eatherley, 2008, Kind, 2011, Knašytė et al., 2012, European Commission, 2014, Bastein and Rietveld, 2015, Hatayama and Tahara, 2015, Glöser-Chahoud et al., 2016, Buchert et al., 2017 and Blengini et al., 2017). For a comprehensive assessment of resource use on the country level in the context of sustainable development, the following dimensions have to be addressed: vulnerability, availability, criticality as well as environmental and social impacts. To determine the dimension vulnerability the aspects *substitutability* followed by *economic importance* and *dependency on imports* are addressed most often. However, more aspects can influence vulnerability as shown by the various aspects addressed in the existing methodologies (Helbig et al., 2016).

As shown in Achzet and Helbig (2013) the most commonly applied indicators for determining the dimension socio-economic availability are *concentration of reserves, production and companies* as well as *by-product dependency, mining capacity* and *demand growth*. The range of considered indicators varies between one (e. g. Buchholz et al., 2012) and eight (e.g. Graedel et al., 2012). However, studies on the product level (e.g. Schneider, 2014, Bach et al., 2016 and Henßler et al., 2016) have shown that more than these eight aspects should be established to reach a comprehensive assessment of socio-economic availability constraints. To calculate the indicator results for the socio-economic dimension, some methodologies use global production data (e. g. Buchholz et al., 2012 and Graedel et al., 2012), while others use a mix of global production and import data, depending on the socio-economic aspect taken into account (e.g. Erdmann et al., 2011, Knašytė et al., 2012, Hatayama and Tahara, 2015, Glöser-Chahoud et al., 2016, Buchert et al., 2017 and Blengini et al., 2017). Whereas some aspects are influenced by the global market and thus are independent from the import mix (e.g. price fluctuations), for other aspects (e.g. political stability) the import structure plays a significant role with regard to the availability of resources and raw materials and thus, should be taken into account. So far import based indicator results are only determined for the categories *concentration of production* and *country risk* (e. g. as done by Erdmann et al., 2011, Knašytė et al., 2012 and Glöser-Chahoud et al., 2016) and no comparison between import based and global results is carried out.

Next to the socio-economic availability, also the physical availability of resources should be addressed. Indicators determining the socio-economic availability consider reserves (identified stocks from which a mineral or metal can be economically extracted as of today (United States Geological Survey, 2015)), whereas the physical availability refers to the long term availability of resources. Thus, all available resource stocks (quantified by the ultimate reserves) are taken into account, assuming that at one point in time they can be extracted as technological development progresses. Existing methodologies focus on socio-economic aspects only, whereas physical aspects are seldom taken into account.

In order to determine the final criticality of raw materials for a country, studies and methodologies either graph the availability and vulnerability dimensions together in a diagram (common two-axis assessment framework as shown by e. g. Eggert et al., 2007, Erdmann et al., 2011, Graedel et al., 2012 and European Commission, 2014) or calculate a single score results by aggregating both dimensions (as shown by e. g. Morley and Eatherley, 2008, Graedel et al., 2012, Knašytė et al., 2012, Bastein and Rietveld, 2015 and Hatayama and Tahara, 2015). So far no common agreement has been reached, which of these is the more favorable approach. However, as shown by Nassar et al. (2012) determining a single score result is challenging as weighting has to be applied, which highly influences the results.

As human beings rely on the environment (and its ecosystem services) it is defined as a resource worthy of protection (European Commission, 2005), and pollution of the environment related to resource use is taken into account in resource use assessment methodologies. Existing methodologies consider environmental implications of resource use either by evaluating pollution of the environment (as done by e.g. Buchert et al., 2017) or by applying the Environmental Performance Index (EPI) (Yale Center for Environmental Law and Policy, 2014) (as done by e.g. Graedel et al., 2012 and European Commission, 2014). When the pollution of the environment is assessed only resource specific impacts (related to resource extraction, processing use and end of life) are taken into account, whereas country specific differences, e.g. different technological standards, are not considered. When EPI is applied only the performance of a country in general and not specific for a resource is taken into account (e. g. processing of aluminum requires more energy and therefore leads to more emissions than steel (Han, 1996)). Further, country specific emissions are determined for the global production mix only, but should also be calculated for the specific import mix of the considered country. Import based results should also be compared to global averages.

Further, when determining resource use in the context of sustainable development also social aspects have to be considered (Jenkins and Yakovleva, 2006; United Nations Environment Programme, UNEP, 2009). Social impacts of a country's resource use are so far taken into account by addressing health impacts applying life cycle impact assessment methods as done by Bensch et al. (2015) or by taken into account aspects addressed in social life cycle assessment as done by Dewulf et al. (2015) and Buchert et al. (2017), e. g violent conflicts, working conditions and corruption of the extracting country. However, country based indicators are determined only for the three countries with the highest global production, therefore neglecting countries with smaller production but possibly higher social violations. Further, social aspects should also be determined based on the import mix and results should be compared to the global average.

Most of the existing methodologies and studies address metals and minerals, with only few ones also taking into account biotic resources and raw materials (e. g. Morley and Eatherley, 2008; Kind, 2011; Knašytė et al., 2012; Oakdene Hollins, 2014) and so far only the publication by (Knašytė et al., 2012)) consider fossil fuels. Assessing availability constraints of biotic and fossil resources and raw materials and comparing them to mineral resources is relevant for a holistic assessment and to identify possible trade-offs (e. g. the use of renewable energy like wind or solar power instead of fossil energy resources leads to a higher demand of specific materials like indium, for which socio-economic availability constraints occur).

Therefore, the aim of the introduced approach is to enhance the assessment of critical resource use at the country level (SCARCE – method) by considering:

- Socio-economic availability,
  - Additional relevant categories (and corresponding indicators) are taken into account
- Environmental impacts,
  - Aspects specific for the considered resource as well as for the production country are taken into account
- Social impacts,
  - Existing indicators are improved with regard to underlying data availability and all production countries are taken into account
- Country specific results for the socio-economic availability as well as environmental and social impacts
  - The specific import mix of the country under investigation is taken into account
  - Results based on the import mix are compared to the global average
- Physical availability,
  - The long term availability of resources based on ultimate reserves is taken into account
- Vulnerability and criticality
  - Existing methodologies and frameworks are applied
- Next to metals also fossil resources.

In the next section, the overall approach to enhance the criticality assessment of a country's resource use is presented and it is shortly explained how relevant categories and indicators are identified (Section 2). Next, the individual dimensions, categories and indicators are explained in more detail (Sections 2.1–2.2) and applied in the case study of Germany (Section 3). Further, challenges of the introduced approach are discussed (Section 4) and conclusions are drawn (Section 5).

2. SCARCE method

In this section the approach to enhance the assessment of critical resource use on country level (SCARCE – method) to enhance the criticality assessment of a country's resource use is introduced. It is established to be used as a stand-alone methodology to analyze aspects of resource use in the context of sustainable development. All three sustainability dimensions are considered (see Fig. 1). The economic dimension is presented by the dimension *criticality*, which is assessed in the sub dimensions *availability* (further divided in *socio-economic availability* and *physical availability*) and *vulnerability*. The dimension *societal acceptance* is divided into the sub *dimensions compliance with social standards* and *compliance with environmental standards*, which reflect the social and environmental dimensions, respectively. For all (sub) dimensions categories and corresponding indicators are displayed. For identifying which indicators will be implemented in the introduced approach a bottom-up & top-down approach based on Bach et al. (2016) is applied (for more details see supplementary material – Section 1).

Following, the considered dimensions, sub dimensions, categories and indicators are explained in more detail.

2.1. Dimension: criticality

In this section the determination of the dimension *criticality* is explained. It consists of the sub dimensions *availability* and *vulnerability*, for both of which a detailed description is provided in the next sections. For evaluation of the *criticality* the two sub dimensions *availability* and *vulnerability* have to be evaluated first. Each sub dimension is calculated by aggregating the indicator results of all associated categories (see Fig. 1). They are then graphed within a matrix (commonly used two-axis approach), where each point represents the specific resource result of the sub dimensions (risk of supply disruption and vulnerability to this disruption). Even though several methodologies provide approaches to determine single score results, in the introduced approach the aggregation of the (sub) dimensions is not carried out. In Section 3 results for Germany are shown (see Fig. 4).

2.1.1. Sub dimension: availability

In this section the categories and indicators used to determine the sub dimension *availability* are introduced. The Integrated Method to Assess Resource Efficiency (hereinafter referred to as ESSENZ) developed by Bach et al. (2016) to evaluate constraints to the availability of resources and raw materials on product level within Life Cycle Assessment, is used as a basis to determine availability constraints on country level. ESSENZ provides indicators for twelve categories. The category *abiotic resource depletion* (based on ultimate reserves) is applied to determine the sub dimension physical availability, whereas the categories *concentration of reserves and production, company concentration, price fluctuation, primary material use, mining capacity, feasibility of exploration projects, occurrence of co-production, trade barriers, political stability* and *demand growth* are used to determine the socio-economic availability. Following, the associated indicators of these categories are introduced (Further details regarding the calculation of the indicator values can be found in the ESSENZ publication by Bach et al., 2016):

- Concentration: The categories *concentration of reserves, production and companies* are determined by squaring the global reserve shares, production shares and company shares respectively and summing each up individually based on Rhoades (1993).
- *Price fluctuations*: The category is quantified by the volatility indicator applied by German Federal Institute for Geosciences and Natural Resources (2014).
- *Primary material use*: To determine the effects of primary material use, the recycled content of the raw material is determined based on the data published by Graedel (2011).
- *Mining capacity*: To quantify this category the reserve of a raw material is set in relation to the annual production based on the data by British Geological Survey (BGS) (2014) and United States Geological Survey (USGS) 2015).
- *Feasibility of exploration projects*: The category is determined by multiplying the raw materials' share of global production per country with the Policy Potential Index (Cervantes et al., 2013).
- *Occurrence of co-production*: To quantify the occurrence of metals with regard to co-production (main or companion product) qualitative values by Angerer et al. (2009) were transformed into quantitative values.
- *Trade barriers*: They are measured by multiplying the raw materials' share of global production with the Enabling Trade Index (Hanouz et al., 2014).
- *Demand growth*: The category is quantified by calculating production increase (or decrease) over the last five years based on annual production based on data provided by BGS (2014) and USGS (2015).
- *Political stability*: For calculating the indicator value for this category the share of global production is multiplied by the Worldwide Governance Indicators (Kaufmann et al., 2011; World Bank Group, 2013).
- *Abiotic resource depletion*: For determining the category of the sub dimension physical availability the characterization factors provided by Guinée et al. (1993) and Oers et al. (2002) for ultimate reserves (crustal content) are applied. They address the quantity of a resource that is ultimately available (van Oers and Guinée, 2016). The indicator is not adapted, but scaled to 0–1.

To determine the indicator results of the categories, first the indicator values have to be determined (as shown above). Then, the indicator value of a material *i* within a category *c* is set in relation to the category specific target value (an exception is the category *physical availability* for which no target value is available) to determine the

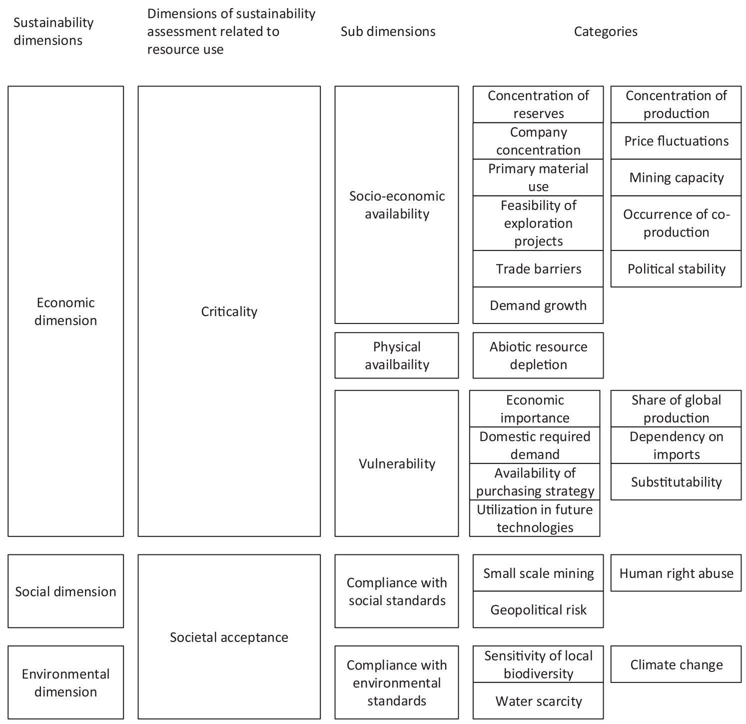

Fig. 1. Overview of considered dimensions, sub dimensions and categories considered within the SCARCE method and their link to the sustainability dimensions.

Distance-to-Target (DtT) value based on the ecological scarcity approach (Müller-Wenk et al., 1990; Frischknecht et al., 2009) (see Eq. (1)).

$$DtT - value_{i,c} = \left(\frac{indicator\ \ value_{i,c}}{target_c}\right)^2 \tag{1}$$

The targets were determined by a stakeholder survey within the ESSENZ project (Bach et al., 2016). The DtT - values reflect to which extent resources and raw materials face availability constraints: a value lower than 1 refers to no availability constraints (and thus is set to zero as the considered aspect does not have any potential limitation on resource availability); a value of 1 or greater than 1 refers to limited availability. Thus, the determined targets are a key element of this approach. The targets as well as a comprehensive explanation and discussion of the approach can be found in the publication by (Bach et al., 2016). Within the product oriented ESSENZ method the next steps include normalization based on global production amounts and scaling of these normalized values to 1.7×10^{13}. Finally, to determine the availability of resources and raw materials in product systems, the indicator values are multiplied with the used amount of raw materials within the product system under consideration.

The approach for the indicator values as applied in ESSENZ can also be used to adequately determine indicator values for a country assessment when the following changes are implemented:

1) Originally all six Worldwide Governance Indicator (Kaufmann et al., 2011; World Bank Group, 2013) were taken into account for the category political stability, whereas within the SCARCE method only four indicators are considered. As *geopolitical risk* is an aspect considered within the sub dimension *availability* as well as within the sub dimension *compliance with social standards* (see Section 2.2.2), the Worldwide Governance Indicators are divided into two sets: indicators quantifying government effectiveness, regulatory quality, rule of law and control of corruption are applied to determine availability constraints due to political stability, whereas the indicators voice & accountability and no violence are applied for the assessment of geopolitical risk (see Section 2.2.2) in the societal dimension.
2) Scale up of the raw material specific DtT - values to 0–1 instead to 1.7×10^{13} (for more explanation[see1]). The indicator results of a raw material *i* are determined by subtracting the smallest DtT – value of the category *c* from the original DtT - value and dividing it by the difference of the highest and smallest DtT - value. (see Eq. (2)).

$$indicator\ \ result_{i,c} = \frac{(DtT - value_{i,c} - DtT - value_{min,c})}{(DtTvalue_{max,c} - DtT - value_{min,c})} \tag{2}$$

3) Global production values as well as the used raw material amounts (which are considered in the *availability* dimension in the product assessment) are taken into account in the sub dimension *vulnerability* for the evaluation of the country's criticality (see Section 3.2).
4) To determine availability constraints specific to a country, import data is used for the calculation of some indicators instead of global production data (see Table 1). Out of the 12 categories considered five four are not influenced by the specific import mix. *Demand growth* as well as *price fluctuations* are predominantly determined based on the global supply and demand balance rather than by exporting countries. The *physical availability* of a resource (defined

[1] The number 1.7×10^{13} was chosen as it presents the highest global production value of the raw material portfolio considered (Bach et al., 2016). Within the product based assessment the amount of materials are multiplied by the indicator values to determine the overall risk to availability. Since some materials can dominate the BoM on mass basis (e. g. steel in cars) (Henßler et al., 2016), the indicator values need to have a certain spreading in order to make critical materials, which are usually present in small amounts (e. g. gold in cars), visible in the results.

Table 1
Overview of indicators used to quantify availability constraints of metals and fossil raw materials.

Category	Indicator	Point of view
Demand growth	Percentage of annual growth based on past developments	Global
Concentration of reserves	Herfindahl-Hirschmann-Index (Rhoades, 1993)	Global
Price fluctuation	Volatility based on (German Federal Institute for Geosciences and Natural Resources, 2014)	Global
Physical availability	Abiotic resource depletion (Guinée et al., 1993; Oers et al., 2002)	Global
Occurrence of co-production	Percentage of production as companion metal (Angerer et al., 2009)	Global
Primary material use	Percentage of new material content (Graedel, 2011)	Global
Company concentration	Herfindahl-Hirschmann-Index (Rhoades, 1993)	Company specific[a]
Mining capacity	Reserve-to-annual-production ratio	Country specific
Feasibility of exploration projects	Policy Potential Index (Cervantes et al., 2013)	Country specific
Trade barriers	Enabling Trade Index (Hanouz et al., 2014)	Country specific
Political stability	Worldwide Governance Indicators (World Bank Group, 2013)	Country specific
Concentration of production	Herfindahl-Hirschmann-Index (Rhoades, 1993)	Country specific

[a] Due to missing data a global perspective is applied.

by its amount within the earth crusts and its extraction rate) also does not depend on the countries the resources are imported from. This also applies to the categories *concentration of reserves* and *occurrence as co-product*, which are determined by the resources appearance in nature. The category *primary material use* is established to determine the pressure on primary materials, which can be reduced by using secondary materials. Due to missing data, the aspect cannot be determined for the considered country and is therefore included as a supply risk based on the global average recycling content of raw materials.

The other categories can be impacted by the choice of importing countries. Thus, instead of indicator values based on global production data, the import mix of the country under consideration is used as a basis for calculation. The categories *trade barriers, political stability* as well as *feasibility of exploration projects* are highly influenced by the governmental structure and practices of the exporting countries. For the category *concentration of production* the number of countries from which resources are imported as well as the amount of raw materials produced in these countries determine the supply risk. If raw materials are only imported from few countries with small materials amounts, the possible supply constraints are higher compared to importing them from several countries with high raw material production. This also applies to the category *company concentration*: being able to trade raw materials with many companies reduces possible supply restrictions, compared to being able to trade with only few companies. However, due to missing data it is not possible to calculate the company concentration for Germany or other countries (as pointed out by an asterisk in Table 1). The category *mining capacity*, which is quantified by the static range (reserve to annual production ratio), assess supply restrictions due to the depletion of currently operating mines and thus the need to establish new mines. As the timeframe for establishing a fully operational mine can add up to around 15 years, a raw materials might not be available in the same amounts as before and is therefore subject to potential availability constrictions. When a country imports its raw materials from countries where the mining capacity is almost exhausted, the risk of possible restrictions is higher than for raw materials imported from countries where the mining capacity is not or less exhausted.

Results based on the country specific import mix can be compared to results based on global production data to determine if the country under consideration performs better, the same or worse than the global average. To determine the difference (Δ) between the import based and global results the import based indicator result (scaled distance to target value) of the considered raw material *i* for category *c* are subtracted from the global indicator result (see Eq. (3)).

$$\Delta_{i,c} = indicator\ \ result_{global,i,c} - indicator\ \ result_{import,i,c} \quad (3)$$

Is the difference greater than zero, the global constraints are larger than for the imported materials. The constraints are equal, if the difference is zero. Is the difference lower than zero, the constraints of the imported raw materials are higher than the global average.

2.1.2. Sub dimension: vulnerability

In this section, the categories and indicators applied for the sub dimension *vulnerability* are introduced. The sub dimension is based on existing methodologies to determine vulnerability, in particular Erdmann et al. (2011), Buchholz et al. (2012), Graedel et al. (2012), European Commission (2014), Klinglmair et al. (2014), Oakdene Hollins (2014) and Sonnemann et al. (2015). It was ensured that only categories were selected for which data is available. Thus, the categories implemented for the assessment of a country's vulnerability are*: economic importance, share of world production, internal required demand, dependency on imports, availability of purchasing strategies, substitutability,* and *utilization in future technologies.* In Table 2 an overview of these categories and the corresponding indicators is shown. The indicators of all categories are scaled to 0–1 (see Eq. (2)) before being weighted to guarantee that comparability is possible. Aggregation of the seven categories to a single score result for every resource and raw material is necessary to plot the results within the criticality matrix.

Table 2
Overview of categories including a short description and indicator for quantification of the dimension vulnerability.

Category	Short description	Indicator
Economic importance	Economic profits of a raw materials	Value added of sectors which utilize the raw material in production according to (Knašytė et al., 2012)
Share of global production	Share of imported raw materials compared to the worldwide production	Imported amounts in relation to global production
Domestically required demand	Imported amount of raw materials	Imported amount
Dependency on imports	Domestic production	Domestic production compared to imported amounts
Availability of purchasing strategies	Purchasing strategies exist between the country under consideration with other countries	Share of the raw material imported from countries, for which purchasing strategies are established
Substitutability	Substitutability of raw materials	Share of raw material, which can be substituted
Utilization in future technologies	Demand of a specific raw material by future technologies	Share of raw material, which will be significant for future technologies

2.2. Dimension: societal acceptance

Next to availability and vulnerability also social and environmental aspects are important when assessing a country´s resource use. Both can lead to availability constraints due to low societal acceptance. Consumers are more and more interested in compliance with social as well as environmental standards (e.g. Tsurukawa and Manhart, 2011; The Guardian, 2015; Eisenhammer, 2015; Osburg et al., 2016; Aitken et al., 2016; Balanay and Halog, 2016; Wan Ahmad et al., 2016; Kemp et al., 2016) and expect companies as well as the government to uphold certain norms. Is the breach too severe, certain material cannot be imported for utilization because of possible consumer boycott. Following, the sub dimensions *compliance with social and environmental standards* are explained in more detail.

2.2.1. Compliance with social standards

In this section the categories and indicators for the sub dimension *compliance with social standards* are introduced, which are established based on ESSENZ as well as the work done by Buchert et al. (2017). Overall three aspects were identified as being significant: *small scale mining, geopolitical risk* and *human rights abuse* (see Fig. 2). These aspects are expressed as categories within the introduced SCARCE method.

Small scale mining is one of the aspects proposed by Organisation for Economic Co-operation and Development (OECD, 2016) to identify minerals mined within risk and conflict zones. Materials mined in small scale mining operations are often used to pay for violent conflicts and wars and are characterized by poor working conditions (Lujala, 2010; Driffield et al., 2013). For quantification data regarding the global share of small scale mining operations, in relation to the considered resources, are identified (e.g. 50% of all chrome worldwide is mined in small scale mining operations) (Ghose, 2003; Dondeyne et al., 2009; Dorner et al., 2012). Furthermore, artisanal small scale mining is considered even more detrimental to human well-being than industrial small scale mining, because of its intense requirement for

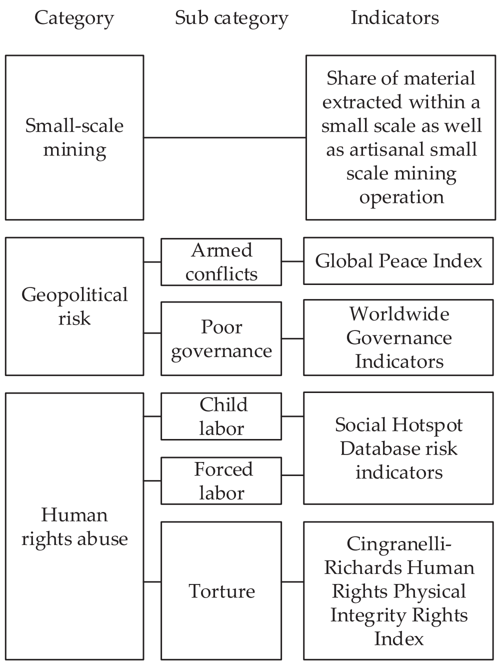

Fig. 2. Overview of considered categories, sub categories and indicators to determine compliance with social standards.

physical labor (Gunson and Jian, 2002). Thus, the share of materials mined in artisanal small scale mining is taken into account as an additional factor. The small scale mining indicator (SCMI), applied in the SCARCE method, is determined of a resource *i* by multiplying the share of the resource extracted in small scale mining operations (share of ssm) with the share extracted in artisanal small scale mining (share of artisanal ssm, which is added to 1 to prevent that the overall indicator becomes zero, when only small scale mining but no artisanal small scale mining occurs) (see Eq. (4)).

$$SCMI_i = share\ \ of\ \ ssm_i \times (share\ \ of\ \ artisanal\ \ ssm_i + 1) \tag{4}$$

The results are scaled to 0–1 according to Equation x. For resources where no data is available, it was assumed that no small scale mining takes places. As data related to small scale mining is collected for over 20 years (e.g. Brower, 1979; Godoy, 1985; Caymo, 2016) this assumption ca be seen as plausible.

Another significant parameter for compliance with social standards is countries displaying unstable governments. Within those countries the likelihood of repression of citizens (with regard to voting, freedom of expression, etc.) as well as politically motivated violence (Bienen and Gersovitz, 1986; Hafner-Burton, 2005; Jong-A-Pin, 2009) is high. To determine the category *geopolitical risk* two of the overall six Worldwide Governance Indicators (voice and accountability and political stability & no violence (GI) (Kaufmann et al., 2011; World Bank Group, 2013)) as well as the global peace index (GPI) (Institute for Economics and Peace, 2015) are taken into account. The GPI ranks countries regarding their level of peacefulness by considering domestic and international conflicts as well as degree of militarization (Institute for Economics and Peace, 2015).

To determine the geopolitical risk the indicators are summed up, squared (this way differences between low and high impacts are more significant), multiplied with the country specific import shares and summed up (see Eq. (5)). Results are scaled to 0–1 according to Eq. (2).

$$Geopolitical\ \ risk\ \ indicator_i = \sum import\ \ shares_{i,x} \times (GI_x + GPI_x)^2 \tag{5}$$

Consideration of *human rights abuse* is essential for determining compliance with social standards. Since small scale mining as well as geopolitical risks already take into account human rights violations, the last category focuses on additional aspects, to which consumers react especially sensitive. These aspects are child labor (CL), forced labor (FL) and overall torture (also including extrajudicial killing and political imprisonment). Child and forced labor are quantified based on data of the Social Hotspot Database (Benoit-Norris et al., 2012; Norris et al., 2013). Torture can be measured by the Cingranelli-Richards Human Rights Physical Integrity Rights Index (PIRI) (Cingranelli and Richards, 2010; Cingranelli et al., 2012). To determine the category *human rights abuse* for a resource *i* the indicators are summed up, squared (to enhance differences between low and high impacts), multiplied with the country specific import shares and summed up (see Eq. (6)). The results then are scaled to 0–1 according to Eq. (2).

$$Human\ \ right\ \ abuse\ \ indicator_i = \sum import\ \ shares_{i,x} \times (CL_x + FL_x + PIRI_x)^2 \tag{6}$$

To determine the final result for the sub dimension *compliance with social standards* the three categories are summed up equally. The results are not plotted in the criticality matrix. Instead, the five raw materials with the highest risks are visually highlighted by frames with a broken line (see Fig. 4).

The indicators are calculated based on country specific import data as default. However, they can also be determined for the global production to be compared in the same way as for the sub dimension availability (see Eq. (3)). Is the difference between the global and import based results greater than zero the compliance with social

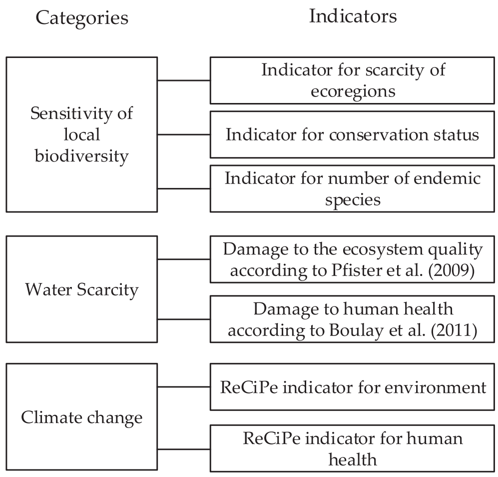

Fig. 3. Overview of categories and indicators considered for compliance with environmental standards.

standards is better for the imported materials compared to the global average. Is the difference lower than zero, the compliance is lower.

2.2.2. Compliance with environmental standards

In this section the categories and indicators for the sub dimension *compliance with environmental standards* are introduced. Overall three aspects are identified as relevant: *sensitivity of local biodiversity, water scarcity* and *climate change*. These aspects are expressed as categories within the SCARCE method (see Fig. 3). The state of the environment can influence the safeguard subjects ecosystem (e.g. biodiversity loss) and human health (e.g. malnutrition). Therefore, impacts on both safeguard subjects are addressed, with the exaptation of biodiversity. So far no reliable indicators exists to measure biodiversity and related ecosystem services of countries with regard to human health impacts (e. g. nutrient recycling to support food production) (Romanelli et al., 2015; Sandifer et al., 2015; Winter et al., 2017).

The *sensitivity of the local biodiversity* is an issue not only related to but of significance for extraction of resources mostly due to transformation of land area to mining areas including expansion of infrastructure (Pascal et al., 2008; Murguía et al., 2016). Within the SCARCE method the protection of biodiversity is quantified by using indicators as proposed in the ecoregions approach by Brethauer et al. (2013). These indicators are scarcity of ecoregions (SE), conservation status (CS) and number of endemic species (ES) and are established based on data provided by World Wildlife Fund (2012) for 827 ecoregions. These ecoregions results are converted (area weighted) into country specific indicator values.

To determine the *sensitivity of local biodiversity* with regard to resources *i* first the three indicators are scaled to 0–1, summed up and squared (to enhance differences between low and high impacts). They are further multiplied with the raw material specific import shares (see Eq. (7)).

$$Impacts\ due\ to\ sensitivity\ of\ local\ biodiversity_i = \sum (import\ share_{i,x} \times (SE_x + CS_x + ES_x)^2) \tag{7}$$

Water scarcity is linked to severe human health issues (e.g. malnutrition (Sophocleous, 2004)) especially in developing countries as well as to impacts on ecosystems (e. g. drying up of rivers (Postel, 2000)). It is necessary for most mining operations and thus, often associated to be in direct competition with environmental and social needs (Camargo and Alonso, 2006; Vörösmarty et al., 2010; Budds and Hinojosa, 2012). Based on Pfister et al. (2009) effects on the ecosystem (expressed in potentially disappeared fractions) are determined. The method of Boulay et al. (2011) is applied to define impacts on human health (expressed in Disability-Adjusted Life Year). Both methods provide indicator values on country level, which can be set in relation to the country specific import share to determine resource *i* specific water scarcity impacts. Therefore, both indicators are scaled to 0–1, summed up and squared (see Eq. (8)).

$$Water\ scarcity\ impacts_i = \sum import\ share_{i,x} \times (impacts\ on\ ecosystem_x + impacts\ on\ human\ health_x)^2 \tag{8}$$

Climate change is the most addressed environmental impact worldwide (Boykoff and Yulsman, 2013; Schmidt et al., 2013; Newman, 2016) and consumers ask about the carbon footprint of their products more and more (Furlow and Knott, 2009; Upham et al., 2011). Thus, the greenhouse gas impacts of resources is also a topic of societal concern (Kolk and Pinkse, 2005; Barrett and Scott, 2012). Greenhouse gas emission data is provided in the databases of GaBi (Thinkstep, 2016) and ecoinvent (Ecoinvent, 2016) for all 40 considered materials. Country specific data is only available for some of the materials as well as countries. Thus, global averages were used to determine the resource *i* specific climate change impact. The impacts of greenhouse gas emissions are determined by applying the ReCiPe methodology (Huijbregts et al., 2017) to determine impacts to human health (CCHH) and the environment (CCE). The results of the two indicators are scaled to 0–1, summed up and squared (see Eq. (9)).

$$Climate\ change\ impacts_i = (CCHH_i + CCE_i)^2 \tag{9}$$

To determine the final result of the sub dimension *compliance with environmental standards* the three categories are summed up. The results are not plotted in the criticality matrix. Instead, the five raw materials with the highest results are visually highlighted with frames with a continuous line (see Fig. 4).

As default the indicators are determined for the country specific import mix of the considered materials. However, they can also be calculated based on global production data and compared as within the sub dimension compliance with social standards (see Eq. (3)). If the difference between the global and import based results is greater than zero the compliance with environmental standards is better for the imported materials compared to the global average. If the difference is lower than zero, the compliance with environmental standards is lower for the imported raw materials compared to the global average.

3. Case study of Germany

Next the SCARCE method is applied to the case study of Germany. First, it is shortly described how the country specific import mix is determined, then results for the individual dimension and sub dimension are presented.

3.1. Determination of the country specific import mix

In this section it is explained how the country specific import mix is determined based on data provided by Ferretti et al. (2013), BGS (2014), United Nations (2015) and USGS (2015). The country specific import mix shall reflect from which fossil raw material and ore producing countries Germany imports raw materials (in the following the term ore is used to refer to metal ores as well as fossil raw materials). Even though data by United Nations (2015) provides import data for Germany based on current trade statistics, some materials imported into Germany are first traded to other European countries and then imported to Germany. For some materials this makes it impossible to trace the original producing countries (e.g. according to the provided data all gold used in Germany is imported

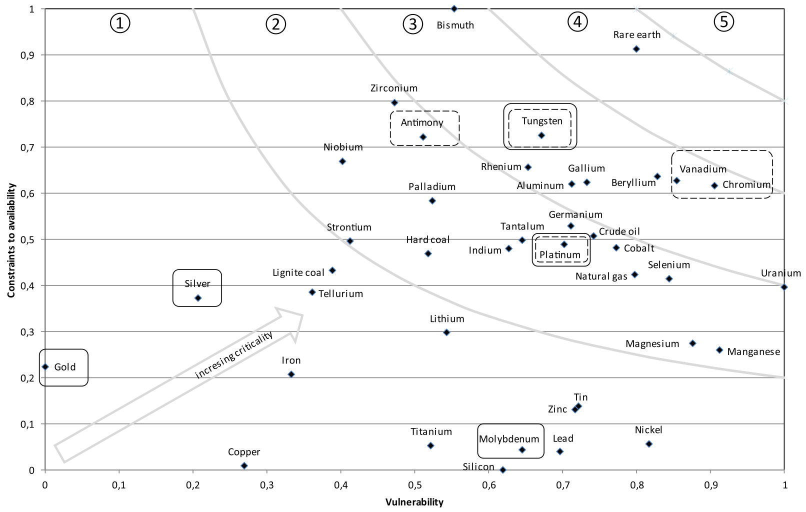

Fig. 4. Results of criticality assessment for Germany; materials with five highest values for compliance with social (frames with a broken line) and environmental standards (frames with a continuous line) are highlighted; contour lines to cluster the results in five criticality levels are shown.

from Switzerland; however within Switzerland no gold is mined). Thus, the import data is corrected using production data by BGS (2014) and USGS (2015). Import statistics and production statistics are compared for every raw material. When all countries importing a raw material into Germany are also producing the raw material, the country specific import mix can be used without adaptation. In case the countries in the import mix are not producing the raw material, the import mix is adapted accordingly as follows. When only few of the exporting countries are not ore producing countries, the overall amount of these countries is allocated to the producing countries based on the global production shares. When none of the exporting countries are producing the ore, the import mix is set equal to the global production mix. Finally, as for some materials purchasing strategies exists (see Section 3.3.2) (e.g. natural gas imported from Kazakhstan (Ferretti et al., 2013)) the country specific import mix is adopted accordingly (e. g. the import share of natural gas from Kazakhstan is set to 17% independently of the import data and global production). The determined import mix thus only covers the import of raw materials but not of products or intermediate products (e.g. metal plates).

3.2. Overall result for Germany

In this section, the overall result for the case study Germany is shown including aggregated results of all four (sub) dimensions (see Sections 3.3.1–3.3.3).

In Fig. 4 the criticality matrix for Germany is displayed showing aggregated results of the raw materials for the dimensions *availability* and *vulnerability*. The materials with the five highest results for the dimension *societal acceptance* are highlighted. Frames with broken lines are used for the category *compliance with social standards,* whereas continuous lines are used for the category *compliance with environmental standards*. To support the interpretation of the results contour lines based on approach by Glöser et al. (2015) are displayed. As the criticality matrix is linearly scaled, the contour lines have a convex shape. By adding contour lines to the criticality matrix, the considered raw materials can be clustered in five areas representing different levels of criticality. Raw materials within level 1 can be seen as least critical, whereas raw materials in level 5 show the highest criticality.

As shown in Fig. 4 none of the considered raw materials has a criticality level of five. The materials with the highest criticality are rare earth metals, which are the only raw materials in level 4. Further, bismuth, tungsten, rhenium, aluminum, gallium, beryllium, vanadium und chromium are classified with a criticality level of 3. Chromium, vanadium and tungsten also have a high risk to be not compliant with social standards (see Section 3.3.3.1 for more details). Tungsten even shows a high risk to violate environmental standards (see Section 3.3.3.2 for more details). Antinomy and platinum, which are classified with a level 2 criticality, are also associated with noncompliance of social standards; platinum even with the noncompliance of environmental standards. Gold, silver and molybdenum, which are classified with the criticality level of 1, are associated with noncompliance of environmental standards. Based on these results, the raw materials rare earth, tungsten, vanadium and chromium are the three materials, which should be further analyzed regarding reducing their criticality as well as social and environmental implications.

3.3. Results of individual dimensions

Next the results for the individual dimension (*availability* – Section 3.3.1; *vulnerability* – Section 3.3.2; *compliance with social standards* – Section 3.3.3.1; *compliance with environmental standards* – Section 3.3.3.2) are introduced in more detail to explain the overall result presented in Fig. 4.

3.3.1. Results of sub dimension availability

In this section the result of the sub dimension *availability* is analyzed in more detail. As shown in Fig. 5 the raw materials silicon, copper, lead, molybdenum und titanium have the lowest risk regarding constraints to availability, whereas bismuth, rare earth, zirconium, tungsten and crude oil are the materials with the highest risks.

Following, the results for bismuth are explained exemplary in more detail. For bismuth the categories with the highest risks are *occurrence of co-production* and *primary material use*. As bismuth is only mined as a by-product of lead, copper and other metals (Campbell, 1985; Ayres et al., 2003) the associated risk for the category *occurrence of co-production* is plausible. It is used as an alloy metal as well as an substance in pharmaceuticals and as a pigment for cosmetics and paints (Anderson, 2014). So far only the share used as an alloy metal can be recycled. Thus, the overall recycling rate is low summing up to a recycled content of around 10% (Graedel et al., 2011). This explains the

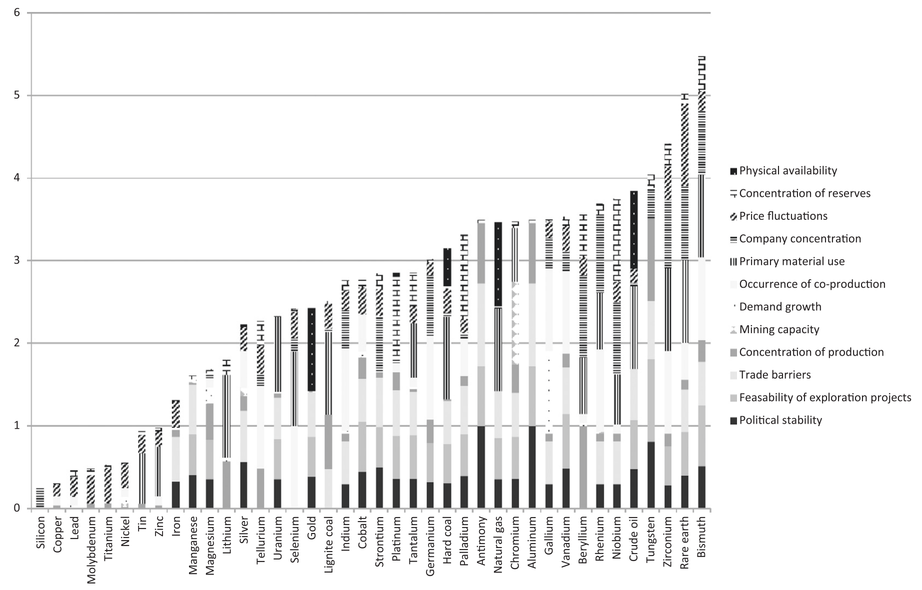

Fig. 5. Result of sub dimension availability.

high potential availability constraints associated with this category. Additionally bismuth shows high risks for four more categories (*company concentration, trade barriers, political stability and feasibility of mining operations*). Overall only for three categories (*physical availability, demand growth and mining capacity*) there are no associated possible availability constraints. A detailed analysis like shown for bismuth should be carried out for all or at least the raw materials with the highest possible availability constraints.

The categories *primary material use, trade barriers* and *political stability* have the highest risk for most raw materials. Low recycling rates for several of the raw materials especially ones used in electronic devices (Graedel, 2011) explain the high risks for the category *primary material use*. Even though there is no direct correlation between *trade barriers* and *political stability*, often both categories are influenced by the country's politics. Studies have shown that even though political stability can be a cause for decreasing trade barriers, the correlation cannot be applied for all countries (Enowbi Batuo and Asongu, 2015; Bonnal and Yaya, 2015; Puig and Chan, 2016). Thus, it is adequate to consider both categories, keeping in mind that similar results are plausible. As Germany has to import almost all of its resources (Huy et al., 2014), it relies on resource rich countries. Several of these resource rich countries are characterized by governments struggling to establish strong governmental institutions (Gylfason, 2001; Hodler, 2006; Venables, 2016; Siakwah, 2017) leading to low political stability and thus often high trade barriers (also called ”resource curse”).

The categories *demand growth, mining capacity* and *physical availability* have a low influence with regard to availability constraints for most raw materials. For most raw materials *demand growth* has not been above average in the last years. Future trends (e.g. e mobility) which could increase the demand of specific raw materials are considered in the sub dimension *vulnerability* (Section 3.3.2). The mining operations of the countries from which Germany imports its raw materials are set up to last longer than 50 years and thus do not display any risk with regard to possible availability constraints. The *physical availability* refers to the total amount of an element in the Earth's crust regardless whether it is economically and technically extractable today or most likely in the future. Thus, the amount characterized as available is very high for the most raw materials and the category only plays a role for natural gas, tungsten, gold and hard coal as these materials show comparably low natural deposits.

As addressed in Section 2.1.1 the results of the import-based categories (*political stability, trade barriers, concentration of production, mining capacity and feasibility of mining operations*) can be compared to the global results (see Fig. 6). Raw materials with values higher than zero have a lower risk with regard to availability constraints compared to the global average. The overall result is marked with a black rhombus sign. For Germany lead performs much better than the global average especially in the categories *feasibility of exploration projects, trade barriers* and *political stability*. China is with 50% the main global producer of lead BGS (2014) and USGS (2015). Its mining industry is characterized by i. a. challenges related to infrastructure, community development conditions as well availability of a sufficient geological database for better exploration strategies. China currently holds the 54th place (out of 104) with regard to attractive jurisdiction (Wederman, 2004; Cervantes et al., 2013). Germany imports only 6% of its lead from China. Most supplies are shipped from Australia, Sweden and USA (United Nations, 2015; USGS 2015), for which the policy potential index (indicator which quantifies the category) performs well. The high global production shares of China and the low amount imported by Germany are also the reason for a better performance of the category *trade barriers* and *political stability*.

Raw materials with a value lower than zero have a higher risk of availability constraints than the global average. For Germany the highest risks compared to the global average occur with regard to aluminum for the categories *political stability* and *concentration of production*.

According to United Nations (2015) Germany imports 92% of its aluminum from Guinea. The country is characterized by political upheaval (Hall, 2015; Dhillon and Kelly, 2015) and is therefore political unstable. From a global perspective Guinea only produces 6% of the global amount of aluminum, 30% is produced in Australia, 19% in Indonesia and 17% in China. These countries have lower worldwide government indicator values (indicator quantifying *political stability*) as Guinea. The large amount of aluminum imported from Guinea also explains the high *concentration of production* for the import mix, whereas on global level several countries contribute to aluminum production (BGS, 2014).

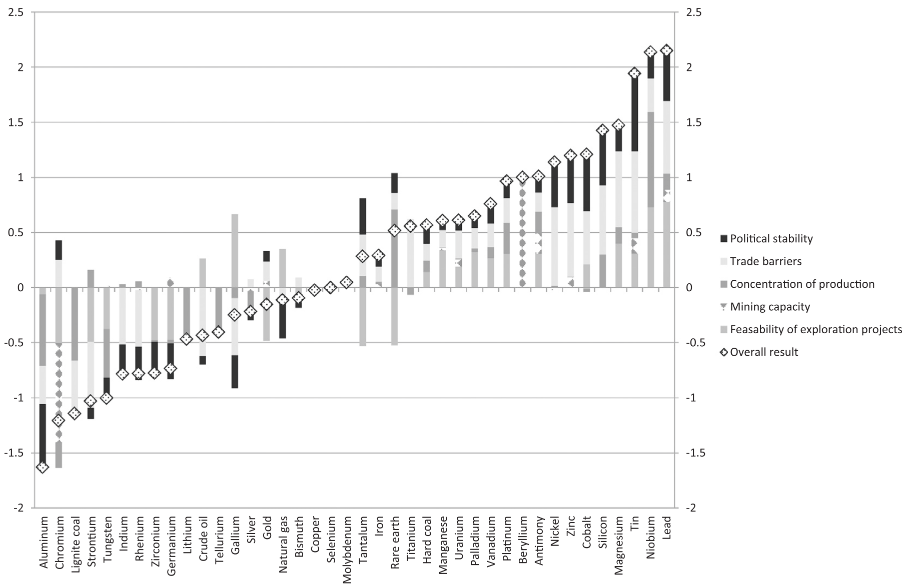

Fig. 6. Comparison of the import based and global results for the categories political stability, trade barriers, concentration of production, mining capacity and feasibility of mining operations.

3.3.2. Results of sub dimension vulnerability

In this section, it is shortly described how the categories are quantified for the case study of Germany. Then the results of the sub dimension are shown and explained.

For the categories *utilization in future technologies* and *substitutability* data by Erdmann et al. (2011) is used. Within this publication *substitutability* and *utilization in future technologies* of raw materials for the German market are assessed and clustered into values from 0 to 1. Thus, these values can be adopted without being converted. As shown in Table 2 the quantification of the category *domestically required demand* is achieved by scaling the imported amounts based on BGS (2014) and USGS (2015) to values from 0 to 1 (1 represents the highest imported amount). The imported amounts are set in relation to the global production to quantify the category *share of global production. Dependency on imports* is identified by determining the amounts of material produced within Germany based on Huy et al. (2014) and comparing them to the imports. To quantify the influence of *availability of purchasing strategies* it is determined how big the import shares are regarding countries for which these strategies exists (Ferretti et al., 2013). The share is subtracted from 1, leading to values from 0 to 1. The *economic importance* of a material is determined based on the added value to German companies utilizing the raw material based on data by Statistisches Bundesamt (2012, 2015) (Federal Statistical Office). The final results of the categories are scaled to 0–1.

The results of the individual categories are aggregated as shown in Fig. 7. The results of the sub dimension *vulnerability* are analyzed in more detail. Germany shows the lowest *vulnerability* with regard to gold, silver, copper, iron and tellurium. Uranium has the highest *vulnerability* followed by manganese, chromium, magnesium and vanadium. Exemplary the results for uranium are analyzed in more detail. Such a detailed analysis should be carried out for all or at least the raw materials with the highest vulnerability.

For uranium the category *share of global production* has the highest results. Even with the Nuclear Phase-Out Act (Gesetz zur Änderung des Atomgesetz (Deutscher Bundestag, 2011)(German parliament) Germany still relies on energy from nuclear power for its electricity, leading up to a share of about 14% nuclear energy (de Menezes and Houllier, 2015; Arbeitsgemeinschaft Energiebilanzen, 2016 (Working Group on Energy Balances)). As Germany only has small amounts of uranium resources, which are currently not extracted, they have to import uranium – 1890 t in 2015 (Statista, 2015). Therefore, the category *share of global production* (imported amount in relation to world production) is the category with the highest contribution. Furthermore, the category *economic importance* shows also high results, since uranium is utilized in the energy sector which generates a high value added (Statistisches Bundesamt, 2012, 2015). Other important categories are *dependency on imports* (due to absence of domestically resources) and *availability of purchasing strategies* (no purchasing strategies for uranium exist in Germany (Ferretti et al., 2013)). However, it can be assumed that the dependency on uranium will decrease significantly in the next years due to the Nuclear Phase-Out Act.

The results of the sub dimension *availability* and *vulnerability* are plotted in a diagram – the criticality matrix (as shown in Fig. 4). The results of both sub dimensions have to be considered to determine the *criticality* of a raw material. For example, even though the *vulnerability* of uranium is high, its risk with regard to availability constraints is rather low.

3.3.3. Results of dimension societal acceptance

The dimension *societal acceptance* consists of the two sub dimensions *compliance with social standards* and *compliance with environmental standards*. For both sub dimensions the overall results are presented. Further, the import based results are compared to the global production shares to determine if raw materials imported to Germany perform better or worse than the global average.

3.3.3.1. Results of sub dimension compliance with social standards. In this section, the results of the sub dimension *compliance with social standards* are explained in more detail. As shown in Fig. 8 lignite coal, tellurium, lithium, beryllium and zinc have the highest *compliance with social standards* (and therefore low indicator values).

Chromium, antimony, platinum, vanadium and tungsten have the lowest *compliance with social standards* (and therefore high indicator

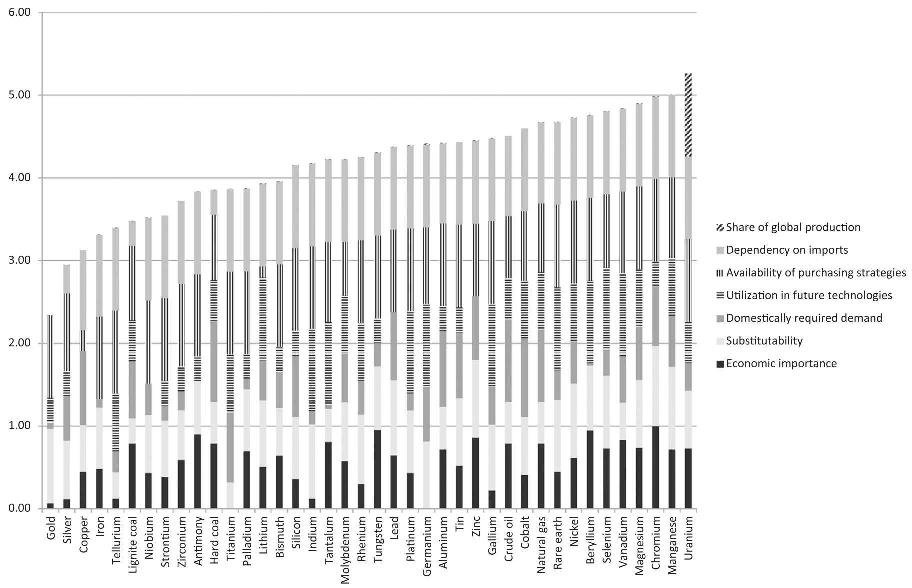

Fig. 7. Results of the sub dimension vulnerability.

values). For tungsten the category *small scale mining* has the highest impact as it is predominately imported from Bolivia (United Nations, 2015), where small scale mining operations are responsible for almost all tungsten extraction (Noetstaller, 1987; Hilson, 2002). For antimony the impact of *small scale mining* is high since it is predominantly imported from China, where it is most likely extracted within small scale mining operations (Noetstaller, 1987; Gunson and Jian, 2002; Shen and Gunson, 2006). Chromium, platinum and vanadium are prominently imported from South Africa, Russia and China – countries where human right violations and political conflicts occur (Seedat et al., 2009; Cingranelli et al., 2012; World Bank Group, 2013; Human Rights Watch, 2016). Thus, the results for these categories are high.

Further, the import based results are compared with the global results to analyze for which raw materials Germany performs better or worse than the global average. As shown in Fig. 9 Germany shows better results for the raw materials silicon, lead, rare earth, zinc and tantalum.

For silicon the categories *human right abuse* and *geopolitical risk* perform better than the global average. Germany imports its silicon mostly from Norway, Poland and France, where human right abuse as well as the geopolitical risk are small. Globally China and Russia are the biggest producer, which have comparably higher risk with regard to the considered categories.

Tin and tungsten perform worse than the global average for the

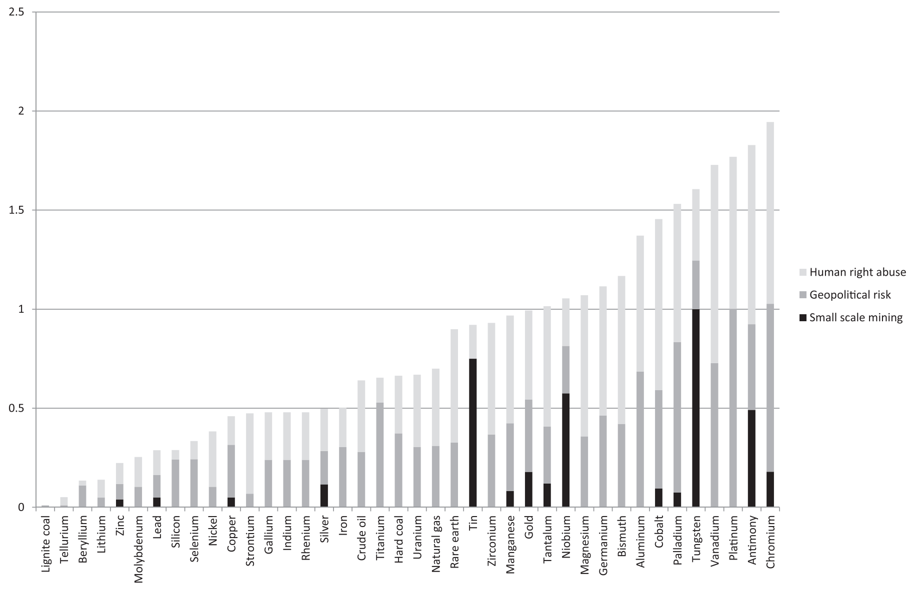

Fig. 8. Results for the sub dimension compliance with social standards.

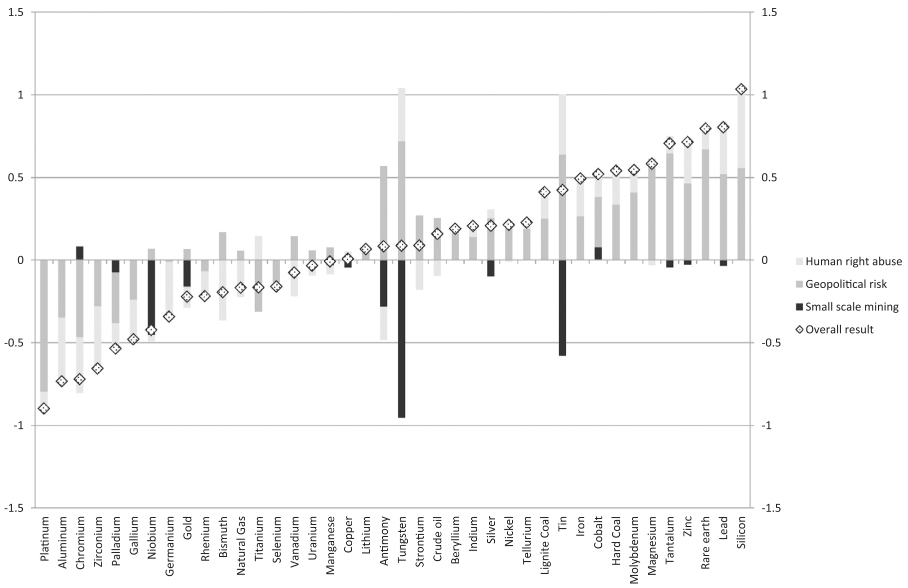

Fig. 9. Comparison of the import based and global results for the sub dimension compliance with social standards.

category *small scale mining*. As described above tungsten is imported predominantly from Bolivia, whereas from a global point of view not all tungsten is extracted within small scale mining. The same applies for tin: it is imported from Thailand and Peru (both countries are associated with small scale mining (Noetstaller, 1987; Labonne, 1996; Hentschel et al., 2002)), whereas globally only a small amount is extracted within small scale mining operations (Dorner et al., 2012). Platinum, aluminum and chromium also perform worse compared to the global average for the categories *geopolitical risk*. This can also be explained by the import mix: aluminum is predominantly imported from Guinea, whereas in the global production mix Guineas share is rather small and large amounts are produced in Australia (with 30%), China (with 18%) and Indonesia (with 19%) (USGS, 2015). As already explained, Guinea has been dealing with by political upheaval in recent years (Hall, 2015; Dhillon and Kelly, 2015) also leading to human rights violations (Human Rights Watch, 2016).

As addressed above imported chromium is predominantly mined in South Africa (a county where human right violations and political conflicts occur (Kynoch, 2005; Seedat et al., 2009)) and therefore performs worse than the global average, where the share extracted in South Africa is much smaller. The imported zirconium comes to 60% from China (and 14% from USA as well as 15% from Australia), whereas for the global production mix Australia is the country with the highest production share (50%; further 10% in USA and only 3% in China). As human right abuses occur in China (Lee, 2007; O'Brien, 2015; Pedersen and Kinley, 2016) a higher imported amount results in a higher difference of the result.

3.3.3.2. Results of sub dimension compliance with environmental standards. Following the results for the sub dimension *compliance with environmental standards* are explained. As shown in Fig. 10 tellurium, lignite coal, aluminum, selenium and lithium perform best, whereas gold, tungsten, platinum, silver and chromium perform worse. Overall the category *climate change* has only an influence on few raw materials: gold, platinum and palladium. Studies as well as current databases show that the climate change impact of these raw materials is especially high (Norgate et al., 2007; Nuss and Eckelman, 2014; Ecoinvent, 2016; Thinkstep, 2016). The high climate change impacts of gold and platinum can be explained by the high energy use for extraction due to their low ore grades (Mudd, 2007; Yang, 2009).

The results for *water scarcity* and *sensitivity of local biodiversity* are determined country specific. The category *water scarcity* is very high for tungsten as it is mined in Bolivia. Even though Bolivia is characterized as a country with medium water scarcity from a resource perspective (Berger et al., 2014), due to governmental regulations and low state of the art with regard to drinking water and wastewater treatment technology in rural areas risks related to human health exists (Spronk and Webber, 2007; Wutich and Ragsdale, 2008; Calizaya et al., 2010).

Silver shows the highest sensitivity with regard to the local biodiversity, because almost 50% of silver is imported from Argentina. Argentina is one of the most biodiverse countries in the world with several areas under protection. However, the transformation of ecosystems to agricultural areas, logging activities and oil and gas prospecting have increased in recent years. Thus, the amount of mammals, amphibians and birds listed under a category of threat has been growing as well (Grau and Diego Brown, 2000; Manrique et al., 2013; Secretariat of the Convention on Biological Diversity, 2014a).

Further, the import based results are compared to the global results to analyze for which raw materials Germany performs better or worse than the global average. As shown in Fig. 11 only for few materials Germany performs better than the global average, for most raw materials it performs worse. The overall result is marked with a black rhombus sign. *Climate change* is zero for all raw materials as the results are not influenced by import mixes, but are established on a global level.

Especially for tungsten and silver the imported raw materials perform worse. This can be explained by the import structure. As already mentioned silver is mostly imported from Argentina which has a high sensitivity with regard to the local biodiversity. On a global scale overall 58 countries mine silver worldwide with individual production shares around 14% (BGS, 2014). Due to this high amount of countries mining silver including many countries with low *sensitivity of the local biodiversity* the global average is smaller than the import based result. Tungsten is predominantly imported from Bolivia, which has a high sensitivity with regard to the local biodiversity (Finer et al., 2008; Secretariat of the Convention on Biological Diversity, 2014b).

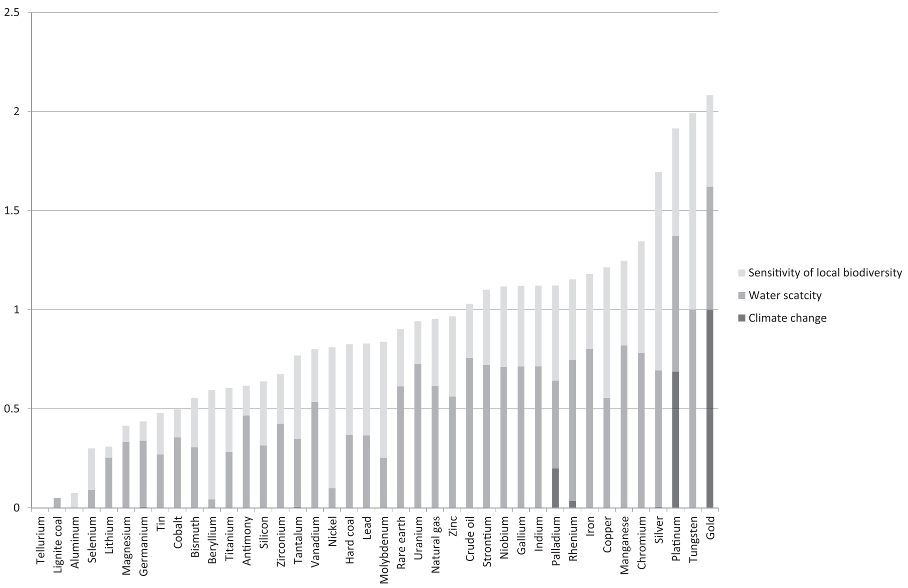

Fig. 10. Results for the sub dimension compliance with environmental standards.

Considering global production Bolivia has a small share of 2% and most of the tungsten is mined in China (80%) and Russia (10%) (BGS, 2014; USGS 2015). The *sensitivity of the local biodiversity* is smaller in China than in Bolivia, because of its smaller biodiversity richness and its higher overall conservation status (based on current trends and conditions) (World Wildlife Fund, 2012).

4. Results and discussion

The aim of the introduced SCARCE method is to provide an improved criticality assessment by considering a comprehensive set of availability indicators, and determine them based on the individual country specific import mix of countries instead of the global production mix. Furthermore, the criticality framework is expanded to include social and environmental aspects and achieve a first step towards sustainability assessment for resources.

Indicator values based on global production are provided for the categories of the sub dimensions *availability, compliance with social standards* and *compliance with environmental standards* for all 40 considered resources and raw materials. Further, the indicator values on country level are provided for the dimension *societal acceptance.* They can be used to calculate import based results for this dimension

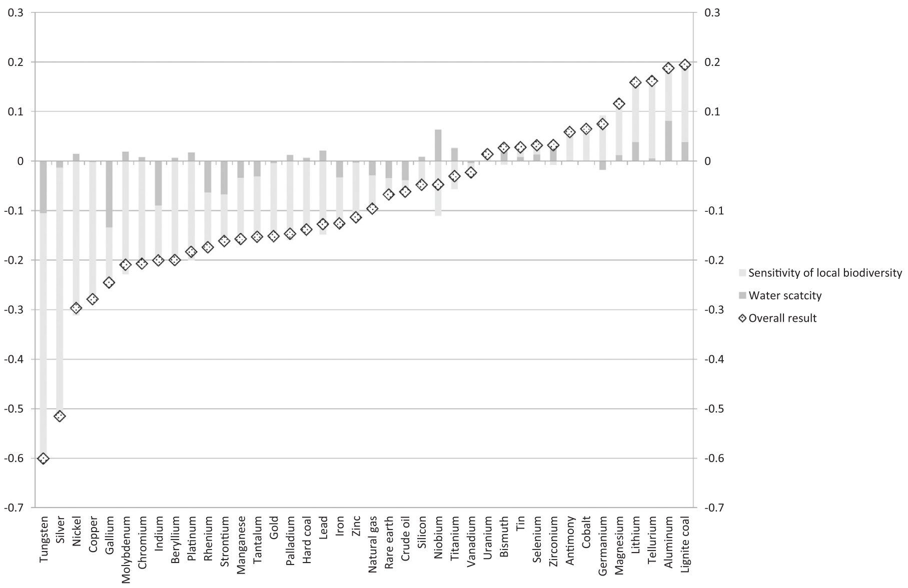

Fig. 11. Comparison of the import based and global results for the sub dimension compliance with environmental standards.

(data are provided within the supplementary material – Section 2). Providing data enhances the applicability of the SCARCE method tremendously.

The SCARCE method is applied on a case study of Germany. The plausibility of results could be shown. However, several challenges exist which are addressed and discussed in the following.

Within all sub dimension aggregation occurs to determine a single score reflecting the corresponding (sub) dimension. This means the twelve categories of the sub dimension *availability*, the seven categories of the sub dimension *vulnerability* as well as the three categories (and the corresponding indicators within the categories) of the sub dimension *compliance with social standards* and the three categories (as well as the corresponding indicators of the categories) of the sub dimension *compliance with environmental standards* are aggregated to one value respectively. For all sub categories equal weighting of all categories (and indicators) is applied as proposed in existing methodologies (e. g. Erdmann et al., 2011, Graedel et al., 2012, VDI (2013), European Commission, 2014 and Bach et al., 2016). As weighting is always subjective (Finkbeiner et al., 2014) it has to be kept in mind that the results could change if different weighting factors are applied. Further, due to the unequal amount of indicators and categories considered within the dimensions, an implicitly weighting occurs. For example, in the sub dimension availability overall 12 categories are considered, whereas in the sub dimension vulnerability only seven categories are taken into account. Thus, the individual categories of the sub dimension vulnerability contribute more to the overall result of this sub dimension. Whereas the individual categories of the sub dimension availability contribute comparably less to the overall result of the sub dimension availability. However, if interpretation of the results is followed as proposed (as shown in the case study for Germany), each category and dimension including the applied weighing schemes should be adequately reflected and can therefore be taken into consideration during interpretation and formulation of possible policy options. Further, none of the proposed approaches to determine a single-score result for criticality was applied due to the uncertainties associated with weighting. Nassar et al. (2012) showed that single score results for criticality vary significantly depending on the weighting approach applied. Thus, the aggregation to one criticality results provides limited additional value for the interpretation of the results.

All indicators applied in the SCARCE method face the challenge of underlying data quality. If the data quality is poor, higher uncertainties are associated with the indicator results. More established indicators (e.g. Enabling Trade Index (Hanouz et al., 2014)) tend to have lower uncertainties because they are improved over time. Further, data for calculating the import mix, global production as well as the indicator values are derived from different years (data used originates from the years 2010–2016).

Furthermore, so far the SCARCE method has only been applied for imported raw materials but does not take into account intermediate products (e.g. metal sheets) and final products (e.g. automotive battery), which can be influenced by availability constraints, vulnerability and societal acceptance as well (Peiró et al., 2013; Lapko et al., 2016).

To fully assess the availability of primary materials anthropogenic stocks have to be taken into account as they can lower the pressure on primary materials. No data with regard to anthropogenic material flows is currently available for Germany (it is assumed this is also the case for most other countries) (Zimmermann and Gößling-Reisemann, 2013; Schiller et al., 2015; United Nations, 2015). Thus, this aspect could not be included in the assessment. To consider this aspect at least to some extent the recycled content was included within the availability assessment. Additionally, the assessment can only be applied to primary materials. The criticality of secondary materials cannot be quantified due to missing data on international and national recycling markets for all considered materials. However, the identified categories and indicators could mostly be applied for secondary materials if sufficient market data were available.

For the sub dimensions *compliance with social and environmental standards* data on country (and sector) level is applied. Thus, no statement with regard to the status of specific mines can be made, because so far there are no indicators and data available. One important aspect missing in the sub dimension compliance with environmental standards is the occurrence of accidents during mining operation, e.g. leaking tailing ponds (Howard, 2015; Schoenberger, 2016). However, no data is available for such accidents on global level yet. Therefore, it is currently not possible to quantify them. Further, climate change impacts were determined based on global data provided by GaBi and ecoinvent. Therefore, different technologies within different countries are not taken into account. To redefine this aspect, country specific technologies should be analyzed so that specific inventory data can be derived.

The results for the category *mining capacity* are determined based on import data, considering the depletion of current mines in the specific countries from which imports occur. However, it could also be argued, that the global market will compensate depleted mines by newly developed ones and thus, that the category should rather be determined on a global level. At this point, it is not known, how and if the global market will balance out country specific mining capacities. As the import mix is compared with the global average, both results are determined and can be analyzed. As shown in Fig. 6 differences between import based and global average results are only significant for few raw materials.

The worldwide governance indicators are applied to determine the *political stability* within the sub dimension availability as well as for the geopolitical risk for the sub dimension *compliance with social standards*. Even though different indicators are used (see Sections 2.1.1 and 2.2.1) and no direct correlation occurs between these two indicator sets (see supplementary material – Section 3), often the same resources and raw materials are being quantified as having a high or respectively low risk. Thus, it should be further analyzed if one of the indicator sets could be excluded to limit the overall amount of considered indicators and categories.

Further, the Abiotic Resource Depletion indicator (Guinée et al., 1993; Oers et al., 2002) using ultimate reserves (crustal content) as a basis is applied to determine the *physical availability* of resources. As the ultimate reserves can never be extracted completely, because some stocks will always remain unavailable under all foreseeable conditions, using the ultimate reserves as a basis to determine the physical availability of resources leads to an overestimation of available resources. However, the ultimate reserves has been evaluated as the most stable and comprehensive dataset and is applied to determine the resource depletion within life cycle assessment case studies (Drielsma et al., 2016; van Oers and Guinée, 2016).

5. Conclusion

The SCARCE method enhances the *criticality* assessment of a country's resource. By establishing the new dimension *societal acceptance* social and environmental aspects are taken into account. Additionally next to metals also fossil fuels are considered, which can directly be compared to metals. This is especially important when systems are compared where fossil energy sources are replaced by renewable systems which require high amounts of certain metals (e.g. lithium and tellurium for solar energy power). Another feature of the SCARCE method is the possibility to determine certain socio-economic availability aspects (e.g. political stability) based on the specific import situation of a country. These country specific results can be compared with results determined based on the global production mix to determine if the country under consideration performs better or worse than the global average. Finally by considering additional categories for the sub dimension availability, which have not been taken into account so far (e.g. *feasibility of exploration projects*), the evaluation of socio-economic constraints to resources is improved. Thus, the SCARCE method goes beyond existing methodologies and considers the use of resources in the context of sustainable development.

Appendix A. Supplementary material

Supplementary data associated with this article can be found in the online version at doi: 10.1016/j.resourpol.2017.07.003.

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Supplementary material

Enhancing the assessment of critical resource use at the country level with the SCARCE method – case study of Germany

Content

1. Bottom-up & top down approach

Following the applied bottom-up & top-down approach is introduced. It is implemented to determine the dimensions, categories and indicators necessary for a comprehensive assessment of resource use in the context of sustainable development.

First in a bottom up approach existing methods assessing resource use are identified and evaluated regarding dimensions, categories and indicators considered. These methods are then analyzed via meta criteria, which are adopted from the publications by (Lehmann et al. 2015; Bach et al. 2016)). Based on these results relevant dimension and categories are determined and indicators are preselected.

In a top-down approach relevant aspects, not addressed by the methods assessed in the bottom-up approach, are determined. A literature review is carried out to identify aspects addressed by media and in the scientific community by analyzing case study specific impacts. These relevant aspects are then allocated to an existing dimension or a new dimension is defined. Within the dimensions categories are established and new indicators are designed.

Finally, data availability was checked for all indicators to guarantee that indicator values for a wide selection of countries and materials can be determined. This ensures that the introduced approach can be implemented into practice.

2. Indicator results

Following the calculated indicator results for 40 considered raw materials (36 metals and 4 fossil energy carriers) are provided for:

- the sub dimension *availability*, calculated based on the global production mix (status: May 2016)
- the sub-dimension *compliance with social standards*, based on the global production mix (status: January 2017)
- the categories *geopolitical risk* and *human rights abuse* of the sub-dimension *compliance with social standards* for 175 countries (status: January 2017)
- the sub-dimension *compliance with environmental standards*, based on the global production mix (status: March 2017)
- the categories *water scarcity* and *sensitivity of the local biodiversity* of the sub-dimension *compliance with environmental* standards for 175 countries (status: March 2017)

These indicators will be updated regularly. The most recent results can be found here: http://www.see.tu-berlin.de/menue/research/results/scarce_method_enhancing_the_assessment_of_critical_resource_use_on_country_level/parameter/en/

2.1 Indicator results of the sub dimension availability

The indicator results of the 12 categories for the sub dimension *availability* (based on global production data) are displayed. They are based on the approach shown in section 2.1.1 of the paper.

Category/ Raw material	Feasibility of exploration projects	Demand growth	Occurrence of co-production	Primary material use	Mining capacity	Company conc.	Price fluctuations	Conc. of reserves	Conc. of production	Trade barriers	Political stability	Physical availability
Lignite Coal	0	0	0	1.00	0	0.04	0.30	0.03	0	0	0	0
Hard Coal	0.62	0	0	1.00	0	0.04	0.30	0.04	0.13	0.67	0.48	0.45
Crude oil	0.86	0	0	1.00	0	0.04	0.17	0	0	0	0.40	0.95
Natural Gas	0.85	0	0	1.00	0	0.04	0	0	0	0.46	0	1.00
Uranium	0.68	0.02	0	0.90	0.07	0	0	0	0.06	0.76	0.45	0

Aluminum	0.67	0	0	0	0	0	0	0.03	0.08	0.66	0.43	0
Antimony	1.00	0.02	0.11	0.72	0.21	0.75	0.18	0.15	0.55	0.70	0.66	0.02
Beryllium	0	0.04	0.11	0.68	1.00	1.00	0.22	0.53	1.00	0	0	0
Bismuth	0.68	0	1.00	1.00	0	0.76	0.24	0.42	0.19	0.62	0.46	0.001
Lead	0.78	0.04	0.11	0	0.11	0	0.24	0.08	0.15	0.66	0.46	0
Chromium	0	0	0	0.66	0.1	0	0	0.07	0.11	0.79	0.54	0
Cobalt	0.82	0.07	0.45	0	0	0	0.33	0.09	0.23	1.00	1.00	0
Iron	0.000	0.031	0	0	0.04	0	0.32	0.03	0.10	0.68	0.43	0
Gallium	0.665	1.00	1.00	0	0	0.37	0.20	0.03	0	0	0	0
Germanium	0	0	1.00	0	0.1	0.74	0.16	0.03	0.25	0	0	0
Gold	0	0	0	0	0.07	0	0	0	0	0.72	0.48	1.00
Indium	0	0.03	1.00	0	0	0.462	0.237	0.116	0.129	0	0	0
Copper	0	0	0.11	0	0.02	0	0.15	0	0	0	0	0
Lithium	0	0.05	0	0.99	0	0	0	0.19	0.11	0	0	0
Magnesium	0.88	0.19	0.11	0	0	0.06	0	0.05	0.58	0.69	0.60	0
Manganese	0.84	0.03	0	0	0.03	0	0	0.04	0.04	0.75	0.50	0
Molybdenum	0	0	0	0	0.02	0	0.35	0.07	0.09	0	0	0
Nickel	0	0.08	0.11	0	0.03	0	0.30	0	0.04	0.72	0.43	0
Niobium	0.73	0	0.11	0.61	0	0.88	0.24	0.99	0.96	0.82	0.54	0
Palladium	0.83	0	0.45	0	0	0.07	0.17	1.00	0.15	0.77	0.51	0.01
Platinum	0.83	0	0.11	0	0	0.04	0	1.00	0.50	0.78	0.51	0.04
Rhenium	0	0	1.00	0.68	0	0.94	0	0.12	0.15	0	0	0.01
Selenium	0	0	1.00	0.90	0	0.17	0.31	0.04	0	0	0	0.01
Rare earth	0	0	0.45	0.99	0	0.90	1.00	0.11	0.83	0.67	0.58	0
Silver	0	0	0.45	0	0.07	0	0.29	0	0	0.69	0.48	0.02
Silicon	0	0.02	0	0	0	0.24	0	0	0.30	0.63	0.50	0
Strontium	0	0	0	0	0	0.68	0.33	0.18	0.22	0	0.40	0
Tantalum	0	0.03	0.11	0.66	0	0	0.21	0.40	0.15	0.89	0.70	0
Tellurium	0	0	1.00	0	0	0.14	0.34	0.29	0.09	0	0	0

4

Titanium	0	0	0	0	0	0	0.43	0.03	0	0.62	0	0
Vanadium	0.93	0	1.00	0	0	0.24	0.31	0.11	0.26	0.78	0.67	0
Tungsten	0.63	0	0	0	0.02	0.37	0	0.16	0.56	0.71	0.61	0
Zinc	0	0	0.11	0.60	0.10	0	0.19	0.03	0.05	0.67	0.43	0
Tin	0.83	0	0	0.61	0.14	0	0.23	0.04	0.11	0.74	0.70	0
Zirconium	0	0	1.00	1.00	0	0.83	0.41	0.27	0.14	0	0	0

2.2 Indicator results of the sub dimension compliance with social standards

The indicator results of the three categories *geopolitical risk, human rights abuse* and *small scale mining* for the sub dimension *compliance with social standards* (calculated based on global production data). They are determined according to the approach shown in section 2.2.1 of the paper.

Category/ Raw material	Geopolitical risk	Human rights abuse	Small scale mining		Category/ Raw material	Geopolitical risk	Human rights abuse	Small scale mining
Lignite Coal	0.26	0.16	0		Manganese	0.42	0.49	0.05
Hard Coal	0.71	0.50	0		Molybdenum	0.51	0.29	0
Crude oil	0.53	0.27	0		Nickel	0.32	0.27	0
Natural Gas	0.37	0.17	0		Niobium	0.31	0.20	0.12
Uranium	0.36	0.27	0		Palladium	0.45	0.54	0
Aluminum	0.34	0.30	0		Platinum	0.20	0.66	0
Antimony	1.00	0.70	0.21		Rhenium	0.17	0.09	0
Beryllium	0.28	0.04	1.00		Selenium	0.11	0.07	0
Bismuth	0.59	0.38	0		Rare earth	1.00	0.70	0
Lead	0.63	0.44	0.01		Silver	0.42	0.27	0.02
Chromium	0.38	0.58	0.26		Silicon	0.80	0.52	0
Cobalt	0.80	1.00	0.17		Strontium	0.34	0.23	0
Iron	0.57	0.43	0		Tantalum	0.93	0.71	0.08
Gallium	0	0	0		Tellurium	0.20	0.08	0
Germanium	0.45	0.32	0		Titanium	0.22	0.27	0

Gold	0.43	0.32	0.02		Vanadium	0.87	0.78	0
Indium	0.38	0.31	0		Tungsten	0.97	0.68	0.05
Copper	0.26	0.20	0.01		Zinc	0.54	0.38	0.01
Lithium	0.10	0.11	0		Tin	0.64	0.53	0.17
Magnesium	0.97	0.68	0		Zirconium	0.09	0.19	0

Next, indicator results for the categories *geopolitical risk* and *human rights abuse* are provided on country level. They can be used to calculate results based on the specific import mix of the country under consideration.

Country key	Geopolitical risk	Human rights abuse		Country key	Geopolitical risk	Human rights abuse		Country key	Geopolitical risk	Human rights abuse
AFG	1.76	0.52		GHA	0.59	0.56		NER	0.99	0.68
ALB	0.58	0.43		GRC	0.65	0.09		NGA	1.39	0.81
DZA	1.06	0.48		GRL	0.15	0.09		NOR	0.15	0.00
ASM	0.87	0.00		GRD	0.58	0.16		OMN	0.79	0.38
ADO	0.25	0.00		GUM	0.92	0.17		PAK	1.54	0.64
AGO	0.96	0.17		GTM	0.94	0.38		PLW	1.05	0.20
AIA	0.86	0.05		GIN	1.01	0.56		PAN	0.57	0.21
ATG	0.58	0.12		GNB	1.05	0.72		PNG	0.80	0.42
ARG	0.65	0.25		GUY	0.75	0.43		PRY	0.77	0.34
ARM	0.90	0.42		HTI	0.93	0.68		PER	0.78	0.48
ABW	0.30	0.06		HND	0.92	0.40		PHL	0.97	0.57
AUS	0.19	0.14		HKG	0.34	0.06		POL	0.29	0.15
AUT	0.08	0.04		HUN	0.38	0.12		PRT	0.22	0.25
AZE	1.15	0.56		ISL	0.06	0.01		PRI	0.52	0.06
BHS	0.42	0.20		IND	0.98	0.78		QAT	0.62	0.25
BHR	1.16	0.44		IDN	0.66	0.45		REU	0.97	0.21

BGD	0.90	0.67		IRN	1.20	0.64		ROM	0.52	0.09
BRB	0.31	0.15		IRQ	1.78	0.51		RUS	1.37	0.69
BLR	0.98	0.52		IRL	0.18	0.09		RWA	1.01	0.57
BEL	0.25	0.00		ISR	0.98	0.27		WSM	0.53	0.22
BLZ	0.85	0.34		ITA	0.43	0.09		STP	0.89	0.17
BEN	0.67	0.45		JAM	0.66	0.34		SAU	1.12	0.70
BMU	0.91	0.00		JPN	0.21	0.12		SEN	0.68	0.64
BTN	0.40	0.27		JEY	0.22	0.00		SRB	0.60	0.29
BOL	0.78	0.40		JOR	0.94	0.54		SYC	0.79	0.43
BIH	0.70	0.39		KAZ	0.88	0.48		SLE	0.71	0.62
BWA	0.39	0.25		KEN	1.03	0.62		SGP	0.41	0.03
BRA	0.72	0.38		KIR	0.64	0.38		SVK	0.30	0.04
BRN	0.69	0.00		PRK	1.51	0.49		SVN	0.26	0.00
BGR	0.52	0.17		KOR	0.54	0.37		SLB	0.87	0.38
BFA	0.89	0.51		KSV	0.79	0.04		SOM	1.85	0.77
BDI	1.15	0.64		KWT	0.73	0.52		ZAF	0.75	0.64
KHM	0.93	0.51		KGZ	1.00	0.50		SSD	1.84	0.12
CMR	1.10	0.60		LAO	0.84	0.13		ESP	0.38	0.14
CAN	0.14	0.04		LVA	0.41	0.12		LKA	0.90	0.69
CPV	0.58	0.21		LBN	1.26	0.55		KNA	0.57	0.08
CYM	0.49	0.00		LSO	0.71	0.34		LCA	0.48	0.16
CAF	1.75	0.87		LBR	0.83	0.59		VCT	0.46	0.09
TCD	1.26	0.77		LBY	1.61	0.74		SDN	1.72	0.78
CHL	0.37	0.17		LIE	0.13	0.00		SUR	0.81	0.20
CHN	1.10	0.68		LTU	0.38	0.08		SWZ	0.97	0.37
COL	1.15	0.52		LUX	0.11	0.00		SWE	0.15	0.04
COM	1.13	0.55		MAC	0.57	0.00		CHE	0.10	0.05
ZAR	1.59	1.00		MKD	0.74	0.17		SYR	1.98	0.64
COG	1.00	0.17		MDG	0.76	0.78		TWN	0.41	0.45

COK	1.04	0.00		MWI	0.65	0.68		TJK	1.11	0.11
CRI	0.36	0.21		MYS	0.59	0.55		TZA	0.76	0.44
CIV	1.01	0.77		MDV	0.85	0.34		THA	1.06	0.55
HRV	0.43	0.20		MLI	1.13	0.47		TMP	0.68	0.04
CUB	0.88	0.37		MLT	0.33	0.05		TGO	0.83	0.49
CYP	0.48	0.11		MHL	1.11	0.20		TON	0.67	0.20
CZE	0.21	0.00		MTQ	0.90	0.06		TTO	0.64	0.12
DNK	0.09	0.00		MRT	1.02	0.64		TUN	0.80	0.47
DJI	1.11	0.36		MUS	0.33	0.46		TUR	1.16	0.65
DMA	0.45	0.00		MEX	1.02	0.36		TKM	1.10	0.56
DOM	0.73	0.32		FSM	0.55	0.30		TUV	0.53	0.17
ECU	0.76	0.36		MDA	0.71	0.63		UGA	0.96	0.62
EGY	1.29	0.46		MNG	0.52	0.23		UKR	1.45	0.38
SLV	0.80	0.44		MNE	0.62	0.27		ARE	0.74	0.30
GNQ	1.02	0.72		MAR	0.89	0.44		GBR	0.41	0.04
ERI	1.27	0.96		MOZ	0.77	0.60		USA	0.55	0.11
EST	0.36	0.04		MMR	1.13	0.46		URY	0.33	0.26
ETH	1.15	0.56		NAM	0.52	0.53		UZB	1.09	0.53
FJI	0.87	0.33		NRU	1.20	0.00		VUT	0.65	0.22
FIN	0.12	0.00		NPL	0.87	0.56		VEN	1.21	0.32
FRA	0.43	0.04		NLD	0.19	0.00		VNM	0.87	0.38
GUF	0.39	0.00		ANT	1.04	0.00		VIR	0.89	0.00
GAB	0.83	0.51		NCL	1.04	0.00		WBG	1.39	0.00
GMB	0.94	0.73		NZL	0.04	0.00		YEM	1.74	0.83
GEO	0.73	0.57		NIC	0.76	0.41		ZMB	0.63	0.49
DEU	0.20	0.05						ZWE	1.09	0.91

2.3 Indicator results of the sub dimension compliance with environmental standards

The indicator results of the three categories *water scarcity*, *sensitivity of the local biodiversity* and *climate change* (calculated based on global production data) for the sub dimension *compliance with environmental standards*. They are determined according to the approach shown in section 2.2.1 of the paper.

Indicator/ Raw materials	Water scarcity	Sensitivity of the local biodiversity	Climate change		Indicator/ Raw materials	Water scarcity	Sensitivity of the local biodiversity	Climate change
Lignite Coal	0.03	0	0		Manganese	0.16	0.43	2.34E-04
Hard Coal	0.09	0.46	4.23E-06		Molybdenum	0.07	0.59	1.16E-03
Crude oil	0.15	0.27	2.47E-06		Nickel	0.04	0.71	1.52E-04
Natural Gas	0.13	0.34	7.90E-06		Niobium	0.14	0.41	2.19E-04
Uranium	0.15	0.22	3.90E-04		Palladium	0.10	0.48	2.00E-01
Aluminum	0.02	0.08	1.38E-04		Platinum	0.14	0.54	6.87E-01
Antimony	0.10	0.15	2.26E-04		Rhenium	0.14	0.41	3.60E-02
Beryllium	0.03	0.55	3.41E-03		Selenium	0.04	0.21	5.37E-05
Bismuth	0.07	0.25	3.39E-04		Rare earth	0.13	0.29	3.14E-05
Lead	0.08	0.46	3.42E-05		Silver	0.14	1.00	5.48E-03
Chromium	0.16	0.56	4.98E-04		Silicon	0.08	0.32	8.05E-05
Cobalt	0.08	0.14	1.68E-04		Strontium	0.15	0.38	6.65E-04
Iron	0.16	0.38	2.85E-05		Tantalum	0.08	0.42	4.93E-03
Gallium	0.14	0.41	3.15E-03		Tellurium	0.02	0	1.26E-04
Germanium	0.08	0.10	4.75E-03		Titanium	0.07	0.32	2.82E-04
Gold	0.13	0.46	1.00E+00		Vanadium	0.11	0.27	1.90E-04
Indium	0.14	0.41	3.61E-03		Tungsten	0.19	0.99	2.21E-04
Copper	0.12	0.66	7.03E-05		Zinc	0.12	0.41	8.21E-05
Lithium	0.06	0.06	2.73E-03		Tin	0.07	0.21	9.25E-05
Magnesium	0.08	0.08	5.81E-04		Zirconium	0.09	0.25	8.10E-05

Next, the results for the categories *water scarcity* and *sensitivity of the local biodiversity* are provided on country level to be able to calculate results based in the specific import mix of the country under consideration.

Country key	Water scarcity	Sensitivity of the local biodiversity		Country key	Water scarcity	Sensitivity of the local biodiversity		Country key	Water scarcity	Sensitivity of the local biodiversity
AFG	0.19	0.26		GMB	0.02	0.15		NOR	0.00	0.26
ALB	0.01	0.37		GEO	0.01	0.27		OMN	0.18	0.19
DZA	0.06	0.27		DEU	0.01	0.15		PAK	0.23	0.28
AGO	0.03	0.31		GHA	0.02	0.16		PLW	0.00	0.70
ATG	0.00	0.54		GRC	0.02	0.30		PAN	0.00	0.58
ARG	0.07	0.65		GRD	0.00	0.55		PNG	0.00	0.60
ARM	0.24	0.25		GTM	0.01	0.36		PRY	0.01	0.61
AUS	0.15	0.37		GIN	0.01	0.17		PER	0.14	0.48
AUT	0.00	0.31		GNB	0.01	0.25		PHL	0.04	0.46
AZE	0.12	0.30		GUY	0.00	0.50		POL	0.02	0.15
BHS	0.00	0.74		HTI	0.01	0.56		PRT	0.02	0.32
BHR	0.35	0.32		HND	0.00	0.41		QAT	0.43	0.18
BGD	0.24	0.25		HUN	0.03	0.17		ROM	0.38	0.21
BRB	0.00	0.55		ISL	0.00	0.55		RUS	0.08	0.32
BLR	0.00	0.10		IND	0.17	0.27		RWA	0.01	0.31
BEL	0.01	0.15		IDN	0.00	0.48		SAU	0.16	0.11
BLZ	0.00	0.36		IRN	0.21	0.25		SEN	0.08	0.15
BEN	0.03	0.10		IRQ	0.25	0.23		SRB	0.01	0.21
BMU	0.00	0.44		IRL	0.01	0.29		SYC	0.00	0.69
BTN	0.02	0.56		ISR	0.05	0.21		SLE	0.00	0.26
BOL	0.19	0.50		ITA	0.01	0.35		SGP	0.00	0.47
BIH	0.01	0.37		JAM	0.01	0.57		SVK	0.01	0.27
BWA	0.24	0.42		JPN	0.00	0.37		SVN	0.00	0.31

BRA	0.23	0.36		JOR	0.61	0.22		SLB	0.00	0.80
BGR	0.36	0.25		KAZ	0.19	0.18		SOM	0.37	0.34
BFA	0.73	0.11		KEN	0.11	0.36		ZAF	0.18	0.36
BDI	0.01	0.26		KIR	0.00	0.46		KOR	0.00	0.30
KHM	0.03	0.36		KWT	0.20	0.30		ESP	0.03	0.27
CMR	0.01	0.41		KGZ	0.22	0.30		LKA	0.19	0.55
CAN	0.01	0.15		LA	0.01	0.46		SDN	0.44	0.24
CPV	0.00	0.71		LVA	0.01	0.10		SUR	0.00	0.50
CAF	0.01	0.27		LBN	0.05	0.29		SWZ	0.06	0.38
TCD	0.38	0.23		LSO	0.16	0.39		SWE	0.01	0.11
CHL	0.06	0.55		LBR	0.00	0.26		CHE	0.00	0.30
CHN	0.09	0.17		LBY	0.08	0.23		SYR	0.10	0.37
COL	0.00	0.30		LTU	0.01	0.10		TWN	0.00	0.59
COM	0.00	0.71		LUX	0.01	0.13		TJK	0.18	0.31
COG	0.00	0.38		MKD	0.02	0.24		TZA	0.68	0.35
CIV	0.01	0.32		MDG	0.02	0.43		THA	0.14	0.45
HRV	0.01	0.29		MWI	0.04	0.28		TMP	0.00	0.47
CUB	0.26	0.50		MYS	0.00	0.48		TGO	0.02	0.12
CYP	0.02	0.43		MLI	0.10	0.21		TON	0.00	0.51
CZE	0.01	0.14		MLT	0.00	0.32		TTO	0.00	0.60
ZAR	0.00	0.16		MRT	0.08	0.26		TUN	0.26	0.26
DNK	0.01	0.24		MUS	0.01	0.54		TUR	0.10	0.33
DJI	0.45	0.47		MEX	0.21	0.39		TKM	0.18	0.18
DMA	0.00	0.54		MDA	0.29	0.11		UGA	0.01	0.18
DOM	0.01	0.56		MNG	0.19	0.32		UKR	0.24	0.13
ECU	0.27	0.53		MNE	0.01	0.46		ARE	0.35	0.13
EGY	1.00	0.24		MAR	0.21	0.27		GBR	0.01	0.28
SLV	0.00	0.36		MOZ	0.07	0.43		USA	0.03	0.34
GNQ	0.00	0.42		NAM	0.26	0.43		URY	0.00	0.17

ERI	0.35	0.32		NPL	0.03	0.49		UZB	0.27	0.21
EST	0.01	0.10		NLD	0.02	0.15		VEN	0.01	0.48
ETH	0.14	0.28		NZL	0.01	0.51		VNM	0.00	0.38
FJI	0.00	0.62		NIC	0.02	0.42		YEM	0.46	0.21
FIN	0.01	0.06		NER	0.13	0.23		ZMB	0.91	0.36
FRA	0.01	0.18		NGA	0.25	0.17		ZWE	0.27	0.39
GAB	0.00	0.34								

3. Correlation of Worldwide Governance Indicators

The Worldwide Governance Indicators (Kaufmann et al. 2011; World Bank Group 2013) are applied to determine the category *political stability* within the sub dimension *availability* as well as for the categories *geopolitical risk* within the sub dimension *compliance with social standards*. Indicators quantifying government effectiveness, regulatory quality, rule of law and control of corruption are applied to determine *availability* constraints due to *political stability*. The indicators voice & accountability and no violence are applied for the assessment of *geopolitical risk* in the *societal dimension*.

The same raw materials are identified as having a high or respectively low risk within both categories. Thus, a correlation analysis was carried out, as presented by (Berger and Finkbeiner 2011) to make sure no direct correlation between the indicator sets occur. The result of the correlation analysis is shown in **Figure 1**. It can be seen that the correlation coefficient (R^2) is low with 0.6. This reflects a low correlation and thus a weak relationship between the two indicator sets.

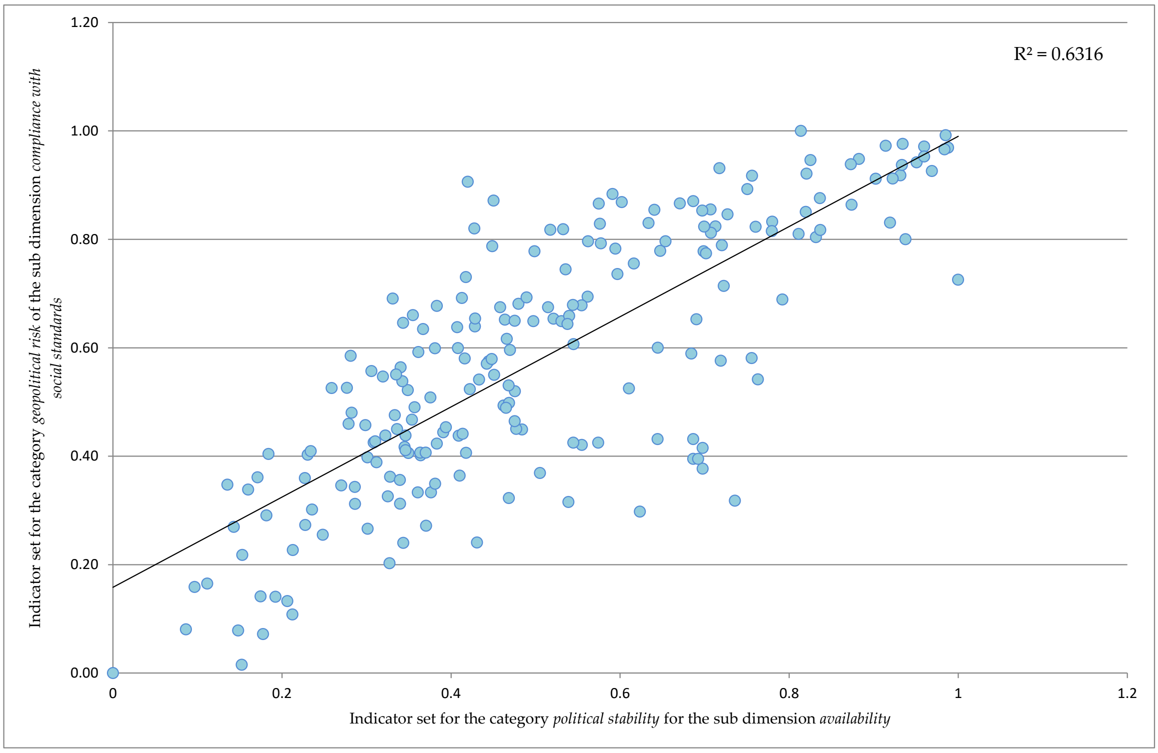

Figure 1: Result of correlation analysis for the indicator set for the categories *political stability* and *geopolitical risk*

4. References

Bach V, Berger M, Henßler M, et al (2016) Integrated method to assess resource efficiency - ESSENZ. J Clean Prod. doi: 10.1016/j.jclepro.2016.07.077

Berger M, Finkbeiner M (2011) Correlation analysis of life cycle impact assessment indicators measuring resource use. Int J Life Cycle Assess 16:74–81. doi: 10.1007/s11367-010-0237-7

Kaufmann D, Kraay A, Mastruzzi M (2011) The Worldwide Governance Indicators: Methodology and Analytical Issues. Hague J Rule Law 3:220–246. doi: 10.1017/S1876404511200046

Lehmann A, Bach V, Finkbeiner M (2015) Product environmental footprint in policy and market decisions: Applicability and impact assessment. Integr Environ Assess Manag 11:417–424. doi: 10.1002/ieam.1658

World Bank Group (2013) The Worldwide Governance Indicators. http://info.worldbank.org/governance/wgi/index.aspx#home.

3.5 Complementary publications

Within this chapter complementary publications - three journal publications, one book chapter and two conference contributions – complementing the five core publications of the thesis, are shortly introduced.

3.5.1 Analyzing existing methods for life cycle impact assessment

The following three publications analyze the state-of-the-art of life cycle impact assessment (LCIA) methods, including the assessment of resource use:

V. Bach, M. Finkbeiner (2016): **Approach to qualify decision support maturity of new versus established impact assessment methods—demonstrated for the categories acidification and eutrophication**, International Journal of Life Cycle Assessment, 22 (3), pp 387–397.

The final publication is available at Springer via: 10.1007/s11367-016-1164-z

This publication proposes an approach to determine maturity of new impact assessment methods compared to established ones and analyses existing methods for theme categories acidification and eutrophication. Based on the analysis, methods are identified to be included in the dimension environmental impacts for ESSENZ and environmental constrains for BIRD (contribution to research targets *1b* and *2b*).

A. Lehmann, V. Bach, M. Finkbeiner (2015): **Product environmental footprint in policy and market decisions – applicability and impact assessment**, Integrated Environmental Assessment and Management, Volume 11, Issue 3, Pages 417–424

DOI: 10.1007/s11367-015-0975-7

Within this publication life cycle impact assessment methods for 14 impact categories, which are predefined by the Product Environmental Footprint method are analyzed with regard to their overall maturity. Based on the analysis indicators are identified to be included in the dimension environmental impacts for ESSENZ and environmental constrains for BIRD (contribution to research targets *1b* and *2b*).

The publication can be found in appendix 10.4.

L. Schneider, V. Bach, M. Finkbeiner (2016): **LCA Perspectives for Resource use Efficiency Assessment**, in: LCA Compendium - The Complete World of Life Cycle Assessment –Special Types of Life Cycle Assessment, Springer: Dodrecht, The Netherlands, pp 179-218, ISBN 978-94-017-7608-0

DOI: 10.1007/978-94-017-7610-3_5

Within this publication, existing methods to assess (abiotic and) biotic resource use (covered by the area of protection *natural resource use* in LCA) are analyzed and methodological gaps are identified. Further, relevant aspects and existing indicators are identified for the assessment of (abiotic and) biotic resource use and raw materials on product level (contribution to research targets *2a* and *2b* as well as *3a*).

Due to copyright issues this publication cannot be displayed here.

3.5.2 Case studies applying ESSENZ

The following publications show the application of ESSENZ within two case studies:

V. Bach, M. Berger, T. Helbig, M. Finkbeiner (2015): **Measuring a product's resource use efficiency – a case study of smartphones**, Proceedings of CILCA 2015 – VI International Conference on Life Cycle Assessment, March 13-16, Lima, Peru, ISBN 978-9972-674-11-2

Within this publication the ESSENZ methodology is applied to the case study of smartphones (contribution to research target *1e*). It could be demonstrated that gold is the most geological scarce metal as its worldwide availability is low. However, considering socio-economic aspects other metals e. g. rare earths are of more concern.

The publication can be found in appendix 10.4.

M. Henßler, V. Bach, M. Berger, M. Finkbeiner, K. Ruhland (2016): **Resource use Efficiency Assessment—Comparing a Plug-In Hybrid with a Conventional Combustion Engine**, Resources, 5(1), 5

DOI: 10.3390/resources5010005

Within this publication ESSENZ is applied to the case study of a Mercedes-Benz C-Class (W 205) passenger car, which can be powered by a conventional petrol engine (C 250) or by a Plug-In Hybrid using electric motor and petrol engine (C 350 e) (contribution to research target *1e*). It could be shown, that the use of more and a broader variety of materials for the Plug-In-Hybrid leads to more potential supply restrictions, whereas advantages with regard to climate change and summer smog arise due to fewer use of fossil energy carriers.

The publication can be found in appendix 10.4.

3.5.3 Enhancement of the ESSENZ methodology

V. Bach, M. Henßler, M. Berger, K. Ruhland, L. Schneider and M. Finkbeiner (2017): **Integrated method to assess resource use use in the context of sustainable development (ESSENZ+)**, LCM Proceedings

The original ESSENZ methodology was developed during the years 2013 until 2015. Since then the author has developed other approaches (BIRD and SCARCE), improving the indicators applied in ESSENZ for the dimension societal acceptance. These improved indicators are implemented into the ESSENZ+ methodology to allow for a better assessment of resource and raw material use in the context of sustainable development (contribution to research target *1c* and *1d*).

Indicator values for the new categories can be found in appendix 10.2

The publication can be found in appendix 10.4.

4. Discussion & outlook

Within this chapter, it is shown how the identified gaps (see chapter 1.4) are addressed in this thesis. Further, with regard to the introduced methodologies remaining challenges are introduced (see chapter 4.1). Several of them are already addressed in the discussion part of the publications of this thesis. They are addressed here again due to their importance and because additional ideas how to solve them are proposed. Further, the approaches published at the same time or later than the methodologies of this thesis as well as methodological trends are discussed (see chapter 4.2). Their significance for the methodologies of this thesis is outlined. Finally, an outlook is given with regard to aspects beyond the challenges addressed in this thesis, but are necessary to achieve a comprehensive assessment of resource and raw material use (see chapter 4.3).

4.1 Key results and remaining challenges

Within this chapter it is shown how the identified gaps of chapter 1.4 are addressed within this thesis. Further, remaining challenges are addressed.

The ESSENZ (and ESSENZ+) as well as the BIRD methodology allow for a more comprehensive assessment of abiotic and biotic resources on product level, because they consider criticality aspects, environmental impacts and social issues additionally to the mass of the used resource.

With regard to the assessment of supply restrictions of abiotic resources ESSENZ (and ESSENZ+) takes into account additional relevant supply restrictions, which have not been accounted for so far (how these aspects are selected is shown in the ESSENZ publication). These include price variation, political stability, feasibility of exploration projects and mining capacity.

Further, the CFs measuring supply risks, determined based on the distance-to-target approach, take into account indicator values, category specific targets and global production amounts of each considered material. They are then scaled up to the highest global production value of the raw material portfolio considered. The target values are determined based on an iterative approach, which includes a comprehensive stakeholder survey (with over 200 people from industry, academia, NGOs, consultancies and others) and expert judgment to make sure that they are reliable in indicating which raw materials might face supply restrictions. However, if the target is set too low, materials with no risk would be classified as risky and an overestimation of the limitations in the product system occurs. If it is set too high, underestimations could occur. Target values should therefore be included in the interpretation of results. Raw materials, which for example are mined in regions with laws and regulations facilitating mining of resource and raw material use (addressed in the category feasibility of mining operations) do not face supply restrictions, and can be excluded from the analysis. The established CFs can be integrated into LCA.by multiplying them with the mass of the resources used in the considered product system, without the mass being the dominating factor.

The indicators identified to determine supply restrictions face the challenge of the underlying data quality. If the data quality is poor, the indicator will have greater uncertainties compared to when the underlying data is good. However, established indicators (e.g. worldwide governance indicators), which are used more frequently, tend to have lower uncertainties as they are improved over time. Correlations and mutual dependencies of supply risk indicators are examined by correlation analysis as presented by Berger and Finkbeiner (2011), but could not be identified. However, interconnections between the aspects the indicators quantify are well known, e.g. in political stable countries the feasibility of mining operations is usually better than for countries with unstable systems (Dondeyne et al. 2009; Spiegel 2012; Prno and Scott Slocombe 2012; Cervantes et al. 2013). These interconnections are so far not considered within the chosen indicators and developed CFs.

Supply risk aspects are determined based on underlying data of past years. It is assumed that these result also reflects the current situation (and partly the situation in the future, because data are only updated every three years). Thus, unpredictable events like revolutions cannot be determined are not reflected in the provided CFs. To address possible future events, forecasts could be included. Forecasts however also have uncertainties. Thus, for every category it has to be further analyzed if indicator values based on forecasts or past events are more adequate.

Further, the assessment of co-products of resource extraction within the category *occurrence of co-products* has to be reevaluated. As shown by Schrijvers (2017) often co-products, for which the market demand is lower than the supply, are stockpiled and therefore do not have a possible higher risk of limited availability compared to materials mined as main materials.

Further, as shown in chapter 1.3 newly established methods consider also vulnerability aspects on product level by considering the same aspects as applied for assessing the vulnerability of companies and regions. Advantages and disadvantages as well as remaining challenges of including vulnerability aspects into the assessment of socio-economic availability of resources are discussed in chapter 4.2.

Within ESSENZ (and ESSENZ+) fossil raw materials (natural gas, crude oil, lignite coal and hard coal) are considered and CFs and indicator values are provided for the same dimensions and categories as for metals. Thus, a direct comparison of fossil raw material use and metal use can be accomplished and trade-offs can be identified (e.g. when comparing different mobility solutions: e.g. conventional engine and e-mobility).

Even though ESSENZ is designed to determine supply risk along the entire supply chain, most case studies applying ESSENZ only consider the first three supply chain stages (occurrence in nature, mining, processing). To cover additional supply chain stages data has to be gathered and additional indicators as well as CFs have to be calculated. This is not done for the published case studies applying ESSENZ (Bach et al. 2015a; Henßler et al. 2016). However, internal studies indicate that ESSENZ can be used to determine supply risk in every supply chain stage.

Further, for a more precise assessment of supply restrictions along the supply chain exact production locations and involved companies have to be known. If this is the case, more precise indicator values could be determined. However, often these data are not available and therefore such a specific assessment is not possible. This aspect is discussed further in chapter 4.2.

The applicability of ESSENZ is proven by being implemented within the current “Umweltzertifikat Mercedes-Benz E-Klasse” of the Daimler AG (Daimler AG 2016). The resource and raw material use and related impacts of three cars (E 350 CGI, conventional petrol engine, E 350 e, plug-in hybrid powered by European electricity grid mix and E 350 e, plug-in hybrid powered by electricity from hydro power) considering the entire life cycle is analyzed and results are displayed. These results are also presented within the publication of Bach et al. (2017c) in appendix 10.4.

Further, ESSENZ and BIRD can be applied to identify opportunities to improve resource and raw material use impacts over the life cycle of products. Additionally companies are able to identify possible availability constraints, which can influence their production and overall efficiency and set up actions to counteract any constraints. SCARCE is able to support policy to define national and international strategies and set up efficient laws and regulations.

The assessment of biotic resource depletion is enhanced by the BRDI provided in BIRD, which next to the renewability rate also considers the extraction rate, vulnerability of the extracted species and resource stocks. Further, BIRD provides a framework considering five dimensions (physical, socio-economic, abiotic, social and environmental constraints) with overall 19 indicators to comprehensively assess availability restrictions along the supply chain. The CFs of the dimensions socio-economic constraints are determined based on the distance-to-target approach (as applied in ESSENZ) and therefore allow for integration into LCA.

One of the abiotic constraints is the availability of phosphorus, which is considered within BIRD only by the sum of phosphorus used in the considered product systems as it is usually done in LCA case studies. However, the availability of the resource use phosphor as such is not taken into account. Further, it should be considered if the phosphorus is taken from primary (extracted from lithospheric deposits) or secondary sources (e.g. recovery from sewage treatment facilities). Loss of phosphorus due to run-off should also be addressed, but is currently not taken into account, because only the used amount is considered. Run-off of phosphorus is indirectly taken into account within impact assessment methods for eutrophication. However, these approaches are based on global or European data and do not consider specific situations on agricultural fields. First solutions to tackle this challenge could be to link existing eutrophication models with specific field data and to include phosphorus sources.

An approach to adequately determine maturity and applicability of assessment methods is developed and is complemented by an in-depth analysis of existing methods to assess eutrophication and acidification. Outcomes of both analyses are used to determine which indicators and methods to use for the assessment of environmental impacts within ESSENZ (and ESSENZ+) as well as BIRD. The applied criteria to determine maturity and applicability of impact assessment methods have to be adapted when LCIA methods and inventory data are further enhanced. For example, LCIA methods for assessing water use do not take the source of water (e.g. ground water, surface water, etc.) into account due to the lack of data in common LCA databases (Berger et al. 2016). If this information is available in the future, LCIA methods addressing the impacts of different water sources used should be applied. This also means that additional criteria with regard to assessing different water sources have to be included in the assessment scheme.

Even though several impact assessment categories are taken into account in ESSENZ and BIRD, important aspects are still missing, e.g. land use, soil quality, toxicity or loss of biodiversity. These categories are excluded as mature impact assessment method are lacking (Joint Research Centre 2011; Lehmann et al. 2015). However, newly developed methods should be evaluated regarding their applicability and maturity and if they can be included in future updates of the methodologies. For land use (Beck et al. 2010; Bos et al. 2016) and biodiversity (Winter et al. 2017) promising approaches were developed recently.

By not considering benchmarks and therefore allowing for the assessment of all occurring social impacts, improvements to the SRSE indicator are made. The improved indicator is applied in ESSENZ and BIRD. As the approach introduced in SCARCE can be seen as an improvement of the approaches presented in BIRD and ESSENZ and should therefore be included in the methodologies in the future. Next to the assessment of compliance with social standards also the fulfillment of environmental standards is addressed in the provided methodologies (ESSENZ, BIRD and SCARCE). Again, the approach provided within SCARCE can be seen as an enhancement of the approaches provided in ESSENZ and BIRD and should be included in the methodologies in the future. First steps to do include these new indicators provided by SCARCE are carried out by setting up the ESSENZ+ methodology (Bach et al. 2017c). Indicator values to apply the new approach are provided in appendix 10.2 of this thesis. However, several challenges still exist: the applied indicators still do not cover all relevant issues, e.g. tailing ponds accidents, which lead to severe impacts within all environmental compartments (Moskowitz 2014; Schoenberger 2016).

Further, an 8-step approach is established to combine assessment methodologies of different resource types in a consistent way and to allow for the comparison of abiotic and biotic resources. This approach is applied to ESSENZ and BIRD setting up a combined methodology able to assess product systems using abiotic (fossils, metals and minerals) as well as biotic resources. However, to apply the provided approach adequately practitioners need to have proper understanding of the methodologies considered.

On regional level the assessment of abiotic resource and raw material use is improved by establishing the SCARCE methodology, which considers criticality aspects, environmental impacts and social issues in addition to the mass of the used resource. SCARCE also takes into account the physical availability of abiotic resources by applying the ADP indicator. Further, the additional relevant supply restrictions company concentration, feasibility of mining operations and primary material use are taken into account for a comprehensive assessment of the socio-economic availability. The indicators are based on the distance-to-target approach and therefore reflect to which extent resources and raw materials face availability constraints. The method also considers fossil fuels and therefore allows for a comparison of fossil raw materials and metals and identify trade-offs.

Existing assessment approaches to include environmental impacts are improved by setting up overarching indicators valid for all countries. These are sensitivity of local biodiversity and water scarcity. Further, climate change impacts are included to account for resource and raw material specific environmental impacts. By establishing applicable indicators for assessing social aspects, impacts can be determined for all ore producing countries. These aspects are small scale mining, geopolitical risk and human right abuse. However, even though the assessment of social and environmental impacts is improved, aspects have to be analyzed further to guarantee they account for all relevant environmental and social impacts.

The presented methodologies of this thesis to assess abiotic resource and raw material use on product level (ESSENZ (Bach et al. 2016a) and regional level

(SCARCE (Bach et al. 2017b)) as well as the methodology to assess biotic resource and raw material use on product level (BIRD (Bach et al. 2017a)) address several dimensions and categories. Thus, the determined overall outcome consists of a large number of indicator results. This makes adequate interpretation as well as communication of outcomes, especially to lay men, challenging. Interpreting multi-indicator results as well as communicating them could be enhanced, by aggregating the results. However, aggregation of results is only possible when weighting factors are applied. Within the SCARCE publication (Bach et al. 2017b) equal weighting is applied to determine single-score results for the dimensions availability and vulnerability, which is necessary to determine the overall criticality. Weighting is per se a subjective choice reflecting preferences of the involved stakeholder group (Finkbeiner et al. 2014). Different weighting sets are proposed for aggregating the results for the dimensions supply restrictions and vulnerability (e.g. Morley and Eatherley (2008), Graedel et al. (2012) and Knašytė et al. (2012)), but Nassar et al. (2012) have shown that their influence on the result is significant. Thus, every criticality assessment shows uncertainties, which are so far not determined sufficiently. To minimize the uncertainties of weighting, SCARCE calls for the interpretation of the individual categories. Within all three methodologies the results of the availability dimension are presented by stacked columns (for SCARCE and BIRD this approach is applied to some of the other dimensions as well). This way, equal weighting of the considered categories can be implied. Currently no commonly agreed upon weighting set is available. Several solutions are possible to tackle this challenge. A weighting set for the different dimensions could be established including all relevant stakeholder groups. However, such a process has been proven to be very difficult (or even deemed to be not possible). Establishing a weighting set for environmental impacts (in the context of LCA) has been an endeavor for several decades now without any commonly agreed outcome (Schmidt and Sullivan 2002; Pizzol et al. 2017). Thus, it might be equally (or even more) challenging to derive weighting factors for the availability dimension. Also within the ESSENZ project a survey of the involved project partners was carried out to establish a weighting set for the dimensions socio-economic availability and environmental impacts. However, even in such a small group (six project partners) the opinions with regard to the importance of the considered aspects varied significantly. Thus, it was decided that stakeholders should determine their own weighting approach when applying ESSENZ. Another possibility could be to determine lead indicators of each dimension, representing the most relevant aspect. Lead indicators could be identified specific to the system under consideration as well as for the materials taken into account. First approaches to determine relevant aspects are carried out by Guinée (2015), Neugebauer et al. (2015b) and European Commission (2017) and can be used as a starting point to develop an adequate approach to determine relevance. Further, the partial order method (Bruggemann et al. 2014) could be applied, which can be used to determine an order of priority of the considered categories. A first approach is introduced by Bach et al. (2015b) and Akter et al. (2017), but needs further refinement to be used as a method to identify lead indicators.

Another key challenge identified is the comparison of resource use depletion (or physical availability) across different resource use types. The depletion of abiotic resource use is mostly assessed by the ADP (baseline approach – based on ultimate reserves) indicator (as addressed in chapter 1.3). However, the results of minerals and metals (referred to as elementary) and fossils cannot be compared, because the corresponding indicators are calculated differently. For metals and minerals (elements), the extraction rate and the ultimate reserves are included in kilogram, whereas for fossil energy carriers the production rate and ultimate reserves are considered in mega joule. This is also apparent in the CFs, which vary significantly: for fossil fuels the CFs range from 10 to 50, whereas the CFs for metals and minerals of most materials are smaller than one (Guinée et al. 2002; van Oers et al. 2002).

Further, also the indicators assessing biotic resource use are established differently than the indicators for the assessment of fossils and mineral raw materials and therefore do not allow for comparison. The depletion of these resource use cannot be compared by taking the same parameters into account. Fossil raw materials are burned in 80% of all cases and can therefore not be recovered, whereas the loss (due to dissipation) of mineral raw materials is comparably low. New methods are proposed to tackle these challenges, e.g. by taking into account anthropogenic stocks (e.g. AADP (Schneider et al. 2015)) or dissipation of materials (e.g. Zimmermann and Gößling-Reisemann (2013) and van Oers and Guinée (2016)). A first solution to determine the depletion of all resource use types consistently so comparison is possible, could be to separate the different aspects associated with resource use depletion, e.g. replenishment rate, accumulation of anthropogenic stocks and extraction from nature and set up indicators which are valid for all resource use types (as already partly done within BIRD (Bach et al. 2017a)).

Additional case studies should be carried out for BIRD and SCARCE for which so far only one case study is conducted each. Further, production systems utilizing abiotic and biotic resource and raw material use, applying the combined methodology of ESSENZ and BIRD should be analyzed in further case studies. Additionally, other sectors as so far addressed, e.g. chemical industry, should be considered in case studies to continuing the testing of the developed indicators and related results.

Further challenges associated with the proposed methodologies of this thesis can be found in the corresponding discussion chapters of the publications.

Within most life cycle sustainability assessment studies the economic pillar is addressed with the life cycle costing (LCC) approach (UNEP/SETAC Life Cycle Initiative 2011). However, this approach rather determines financial costs expressed in monetary values instead of evaluating the economic sustainability of products (Klöpffer and Ciroth 2011; Swarr et al. 2011; Neugebauer et al. 2016). Thus, the LCC approach is not applied with the developed methodologies. However, as the availability of resources and raw materials is assessed, which are linked to the financial success of a company or region, some relevant economic aspects are included. Further, as applying additional economic values might be reasonable in some cases (e.g. to evaluate investment decisions) the use of additional economic measures is accounted for in the developed methodologies.

Finally, the developed methods are discussed in the context of resource efficiency, a concept, which is broader than the indicator frameworks discussed so far. It relates value added by input, e.g. resource use. Key objectives of resource efficiency comprise of guaranteeing supply of resource use and to reduce environmental impacts of resource and raw material use (Giljum and Polzin 2009; BIO Intelligence Service 2012). As addressed by Schneider et al. (2016a) indicators applied in LCA can also be used to measure the "input" of the resource and raw material use efficiency equation. The focus of all publications of this thesis is the adequate assessment of resource and raw material use and their impacts, which is one part of resource efficiency assessment. The methodologies can further be applied without the context of resource efficiency as part of LCA or LCSA. The "added value" is defined in ESSENZ as the functional unit, because several challenges exist with regard to monetary values, which are usually applied. Monetary values related to a product system can for example be identified by applying life cycle costing indicators, e.g. price of product. However, depending which actor in the supply chain is considered the monetary value can differ significantly (this is discussed further in the ESSENZ publications (Bach et al. 2016a; Bach et al. 2016b). The approach to determine resource efficiency on product level proposed in ESSENZ could also be applied for BIRD. Determining resource efficiency on regional level is more challenging as resource and raw material use cannot be easily related to a function (as on product level, expressed through the functional unit).

The recycling of products containing metals is promoted in existing resource efficiency as well as sustainability strategies on international and national level. Due to higher amounts of secondary materials, less primary materials have to be mined, which can lead to lower environmental burdens. Additionally, the recovered materials would lower possible availability constraints for regions and companies producing in these regions. However, implementation of recycling quotas is challenging, because it is mainly influenced by market price of materials (UNEP 2013). Is the market price of primary materials too low, there is no incentive for recycling. Thus, if recycling is not economic unprofitable, policymakers have to set up incentives, e.g. like the circular economy strategy and its measurements (European Commission 2015), to encourage recycling of materials nevertheless. However, it has to be guaranteed that the promoted recycling technologies are indeed more environmental friendly (UNEP 2013). Recycling processes also cause environmental burdens, especially considering the increased material complexity of modern technologies. Thus, burden shifting from one environmental problem to another has to be avoided (Schrijvers 2017). The methods developed in this thesis can support the assessment of product recycling with regard to environmental impacts, social aspects and the availability and criticality of materials.

4.2 Existing methods and methodological trends

In this chapter additional relevant methods are discussed (that are not addressed in chapter 1.3 and 1.4), which are published at the same time or later than the publications of this thesis. Further, methodological trends are introduced and their relevance for the methodologies of this thesis are discussed.

The geopolitical supply risk (GeoPolRisk) method (Gemechu et al. 2016; Helbig et al. 2016; Cimprich et al. 2017) shows the following differences compared to ESSENZ:

i) considering import data instead of global production data,

ii) taking into account just two supply chain restrictions instead of eleven,

iii) assuming that domestic production is risk free,

iv) integrating producer's and supplier's vulnerability,

v) setting up a cause-effect-chain and

vi) proposing to neglect the mass of the used materials.

Significant differences between ESSENZ and these newly developed methods is the consideration of import data as well as the neglect of the used mass of the resource and raw material in the product system. The use of import data has advantages compared to applying global production data, because the individual impacts for a specific material or intermediate product can be determined, which provide a more realistic picture of possible supply chain restrictions. Further, the consideration of domestic products as well as vulnerability aspects can be included. However, most of the time it is difficult to determine where the material used in a product originates from (this is confirmed in the case studies carried out during the development of ESSENZ). Applying global trade data as done by Gemechu et al. (2016) is challenging as the provided data reflect trade patterns rather than providing information where the imported materials are actually extracted and refined.

For example, materials imported into Germany are often imported via another transit region before, e.g. Switzerland imports gold and exports it then all over the world, but does not have own mines. Thus, for an approach based on global trade patterns it has to be made sure, that all regions importing and exporting a certain material as well as the extracting region are taken into account. Further, specific indicators in every supply chain stage should be applied, e.g. in the presented case of Switzerland the export of gold might not be a political challenge (as Switzerland is a political stable country), but rather an economic one (e.g. price increase). Within ESSENZ CFs and indicator values based on global production data are provided so that every company worldwide can apply them. However, ESSENZ also states that additional analyses, e.g. determining specific trade relations of the company carrying out the assessment or of the region they produce in, are necessary before determining strategies to lower the possible supply restrictions associated with a product.

To set materials of domestic production to not critical as done by Helbig et al. (2016) is valid and is also done within SCARCE, where import data are applied. It would be possible to include this assumption into ESSENZ for the company carrying out the assessment. It should be noted, that the boundary between product and company level is not straightforward, because products are used and produced by companies. The domestic production of resource and raw material use produced within the country in which the company produces, could be set to zero in the calculation. This however means that the characterization factors have to be recalculated for every case study individually. The vulnerability of the company or/and the country it produces in can also be determined for the company carrying out the assessment. In fact, ESSENZ clearly states that further evaluations are necessary, e.g. determining the vulnerability of the company carrying out the assessment, to set up effective measures reducing possible supply restrictions.

As already discussed in chapter 4.1 ESSENZ is developed to determine supply restriction along the entire supply chain. However, due to lack of inventory data in most case studies only the first and second supply chain stage, for which CFs are provided, are addressed. Thus, incorporating the supply risk of intermediate products as done by Helbig et al. (2016) and Gemechu et al. (2017) is not seen as a discrepancy compared to ESSENZ, but rather as a welcomed pioneer work to show how case studies addressing supply risks should ideally be carried out.

Further, within the newly developed method by Gemechu et al. (2016), Helbig et al. (2016), Gemechu et al. (2017) and Cimprich et al. (2017) supply risks are determined by considering the aspects political stability (measured by Worldwide Governance Indicators (Kaufmann et al. 2011)) and concentration of production. In ESSENZ overall eleven aspects are taken into account including political stability and concentration of production. During the development of ESSENZ an approach is established to determine all relevant supply chain restrictions (see ESSENZ publication (Bach et al. 2016a)) . Even though considering only few aspects would reduce the challenges associated with weighting of multiple aspects (as discussed in chapter 4.1), the consideration of a large set of socio-economic aspects is useful in order to determine potential availability constraints.

The cause effect mechanism for supply risks proposed by Cimprich et al. (2017) can be seen as a first step to determine a comprehensive set of impact pathways The fact that geopolitical factors can lead to supply disruptions and therefore to an impaired product function and impacts on human welfare is undoubtedly correct, but still rather simplified. Especially within the first step of the cause effect chain (geopolitical factors cause supply disruptions) a more detailed description should be achieved with regard to the specific supply restrictions possible, e.g. current extraction of resource and raw material use is translated into few regions producing raw materials (concentration of production), which can possibly lead to supply disruptions. The need of individual cause-effect chains for the various supply restrictions (expressed as different categories within ESSENZ) is addressed within the publication of ESSENZ (within the book on pages 27+28 as well as in the supplementary material shown in chapter 3.1).

Finally, Cimprich et al. (2017) propose to not account for the mass of the resource and raw material use used in a product system, because possible supply restrictions are independent of the mass. However, the mass of the material is not excluded completely, but addressed as a vulnerability aspect (the same approach is used within SCARCE). Within ESSENZ the mass of the material plays a vital role in determining supply restrictions as the CFs expressing the supply restriction of every raw material are multiplied with their amount used in the product system under consideration. As ESSENZ does not include vulnerability aspects (see discussion in paragraph above) the mass is taken into account on product level. Even though the argument that the supply restriction as such is not impacted by the mass of the material used is valid, the impacts of the supply restrictions on products under consideration also depends on the mass of the material used. Thus, incorporating vulnerability aspects within ESSENZ and moving the influence of the mass from the assessment of supply restrictions to the vulnerability evaluation (as done in SCARCE) might be possible. However, taking into account methodological consistency ESSENZ is designed to complement existing impact assessment methods and therefore the same application as in LCA was thought to be beneficial for the practitioner. The above discussed aspects with regard to ESSENZ are also valid for BIRD, because it determines supply restrictions based on global production data and therefore does not considers domestic production as well as vulnerability aspects.

Mancini et al. (2016) propose a method to include supply restrictions (for the categories political stability and concentration of production) into LCA. Different calculation principles for determining CFs are applied and it is analyzed which of these principles should be applied for determining supply restrictions within LCA. The authors come to the conclusion that the indicator values of each material should be divided by the respective mine production as otherwise the mass of the inventory dominates the results. Dividing the indicator value by production data is an element also applied in the approach to determine supply restrictions within ESSENZ, BIRD and SCARCE. They further state, that applying threshold to determine the supply chain risk is subjective and should not be used to determine characterization factors. Thresholds are considered within ESSENZ, BIRD and SCARCE to determine for which raw materials supply risk actually occur. It is discussed in chapter 4.1 that thresholds are subjective, but applying them has the benefit of being able to focus on the materials facing high and multiple supply restrictions. Thus, the focus can be on raw materials, which do show risks of availability constraints. However, during the interpretation of results the defined thresholds have to be included in the analysis as well.

Crenna et al. (2017b) introduce a database, which gives a comprehensive overview of biotic resources used in product systems including species name, family name, distribution and vulnerability scores as well as renewability rates of biotic resources used in product systems. Next to the database, the renewability rate is proposed as the category indicator to measure biotic resources. It is stated that other aspects such as extraction over availability or ecological features such as vulnerability of species should be included in the future. The renewability rate is considered within the BRDI, together with other aspects such as vulnerability of species. Considering the current vulnerability of the extracted species (as done with the threatened species indicator) is important to guarantee that the species will not get extinct by extracting the species. Further, the approach by Crenna et al. (2017a) is only valid to biotic resources, but is not applicable for biotic raw materials for which availability constraints due to replenishment should also be accounted for.

As already addressed before (in chapters 1.3 and 1.4) the assessment of abiotic resource use depletion has been a topic of discussion recently, amongst others due to the findings within the Product Environmental Footprint (PEF) pilot phase (European Commission 2016). Within the PEF method, the ADP indicator based on the reserve base is recommended, which is an economic indicator. It is not static, but changes over time based on material prices, new technologies and other factors (Graedel et al. 2011; Schneider 2014; Drielsma et al. 2016b). Thus, the results of the PEF studies surprised the pilots and the PEF method developers, because results varied significantly compared to existing results for which the ADP baseline approach is applied (Lehmann et al. 2016; European Commission 2017). Further, recently a UNEP/SETAC task force for the assessment of mineral resource use has been set up (Berger and Sonderegger 2017), where existing methods are evaluated and discussed with regard to their scope, scientific robustness and applicability in LCA. Currently the most promising approach to achieve this, seems to be the evaluation of dissipation of abiotic resource use (van Oers and Guinée 2016). This thesis only determined the most suitable indicator to be included in the methodologies, but did not provide a method itself. Thus, newly developed indicators might substitute the currently applied ADP (baseline approach – based on ultimate reserves) indicator, when they are evaluated as a better fit.

4.3 Outlook

This chapter provides an outlook, which aspects beyond the key challenges should be addressed in future work to enhance the comprehensive assessment of resource and raw material use on all levels.

Some of these key challenges described here will be revisited in an upcoming project founded by the Deutsche Forschungsgesellschaft (DFG). The goal of the project is to develop an integrated methodology to assess resource and raw material use of different mobility solutions based on ESSENZ, BIRD, SCARCE and the proposed approach for evaluating different resource types. It is hoped that some of the challenges can be tackled in this project.

The CFs and indicator values for ESSENZ (and ESSENZ+) are provided at: www.see.tu-berlin.de and will be updated regularly every 3 years. During the work of this thesis, they were already updated once. The old values can be found in chapter 3.1 in the supplementary material of the ESSENZ publication and book. The updated values can be found in 10.1 and 10.2.

So far, the meso level (company) is not addressed in the publications of this thesis. However, the methodologies proposed for the product and regional level can be used as starting points. For the assessment on company level the organizational life cycle assessment (OLCA) method (Martínez-Blanco et al. 2015) is established, which can be linked with the methodologies introduced within this thesis. Lessons learned from combining ESSENZ and BIRD (e.g. wording of dimensions and categories) should be considered when new methodologies are established. Further, a methodology should be developed to assess biotic resources on regional level based on BIRD.

Further, an assessment methodology for secondary raw materials, especially for supply chain restrictions, should be established. So far secondary materials are only indirectly considered (by the categories primary material use and anthropogenic availability as well as by their environmental impacts on product level). A first approach by Finkbeiner and Schneider (2012) can be used as a starting point.

Further, availability of anthrophonic stocks is not adequately considered, because available data are insufficient. Several research projects (e.g. Schiller et al. (2015)) have started to map anthropogenic flows determining regional and global flows to better understand how materials within products are distributed. When this information is available, it should be included in the proposed methodologies.

Next to the assessment of terrestrial biotic resource and raw material use (which are considered in BIRD) also the use of aquatic biotic resource and raw material use beyond fish should be addressed. First pilot projects have started analyzing the use of algae as an possible fuel for cars and airplanes (Pate et al. 2011; Carneiro et al. 2017). Thus, their use and associated impacts should be included in future assessment methodologies. As the assessment of algae based fuels is part of the project funded by DFG it is likely that assessing aquatic biotic resource and raw material use can be accomplished in the project.

The models applied within the methodologies of this thesis determining supply chain restrictions are based on the economic and political developments in recent years (only exception is the category utilization in future technologies within SCARCE). However, a projection of current situations and events to the future always holds an uncertainty that then may lead to inaccurate recommendations. Thus, future scenarios should be included in the assessment of supply chain restrictions as well, as done by Helbig et al. (2017). However, as future scenarios also rely on assumptions, their uncertainty could be equal or even higher. CFs based on future scenarios can be compared with the results obtained from ESSENZ and SCARCE and conclusions can be drawn with regard to uncertainty of both modeling approaches.

Further, to achieve the goal to make the CFs of ESSENZ and BIRD applicable like other LCIA methods, common LCA databases have to be adapted. Currently the databases apply economic allocation to assign metal contents from mixed ores to metal datasets. To be able to assess availability aspects with the provided CFs the databases need to provide the actual material flow, this means the actual amount of resources leaving the mine and of the raw materials in every supply chain stage.

5. Conclusion

Due to the increasing demand of abiotic and biotic resource and raw material use, their availability, environmental pollution as well as related societal impacts have been a topic of concern in the last decades. Thus, to evaluate the success of strategies implemented to efficiently manage resource and raw material use, methodologies are required to comprehensively assess resource and raw material use and associated impacts for products and regions.

The analysis of existing methods has shown that several challenges exist with regard to the assessment of resource and raw material use on product and regional level (see chapters 1.3 and 1.4). Thus, the overall goal of the thesis is to enhance the assessment of abiotic and biotic resources (excluding water and land) by evaluating resource and raw material use across all three sustainability dimensions. This goal is met by establishing four methodologies enhancing the assessment of abiotic and biotic resource and raw material use on product and regional level considering multiple aspects.

ESSENZ allows for an assessment of abiotic resource and raw material use by determining overall 21 indicators in three dimensions. To assess availability constraints of terrestrial biotic resource and raw material use in BIRD 24 indicators within five dimensions are considered. SCARCE provides 25 indicators for two dimensions to assess abiotic resource and raw material use on regional level. In these three existing indicators are evaluated with regard to their maturity and applicability. These are then combined with newly developed indicators to an integrated methodology to assess abiotic and biotic resource and raw material use on product and country level, respectively.

New approaches are established for the assessment of socio-economic availability and societal acceptance of abiotic and biotic resource and raw material use on product and regional level, as well as abiotic and physical constraints of biotic resource and raw material use. CFs for eleven categories based on the distance-to-target approach are established, which can be integrated on product level in existing LCA based schemes. Thus, companies already applying LCA for determining their environmental impacts can integrate the assessment of supply restrictions more easily.

Within ESSENZ and SCARCE metals as well as fossil raw materials are taken into account, allowing for a comparison of both resource types and identifying trade-offs. To enhance the assessment of different resource types even further, an approach is proposed which combines existing methods addressing different resource types in a consistent way, allowing for the direct comparison of abiotic (fossils, metals and minerals) and biotic resource and raw material use. The applicability of the approach is proven by combining ESSENZ and BIRD to a combined methodology. This enhances the assessment of resource types significantly, because overall 5 dimensions and 25 of the 27 considered categories and indicators could be included in the combined methodology.

Further, the application of the methodologies is significantly enhanced by providing results of CFs and indicator values for the methodologies ESSENZ and SCARCE. For ESSENZ CFs for 36 metals and four fossil raw materials are provided to determine the socio-economic availability. Further, indicator results for the dimension societal acceptance are provided. For SCARCE the global indicator results for the dimensions availability as well as societal acceptance are provided as well as indicator values for regions to determine the import based results of these dimensions.

Several remaining key issues are addressed in this thesis to enhance the applicability of the developed approaches.

These includes a meaningful reduction of the considered categories to enhance the interpretation and communication of the results, e.g. by aggregating of results or identifying lead indicators, enhancing the assessment of resource use depletion across different resource types, improving the assessment of the resource phosphorus and complementing the considered methods for assessing environmental impacts with newly developed methods.

By establishing four scientifically robust yet applicable methodologies, which consider multiple aspects of abiotic and biotic resource and raw material use on product and regional level in all three sustainability dimensions, the assessment of resource and raw material use is improved significantly.

Thus, opportunities to improve resource and raw material use impacts over the life cycle of products can be identified and companies are able to establish strategies to counteract any supply restrictions. Finally, SCARCE is able to support policy to define national and international strategies and set up efficient laws and regulations.

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7. Glossary

Area of protection (AoP)	Safeguard subject to be protected, typically three AoPs are defined in LCA: human health, natural resource use and natural environment (Guinée et al. 2002) Resource use has so far been addressed within the AoP natural resource use. For criticality related approaches the new AoP "Resource use provision capability for human welfare" is suggested by Schneider (2014).
Availability	The existence and accessibility of resources (Graedel et al. 2012)
• Physical availability	Existence and accessibility of resources and raw materials due to deposits in nature and stocks in the anthroposphere (Schneider 2014)
• Socio-economic availability	Accessibility of resources and raw materials due to socio-economic aspects (geopolitical, political and economic factors) occurring in the supply chain (Graedel et al. 2012)
Characterization factor	Factor derived from a characterization model which is applied to convert an assigned life cycle inventory analysis result to the common unit of the category indicator (ISO 14044 2006).
Criticality	The use of resources or raw materials showing high supply risks within a system, which is vulnerable to these potential supply disruptions (Dewulf et al. 2016).
Deposit	A body of mineralization, which can become an ore, depending on the technological progress and economic conditions (Schneider 2014)

Indicator	A pointer that indicates something Indicators are applied to quantify and assess a specific situation or development, e.g. the use of resource use. They simplify, clarify and make information accessible to various stakeholder, especially policy makers (United Nations 2007).
Impact category	Class representing environmental issues of concern to which life cycle inventory analysis results may be assigned (ISO 14044 2006).
Life cycle assessment	Compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (ISO 14044 2006).
Life cycle impact assessment	Phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system throughout the life cycle of the product (ISO 14044 2006).
Life cycle inventory	Phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a product throughout its life cycle (ISO 14044 2006).
Product system	Collection of unit processes with elementary and product flows, performing one or more defined functions, and which models the life cycle of a product (ISO 14044 2006).
Raw material	Primary or secondary material that is used to produce a product (ISO 14044 2006).
Reserve	Part of the deposits that are known and profitable to exploit at current prices, state of technology, etc. (USGS 2015)

Resource	Resource use are material and non-material assets occurring in nature that are at some point in time deemed useful for humans (Sonderegger et al. 2017). This includes material assets as well as the environment. For abiotic and biotic resources a distinction is made between resources and raw materials according to Lindeijer et al. (2002). Additionally, social aspects related to resource extraction and use are taken into account to achieve an assessment of resources in line with sustainable development.
• abiotic	Refers to non-living natural resource use (including energy resource use), elements and minerals from the earth's crust
• biotic	Refers to living objects obtained from the biosphere
Societal acceptance	Refers to the acceptance of consumers with regard to environmental and social impacts arising due to non-compliance with standards (social norms as well as laws and regulations) over the supply chain (own definition)
Stock	Collective amount available of one specific resource use or raw material (USGS 2015)
Social life cycle assessment	SLCA is an assessment method that evaluates the potential social negative (and positive) impacts of products in relation to different stakeholder groups along the life cycle (UNEP 2009)
Supply risk	Probability of an incident due to geopolitical, political and economic aspects along the supply chain, resulting in the inability of the purchasing resources, materials or products (Graedel et al. 2012)

	See also definition of socio-economic availability
Sustainable development	Development until the ideal state is reached, where the needs of the present as well as future generations with regard to economic development, social development and environmental protection can be fulfilled (United Nations 1987)
Vulnerability	Inability of a system to withstand harmful effects (Dewulf et al. 2016)

8. List of figures

9. List of tables

10. Appendix

10.1 Update of CFs and indicator values for ESSENZ

Following the results for the dimensions *socio-economic availability* and *societal acceptance* are presented.

10.1.1 Updated CFs for the dimension socio-economic availability

In this section, the results of the updated CFs for the categories demand growth, mining capacities, price variation and country concentration reserves in **Table 2** and country concentration of mine production, trade barriers, feasibility of exploration projects and political stability in **Table 3** are shown. Data for the categories companion metal, primary material use and company concentration could not be updated, because updated underlying data are not available.

Table 2: Updated CFs of ESSENZ method for the categories demand growth, mining capacities, concentration of reserves and price variation of the dimension socio-economic availability

Category/raw material	Demand growth	Mining capacities	Concentration of reserves	Price variation
Aluminum	1,37E+07	0,00E+00	1,77E+06	0,00E+00
Antimony	0,00E+00	8,00E+05	1,73E+06	0,00E+00
Beryllium	0,00E+00	1,82E+13	1,06E+13	6,33E+12
Bismuth	1,04E+11	9,54E+08	9,09E+10	9,10E+10
Boron	5,70E+07	8,20E+05	7,58E+07	0,00E+00
Chromium	1,24E+07	1,79E+06	1,84E+07	0,00E+00
Cobalt	0,00E+00	3,83E+07	2,29E+09	0,00E+00

Category/raw material	Demand growth	Mining capacities	Concentration of reserves	Price variation
Copper	0,00E+00	4,69E+05	6,46E+06	0,00E+00
Crude oil	0,00E+00	1,18E+03	0,00E+00	6,27E+05
Gallium	1,82E+13	0,00E+00	2,34E+11	0,00E+00
Germanium	4,67E+12	2,85E+11	7,34E+11	0,00E+00
Gold	0,00E+00	1,21E+10	0,00E+00	0,00E+00
Graphite	4,53E+08	0,00E+00	2,62E+08	7,94E+08
Hard Coal	0,00E+00	0,00E+00	1,84E+04	1,86E+05
Indium	5,36E+11	0,00E+00	5,60E+11	0,00E+00
Iron	1,26E+05	6,27E+03	3,27E+04	7,52E+05
Lead	0,00E+00	8,10E+06	4,14E+07	0,00E+00
Lignite Coal	0,00E+00	0,00E+00	1,22E+05	1,23E+06
Lithium	0,00E+00	0,00E+00	1,94E+10	3,39E+10
Magnesium	0,00E+00	0,00E+00	3,10E+06	0,00E+00
Manganese	0,00E+00	4,66E+05	8,31E+06	6,45E+07
Molybdenum	0,00E+00	1,65E+07	1,98E+09	6,65E+09
Natural Gas	0,00E+00	0,00E+00	0,00E+00	0,00E+00
Nickel	3,38E+08	4,36E+06	0,00E+00	4,81E+08
Niobium	0,00E+00	0,00E+00	5,68E+10	0,00E+00
Palladium	0,00E+00	0,00E+00	1,59E+13	4,66E+12
Platinum	0,00E+00	0,00E+00	1,82E+13	0,00E+00
Rare earth	0,00E+00	0,00E+00	1,77E+09	1,61E+10
Rhenium	0,00E+00	1,00E+11	9,07E+12	0,00E+00
Selenium	3,11E+11	3,16E+09	2,93E+10	9,30E+11
Silicon	0,00E+00	0,00E+00	1,15E+08	0,00E+00
Silver	0,00E+00	1,07E+09	0,00E+00	3,59E+10
Strontium	0,00E+00	0,00E+00	5,49E+09	6,21E+09
Tantalum	2,90E+12	0,00E+00	1,47E+12	0,00E+00
Tellurium	1,04E+13	0,00E+00	5,44E+12	1,22E+13

Category/raw material	Demand growth	Mining capacities	Concentration of reserves	Price variation
Tin	0,00E+00	1,96E+08	0,00E+00	0,00E+00
Titanium	0,00E+00	0,00E+00	1,46E+07	2,51E+08
Tungsten	5,40E+09	1,07E+08	1,03E+10	1,18E+10
Uranium	0,00E+00	4,36E+11	1,64E+12	1,82E+13
Vanadium	0,00E+00	0,00E+00	6,92E+09	1,86E+10
Zinc	0,00E+00	3,46E+06	8,90E+06	0,00E+00
Zirconium	0,00E+00	0,00E+00	6,82E+08	7,44E+08

Table 3: Updated CFs of ESSENZ method for the categories political stability, concentration of production, trade barriers, feasibility of exploration projects and of the dimension socio-economic availability

Category/raw material	Political stability	Concentration of production	Trade barriers	Feasibility of exploration projects
Aluminum	1,97E+07	1,09E+07	0,00E+00	1,38E+08
Antimony	1,42E+07	1,70E+07	0,00E+00	0,00E+00
Beryllium	0,00E+00	1,82E+13	0,00E+00	0,00E+00
Bismuth	1,94E+11	2,07E+11	0,00E+00	0,00E+00
Boron	1,72E+08	2,61E+08	0,00E+00	0,00E+00
Chromium	4,00E+07	1,29E+07	0,00E+00	2,18E+08
Cobalt	1,65E+10	4,99E+09	9,75E+12	0,00E+00
Copper	4,54E+07	0,00E+00	0,00E+00	0,00E+00
Crude oil	2,58E+05	0,00E+00	0,00E+00	2,39E+06
Gallium	4,32E+12	1,07E+13	0,00E+00	1,82E+13
Germanium	8,10E+12	1,32E+13	0,00E+00	0,00E+00
Gold	4,38E+11	0,00E+00	0,00E+00	0,00E+00

Category/raw material	**Political stability**	**Concentration of production**	**Trade barriers**	**Feasibility of exploration projects**
Graphite	1,63E+09	1,18E+09	0,00E+00	0,00E+00
Hard Coal	2,07E+05	8,11E+04	0,00E+00	0,00E+00
Indium	1,14E+12	6,54E+11	0,00E+00	0,00E+00
Iron	3,34E+05	1,23E+05	0,00E+00	0,00E+00
Lead	2,39E+08	8,76E+07	0,00E+00	0,00E+00
Lignite Coal	7,46E+05	0,00E+00	0,00E+00	0,00E+00
Lithium	0,00E+00	2,24E+10	0,00E+00	0,00E+00
Magnesium	4,07E+07	4,40E+07	0,00E+00	0,00E+00
Manganese	6,43E+07	1,28E+07	0,00E+00	4,05E+08
Molybdenum	3,63E+09	1,69E+09	0,00E+00	0,00E+00
Natural Gas	0,00E+00	0,00E+00	0,00E+00	2,69E+06
Nickel	4,58E+08	0,00E+00	0,00E+00	0,00E+00
Niobium	2,08E+10	6,83E+10	1,82E+13	0,00E+00
Palladium	5,64E+12	2,45E+12	0,00E+00	0,00E+00
Platinum	6,53E+12	1,09E+13	0,00E+00	0,00E+00
Rare earth	1,23E+10	2,21E+10	0,00E+00	0,00E+00
Rhenium	0,00E+00	1,52E+13	0,00E+00	0,00E+00
Selenium	0,00E+00	6,45E+10	0,00E+00	0,00E+00
Silicon	1,89E+08	1,70E+08	0,00E+00	0,00E+00
Silver	4,88E+10	0,00E+00	0,00E+00	0,00E+00
Strontium	1,23E+10	8,11E+09	6,57E+12	4,38E+10
Tantalum	1,86E+12	3,97E+11	0,00E+00	0,00E+00
Tellurium	0,00E+00	1,94E+12	0,00E+00	0,00E+00
Tin	6,00E+09	9,93E+08	0,00E+00	0,00E+00
Titanium	1,37E+08	0,00E+00	0,00E+00	0,00E+00
Tungsten	2,03E+10	3,38E+10	0,00E+00	0,00E+00
Uranium	1,70E+13	5,44E+12	0,00E+00	0,00E+00

Category/raw material	Political stability	Concentration of production	Trade barriers	Feasibility of exploration projects
Vanadium	2,80E+10	1,54E+10	0,00E+00	0,00E+00
Zinc	9,18E+07	1,42E+07	0,00E+00	0,00E+00
Zirconium	0,00E+00	4,10E+08	0,00E+00	0,00E+00

10.1.2 Updated indicator values for the dimension societal acceptance

In this section, the updated results for the categories compliance with social and environmental standards are shown (see **Table 4**).

Table 4: Updated indicator values for the dimension societal acceptance

Category/raw material	(Non) compliance with social standards	(Non) compliance with environmental standards
Aluminum	10,12	23,75
Antimony	20,16	37,45
Beryllium	0,41	9,30
Bismuth	18,24	33,33
Boron	13,09	19,51
Chromium	10,61	24,14
Cobalt	15,62	53,59
Copper	1,02	18,38
Crude oil	0,86	18,20
Gallium	28,18	36,94
Germanium	18,80	26,52

Category/raw material	(Non) compliance with social standards	(Non) compliance with environmental standards
Gold	1,19	24,45
Graphite	15,02	40,82
Hard Coal	8,90	29,66
Indium	8,14	19,33
Iron	6,07	20,73
Lead	7,63	23,17
Lignite Coal	0,95	14,69
Lithium	0,99	5,61
Magnesium	18,21	33,44
Manganese	7,14	25,79
Molybdenum	6,84	22,07
Natural Gas	0,96	12,01
Nickel	2,01	24,84
Niobium	9,62	18,84
Palladium	10,43	19,03
Platinum	26,72	23,44
Rare earth	21,76	31,89
Rhenium	1,45	10,84
Selenium	3,96	14,93
Silicon	14,56	30,58
Silver	1,86	23,16
Strontium	9,76	47,64
Tantalum	7,23	33,24
Tellurium	1,20	8,44
Tin	6,95	40,78
Titanium	2,93	20,68
Tungsten	21,55	31,68
Uranium	2,50	16,22
Vanadium	13,55	34,00

Category/raw material	(Non) compliance with social standards	(Non) compliance with environmental standards
Zinc	4,55	24,75
Zirconium	4,48	10,51

10.2 CFs and indicator values for ESSENZ+

Following, the CFs and indicator values for the ESSENZ+ methodology are provided. For the dimension *socio-economic availability* only the category *political stability* is determined differently compared to the original ESSENZ approach (see **Figure 5**). Therefore, only this category is displayed here. Results for the other categories can be found in 10.1.1. Further, the indicator values for the dimension societal acceptance are presented.

Table 5: CFs and indicator values for ESSENZ+ for the category political stability of the dimension socio-economic availability and the categories compliance with social and environmental standards of the dimension societal acceptance

Category/ raw material	Political stability	(Non) compliance with social standards	(Non) compliance with environmental standards
Aluminum	1,85E+07	0,34	0,15
Antimony	1,48E+07	0,80	0,17
Beryllium	2,54E+12	0,11	0,15
Bismuth	2,04E+11	0,57	0,15
Boron	1,90E+08	0,25	0,31
Chromium	3,65E+07	0,63	0,38
Cobalt	1,30E+10	0,71	0,14
Copper	5,21E+07	0,26	0,37
Crude oil	2,76E+05	0,27	0,20

Category/ raw material	**Political stability**	**(Non) compliance with social standards**	**(Non) compliance with environmental standards**
Gallium	4,85E+12	0,55	0,13
Germanium	9,58E+12	0,39	0,09
Gold	4,19E+11	0,46	1,00
Graphite	1,65E+09	0,24	0,21
Hard Coal	2,30E+05	0,41	0,19
Indium	1,53E+12	0,28	0,10
Iron	3,95E+05	0,35	0,28
Lead	2,80E+08	0,39	0,21
Lignite Coal	9,06E+05	0,16	0,15
Lithium	1,90E+10	0,12	0,44
Magnesium	4,47E+07	0,53	0,16
Manganese	7,42E+07	0,49	0,30
Molybdenum	4,53E+09	0,31	0,23
Natural Gas	3,45E+05	0,18	0,18
Nickel	4,55E+08	0,31	0,30
Niobium	1,48E+10	0,67	0,57
Palladium	5,20E+12	0,57	0,37
Platinum	5,59E+12	0,70	0,91
Rare earth	1,45E+10	0,48	0,16
Rhenium	1,19E+13	0,09	0,25
Selenium	2,59E+11	0,19	0,10
Silicon	2,14E+08	0,42	0,13
Silver	4,60E+10	0,42	0,37
Strontium	1,19E+10	0,76	0,61
Tantalum	1,63E+12	0,61	0,08
Tellurium	4,27E+12	0,12	0,11
Tin	5,22E+09	0,86	0,31

Category/ raw material	Political stability	(Non) compliance with social standards	(Non) compliance with environmental standards
Titanium	1,45E+08	0,39	0,25
Tungsten	2,53E+10	1,00	0,12
Uranium	1,82E+13	0,32	0,25
Vanadium	2,78E+10	0,62	0,21
Zinc	1,02E+08	0,36	0,28
Zirconium	5,27E+08	0,28	0,40

10.3 Update of indicator values for SCARCE

In this section, the results of the updated indicator values for the categories *demand growth, mining capacities, price variation* and *country concentration reserves* in Table 6 and *country concentration of mine production, trade barriers, feasibility of exploration projects* and *political stability* in Table 7 are shown. Data for the categories companion metal, primary material use and company concentration could not be updated, because updated underlying data are not available. Further, the updated indicator values for the dimension societal acceptance are shown in chapter 10.1.2.

Table 6: Updated indicator values of SCARCE for the categories demand growth, mining capacities, price variation and country concentration reserves of the dimension socio-economic availability

Category/raw material	Demand growth	Mining capacities	Price variation	Concentration of reserves
Aluminum	7,54E-07	0,00E+00	0,00E+00	1,80E-28
Antimony	0,00E+00	3,11E-26	0,00E+00	1,77E-28

Category/raw material	Demand growth	Mining capacities	Price variation	Concentration of reserves
Beryllium	0,00E+00	7,09E-19	3,47E-01	1,08E-21
Bismuth	5,71E-03	3,71E-23	5,00E-03	9,27E-24
Boron	3,13E-06	3,19E-26	0,00E+00	7,73E-27
Chromium	6,83E-07	6,96E-26	0,00E+00	1,87E-27
Cobalt	0,00E+00	1,49E-24	0,00E+00	2,34E-25
Copper	0,00E+00	1,83E-26	0,00E+00	6,59E-28
Crude oil	0,00E+00	4,61E-29	3,44E-08	0,00E+00
Gallium	1,00E+00	0,00E+00	0,00E+00	2,38E-23
Germanium	2,56E-01	1,11E-20	0,00E+00	7,49E-23
Gold	0,00E+00	4,72E-22	0,00E+00	0,00E+00
Graphite	2,49E-05	0,00E+00	4,36E-05	2,67E-26
Hard Coal	0,00E+00	0,00E+00	1,02E-08	1,88E-30
Indium	2,94E-02	0,00E+00	0,00E+00	5,71E-23
Iron	6,94E-09	2,44E-28	4,13E-08	3,34E-30
Lead	0,00E+00	3,15E-25	0,00E+00	4,22E-27
Lignite Coal	0,00E+00	0,00E+00	6,77E-08	1,25E-29
Lithium	0,00E+00	0,00E+00	1,86E-03	1,98E-24
Magnesium	0,00E+00	0,00E+00	0,00E+00	3,16E-28
Manganese	0,00E+00	1,81E-26	3,54E-06	8,48E-28
Molybdenum	0,00E+00	6,40E-25	3,65E-04	2,02E-25
Natural Gas	0,00E+00	0,00E+00	0,00E+00	0,00E+00
Nickel	1,85E-05	1,70E-25	2,64E-05	0,00E+00
Niobium	0,00E+00	0,00E+00	0,00E+00	5,80E-24
Palladium	0,00E+00	0,00E+00	2,56E-01	1,63E-21
Platinum	0,00E+00	0,00E+00	0,00E+00	1,86E-21
Rare earth	0,00E+00	0,00E+00	8,86E-04	1,81E-25
Rhenium	0,00E+00	3,90E-21	0,00E+00	9,25E-22
Selenium	1,71E-02	1,23E-22	5,10E-02	2,99E-24

Category/raw material	Demand growth	Mining capacities	Price variation	Concentration of reserves
Silicon	0,00E+00	0,00E+00	0,00E+00	1,18E-26
Silver	0,00E+00	4,18E-23	1,97E-03	0,00E+00
Strontium	0,00E+00	0,00E+00	3,41E-04	5,59E-25
Tantalum	1,59E-01	0,00E+00	0,00E+00	1,50E-22
Tellurium	5,69E-01	0,00E+00	6,69E-01	5,55E-22
Tin	0,00E+00	7,61E-24	0,00E+00	0,00E+00
Titanium	0,00E+00	0,00E+00	1,38E-05	1,49E-27
Tungsten	2,97E-04	4,15E-24	6,50E-04	1,05E-24
Uranium	0,00E+00	1,70E-20	1,00E+00	1,67E-22
Vanadium	0,00E+00	0,00E+00	1,02E-03	7,06E-25
Zinc	0,00E+00	1,34E-25	0,00E+00	9,08E-28
Zirconium	0,00E+00	0,00E+00	4,08E-05	6,96E-26

Table 7: Updated indicator values of SCARCE for the categories concentration of production, trade barriers, feasibility of exploration projects and political stability of the dimension socio-economic availability

Category/raw material	Concentration of production	Trade barriers	Feasibility of exploration projects	Political stability
Aluminum	6,01E-07	0,00E+00	7,57E-06	1,01E-06
Antimony	9,34E-07	0,00E+00	0,00E+00	8,14E-07
Beryllium	1,00E+00	0,00E+00	0,00E+00	1,39E-01
Bismuth	1,14E-02	0,00E+00	0,00E+00	1,12E-02
Boron	1,43E-05	0,00E+00	0,00E+00	1,04E-05
Chromium	7,08E-07	0,00E+00	1,20E-05	2,00E-06
Cobalt	2,74E-04	5,35E-01	0,00E+00	7,11E-04

Category/raw material	Concentration of production	Trade barriers	Feasibility of exploration projects	Political stability
Copper	0,00E+00	0,00E+00	0,00E+00	2,86E-06
Crude oil	0,00E+00	0,00E+00	1,31E-07	1,51E-08
Gallium	5,89E-01	0,00E+00	1,00E+00	2,66E-01
Germanium	7,27E-01	0,00E+00	0,00E+00	5,26E-01
Gold	0,00E+00	0,00E+00	0,00E+00	2,30E-02
Graphite	6,48E-05	0,00E+00	0,00E+00	9,06E-05
Hard Coal	4,45E-09	0,00E+00	0,00E+00	1,26E-08
Indium	3,59E-02	0,00E+00	0,00E+00	8,39E-02
Iron	6,77E-09	0,00E+00	0,00E+00	2,17E-08
Lead	4,81E-06	0,00E+00	0,00E+00	1,54E-05
Lignite Coal	0,00E+00	0,00E+00	0,00E+00	4,97E-08
Lithium	1,23E-03	0,00E+00	0,00E+00	1,04E-03
Magnesium	2,41E-06	0,00E+00	0,00E+00	2,46E-06
Manganese	7,03E-07	0,00E+00	2,23E-05	4,07E-06
Molybdenum	9,25E-05	0,00E+00	0,00E+00	2,49E-04
Natural Gas	0,00E+00	0,00E+00	1,48E-07	1,89E-08
Nickel	0,00E+00	0,00E+00	0,00E+00	2,50E-05
Niobium	3,75E-03	1,00E+00	0,00E+00	8,13E-04
Palladium	1,34E-01	0,00E+00	0,00E+00	2,85E-01
Platinum	6,00E-01	0,00E+00	0,00E+00	3,07E-01
Rare earth	1,21E-03	0,00E+00	0,00E+00	7,93E-04
Rhenium	8,34E-01	0,00E+00	0,00E+00	6,53E-01
Selenium	3,54E-03	0,00E+00	0,00E+00	1,42E-02
Silicon	9,35E-06	0,00E+00	0,00E+00	1,17E-05
Silver	0,00E+00	0,00E+00	0,00E+00	2,53E-03
Strontium	4,45E-04	3,60E-01	2,40E-03	6,55E-04
Tantalum	2,18E-02	0,00E+00	0,00E+00	8,97E-02

Category/raw material	**Concentration of production**	**Trade barriers**	**Feasibility of exploration projects**	**Political stability**
Tellurium	1,06E-01	0,00E+00	0,00E+00	2,34E-01
Tin	5,45E-05	0,00E+00	0,00E+00	2,87E-04
Titanium	0,00E+00	0,00E+00	0,00E+00	7,98E-06
Tungsten	1,85E-03	0,00E+00	0,00E+00	1,39E-03
Uranium	2,99E-01	0,00E+00	0,00E+00	1,00E+00
Vanadium	8,44E-04	0,00E+00	0,00E+00	1,53E-03
Zinc	7,79E-07	0,00E+00	0,00E+00	5,59E-06
Zirconium	2,25E-05	0,00E+00	0,00E+00	2,89E-05

10.4 Complementary publications

In this chapter the complementary publications of the thesis are provided:

i. V. Bach, M. Finkbeiner (2016): Approach to qualify decision support maturity of new versus established impact assessment methods—demonstrated for the categories acidification and eutrophication, International Journal of Life Cycle Assessment, 22 (3), pp 387–397, The final publication is available at Springer via: 10.1007/s11367-016-1164-z

page 277

ii. A. Lehmann, V. Bach, M. Finkbeiner (2015): Product environmental footprint in policy and market decisions – applicability and impact assessment, Integrated Environmental Assessment and Management, Volume 11, Issue 3, Pages 417–424, DOI: 10.1002/ieam.1658

page 289

iii. M. Henßler, V. Bach, M. Berger, M. Finkbeiner, K. Ruhland (2016): Resource Efficiency Assessment—Comparing a Plug-In Hybrid with a Conventional Combustion Engine, Resources, 5(1), DOI: 10.3390/resources5010005

page 297

iv. V. Bach, M. Berger, T. Helbig, M. Finkbeiner (2015): Measuring a product's resource efficiency – a case study of smartphones, Proceedings of CILCA 2015 – VI International Conference on Life Cycle Assessment, March 13-16, Lima, Peru, ISBN 978-9972-674-11-2

page 309

v. V. Bach, M. Henßler, M. Berger, K. Ruhland, L. Schneider, M. Finkbeiner (2017): Integrated method to assess resource use in the context of sustainable development (ESSENZ+), Proceedings of LCM 2017 (postprint)

page 313

Int J Life Cycle Assess (2017) 22:387–397
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LCIA OF IMPACTS ON HUMAN HEALTH AND ECOSYSTEMS

Approach to qualify decision support maturity of new versus established impact assessment methods—demonstrated for the categories acidification and eutrophication

Vanessa Bach[1] **· Matthias Finkbeiner**[1]

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Abstract
Purpose Initiatives like the EU Product Environmental Footprint have been pushing the discussion about the choice of life cycle impact assessment methods. Practitioners often prefer to use established methods for performance tracking, result stability, and consistency reasons. Method developers rather support newly developed methods. As case studies must provide consistent results in order to ensure reliable decision-making support, a systematic approach to qualify decision support maturity of newly developed impact assessment methods is needed.
Methods A three step approach referring to key aspects for decision maturity was developed which takes the established life cycle impact assessment methods as a benchmark. In the first step, the underlying models of the methods and their respective differences are analyzed to capture the scope and detail of the characterization models. Second, the considered and available elementary flows covered by the methods are identified and compared to reveal consistent coverage, respectively, gaps between alternatives. In the third step, neglected elementary flows are evaluated with regard to their potential impact to the particular impact category. Furthermore, the characterization factors of concurring elementary flows are analyzed for significant differences in their shares. The developed approach was tested for LCIA methods for eutrophication and acidification in Europe.
Results and discussion A systematic and practical qualification of decision support maturity can be achieved by a three-step approach benchmarking model scope, quantitative and qualitative coverage of elementary flows for new methods with established ones. For the application example, the established CML-IA method was compared with the ReCiPe method and the method of accumulated exceedance. These models vary regarding subdivision of environmental compartments, consideration of fate, as well as regionalization of characterization factors. The amount of covered elementary flows varies significantly as CML-IA covers about 28 more flows within the category acidification and about 35 more flows within the category eutrophication compared to ReCiPe and accumulated exceedance. The significance of all neglected elementary flows for the categories eutrophication and acidification is significant and represents a gap of up to 80 %. Furthermore, it was shown that the shares of some concurring elementary flows differ significantly.
Conclusions The introduced approach allows the benchmarking of newly developed against established methods based on application-oriented criteria. It was demonstrated that significant differences between the methods exist. To guarantee reliable decision-making support, newly developed methods should not replace established ones until a minimum level of decision support maturity is reached.

Keywords Accumulated exceedance · Acidification · Characterization factors · CML-IA · Eutrophication · Life cycle impact assessment · ReCiPe

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✉ Vanessa Bach
vanessa.bach@tu-berlin.de

[1] Chair of Sustainable Engineering, Technische Universität Berlin, Strasse des 17. Juni 135, 10623 Berlin, Germany

1 Introduction

A default list of impact categories or even uniform characterization models for the life cycle impact assessment (LCIA) phase of life cycle assessment (LCA) is an issue of debate in the LCA community since years. One flagship project of the United Nations Environment Programme (UNEP)/Society for Environmental Toxicology and Chemistry (SETAC) initiative tries again to establish a global consensus on certain impacts and their modeling in LCIA. Recent initiatives like the EU Product Environmental Footprint (PEF) (European Commission 2014a) pushed a discussion about LCIA within a concrete application context. (e.g., Finkbeiner 2013; Rack et al. 2013; Jolliet et al. 2014; Lehmann et al., 2015).

Both processes revealed different understandings and preferences between practitioners and real world decision makers using LCA and the scientific community of method developers. Users typically prefer established LCIA methods as important criteria are consistency, understandability, and decision support robustness. LCIA method developers are supporting newly developed methods and promote their application already in status nascendi. They typically think that an LCIA method is better, if the characterization model and impact pathway are more sophisticated and detailed, if there is a higher resolution in the scope of impact categories, and if the impacts are modeled on a regional basis etc.

Some of the 14 predefined LCIA methods of PEF are rather new and have so far not been applied in many case studies, let alone real world decision-making processes. However, based on studies of method developers for JRC, the EU Commission requires using these newly developed ones. Established methods preferred by practitioners can be used additionally. However, due to the already very high effort of the pilot phase, almost none of the pilots have resources to test other methods as well. (Joint Research Centre 2011; European Commission 2014b; Laget and Carpentier 2015; Lehmann et al. 2016).

As LCA is a tool to support decision-making in industry and politics (ISO 14044 2006), results have to be reliable and robust. They should mainly depend on the studied product system and not be primarily influenced by the applied impact assessment method. Any new method needs to be assessed and benchmarked with existing ones to guarantee actual improvement and reliable decisions as already raised by Baitz et al. (2012). This aspect of application readiness or decision support maturity has so far never been sufficiently included in ongoing discussions. Newly developed impact assessment methods do not automatically improve decision-making. They have to be compared to establish ones to identify strengths and weaknesses. Such testing or verification processes of actual improvements for decision-making are lacking. So far, newly developed methods are not comprehensively analyzed but rather recommended solely based on reflecting the cause-effect chain in more detail. (Margni et al. 2008; Joint Research Centre 2010; Jolliet et al. 2014).

The aim of this paper is to fill this gap by proposing a systematic approach for the benchmarking of established against newly developed LCIA methods by focusing on their suitability, respectively, maturity for decision making. The introduced approach is however only applicable to compare existing methods of the same impact category. It cannot be applied to assess fully new methods for impact categories for which so far not impact assessment methods or models exist, e.g., the newly developed model for ocean acidification by Bach et al. (2016). The introduced approach is demonstrated and tested for different methods of the impact categories acidification and eutrophication in a European context. The identification of the most relevant methods for these categories is based on recent publications of case studies (e.g., Buyle et al. 2013), product category rules (e.g., Institut Bauen und Umwelt e.V. 2009), environmental product declarations (e.g., Bombardier Transportation 2011), harmonization papers (e.g., PE International 2014), and publications providing a review of LCIA methods (e.g., International Reference Life Cycle Data System Handbooks addressing LCIA (Joint Research Centre 2010; Joint Research Centre 2011)). Overall within the European context, three methods were identified as being relevant for assessing acidification: Huijbregts (1999) and Hauschild & Wenzel (1998) known as part of the CML-IA method (referred to as CML-IA in the following) (Guinée et al. 2002), van Zelm et al. (2007) published within the ReCiPe method (Goedkoop et al. 2009) (referred to as ReCiPe in the following) and the accumulated exceedance method by Seppälä et al. (2006) (referred to as accumulated exceedance (AE) in the following). Three methods were identified as being relevant for assessing eutrophication: Heijungs et al. (1992a, b) published as part of CML-IA (Guinée et al. 2002), Struijs et al. (2008) as part of ReCiPe (Goedkoop et al. 2009), and AE (Seppälä et al. 2006).

2 Methods

A three-step approach has been developed to analyze the LCIA methods in the context of adequate decision-making support. Key criteria include the scope and detail of the characterization models and the quantitative and qualitative coverage of elementary flows. As shown in Fig. 1, the first step consists of an evaluation regarding the underlying models of each method to identify the maturity of reflecting the cause-effect chain, but also possible challenges regarding application and interpretation.

Within the second step, the covered elementary flows of each method are determined. Even though the mere number of characterization factors (CFs) does not necessarily reflect the quality of the model, missing CFs can be a significant gap and a possible restriction regarding adequate decision-making support for the identification of potential impacts. Therefore, in the steps three (a) and (b), the elementary flows are

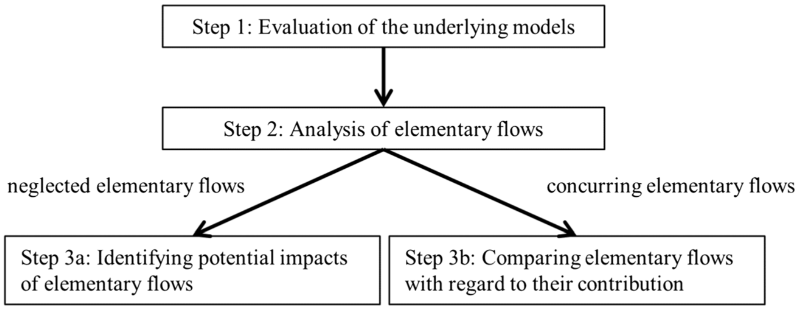

Fig. 1 Overview of the developed three-step approach to evaluate LCIA methods

analyzed further in detail. The neglected flows are analyzed in step three (a) with regard to their importance for the considered impact assessment category. When LCIA methods do not provide CFs for certain elementary flows, which are covered by other methods, it has to be determined whether the associated potential impact neglected is significant or not.

In step three (b), an analysis is carried out regarding the concurring elementary flows with regard to the considered impact assessment category, i.e., those flows that are covered by all the methods. Here, the question arises, whether the same substance has different CF values in different methods and whether there is then a plausible and transparent explanation, if and why the new factor is scientifically more robust than the existing one. Adopted from the approach of Posch et al. (2008), the share of different CFs are compared on a percentage basis as often different reference units exist making a direct comparison impossible. A relative comparison can also reveal, if concurring elementary flows have similar potential impacts. If shares of elementary flows differ, the results and recommendations of LCA studies vary accordingly.

3 Results and discussion

Within step one of the three-step approach, the underlying models of the identified methods are analyzed. Within CML-IA, a baseline and a non-baseline approach are provided, the baseline approach representing the most common impact assessment model (within this paper, the CML-IA version 4.2 is applied). Practitioners usually apply both as the non-baseline complementing the baseline approach. For acidification, the baseline approach by Huijbregts (1999) considers fate and deposition of acidifying substances and critical load[1] data for Europe. Within the non-baseline method by Hauschild and Wenzel (1998), CFs are determined stoichiometrically based on the number of hydrogen ions, which are potentially produced by an acidifying substance. For eutrophication, the baseline method by Heijungs et al. (1992a, b) determines CFs stoichiometrically based on potential contributions of nitrogen and phosphorus emissions as limiting factors to biomass formation. Distribution and deposition of substances within environmental compartments are not considered.

Within ReCiPe, the impact categories acidification and eutrophication are subdivided into several separate impact categories based on the environmental compartments (within this paper, the ReCiPe version 1.07 is taken into account). For acidification, only the terrestrial compartment is considered. Freshwater and marine ecosystems are taken into account with regard to eutrophication. CFs for terrestrial acidification are determined considering the fate of the substances using simulation models, tracking the changes in base saturation of soil based on changes in acid deposition in Europe. For aquatic eutrophication, CFs are determined based on increased concentration of phosphor and nitrogen emissions in water bodies based on nutrient changes via air emissions and from emission sources like agriculture. As phosphorus is the limiting substance for freshwater bodies and nitrogen for marine water bodies, only substances containing phosphorus or nitrogen are considered within the related impact assessment category, respectively (Goedkoop et al. 2009).

Within AE, only the terrestrial compartment is taken into account. Impacts on freshwater and marine compartments are not considered (within this paper CFs for AE are used as published by Seppälä et al. (2006)). The characterization model is taking fate and background load into account. An additional sophistication is introduced by including regional differences. Sensitive areas at and above critical load based on emission and critical load data within Europe are considered when determining CFs for terrestrial eutrophication and terrestrial acidification. Three site specific CFs for acidification and two for terrestrial eutrophication are provided by Seppälä et al. (2006) for several European countries (these include Finland, Germany, France, UK, Spain, Netherlands, and Sweden). As LCA case studies are usually performed with general not with site-specific inventory data, the emission profiles in the LCI do not contain any regional resolution. As a consequence, the method has so far not been applied as it was originally intended. To make it applicable

[1] Critical load data is used to characterize the sensitivity of the ecosystem for additional acidifying depositions due to limited buffer capacity. (Margni et al. 2008; Joint research Centre 2011; Seppälä et al. 2006).

with existing inventory databases, European weighted CFs were provided by Posch et al. (2008) and Joint Research Centre (2011). However, using these average values means that the key advantage of the method in terms of a regionalized resolution of impacts is basically lost.

Analyzing the identified methods, the following features became apparent as main differences:

- subdivision of impact categories by separation of environmental compartments and
- consideration of fate and regionalization of characterization factors.

Whereas the established CML-IA method assesses terrestrial and aquatic impacts together, ReCiPe and AE split the environmental compartments. Thus, to assess all compartments, several models and indicators have to be applied. However, for acidification neither ReCiPe nor AE consider the aquatic compartment. Furthermore, by subdividing the compartments, not only one result but also up to three results are presented for originally one impact category. This might lead to implicit over weighting in the interpretation of the categories acidification and eutrophication compared to other categories, which are not subdivided. In addition, separation of the compartment leads to the fact that always both nitrogen and phosphorus emissions dominate one category. In CML-IA, the user could identify whether nitrogen or phosphorus is more relevant for a certain product system. By separating the compartments, this information is lost.

ReCiPe, AE and, the acidification model by Huijbregts (1999) consider the fate of the substances within the environmental compartments, but in different ways. Furthermore, the background load is taken into account. Within Hauschild and Wenzel (1998) and Heijungs et al. (1992a, b), neither fate nor background contamination are considered. Modeling detailed cause-effect chains including fate of substances and background load supports a more sophisticated and realistic reflection of acidifying and eutrophying substances and their specific impacts on the environment. As fate and background information are region specific, the determined CFs are only valid for the specific region of Europe—either provided as one overall value for Europe (ReCiPe and Huijbregts (1999)) or divided for several European countries (AE). However, the European based CFs are typically applied for the entire life cycle. Most supply chains however also include regions outside of Europe, where regionalized inventory data are only partly available (Berger et al. 2012; Martínez-Blanco et al. 2013). For emissions occurring outside of Europe, the evaluation based on European conditions might be misleading as it results in either over or underestimations of the impacts occurring.

In the second step of the approach, the coverage of elementary flows by each method is determined including a gap analysis of missing elementary flows. The results of these analyses are shown in Table 1. For acidification, CML-IA considers 35 elementary flows. ReCiPe and AE only consider 14 to 20 % of these flows with five elementary flows by ReCiPe and seven by AE. However, as ReCiPe and AE are only assessing terrestrial acidification and CML-IA additionally considers aquatic acidification, only elementary flows contributing to terrestrial acidification can be compared. For terrestrial acidification, 23 flows are covered within CML-IA (identified by not accounting for emissions into freshwater and marine water), leading to a coverage by ReCiPe and AE of 20 to 30 %.

With regard to the category, eutrophication ReCiPe covers 12 flows for the freshwater and 17 flows for the marine compartment. For the terrestrial compartment, AE considers seven flows. Thus, contrary to acidification where ReCiPe and AE together only considering the aquatic compartment, for eutrophication, the aquatic and terrestrial compartments are taken into account when applying both methods together. Thus, the overall amount of CFs provided by ReCiPe and AE can be compared to the overall amount of CFs made available by CML-IA. Whereas CML-IA provides CFs for 52 elementary flows, ReCiPe and AE even together only consider 36 flows. This means that the potential impacts of 16 elementary flows not taken into account.

In step three (a), the relevance of missing CFs is assessed. For acidification, ReCiPe and AE only evaluate terrestrial acidification. As a consequence, only neglected elementary flows contributing to terrestrial acidification are analyzed. However, it should be highlighted that aquatic acidification is an important environmental problem that should not be dismissed within LCA (Guinée et al. 2002; Wright et al. 2011; Dunford et al. 2012; Finstad et al. 2012).

Theoretically, all substances potentially contributing to terrestrial acidification have to be considered within the "perfect" LCIA model. This includes substances emitted directly into soil as well as indirectly due to wet and dry depositions of airborne emissions (Jacobson 2004; Meyer 2013). Nitrogen-based substances are widely accepted as major contributors to acidification and should therefore not be neglected in any impact assessment method evaluating acidifying impacts on the environment (World Health Organization 2006; OECD 2013; European Union 2014). Nitrogen monoxide is not considered within ReCiPe. However, it is a big contributor to acidification as it is emitted during the burning of fossil fuels, which is a vital part of several product systems (Seinfeld and Pandis 2006). Next to nitrogen, sulfur compounds contribute to acidification as well (Dunford et al. 2012; Posch et al. 2015). Sulfur trioxide, which is not covered by ReCiPe, can convert into sulfuric acid when released into air. By wet and dry deposition over land, it can contribute to terrestrial acidification. It is mainly emitted during the use of fossil fuels (Kikuchi 2001; Wright and Welbourn 2002; Jacobson 2004;

Table 1 Overview of considered impact assessment methods, underlying models, reference substance, amount of characterization factors provided, and list of not considered elementary flows for eutrophication and acidification methods CML-IA, ReCiPe, and AE

Impact category		Method	Model	Amount of CFs	Elementary flows not covered
Acidification	Terrestrial and aquatic	CML-IA—Huijbregts (1999) and Hauschild & Wenzel (1998)	Hazard index and stoichiometry	35	- Sulfur oxides to air
	Terrestrial	ReCiPe—Goedkoop et al. (2009)	Base saturation	5	- Chloride, fluoride, and sulfide hydrogen to air and soil - Nitric, phosphoric and sulphuric acid to soil - Nitrogen monoxide to air - Sulfur trioxide to air
	Terrestrial	Seppälä et al. (2006) and Posch et al. (2008)	Accumulated exceedance	7	- Chloride, fluoride, and sulfide hydrogen to air and soil - Nitric, phosphoric and sulfuric acid to soil
Eutrophication	Terrestrial and aquatic	CML-IA—Heijungs et al. (1992a, b)	Stoichiometry	52	- Nitrite to air - Cyanide to water
	Aquatic freshwater	ReCiPe—Goedkoop et al. (2009)	Concentration change	12	- Phosphate, phosphoric acid, phosphorus, and phosphorus pentoxide to air - Phosphate and phosphorus pentoxide to fresh water - Phosphorus and phosphorus pentoxide to soil
	Aquatic marine water	ReCiPe—Goedkoop et al. (2009)	Concentration change	17	- Ammonium, nitric acid and dinitrogen acid to air - Nitric acid to freshwater - Nitric acid to marine water - Ammonia, ammonium, nitric acid, and nitrate to agric. soil - Ammonia, ammonium, nitrate, and nitric acid to indus. soil - (Phosphate, phosphoric acid and phosphorus into water, soil and air)
	Terrestrial	Seppälä et al. (2006) and Posch et al. (2008)	Accumulated exceedance	7	- Ammonium, nitrate, nitric acid, nitrogen, phosphate, phosphoric acid, phosphorus, and phosphorus oxide flows into soil - Dinitrogen oxide to air - Nitric acid to air - Nitrogen to air - Phosphate, phosphoric acid, and phosphorus to air

Apsimon 2014). Furthermore, also hydrogen chloride, hydrogen fluoride, and hydrogen sulfide neglected by ReCiPe as well as AE can contribute to acidification when released to the environment. They are used in different industry sectors like the electronic and waste management industries and can be released when coal, waste, and plastic are burned or during sewage treatment (EPA 2000; Tylenda 2003; Meyer 2013). Additionally, nitric, sulfuric, and phosphoric acids are not considered in ReCiPe and AE, even though their impact on acidification has been proven to be significant (Poor et al. 2001; Bouwman et al. 2002). Nitric acid can be emitted directly by processes using nitric acid like electroplating, fertilizer, or during fossil fuel production. Sulfuric and phosphoric acid can be released during processing of sulfur containing metal ores, phosphate rock processing, or during phosphate acid production (Wright and Welbourn 2002; Hill 2010; Apsimon 2014; Tyrell 2014). Within CML-IA, the CF for sulfur oxides as a sum parameter is missing. However, as sulfur oxides are the sum parameter for sulfur dioxide and sulfur trioxide and are often included in inventory data, an acidification potential should be allocated.

For eutrophication, all substances, which can potentially contribute to oversupply of nutrients in water and soil leading to overgrowth of e.g., algae in water bodies or shifts in plant species towards nitrophilic species, have to be considered. Dry and wet depositions of airborne emissions over soil or water bodies need to be taken into account as an additional source of pollution next to direct release into soil and water (Bergestrom and Jansson 2006; Baron et al. 2014; Palani et al. 2014; Kolzau et al. 2014; Mischler et al. 2014; Trochine et al. 2014). Elementary flows neglected are identified with regard to the considered environmental compartment: for AE, only elementary flows contributing to terrestrial eutrophication (emissions to air and soil); for ReCiPe, only flows contributing to freshwater (emissions to air, soil, and freshwater) and marine (emissions to air, soil, freshwater, and marine water) eutrophication; and for CML-IA, flows contributing to all compartments are analyzed. Nitrogen flows (e.g., dinitrogen oxide, nitric acid, and nitrogen, etc.) into soil are identified as leading substances regarding terrestrial eutrophication (Poor et al. 2001; Bouwman et al. 2002; Pecka and Mill 2012; Baron et al. 2014). However, AE does not provide CFs for ammonium, nitrate, nitric acid, dinitrogen oxide, and nitrogen. Even though phosphate, phosphoric acid, phosphorus, and phosphorus oxide flows have been proven to change soil fertility, these substances are not considered within AE for terrestrial eutrophication (Bennett et al. 2001; Elser et al. 2007; Conley et al. 2009; Palani et al. 2014). ReCiPe does not take into account phosphate, phosphoric acid, phosphorus pentoxide, and phosphorus flows to air, freshwater, and soil although phosphor flows are identified as the main contributor to freshwater eutrophication (Goedkoop et al. 2009). For marine eutrophication, ReCiPe only focuses on nitrogen flows as these were identified as the limited substance. However, several nitrogen flows are not considered, including ammonium, nitric acid, dinitrogen acid, and ammonia. Furthermore, recent studies show that next to nitrogen flows also other emissions e.g., phosphor substances may play a bigger role in marine water eutrophication than originally assumed (Howarth and Marino 2006; Conley et al. 2009). Thus, phosphorus emissions should be included in the model. As the category is subdivided into several subcategories, there is a severe risk that phosphorous emissions into water are neglected at all when freshwater acidification is not a prioritized category. Nitrite as part of common fertilizers leading to eutrophication (Nollet and Gelder 2000; Camargo and Alonso 2006; Shinn et al. 2013) is neglected by CML-IA. Furthermore, cyanide is not considered within CML-IA, even though it can oxidize into the eutrophying substance nitrogen dioxide. They are used by e.g., gold mining or agricultural herbicides and can be emitted additionally during petroleum refining or coal gasification (Dzombak et al. 2005; Cui et al. 2012).

To demonstrate the relevance of the gaps in characterized elementary flows in AE and ReCiPe, a benchmark with CML-IA was performed. For this purpose, the CFs available within CML-IA are divided into two groups: “elementary flows covered by all methods” and “elementary flows covered by CML-IA only” (which equals the flows neglected by AE and ReCiPe). For both groups, the flows and their corresponding CML-IA CFs are shown in Fig. 2 for terrestrial acidification (left), terrestrial eutrophication (center left), freshwater eutrophication (center right), and marine water eutrophication (right). Here, only the values of the CFs are examined; no statement is made in regard to the importance of the elementary flow itself. The impact of an elementary flow with regard to case study results is not only determined based on the value of the CFs but also based on the retrospective amount of the elementary flow. For example, an elementary flow can have a high CF but is barley emitted into the environment and thus, its importance for the examined product systems is low. However, just because certain elementary flows might not be important for one or even the majority of product systems, this does not mean that they are not of upmost importance to some product systems. Thus, determining the importance of an elementary flow is not a straightforward task. However, here, the elementary flows only covered by CML-IA, but not the other methods are ranked with regard to their overall value not to determine their overall importance but to give a first impression how significant the neglect of these elementary flows might be. By not considering elementary flows with high values, the possibility of not considering significant impacts to the environment is higher. However, also elementary flows with only small CFs

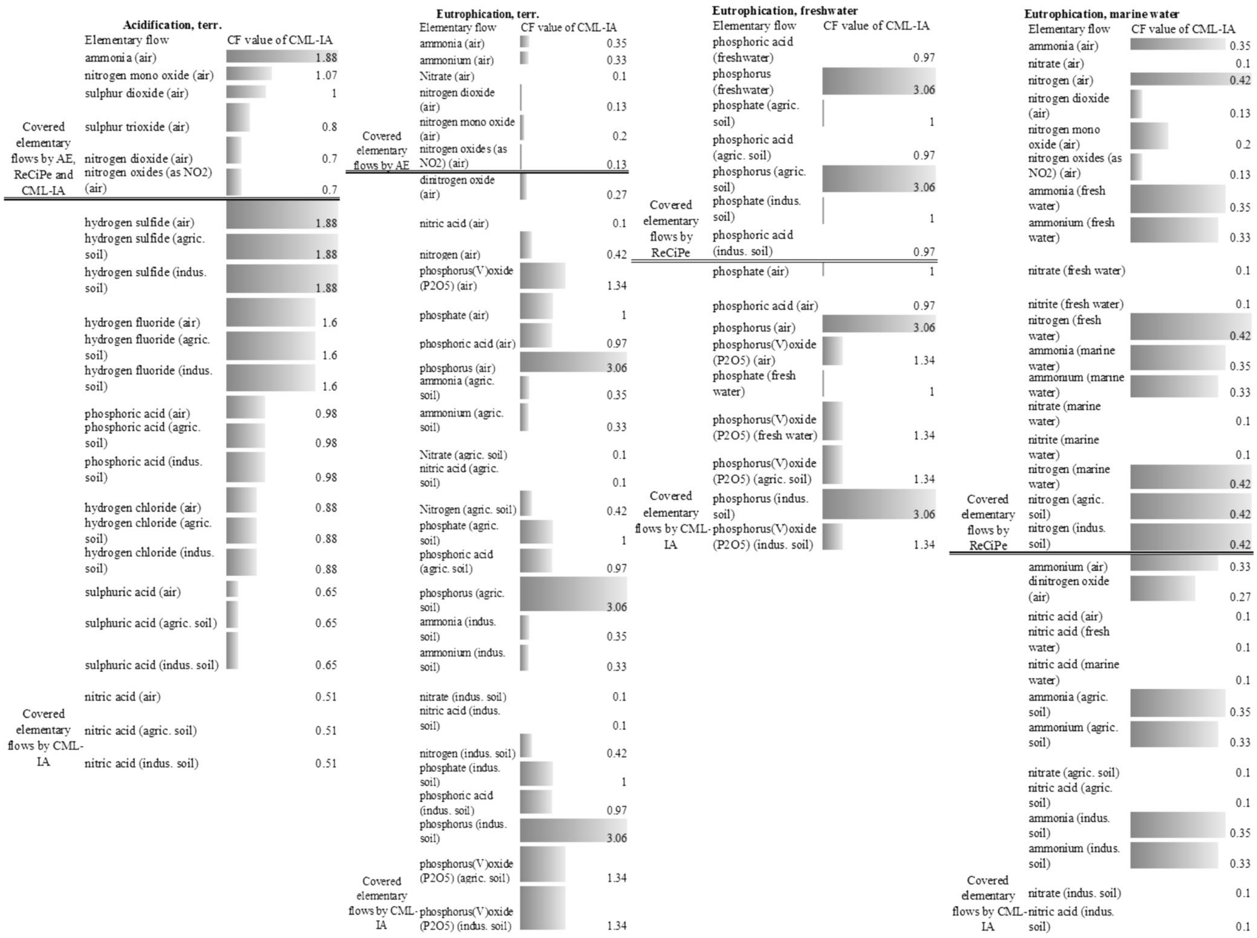

Fig. 2 Overview of elementary flows covered and neglected and their associated characterization factors of CML-IA for terrestrial acidification (*left*), terrestrial eutrophication (*center left*), freshwater eutrophication (*center right*), and marine water eutrophication (*right*)

should not be neglected. For terrestrial acidification, six of the overall 18 neglected substances have a CF higher than one and three of almost one, whereas only three out of six of the covered substances show a CF higher than one. For terrestrial eutrophication, 15 of the 25 neglected substances have a higher CF as all of the 11 covered substances. For freshwater eutrophication, only small differences occur. For marine eutrophication, none of the neglected substance has a CF as high as the considered flows. However, six of the 13 neglected flows show similar CFs.

As shown within the detailed analysis of step three (a), the neglected flows can contribute to acidification and eutrophication and should be included in a comprehensive LCIA method to avoid underestimation of impacts and undesired burden shifting. It is apparent that the choice of the best method suffers from trade-offs between the breadths and the depths of the characterization model and resulting factors. The methods that employ a more specific and sophisticated model of the impact pathway suffer from significant gaps in the coverage of contributing substances. Vice versa, the method which has the broadest coverage of substances has a limited depth with regard to model to environmental mechanisms.

Within step three (b), concurring elementary flows of all three methods are compared with regard to their shares to acidification and eutrophication. For acidification, the elementary flows identified as concurring are just four emissions into air, i.e., ammonia (NH_3), nitrogen dioxide (NO_2), nitrogen oxides (NO_x), sulfur trioxide (SO_3), and sulfur dioxide (SO_2). As shown in Fig. 3 (top left), the relative shares of the four concurring elementary flows differ depending on the LCIA method. Within AE, the highest CF values originate from NO_x and NO_2, whereas for CML-IA and ReCiPe NH_3, SO_2, and SO_3 have the biggest CF values. However, for ReCiPe, the CF value of SO_3 is even higher as for CML-IA. For NH_3, the biggest variance between AE and ReCiPe occurs. When comparing the CFs of NH_3 in both methods, the characterization factor in ReCiPe (1.99) is around 83 % higher as for AE (0.33). All three models consider the fate of the substances within the environmental compartments, but the acidification model applied within CML-IA does not use the most current data. ReCiPe and AE both take background

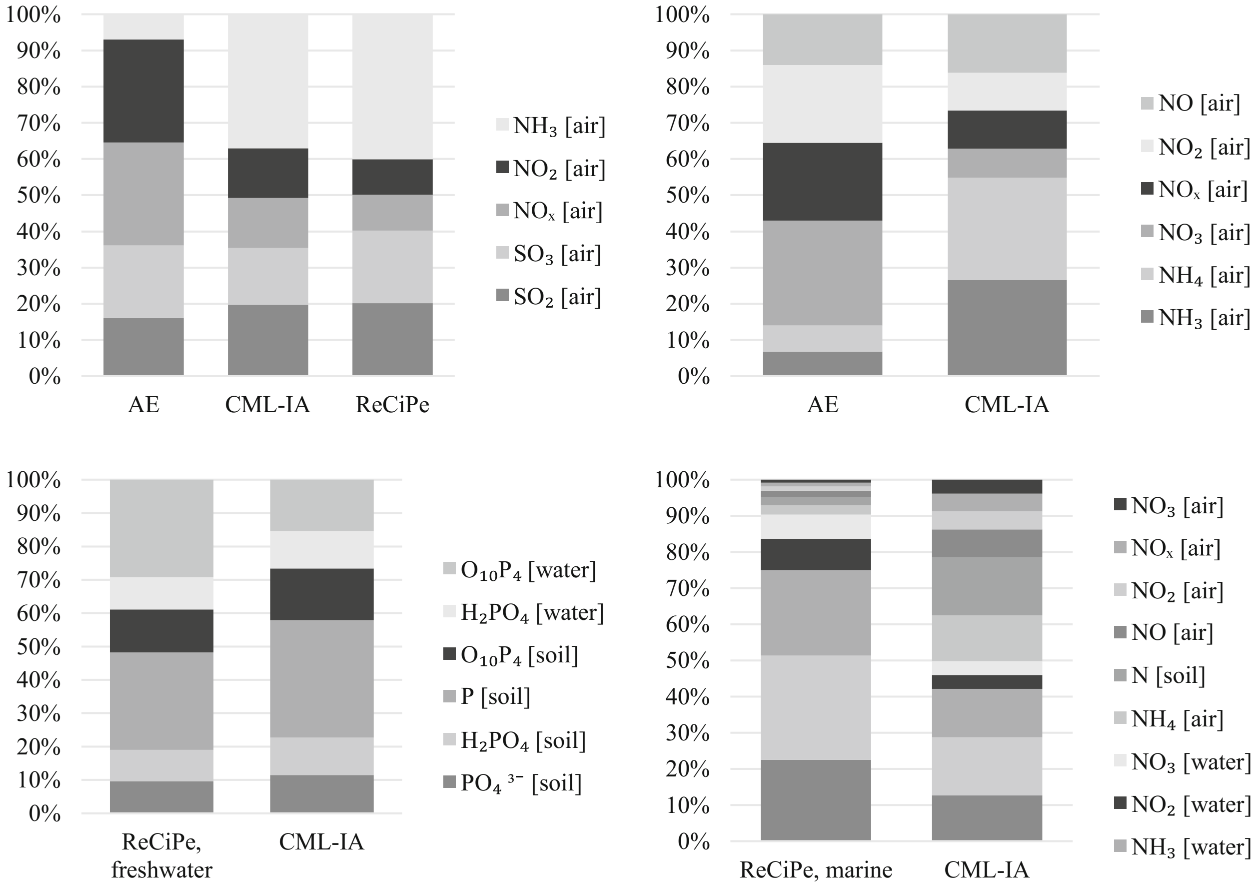

Fig. 3 Shares of acidifying elementary flows (*top left*) to indicator results and of terrestrial (*top right*), marine water (*lower right*), and freshwater (*lower left*) eutrophying elementary flows to indicator results for CML-IA, ReCiPe, and AE

contamination into account. The AE model even goes one step further by including the sensitivity of the environment. Thus, it might be more precise when it comes to determining the impacts of these substances and their impacts on European soils.

For eutrophication, the three compartments terrestrial, freshwater, and marine have to be considered separately (see Fig. 2 top right and bottom graphs). For terrestrial eutrophication, assessed by CML-IA and AE, six concurring elementary flows are identified, i.e., nitrogen monoxide (NO), NO_2, NO_x, nitrate (NO_3), ammonium (NH_4), and ammonia (NH_3) to air. CF values of NH_3 and NH_4 are high within CML-IA, but low within AE. For AE, however, CF values of NO_3, NO_x, and NO_2 are high, whereas they are low within CML-IA. The biggest difference occurs for NH_4. As AE considers the background load as well as the ecosystem sensitivity, it can be seen as more precise for impacts on European soils. However, as most supply chains and associated emissions are not only European but include manufacturing locations all over the world, the CFs of AE can lead to over or underestimation of impacts. Comparing again the six concurrent elementary flows (i.e., phosphorusoxide (P_4O_{10}) to water, dihydrogen phosphate ion (H_2PO_4) to water, phosphorus (P) to soil, H_2PO_4 to soil, phosphate (PO_4^{3-}) to soil) for freshwater eutrophication of CML-IA and ReCiPe shows the smallest differences compared to the other compartments. Both methods identify P as the main contributor for freshwater eutrophication. However, for P_4O_{10} to water, bigger differences occur. As ReCiPe considers fate of substances and background load, it might be more precise for determining the impacts on European soils and in European freshwater bodies. However, for global supply chains, CML-IA might be the better choice. For marine eutrophication, 11 concurrent elementary flows were identified for CML-IA and ReCiPe i.e., NO_3, NH_4, NO, NO_2, and NO_x to air; nitrogen (N) to soil; and NH_4, N, NH_3, NO_2, and NO_3 to water. The CF values of NH_4, NO_3, NO_2, N, and NH_3 flows into water are higher for ReCiPe than for CML-IA. Within CML-IA, the CF value of N into soil is higher as for ReCiPe. ReCiPe considers the fate of substances as well as background load and thus might be more precise for determining the impacts on European marine water bodies. However, for assessing the impacts in global supply chains, CML-IA might be a better choice.

In step three (b), it was shown that the share of the concurring elementary flows is different depending on the chosen LCIA method. The overall effect of these differences might even be amplified when neglected flows would be taken into account as well. Taking into account the concurrent flows, only for a few flows significant differences occur. In that sense, the more sophisticated models did not lead to fundamentally different results. Thus, using a less sophisticated method, such as CML-IA, that has a much broader coverage of elementary flows might be more suitable for assessing potential environmental impacts within LCA than methods that appear more sophisticated, but which cover only a rather small part of the relevant emissions.

4 Conclusions and outlook

The developed three-step approach has shown that newly developed LCIA methods for the impact categories acidification and eutrophication tend to do the following:

- separate impact pathways according to their environmental compartments,
- consider fate of the substances,
- include background load,
- take into account substance specific behavior,
- provide regionalized CFs (so far limited to Europe), but
- consider fewer substances and therefore have gaps in the coverage of elementary flows.

Furthermore, it was demonstrated that these neglected elementary flows have in reality acidifying and eutrophying impacts, and therefore, their neglection represents a significant gap of these models. It was also demonstrated that only some concurring elementary flows differ significantly. All identified aspects might lead to substantial differences in case study results depending on the LCIA methods defined in the goal and scope. Thus, for adequate decision-making support, newly developed method cannot just replace established ones, rather a transparent transitioning process has to take place.

In a nutshell, the more recent methods do lead to different results for concurrent flows, but have a much smaller coverage of elementary flows. As mentioned earlier, the subdivision into more granular impact categories is not really an application advantage as such. Based on the assessment presented above, switching to AE and ReCiPe does not automatically improve the decision support quality of LCIA. Explaining method switches and associated differences to decision makers without significant benefits or improvements is usually not recommended. The decision support maturity of AE and ReCiPe is not yet sufficient to make the case for a method switch from CML-IA. It just makes LCA application more complex without a proven improvement in decision support quality.

The introduced approach is a first step in setting up a benchmark or testing procedure of established against newly developed methods to transparently support the application of newly developed LCIA methods for decision-making processes. Within this paper, the focus is on methods used within Europe. However, identified results and associated consequences of the analysis can be transferred to other regions and models like TRACI (Bare 2002) used within the USA or the Japanese LIME model (Itsubo and Inaba 2012).

At the end of the day, the selection of the preferred LCIA method is a value choice. If a new method is preferred due to a more sophisticated impact pathway or simply to document that a practitioner is up-to-date, this is acceptable as long as the inherent gaps of the new methods are accounted for. One option could be that next to the newly developed methods also established methods like CML-IA are used during the interpretation of the results. Thus, neglected elementary flows by AE and ReCiPe are taken into account, and their influence on the production system can be analyzed. Furthermore, newly developed methods should be sufficiently tested in case studies of several sectors and results have to be compared to established methods until application readiness is proven. This should include explicit application of new versus established methods in a range of case studies that should be selected from the perspective of covering the full range of low to high expected differences in the results. It should be the duty of the developers of the new method to transparently explain the differences—positive and negative. Changes in results just based on method switches need to be made plausible to the responsible stakeholders using or judging upon the LCIA methods, like implementers of LCIA methods into software and databases, reviewers, as well as in the end to decision makers.

International standards, schemes, and groups (e.g., ISO, national standardization bodies, scientific working groups like UNEP/SETAC life cycle initiative, EPD operators, labeling procedures, as well as the PEF pilot process) should do better in serving this purpose, by assessing different LCIA methods in a transparent way. Unfortunately, no initiatives so far succeeded to do so in a proper way, and new methods are rather applied or recommended without proper reflection or comparison with existing methods. These exercises generate a lot of numbers, but rather limited information with regard to the decision support maturity of new methods. Thus, the results of LCA projects and studies in different sectors might be distorted as the proposed methods lead to other results than the established methods.

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Product Environmental Footprint in Policy and Market Decisions: Applicability and Impact Assessment

Annekatrin Lehmann,† Vanessa Bach,*† and Matthias Finkbeiner†*
†Chair of Sustainable Engineering, Department of Environmental Technology, Technische Universität Berlin, Berlin, Germany

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EDITOR'S NOTE:

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Special Series

ABSTRACT

In April 2013, the European Commission published the Product and Organisation Environmental Footprint (PEF/OEF) methodology—a life cycle-based multicriteria measure of the environmental performance of products, services, and organizations. With its approach of "comparability over flexibility," the PEF/OEF methodology aims at harmonizing existing methods, while decreasing the flexibility provided by the International Organization for Standardization (ISO) standards regarding methodological choices. Currently, a 3-y pilot phase is running, aiming at testing the methodology and developing product category and organization sector rules (PEFCR/OEFSR). Although a harmonized method is in theory a good idea, the PEF/OEF methodology presents challenges, including a risk of confusion and limitations in applicability to practice. The paper discusses the main differences between the PEF and ISO methodologies and highlights challenges regarding PEF applicability, with a focus on impact assessment. Some methodological aspects of the PEF and PEFCR Guides are found to contradict the ISO 14044 (2006) and ISO 14025 (2006). Others, such as prohibition of inventory cutoffs, are impractical. The evaluation of the impact assessment methods proposed in the PEF/OEF Guide showed that the predefined methods for water consumption, land use, and abiotic resources are not adequate because of modeling artefacts, missing inventory data, or incomplete characterization factors. However, the methods for global warming and ozone depletion perform very well. The results of this study are relevant for the PEF (and OEF) pilot phase, which aims at testing the PEF (OEF) methodology (and potentially adapting it) as well as addressing challenges and coping with them. *Integr Environ Assess Manag* 2015;11:417–424. ©2015 SETAC

Keywords: Comparability Harmonization Impact assessment ISO 14044 ISO 14025 OEF PEF

INTRODUCTION

International Organization for Standardization (ISO) 14044 (2006) is currently the only standardized method to assess a range of potential environmental impacts of products and services. It is transparent, widely used, and accepted throughout policy and in the market. Life Cycle Assessment (LCA) is applied both internally for environmentally oriented product and process development as well as externally to document environmental performance in environmental product declarations or in the development of type I label criteria. In 2013, the European Commission published the Product and Organisation Environmental Footprint (PEF/OEF) methodology—a life cycle-based multicriteria measure of the environmental performance of products, services, and organizations (European Union 2013a) (PEF/OEF Guide), developed as part of the 2020 European strategy "A resource efficient Europe" (European Union 2013b). With its approach of "comparability over flexibility," the PEF/OEF methodology aims at harmonizing existing methods, while decreasing the flexibility provided by the ISO standards regarding methodological choices. This is reflected, for example, in predefined life cycle impact assessment (LCIA) methods.

The PEF/OEF methodology is currently widely discussed amongst stakeholders, and skepticism exists in industry and consumer organizations, as well as in the scientific community. For example, concerns expressed by the European Automobiles Manufacturers Association (ACEA), Orgalime (European Engineering Industries Association), and the European consumer voice in standardization (ANEC) (Orgalime, ACEA, ANEC 2013) refer to potential unfair competition and market distortion at the consumer level. Concerns from a scientific perspective are described in Finkbeiner (2014) including a lack of conformity with international standards, predefined and partly untested impact assessment methods, and objections to the PEF/OEF "comparability over flexibility" approach.

Therefore, the methodology is also a challenge for existing environmental labeling schemes such as The Blue Angel

Lehmann and Bach contributed equally to the work.
* Address correspondence to: vanessa.bach@tu-berlin.de
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(https://www.blauer-engel.de/) or the EU-Flower (http://ec.europa.eu/environment/ecolabel/index_en.htm). Altogether, this reveals risks of impeding a life cycle-based environmental analysis of products and services.

The first wave of a 3-y PEF/OEF pilot phase started in November 2013 with the aim of testing the methodology in practice and developing product category and organization sector rules (PEFCR/OEFSR) for selected products and sectors; a second wave started in June 2014. PEFCRs and OEFSRs are life cycle-based rules that complement the general PEF/OEF Guide by providing further specification at the level of a specific product or sector category (European Commission 2014a). In the pilot phase, challenges regarding its applicability—known as well as new ones, likely to appear during the pilot phase—need to be addressed and coped with.

The goal of this article is to present and discuss the main differences between the PEF and the ISO methodologies (ISO 14044 [2006] and ISO 14025 [ISO 14025 2006]), identify and highlight related challenges regarding the PEF methodology and its applicability in practice, with a focus laid on impact assessment. Our report complements the previous analyses in the PEF Guide (European Union 2013a, Annex X) and Finkbeiner (2014) by further discussing challenges and the applicability of the PEF requirements. Regarding the impact assessment, 14 methods on midpoint level that were adopted by the PEF/OEF methodology and have to be applied when carrying out a PEF/OEF study were evaluated. Even though an analysis of these methods exists, conducted within the ILCD (International Reference Life Cycle Data System) Handbook (European Commission—Joint Research Centre—Institute for Environment and Sustainability 2011) in a comprehensive way, important criteria regarding their applicability are underrepresented in that analysis and thus picked up in this paper. Hence, our evaluation complements the existing assessment and highlights challenges of the proposed impact assessment methods. Additionally, selected challenges of the PEF methodology identified in our analyses are illustrated by using a theoretical example.

METHODOLOGICAL APPROACH

This study followed a 2-step approach: 1) summary and discussion of differences between the ISO and the PEF and highlighting challenges of the methodology, and 2) a thorough analysis of the PEF/OEF impact assessment methods and description of their challenges. The methodological approach of the latter is further described in the next subsection. A theoretical example (comparing biobased with conventional plastics) was used to test selected methodological requirements of the PEF methodology and to illustrate the identified challenges. As an illustrative example, we compare a thermoplastic cellulose acetate and a thermoplastic polymethylmethacrylate for biobased and conventional plastics, respectively (with a functional unit of 1 kg plastic).

This example addresses all challenges related to the impact assessment and the following selected other challenges: identification of a suitable CPA/NACE code (Classification of Products by Activity/Nomenclature statistique des activités économiques dans la Communauté européenne) and the ban of cutoffs.

Criteria-based evaluation of impact assessment methods

To assess the suitability of the methods, a new evaluation scheme based on the evaluations from the European Commission Joint Research Centre (2011) and Reimann et al. (2010) was developed. Overall, 4 main criteria are considered for evaluation: "stakeholder acceptance," "documentation and review," "environmental relevance," and "applicability." As shown in Figure 1, each criterion is specified by 2 to 5 subcriteria.

For the criterion "stakeholder acceptance," 3 subcriteria were considered. They were chosen to represent the following requirements: To communicate the results to the broad public

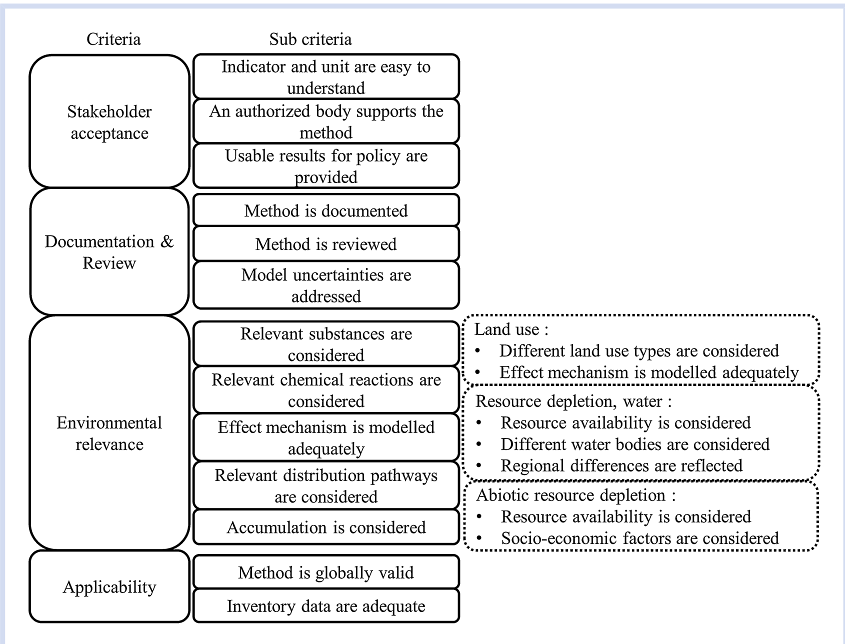

Figure 1. Assessment scheme for evaluating 14 impact assessment methods considering 4 main criteria each with 2–5 subcriteria.

or layperson, politics, and industry, the impact indicator as well as the corresponding unit has to be easy to understand. Moreover, when being supported by an authorized body, it is more likely that the method will be accepted by other stakeholders as well. For the criterion "usable results for policy are provided," all considered impact assessment categories are assumed to be relevant for politics or policy. Thus, the assessment focuses on whether the models are able to provide results that are useful on a policy level.

The criterion "documentation and review" has been evaluated based on whether the method is documented and reviewed and uncertainties are addressed. A thorough documentation of the method itself, background models, and resulting available characterization factors other than a reviewed journal paper is required for a holistic understanding. Furthermore, the scientific quality of the method is assumed to be higher when scientists dealing with the same or related topics have reviewed the method thoroughly. Addressing uncertainties regarding, for example, model assumptions—qualitatively as well as quantitatively (as done within the USEtox model [Rosenbaum et al. 2011])—improves the understanding of potential uncertainties of the impact assessment results, which therefore supports the interpretation.

The third criterion, "environmental relevance," evaluates the ability of a method to reflect the impacts on the environment adequately. Therefore, it is evaluated in the sub criteria if relevant substances and chemical reactions are considered, if effect mechanisms are modeled adequately, and if relevant distribution pathways and accumulation are considered. Furthermore, in a method properly reflecting the environmental impacts, characterization factors are assumed to be provided for all relevant substances. Missing factors would lead to a neglect of substances relevant for the impact category. Thus, as a first step, a classification of all relevant substances has to take place. Relevant substances can be identified based on the current knowledge in this field. Moreover, existing impact assessment methods should be included, because the elementary flows considered up to now are assumed to present relevant substances for this category. Based on the identified relevant substances, related chemical reactions and distribution pathways should be identified considering state-of-the-art research in the individual science disciplines. The work of the Intergovernmental Panel on Climate Change (IPCC 2007), for example, recently highlighted the growing problem of ocean acidification, which is so far not covered in LCA. Based on this research, an LCIA method can be developed considering relevant elementary flows such as carbon dioxide as well as relevant reactions such as the formation of hydrogen ions in water. Because the impact categories land use, depletion of water, and depletion of abiotic resources are input based, they have to be evaluated with a different set of criteria. For a comprehensive assessment of land use, the different land use types have to be considered, because their contribution to biodiversity or soil quality varies. Also, several effect mechanisms attributable to different land use types have to be modeled adequately. For water depletion, the resource availability and differentiation of water bodies need to be considered within the model, and regional aspects have to be reflected. To determine abiotic resource depletion, both the physical availability as well as socioeconomic aspects have to be considered.

The fourth criterion assesses the applicability of the impact assessment method. A sufficient amount of inventory data has to be available to use the method in practice. Global validity is also important, because supply chains have a global, not a European, scope.

As a result, the methods are ranked between levels 1 and 3 for each criterion. A 3-level approach was chosen for the classification, following an easy-to-understand traffic light rating system. Such systems are already widely applied in literature, for example, for impact assessment in Social LCA (e.g., Franze and Ciroth 2011; Martínez-Blanco et al. 2014). A classification of 1 indicates that the method fulfils all requirements of the criterion, and a ranking of 3 means that the demands could not be reached. Because many methods can only partly achieve the criteria requirements, an intermediate ranking was included (level 2).

The concluding recommendation is based on the results of this assessment and supported by an example comparing bio-based and conventional plastics.

RESULTS AND DISCUSSION

In the following sections, we summarize the differences between ISO and PEF methodologies and the identified challenges as well as the results from the evaluation of impact assessment methods, using the previously mentioned example for illustration.

Differences between ISO and PEF and challenges of the PEF methodology

The PEF differs regarding goal and scope, mainly because it introduces a set of new terminology for globally accepted terms (e.g., *unit of analysis* instead of *functional unit*), it is communication driven, whereas ISO is not, and it specifies how the unit of analysis shall be defined (amongst other methods, linked with CPA/NACE codes) (see also Finkbeiner [2014]). The introduction of (in our opinion unnecessary) new terms was already reviewed by the European Commission, noting that renaming common terms found limited support by the stakeholders (Galatola and Pant 2014). With regard to communication, appropriate methods or vehicles are yet missing. In the context of communication (and intended application), in the PEF overview on key requirements of PEF methodology (European Union 2013a, Annex X), business-to-business and business-to-consumers are claimed as communication targets of ISO 14044, whereas this is not directly stated in ISO 14044 (2006). The PEF requirement of linking the unit of analysis with CPA/NACE codes appears not to be a straightforward task: Taking the theoretical example on bio-based and conventional plastics, the category "manufacture of plastic products" (CPA/NACE code 25.2) would apply most. However, more than 1 category would be needed to cover the product system adequately—the production of raw materials for bio-based plastic (here mainly cellulose from recycled paper and pulpwood), for example, differs from raw material production of conventional plastic, but it is not reflected in the previously mentioned category and should thus be described in an additional category (maybe CPA/NACE code 17.12, "manufacture of paper and paperboard").

Regarding the inventory, one main difference is that the PEF Guide does not allow cutoffs. This would mean that all processes have to be modeled—in the plastics example, all machinery, even the extruder, which has a negligible contribution to the overall environmental impact. This is hardly feasible in practice, because it causes extra effort and costs without providing additional information. However, this requirement is relativized in the PEFCR Guide: although not using the term "cutoff," it states that processes identified (in the screening study) as being not relevant—thus neglible—do not need to be considered in the subsequent PEF study. As in ISO 14044 (2006), fixed criteria defining when a process is considered as relevant or not (e.g., contribution of less than 5% to the impact category) are not provided in the PEF/PEFCR Guides. As a result of the screening study during the PEFCR development, one may exclude certain impact categories in the PEFCR, if identified as not being relevant (European Commission 2014b), without knowing yet how relevance should be measured. Methodologically, excluding impact categories, however, means that elementary flows are cut off. Moreover, the PEF methodology provides numerous specifications, for example, on how to deal with fossil and biogenic carbon (C), land use change, C storage, and so forth, or on quantitative minimum data quality requirements and data sources as well as regarding allocation by proposing a new "recycling formula."

Regarding impact assessment and interpretation, the PEF methodology differs because it provides a default set of 14 impact assessment methods (further explained in the following section). Normalization is a recommended step and has to be tested within the PEF pilot phase. Therefore, European normalization factors are provided by the European Commission (European Commission 2014c). Considering only European emission data disregards the fact that most raw materials are produced outside of Europe. Furthermore, except for some aspects such as greenhouse gas emissions, global as well as regional emission data necessary to determine normalization factors are not yet available. Hence, the provided normalization factors are so far based on estimations, and normalized results should be interpreted with care, such as the significance of impact assessment categories for a specific product system (Heijungs et al. 2006; Benini et al. 2014). To calculate credits or burdens for the end of life (EoL), a specific formula to deal with multifunctionality in recycling situations is given (European Commission 2014b), which requires the collection of 18 factors regarding the EoL product system. Generic data should be used if specific data are not available. For bio-based plastic, in which so far generic processes are lacking, not all requested factors can be provided by the modeled system, and generic data from conventional plastic have to be considered. Thus, in this example, a distinction between the impacts of bio-based and conventional plastic in the EoL stage is not possible. Moreover, the PEF methodology allows weighting as an optional step (results provided as "additional environmental information") for business-to-business or business-to-consumers communication with comparisons and comparative assertions. This clearly contradicts ISO 14044 and ISO 14025 as well as the fact that a PEF study can be used as claim about the environmental superiority or equivalence of a product compared with another product (European Commission 2014b), whereas ISO states that LCIA shall not be the sole basis for interpretation in comparative assertions (ISO 14044 2006).

Criteria-based evaluation of impact assessment methods

In this section, the evaluation results for all 14 impact assessment methods are presented. Because of the complexity of the analysis, not all criteria can be discussed in detail. Key points that reflect the overall maturity of the method are highlighted.

Table 1 shows that stakeholder acceptance has been rated very well for most methods. Environmental topics discussed in the public for years such as global warming (Barker 2007; IPCC 2007) have the advantage of being understood even by laymen, although underlying indicators might be complex. Stakeholder acceptance of methods for ionizing radiation, toxicity, water, and abiotic resource depletion is lacking as several sub criteria are evaluated with level 3. Because the methods for assessing abiotic resource depletion based on reserves instead of ultimate reserves (Guinée et al. 2001; Oers et al. 2002) and ionizing radiation (Frischknecht et al. 2000) have never been used in LCAs so far and therefore have never been tested, their stakeholder acceptance is rated as minor; however, for policy the abiotic depletion potential (ADP) method (Guinée et al. 2001) might be of interest as the economic resource availability is assessed. Because of its complexity and high uncertainties, the UseTOX method for assessing human and ecosystem toxicity (Rosenbaum et al. 2008; Rosenbaum et al. 2011) is limited in its comprehensibility and applicability for policy. Even though it becomes more reliable by addressing the uncertainties within a method, when these uncertainties are very high as is the case for USEtox, where the result varies by 1 or 2 orders of magnitude, policy cannot rely on the results. Furthermore, because USEtox has never been applied in a policy context before—contrary to, for example, the accumulated exceedance method for acidification and eutrophication, which has been used in mitigation strategies in Europe for years—real life experience using and interpreting this method is missing. The ecological scarcity method (Frischknecht et al. 2009) is easy to understand, but because of its severe flaws, such as modeling artefacts that lead to a higher water scarcity in Germany than in Australia (Finkbeiner 2014), it is not suitable for use in policy in its current form.

The documentation is evaluated as very well for most methods. However, some methods are using background models, which are not publicly accessible and are therefore ranked at level 2. All methods besides the Riskpoll model for particulate matter (Rabl and Spadaro 2004) are published in scientific journals. Furthermore, only 5 of the 14 methods provide uncertainty assessments.

Many methods such as the World Meterological Organization method for ozone depletion (WMO 2010) or the IPCC (2007) for climate change perform very well regarding their environmental relevance. Because both methods are very mature, they could be used, not exclusively though, for a comparative assertion such as bio-based and conventional plastics. The ionizing radiation method (Frischknecht et al. 2000) only considers effect mechanism regarding nuclear power production and only within France. Thus, its reflection of the environmental relevance is not satisfying. The plastics example shows that ionizing radiation is predominantly related to electricity production. Because the method has not been tested sufficiently so far, that other pathways and related inventory data for, for example, fertilizer production (Scholten and Timmermans 1996), are considered adequately as well cannot be ensured. For the example chosen, this means that excluding this aspect could lead to an inequitable advantage for the bio-based material. The USEtox model (Rosenbaum et al. 2008) for assessing human toxicity and ecotoxicity only covers a small amount of the potentially toxic chemicals (although more than 3000 characterization factors are included in the latest version [Rosenbaum et al. 2011] and cumulative toxic effects cannot be assessed so far [Finkbeiner et al. 2014]).

Table 1. Evaluation results of impact assessment methods shown for main criteria and subcriteria

Impact Assessment Categories	Stakeholder Acceptance[a]	Documentation & Review[b]	Environmental Relevance[c]	Applicability[d]
Climate change	1 # 1 # 1	1 # 1 # 1	1 # 1 # 1 # 1 # 1	1 # 1
Ozone depletion	1 # 1 # 1	1 # 1 # 1	1 # 1 # 1 # 1 # 1	1 # 1
Particulate matter	1 # 2 # 2	2 # 1.5 # 3	2 # 1 # 1 # 1 # 1	2 # 2
Ionizing radiation	1 # 3 # 3	1 # 1 # 2	1 # 1 # 2 # 2 # 2	3 # 2
Photochemical ozone creation	1 # 2 # 2	2 # 1 # 3	1 # 1 # 1 # 1 # 1	2 # 1
Eutrophication, terrestrial	1 # 2 # 1	2 # 1 # 3	3 # 1 # 1 # 1 # 1	2 # 1
Eutrophication, aquatic	1 # 2 # 2	2 # 1 # 3	3 # 1 # 1 # 1 # 1	2 # 1
Acidification, aquatic	1 # 2 # 1	2 # 1 # 3	3 # 1 # 1 # 1 # 1	2 # 1
Human toxicity, carcinogen	3 # 1 # 2	1 # 1 # 1	3 # 2 # 1 # 1 # 2	1 # 2
Human toxicity, noncarcinogen	3 # 1 # 2	1 # 1 # 1	3 # 2 # 1 # 1 # 2	1 # 2
Eco toxicity	3 # 1 # 2	1 # 1 # 1	3 # 2 # 2 # 2 # 2	1 # 2
Land use	1 # 2 # 2	2 # 1 # 3	2 # 3[e]	2 # 2
Water depletion	1 # 3 # 3	1 # 1 # 3	2 # 3 # 3[f]	2 # 3
Abiotic resource depletion	1 # 3 # 2	1 # 1 # 3	2 # 3[g]	1 # 3

[a]Subcriteria: indicators and unit are easily understandable, an authorized body supports the method, usable results for policy are provided
[b]Subcriteria: method is documented, method is reviewed, model uncertainties are assessed
[c]Subcriteria: relevant substances are considered, relevant chemical reactions are considered, effect mechanism is modelled adequately, relevant distribution pathways are considered, accumulation is considered
[d]Subcriteria: method is globally valid, inventory data are adequate
[e]Subcriteria: different land use types are considered, effect mechanism is modelled adequately
[f]Subcriteria: resource availability is considered, different water bodies are considered, regional differences are reflected
[g]Subcriteria: resource availability is considered, socio-economic factors are considered

Thus, its environmental relevance is not satisfying. Some characterization factors such as those for metals are marked as interim; for them, proceeding with caution is advised (Rosenbaum et al. 2011). Hence, this method should be used with care in comparative LCA studies. The accumulated exceedance method (Seppälä et al. 2006) for acidification and eutrophication only provides few characterization factors (Seppälä et al. 2006; Finkbeiner 2014). Within the plastics example, the accumulated exceedance method (Seppälä et al. 2006) covers 7 elementary flows less for the acidification potential than the impact assessment method developed under the lead of the Center of Environmental Science of Leiden University (CML) (Guinée et al. 2001). For the products compared here, these neglected emissions only sum up to 2% of the overall impact, but for other product systems their contribution may be more significant, especially when metals are considered. The land use method (Milà i Canals et al. 2007) is the third evaluated as not being satisfactory. Soil organic matter only serves as a headline indicator for soil fertility and climate regulation; other aspects such as loss of biodiversity have to be considered by applying additional impact assessment methods (Milà i Canals et al. 2007; Brandao et al. 2011). Thus, this method should be applied with care within comparative studies as well, especially when the assessed systems differ significantly (such as for bio-based vs conventional plastics). The ecological scarcity method (Frischknecht et al. 2009) used to assess water depletion does not adequately consider all environmental relevant factors for the resource freshwater, because it is only based on relative scarcity (Finkbeiner 2014). Because no regional inventory data are available in LCA databases so far, water footprint case studies are very time consuming and difficult to carry out, especially for complex industry products (Berger et al. 2012). For PEF studies, European weighted characterization factors are provided, which consider different water bodies but are not geographically explicit. Especially for the evaluation of water depletion, regional differences are of upmost importance (Pfister et al. 2009; Berger et al. 2014; Finkbeiner et al. 2014). Thus, applying results of average characterization factors cannot be used to support the decision making for a product system. This is shown in the example of bio-based plastics compared with conventional plastic, where regional differences are not considered and only the amount of water consumed is the deciding factor. Conventional plastic is produced from crude oil, which is extracted from countries all over the world. High amounts are obtained in Saudi Arabia (Central Intelligence Agency 2012), which is a water-scarce country. The bio-based plastic example cellulose acetate is made mainly from wood pulp (Sustainable Forestry Initiative 2014; Balser et al. 2015). Current trade statistics (United Nations Economic Commission for Europe 2013) show that low amounts of industrial round wood used for wood pulp are exported within Europe. Thus, the 30% cellulose acetate produced in Europe (Dammer et al. 2013) can be assumed to be manufactured from the wood of European trees. European countries have on average less water scarcity than most oil-producing countries outside of Europe. Thus, a regionalized impact assessment method should be recommended. The method for ADP (Guinée et al. 2001) is also not sufficient regarding its environmental performance, because it evaluates resources only from an economic point of view considering reserves. Neither physical availability nor proper socioeconomic factors addressing scarcity are considered. Furthermore, regarding the example, ADP indicator results for conventional plastic would probably be worse than for bio-based plastic because ADP only considers abiotic resources. So far, no impact assessment methods are available for evaluating biotic resource depletion (European Commission—Joint Research Centre—Institute for Environment and Sustainability 2011; Finkbeiner et al. 2014). Thus, a comparison of abiotic and biotic resource depletion is not possible.

The biggest challenge regarding the applicability of these methods is the availability of inventory data. Most of the methods including accumulated exceedance method (Seppälä et al. 2006) are not valid worldwide because only European background data are considered. When assessing silvicultural systems such as forestry for bio-based plastic, regional aspects might be of importance for the overall environmental profile. Acidifying substances, which are caused by transports, might be considered less or more critical when emitted, for example, in Canada instead of Europe. Because supply chains are spread out around the world, a uniform world model as used within RiskPoll (Rabl and Spadaro 2004) or USEtox (Rosenbaum et al. 2008) has clear advantages. Because eutrophication and acidification are assessed within LCA for many years, the inventory data can be assumed as more or less complete.

RECOMMENDATIONS AND OUTLOOK

In summary, half of the impact assessment methods evaluated here cannot be seen as adequate to be used solely for decision support such as comparative assertions. However, methods for global warming and ozone depletion are very good, as evaluated in all subcategories.

The idea of the PEF/OEF to harmonize existing life cycle-based methods is good, but although all methods reference ISO 14044 as their basis (except for the C footprint standards with limited focus of climate change) (Finkbeiner 2014), the PEF/OEF methodology shows several differences and even contradiction, such as allowing for weighting of LCA results to single scores. Besides the fact that the new methodology may cause confusion amongst stakeholders and bears the risk of impeding a life cycle-based environmental analysis of products and services as such, the applicability (as well as the acceptance) of the method is not yet proven. Questions remain, for example, regarding the identification of suitable NACE/CPA codes, dealing with the ban of cutoffs, and impact assessment methods, especially for water depletion and land use. These questions and related challenges were discussed in this article and illustrated using a theoretical example comparing bio-based and conventional plastic. Identifying a suitable NACE/CPA code is not a straightforward task, because more than one category would be needed to cover the product system adequately.

The evaluation of the PEF/OEF impact assessment methods and their application in the example showed that using the proposed methods for land use and water depletion is not recommended in general, but especially not for comparing different product systems. For water depletion, more commonly used and better tested methods are available (Pfister et al. 2009; Berger et al. 2014; Finkbeiner 2014). Other relatively new methods such as ionizing radiation (Frischknecht et al. 2000) or accumulated exceedance (Seppälä et al. 2006; Posch et al. 2008) should be tested regarding their applicability and necessity within the pilot phase. Applying USEtox (Rosenbaum et al. 2008) presents a challenge because of several issues mentioned within this paper. The pilot phase should be used to determine which product systems should use this method. The IPCC method (IPCC 2007) as well as the World Meterological Organization method (WMO 2010) can be applied without any difficulty. The results of this evaluation of impact assessment methods can be taken into account within the PEF/OEF pilot phase, together with the original evaluation and classification of the European Commission (2011).

Furthermore, important environmental effects such as loss of biodiversity, which cannot be addressed adequately within LCA so far (Finkbeiner et al. 2014), might have a big influence on the result and should be considered qualitatively, maybe within the section Additional Environmental Information in the PEFCRs/OEFSRs.

Further challenges of the PEF methodology beyond the scope of this paper include the applicability of the data quality assessment scheme, the suitability of the provided allocation approach for recycling, and weighting or the identification of appropriate measures to communicate PEF/OEF results. Benchmarks and performance classes shall be defined during PEFCR development to enable comparisons between products, but for now it is not clear how they should be defined practically nor theoretically if a benchmark system is to be meaningful.

The European Commission promotes the use of PEF and OEF for measuring and communicating environmental life cycle performance of products and organizations. Thus, it provides opportunities to support decision-making processes at all levels, that is, industry and policy. However, for this, information obtained must be based on a solid methodology, and appropriate communication tools must be used, but both of these conditions are not yet ensured.

The ongoing pilot phase intends to test the PEF/OEF methodology, to deal with remaining questions, and to solve problems. This includes further clarifying the goal of PEF/OEF (PEFCR/OEFSR), testing normalization factors, reconsidering whether weighting is indeed an appropriate way for communicating LCA/PEF results, and adapting some of the predefined impact assessment methods (Finkbeiner 2014). The results of this paper are relevant for the pilot phase, because they highlight theoretical and practical challenges and propose possibilities to cope with them.

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Article

Resource Efficiency Assessment—Comparing a Plug-In Hybrid with a Conventional Combustion Engine

Martin Henßler [1,*], Vanessa Bach [2,*], Markus Berger [2], Matthias Finkbeiner [2] and Klaus Ruhland [1]

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1 Group Environmental Protection, Daimler AG, Stuttgart 70546, Germany; klaus.ruhland@daimler.com
2 Chair of Sustainable Engineering, Technische Universität Berlin, Straße des 17. Juni 135, Berlin 10623, Germany; markus.berger@tu-berlin.de (M.B.); matthias.finkbeiner@tu-berlin.de (M.F.)
* Correspondence: martin.henssler@daimler.com (M.H.), vanessa.bach@tu-berlin.de (V.B.); Tel.:+49-30-314-27941 (V.B.)

Abstract: The strong economic growth in recent years has led to an intensive use of natural resources, which causes environmental stress as well as restrictions on the availability of resources. Therefore, a more efficient use of resources is necessary as an important contribution to sustainable development. The ESSENZ method presented in this article comprehensively assesses a product's resource efficiency by going beyond existing approaches and considering the pollution of the environment as well as the physical and socio-economic availability of resources. This paper contains a short description of the ESSENZ methodology as well as a case study of the Mercedes-Benz C-Class (W 205)—comparing the conventional C 250 (petrol engine) with the C 350 e Plug-In Hybrid (electric motor and petrol engine). By applying the ESSENZ method it can be shown that the use of more and different materials for the Plug-In-Hybrid influences the dimensions physical and socio-economic availability significantly. However, for environmental impacts, especially climate change and summer smog, clear advantages of the C 350 e occur due to lower demand of fossil energy carriers. As shown within the case study, the when applying the ESSENZ method a comprehensive evaluation of the used materials and fossil energy carriers can be achieved.

Keywords: resource efficiency; life cycle assessment; physical availability; socio-economic availability; environmental impact

1. Introduction

The demand of abiotic resources like metals, or fossil fuels has increased significantly in recent decades due to the global industrial and technological development. Additionally, the pollution of natural resources like water has risen as well. As these patterns of resource consumption will lead to an exceedance of the sustainable capacity of ecosystems worldwide, enhancing resource efficiency is a key goal of national and international strategies (e.g., [1,2]).

There is no commonly accepted definition of the term "resources" yet. Often it is defined as materials and energy as well as knowledge, services, or staff. With regard to the assessment of efficiency of a resource the term is mostly used evaluating the use of minerals, metals, and fossil energy carriers [2–5]. The Strategy on the sustainable use of natural resources [6] includes the environment in the definition as well, leading to a more comprehensive view of resources.

Resource efficiency is mostly regarded as a macro economic strategy because resources are key components of every society to sustain production and the wellbeing of current and future generations [5,7]. However, often resource efficiency is implemented on a micro economic level e.g., by reducing material inputs. Thus, measuring resource efficiency in a methodologically correct yet applicable way on a product level enables companies to address scarcity of resources and decrease environmental impacts on a corporate level.

Evaluating the resource efficiency of products can be established by the ESSENZ method (Integrated method to assess/measure resource efficiency) [8–11].

In the following sections, the applied ESSENZ method is introduced as well as the subjects of the case study. Results are shown for the individual dimensions as well as for the summarized results.

2. Method

In cooperation with TU Berlin (Chair of Sustainable Engineering), Daimler AG, Deutsches Kupferinstitut Berufsverband e. V., Evonik Industries AG, Siemens AG, ThyssenKrupp Steel Europe AG, and Wissenschaftlicher Gerätebau Dr.-Ing. Herbert Knauer GmbH a comprehensive method has been developed to measure resource efficiency of products. Overall, 18 categories and corresponding indicators were established to enable a holistic assessment of resource efficiency in the context of sustainable development considering the three dimensions "physical availability", "socio-economic availability", and "environmental impacts" (Figure 1) [8–11]. Within the ESSENZ approach next to the environment (including all environmental compartments e.g., water, air, soil), raw materials are considered as a resource as well. Even though the developed approach can be applied to all resources in theory, practical experience has so far been limited to metals, fossil energy carriers, as well as parts of the environment.

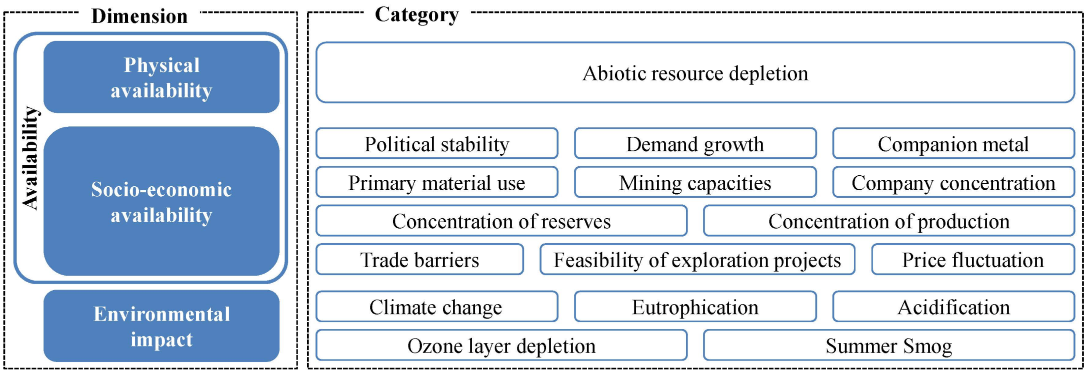

Figure 1. Dimensions and categories to assess resource efficiency within the ESSENZ method.

Existing geological deposits (physical availability) as well as socio-economic factors (socio-economic availability) might be restricting the availability of resources and thus influencing their supply security. The physical availability is evaluated by means of the abiotic depletion potential (ADP) indicator (baseline approach—ultimate reserves), which is subdivided to assess resource depletion of raw materials ($ADP_{elemental}$) and resource depletion of fossil fuels (ADP_{fossil}) [12]. To evaluate the socio-economic availability of resources, economic constraints leading to supply shortages along the product's value chain are quantified [13]. Possible constraints include, for example, the political stability of countries, which can be impaired due to factors such as corruption within the government, disrupting the capacity to effectively implement robust policies [14]. Based on existing work [15–17] 11 categories with corresponding category indicators are identified (Table 1). Characterization factors for all categories are determined based on the ecological scarcity approach [18,19]. Indicator values are compared with regard to a category specific target. These targets have been determined based on a stakeholder survey and expert interviews. Applying normalization and scaling (up to 1.5×10^{19} representing the overall production of all considered materials within the year 2013) the calculation of the final characterization factors is concluded. They are provided for a portfolio of 36 metals and four fossil energy carriers [8–11]. So far, the physical and socio-economic availability of a product can only be determined for the BoM of the considered product. Due to the fact that current LCA databases use economic allocation to assign metal contents from mixed ores to metal datasets, the mass of metals in the datasets does not reflect the physically present metal content but rather represents an over- or underestimation depending on the economic value of the considered metals.

Table 1. Socio-economic categories and related category indicators.

Category	Description	Category Indicator
Political stability	Governance stability of producing countries	World Governance Indicators [14]
Demand growth	Increase of demand over the last five years	Percentage of annual growth based on past developments (based on data from British Geological Service [20])
Companion metal	Companion metals within host metal ore bodies	Percentage of production as companion metal [21]
Primary material use	Recycling content of a material	Percentage of new material content [22]
Mining capacities	Overall mining time of a material considering current production	Reserve-to-annual-production ratio (based on data from United States Geological Service [23] and British Geological Service [20])
Company concentration	Company concentration based on producing companies	HHI (1)—index is calculated by squaring the market share of each company or country with regard to the production or reserves [24]
Concentration of reserves	Reserve concentration of certain materials based on reserves in countries	
Concentration of production	Concentration of mine production based on production in countries	
Trade barriers	Materials underlying trade barriers	Enabling Trade Index [25]
Feasibility of exploration projects	Political and societal factors influencing opening of mines	Policy Potential Index [26]
Price fluctuation	Unexpected price fluctuations	Volatility [27]

Note: (1) HHI: Herfindahl-Hirschman-Index.

For determining impacts of resource use on the environment five different indicators are applied using the CML-IA impact assessment method (CML 2001—Version: April 2013, baseline approach) [28] for the subjects' climate change, acidification, eutrophication, ozone layer depletion, and summer smog. The considered environmental impact categories as well as methods are chosen based on their applicability and maturity [29]. Thus, biodiversity and land use are not included as no adequate and applicable methods exist so far [30].

For an overall result regarding the resource efficiency of a product system the considered categories are analyzed together to achieve a comprehensive evaluation enabling meaningful decision making [8–11].

3. Case Study of Mercedes-Benz C-Class

In the following section, the resource efficiency assessment according to the ESSENZ method is presented for the example of the C-Class (W 205). The study compares the conventional C 250 (petrol engine) with the C 350 e Plug-In Hybrid (electric motor and petrol engine). First, in Section 3.1

technical data and a detailed analysis of materials for the considered passenger cars are provided. Results of the resource efficiency assessment with the ESSENZ method are shown in Section 3.2.

3.1. Product Documentation of the C 250 and C 350 e

This section documents significant specifications of the different variants of the C-Class analyzed in this study. Section 3.1.1. provides an overview of the technical data of the C 250 and C 350 e. The material composition is discussed in Section 3.1.2.

3.1.1. Technical Data

The Plug-In Hybrid model in the current C-Class, the C 350 e, combines a 60 kW electric motor and an externally rechargeable battery with a four-cylinder petrol engine with 155 kW (Table 2). The high voltage lithium-ion battery of the C 350 e has an energy content of 6.38 kWh. With the aid of the synchronous electric motor, the C 350 e has an all-electric range of 31 km. The certified combined consumption according to the New European Driving Circle (NEDC) of the C 350 e is 2.1 l and 11.0 kWh per 100 kilometer (ECE-R101). This corresponds to CO_2-emissions of 48 g/km.

Table 2. Technical data of C 250 and C 350 e [31].

Technical Data	C 250	C 350 e
Weight (kg)	1435 [1]	1705
Output (kW)	155	155 + 60 (electric motor)
Fuel consumption NEDC [2] combined (l/100 km)	5.3	2.1
Electric energy consumption NEDC [2] combined (kWh/100 km)	-	11.0
Electric range (km)	-	31
Driving share petrol engine (%)	100	45 [3]
Driving share electric motor (%)	-	55 [3]
CO_2 (g/km)	123	48

Notes: [1] Comparably equipped as C 350 e; [2] NEDC: New European Driving Circle; [3] Determination of electric driving share according to type approval directive ECE-R101; percentages related to driving distance.

The C 250 is powered by the 155 kW four-cylinder petrol engine (Table 2). The fuel consumption is 5.3 l/100 km (NEDC). This causes CO_2-emissions of 123 g/km which are more than twice as high as the CO_2-emissions of the C 350 e.

The C 250 and the C 350 e can be assumed to be functionally equivalent as they have similar driving performance as well as safety and comfort features (Table 2). The use phase is calculated on the basis of a mileage of 200,000 km. The key components of both vehicles (incl. battery) do not require replacement over the life cycle.

3.1.2. Material Composition

The weight and material data for the C 250 and C 350 e are determined on the basis of internal documentation of the components used in the vehicle (parts list, drawings). The kerb weight according to DIN 70020 (without driver and luggage, fuel tank 90 percent full) serves as a basis for the life cycle assessment (LCA). Figure 2 shows the material composition of the C 250 and C 350 e.

The weight of the C 250 is 1435 kg. The weight of the C 350 e is 1705 kg and thus 270 kg heavier than the C 250. Steel/ferrous materials account for slightly less than half the vehicle weight (approximately 47 percent) in both cars. The next largest shares are light alloys at 21 percent and polymer materials at 19 (C 250) and 21 percent (C 350 e). Service fluids and other metals comprise around 5 (C 350 e) and 6 percent (C 250); and 2 (C 250) and 5 percent (C 350 e), respectively. The proportions of other materials are somewhat lower, at about 4 percent.

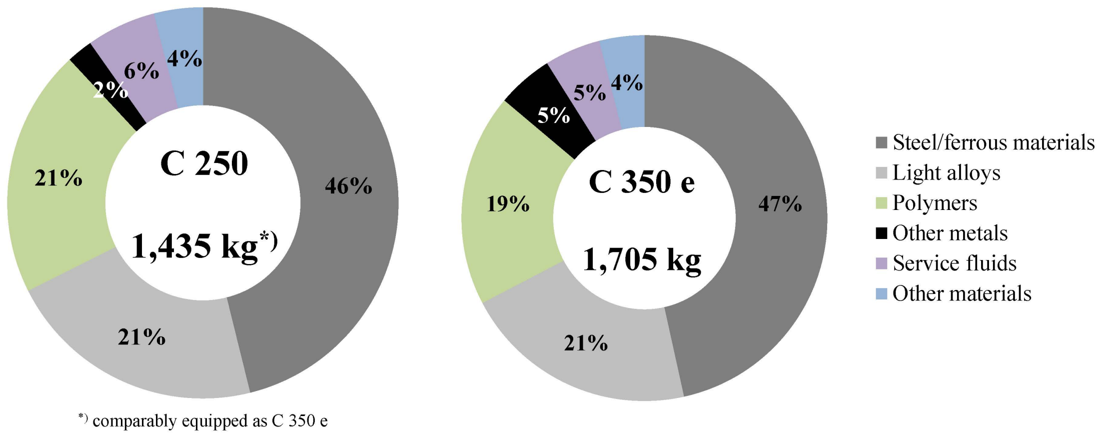

Figure 2. Material composition of C 250 and C 350 e [31].

Figure 3 shows the main differences in weight and material mix in the modules exterior, interior, chassis, powertrain, and electric comparing the C 350 e with the C 250. The biggest difference can be found regarding electric constituents. Due to the hybrid components, especially the high voltage battery, the power electronics, and the cabling, the additional weight is about 140 kg. In the powertrain, an extra weight of about 76 kg is derived primarily from the electric motor. Larger breaks and tires as well as air suspension cause an extra weight of the chassis of about 66 kg. The weight of the exterior is about 25 kg higher due to the high voltage crash package of the battery. The alternative drive components and the related mix of materials—especially the material group of other metals—used in the C 350 e change the weight substantially compared to the conventional C-Class.

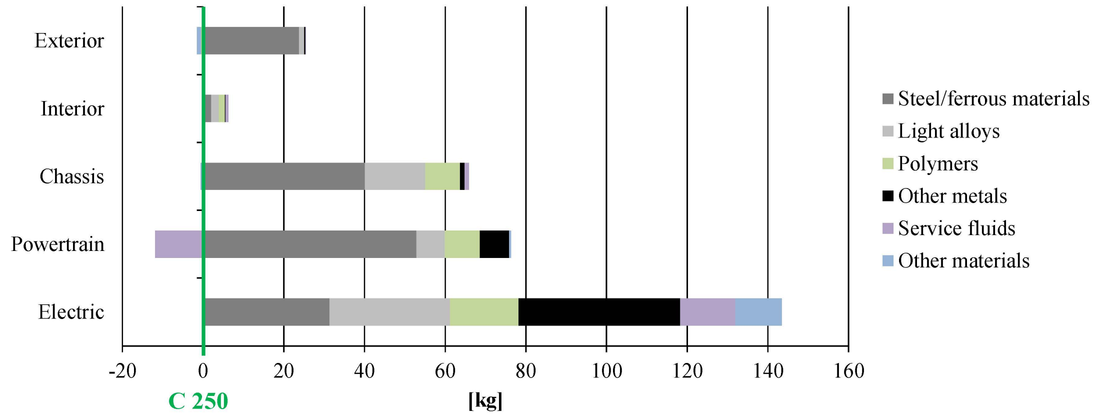

Figure 3. Main weight differences—C 350 e compared to the C 250.

3.2. *Assessment of Different Resource Efficiency Dimensions Considered in the ESSENZ Method*

The ESSENZ method described in Section 2 was applied, leading to the following results regarding physical availability (Section 3.2.1.), socio-economic availability (Section 3.2.2.), and environmental impacts (Section 3.2.3.). The section concludes by summarizing the results of all considered dimensions of the C 250 compared to the C 350 e (Section 3.2.4.).

For the C 350 e, two energy consumption scenarios for the use phase are considered. In addition to the EU electricity grid mix, electricity from hydro power is accounted for. The results of the use phase (electricity generation, fuel production, and operation) are based on the certified NEDC electricity/fuel consumption and the certified specific emissions of each car via a mileage of 200,000 km. The study includes environmental impacts of the recovery phase on the basis of the standard processes of drainage, shredding, and recovery of energy from the shredder light fraction. Environmental credits are not considered.

3.2.1. Physical Availability

Figure 4 shows the results for the category abiotic resource depletion (quantified by $ADP_{elemental}$ in kg Sb_{eq} and ADP_{fossil} in GJ). For the calculation of the $ADP_{elemental}$ only the BoM of the vehicle is taken into account since no consistent background data for the whole supply chain are available andno materials are required in the use phase. Assessing resource depletion of fossil energy carriers (ADP_{fossil}) the whole life cycle (car production, fuel production, operation, electricity generation, and end of life) of the vehicle is included. The additional hybrid-specific components (Section 3.1.2.) lead to a higher resource depletion potential of metals for the C 350 e, which is about 170 percent higher (0.63 kg Sb_{eq}) compared to the C 250 (0.23 kg Sb_{eq}).

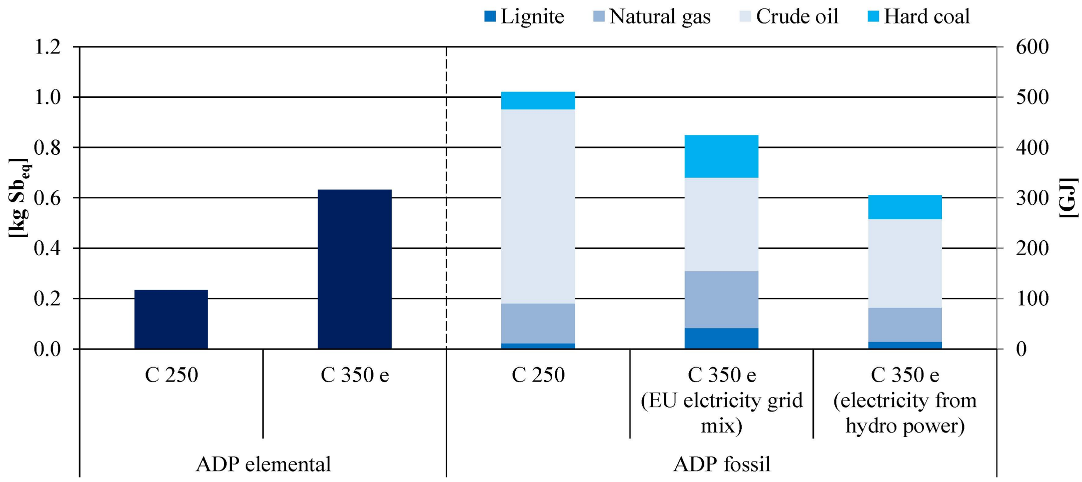

Figure 4. $ADP_{elemental}$ and ADP_{fossil}—C 250 compared to the C 350 e.

The abiotic depletion potential (ADP_{fossil}) of the C 350 e with electricity from hydro power is 40 percent lower (205.6 GJ) than of the C 250 (511 GJ—comprised of 385.6 GJ crude oil, 78.8 GJ natural gas, 34.6 GJ hard coal, and 12.7 GJ lignite). Due to additional hybrid-specific components in the car production and the generation of electricity during the operation phase, the consumption of natural gas, hard coal, and lignite rises for the C 350 e using EU electricity grid mix to 112.9 GJ (natural gas), 84.1 GJ (hard coal), and 41.8 GJ (lignite). Crude oil consumption can be reduced by over 50 percent to 185.6 GJ due to the high efficiency of the Plug-In Hybrid. When the vehicle is charged with renewably generated electricity, the consumption of lignite, natural gas, crude oil, and hard coal can be reduced further.

Evaluating the abiotic resource depletion enables a comprehensive assessment of the physical availability of metals and fossil energy carriers. Advantages in the production of conventional vehicle concepts compared to alternative vehicle concepts due to the reduced consumption of materials could be shown. Furthermore, alternative engines show benefits in the use phase (electricity generation, fuel production, and operation) due to lower fuel consumption.

3.2.2. Socio-Economic Availability

The dimension "socio-economic availability" is quantified by 11 categories (Section 2). Regarding the socio-economic availability of metals the BoMs of the respective cars are taken into account. As most fossil energy carriers are consumed in the use phase, the whole life cycle is considered when analyzing their socio-economic availability. The results of the comparison of C 250 and C 350 e are shown in Figure 5 (C 250 is scaled to 100 percent).

Overall, for the calculation of the socio-economic availability, 33 metals (Section 2) are taken into account in addition to the fossil energy carriers' lignite, natural gas, crude oil, and hard coal. As shown in Figure 5, the fossil energy carriers have little influence on the 11 categories. Exceptions are "Primary material use" (19 to 39 percent), "Price fluctuation" (12 to 28 percent), and "Company concentration" (2 to 5 percent). Due to the higher material consumption of the C 350 e the C 250 performs far better in all categories except for "Mining capacities". The source of electricity used to charge the C 350 e has no impact regarding the socio-economic availability. The C 350 e using EU electricity grid mix is almost on par with C 350 e using electricity from hydro power—except differences in the categories "Primary material use" and "Price fluctuation" due to higher amounts of used fossil energy carriers (see Section 3.2.1.).

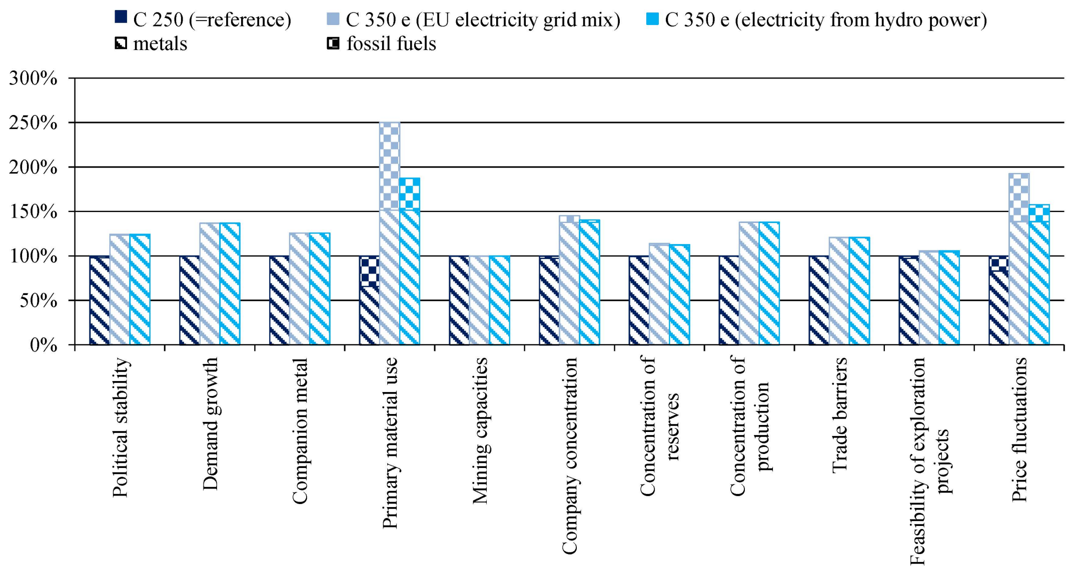

Figure 5. Assessment of the socio-economic availability—C 250 compared to the C 350 e.

The greatest differences between the C 250 and the C 350 e (EU electricity grid mix) occur in the categories "Primary material use" (+ 150 percent), "Price fluctuation" (+ 93 percent), "Company concentration" (+ 45 percent), "Concentration of production" (38 percent), and "Demand growth" (+ 37 percent). Regarding the categories "Demand growth" and "Primary material use", the differences are caused by the use of lithium, which is an essential part of the high voltage battery of the Plug-In Hybrid. The categories "Company concentration", "Concentration of production", and "Price fluctuation" are primarily affected by rare earth elements, which are mainly required for the magnets of the electric motor.

For the categories "Political stability", "Companion metal", "Mining capacities", "Concentration of reserves", "Trade barriers", and "Feasibility of exploration projects" the C 350 e performs up to 26 percent worse than the C 250. These categories are mainly affected by platinum and palladium (exhaust catalyst), magnesium (alloy material), lithium (high voltage battery), rare earth elements (electric motor), and tantalum (condensers).

Overall, it can be concluded that the socio-economic categories in this case study are particularly affected by platinum, palladium, magnesium, lithium, rare earth and tantalum. Except for magnesium, all materials occur in the car in very small amounts.

3.2.3. Environmental Impacts

Figure 6 shows the results of the considered environmental categories climate change (CO_{2eq}-emissions), eutrophication (phosphate$_{eq}$-emissions), acidification (SO_{2eq}-emissions), and summer smog (ethene$_{eq}$-emissions) over the individual life cycle phases (car production, fuel production, operation, electricity generation, and end of life) of the C 250 and C 350 e.

The production of the C 350 e entails visibly higher CO_{2eq}-emissions on account of the additional hybrid-specific components. The CO_{2eq}-emissions in the production phase (11.3 t CO_{2eq}) are 32 percent higher than those of the C 250 (8.6 t CO_{2eq}). Over the entire life cycle the Plug-In Hybrid has clear advantages as external charging with the EU electricity grid mix can cut overall CO_{2eq}-emissions by about 13 percent (4.9 t CO_{2eq}) compared to the C 250. A reduction of 39 percent (15.3 t CO_{2eq}) is possible through the use of renewably generated electricity from hydro power.

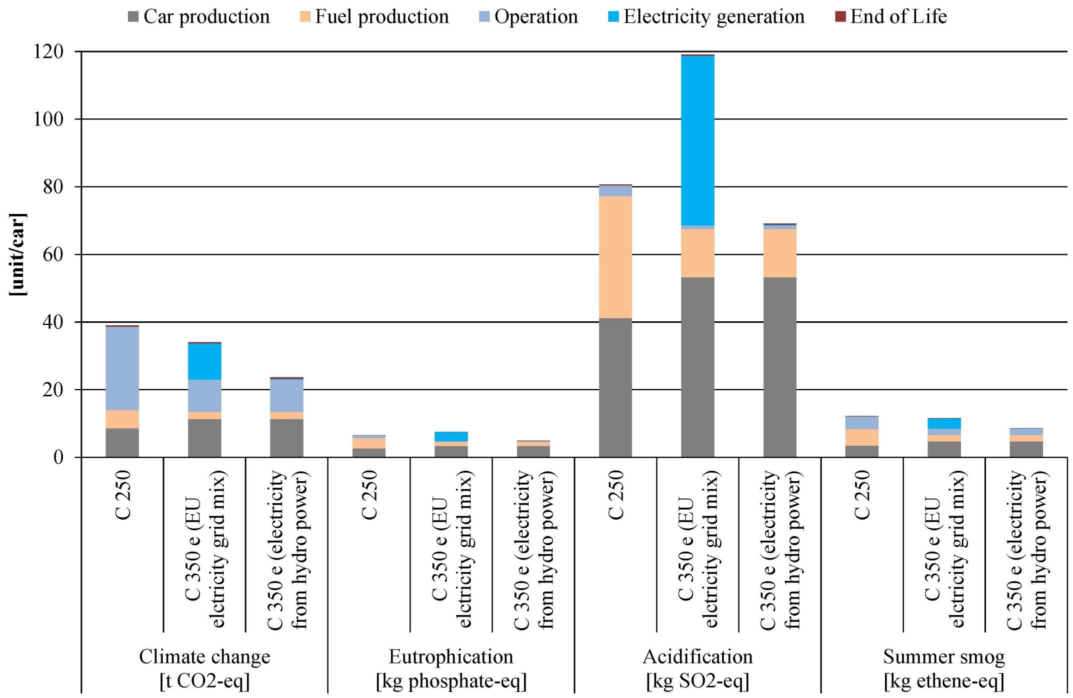

Figure 6. Selected environmental categories—C 250 compared to the C 350 e (unit/car) [24].

Considering the category eutrophication, the C 350 e using electricity from hydro power causes the lowest emissions with 4.9 kg phosphate$_{eq}$. Charging the C 350 e with EU electricity grid mix the phosphate$_{eq}$-emissions add up to 7.5 kg phosphate$_{eq}$, thus being respectively 15 and 53 percent higher than the phosphate$_{eq}$-emissions of the C 250 (6.5 kg phosphate$_{eq}$) or the C 350 e with electricity from hydro power.

In the production phase of the C 350 e (53.2 kg SO_{2eq}) the SO_{2eq}-emissions are 29 percent higher than those of the C 250 (41.2 kg SO_{2eq}). In the use phase (electricity generation, fuel production, and operation) most SO_{2eq}-emissions (65.5 kg SO_{2eq}) are produced during the charging of the vehicle with EU electricity grid mix. Thus, 40 and 76 percent more emissions occur than for the C 250 or the C 350 e with electricity from hydro power respectivly. Over the entire life cycle, the C 350 e with electricity from hydro power saves 42 percent (50.0 kg SO_{2eq}) in comparison to the C 350 e with EU electricity grid mix and 14 percent (11.6 kg SO_{2eq}) compared to the C 250.

The summer smog emissions during production of the C 350 e (4.7 kg ethene$_{eq}$) are 36 percent higher than those of the C 250 (3.4 kg ethene$_{eq}$). Regarding the use phase (electricity generation, fuel production and operation) summer smog emissions can be reduced by 22 percent charging the C 350 e with EU electricity grid mix respectively by 55 percent using electricity from hydro power compared to the C 250. The highest summer smog emissions with 12.1 kg ethene$_{eq}$ are caused by the C 250.

Compared to the C 350 e charged with EU electricity grid mix as well as renewable generated electricity summer smog can be reduced by 5 or 29 percent, translating in a reduction of 11.5 or 8.6 kg ethene$_{eq}$.

In conclusion, it can be stated that over the entire life cycle the C 350 e using electricity from hydro power has clear benefits in all considered categories (shown in Figure 6) compared to the C 250. If the EU electricity grid mix is used for charging advantages with respect to climate change and summer smog occur. However, with regard to eutrophication and acidification, the C 350 e respectively has 15 percent (1.0 kg ethene$_{eq}$) and 48 percent (38.4 kg SO_{2eq}) more impacts than the C 250.

3.2.4. Summary of the Results

Figure 7 shows the summary of the three dimensions considered in this case study. The reference C 250 is scaled to 100 percent.

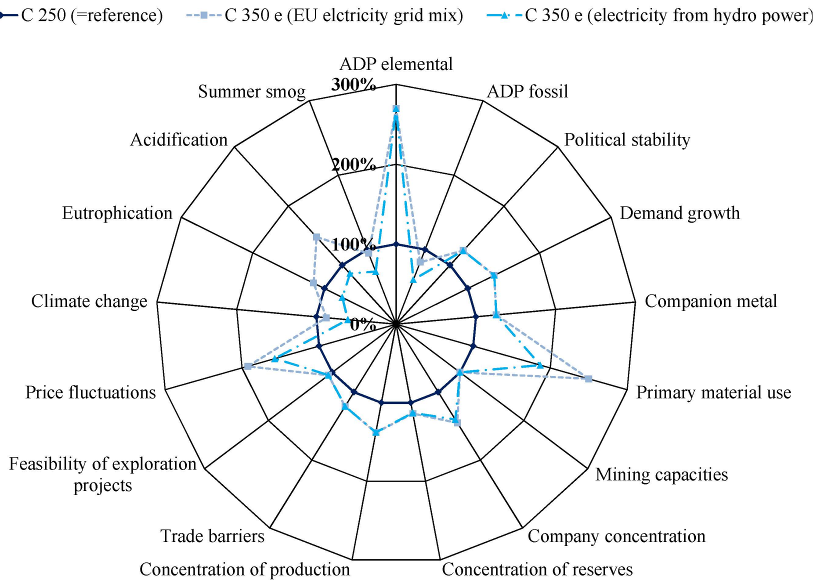

Figure 7. Summary of resource efficiency dimensions of the ESSENZ method—C 250 compared to the C 350 e.

It can be seen that the C 250 performs better within the category abiotic resource depletion of metals (ADP$_{elemental}$) compared to the C 350 e due to its lower overall use of metals. This is also reflected in the dimension "socio economic availability". As the C 250 has a higher fossil energy carrier consumption in the use phase, it performs worse in the category resource depletion of fossil energy carriers (ADP$_{fossil}$). Both C 350 e's have advantages in the use phase (electricity generation, fuel production and operation) due to their lower fuel consumption.

Only minor differences for the dimension "socio-economic availability" as well as the category resource depletion of raw materials (ADP$_{elemental}$) occur when comparing the C 350 e charged with electricity from hydro power with the C 350 e charged with EU electricity grid mix. The metals used for car production influence the categories more significantly than the energy consumption in the use phase. For the category resource depletion of fossil energy carriers (ADP$_{fossil}$) the C 350 e (electricity from hydro power) performs only slightly better.

The results of the environmental impacts measured over the entire life cycle show clear advantages for the C 350 e especially for the categories climate change and summer smog, regardless of the kind of electricity used for external charging of the Plug-In Hybrid. The results for the categories eutrophication and acidification, however, depend on the electricity used for charging the vehicle. Using EU electricity grid mix eutrophying or acidifying emissions of C 350 e are higher than those of C 250. Using electricity from hydro power, the eutrophication potential or acidification potential are lower compared to C 250.

4. Conclusions

The ESSENZ method allows a transparent evaluation of product systems with regard to the physical and socio-economic availability of fossil energy carriers and metals as well as related environmental impacts over the life cycle.

The case study presented in this article—comparing a plug-in hybrid with a conventional engine—is a good example of why such a comprehensive assessment is necessary. The higher use of material resources for the Plug-In Hybrid vehicle has a strong influence on socio-economic and physical availability, whereas some of the environmental impact categories (e.g., climate change and summer smog) show clear advantages for the C 350 e due to the lower demand of fossil energy carriers. As a consequence, it is necessary for a scientifically robust and sustainability oriented resource assessment to consider both materials and energy resources as well as the whole lifecycle of the product.

The comprehensive ESSENZ method enables the user to transparently evaluate various views on the multitude of parameters applicable. It further empowers companies to take appropriate actions regarding the specific materials used in their products, e.g., material specific sourcing strategies or development of recycling technologies.

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Author Contributions: Martin Henßler was the lead in carrying out the case study. Vanessa Bach was leading in the development of the ESSENZ method. Markus Berger and Matthias Finkbeiner both provided substantial contributions to the design of the ESSENZ method as well as valuable comments to deepen the conclusion. Klaus Ruhland gave substantial input to the design of the case study as well as valuable comments to draw the conclusion. All authors proofread and approved the final manuscript.

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Measuring a product's resource efficiency – a case study of smartphones

Vanessa Bach[1]*, Markus Berger[1], Tobias Helbig[2], Matthias Finkbeiner[1]

[1]*Technische Universität Berlin, Chair of Sustainable Engineering, Office Z1, Strasse des 17. Juni 135, 10623 Berlin, Germany, www.see.tu-berlin.de/*

[2] *Malmö City Planning Office, Planning department, August Palms plats 1, 205 80 Malmö, Sweden, www.malmo.se/stadsbyggnadskontoret*

*vanessa.bach@tu-berlin.de

Abstract

Natural resources, such as metals, or clean airserve as vital inputs for a stable economy and society. In recent decades, the demand for many resources has increased significantly.Sales for smartphones were summed up to 1.8 billion units in 2013.To measure resource efficiency of products a comprehensive methodology has been developedin cooperation with the European industry. Overall five indicators are applied measuring impacts on the natural resources water, air, and soil.Eleven indicators are used to determine potential risks ofthe availability of raw materials. Two indicators are included to estimate potential decreasing acceptance by consumers for the product under assessment regarding the lack of social and environmental standards. In order to illustrate the implementation of this approach, a case study for smartphones is presented. For smartphones gold is the most geological scarce metal as its worldwide availability is low. However, considering socio-economic aspects other metals e.g. rare earthsare of more concern.Regarding the indicator reflecting possible decreasing acceptance by the consumers due to missing social standards, cobalt, predominantly mined in Congo, is a hotspot for smartphones.The proposed method supports a more holistic assessment of resourcesas supply risks and impactson the environment can be identified.

Keywords: resource efficiency, mobile phones, sustainable development

Purpose

In recent decades, the demand for many resources has increased significantly due to their growing importance for industrial and technological development. This includes the worldwide increasing demand for mobile phones - sales summed up to 1.8 billion units in 2013. As neither the availability of natural resources as input for production processes nor the capacity of natural resources to absorb pollution are endless, the sustainable use of resources by increasing resource efficiency is the commonly mentioned strategy for sustainable development(European Commission 2011; Klinglmair et al. 2014). To adequately assess the efficiency of resource use all three sustainability dimensions have to be considered. Furthermore, companies need applicable methods to use in their daily operations, which so far are lacking (Schneider et al. 2015a).

Methods

To measure resource efficiency (RE) of products a comprehensive methodology (*ESSENZ - Integrated method to assess/measure resource efficiency*) has been developed in cooperation with the European industry. The approach is based on Schneider et al. (2014)and aims at considering all three sustainability dimensions.This includes ensuring security of resource supply (economic), decreasing environmental impacts due to resource extraction and use (environment) and guaranteeing compliance with social and environmental standards (social dimension)(see Figure 1). However, the social dimension cannot be reflectedcomprehensively due to general challenges (e.g. missing of inventory data as well as sufficient impact assessment methods) of social life cycle assessment (SLCA)(Lehmann et al. 2013; Neugebauer et al. 2014).

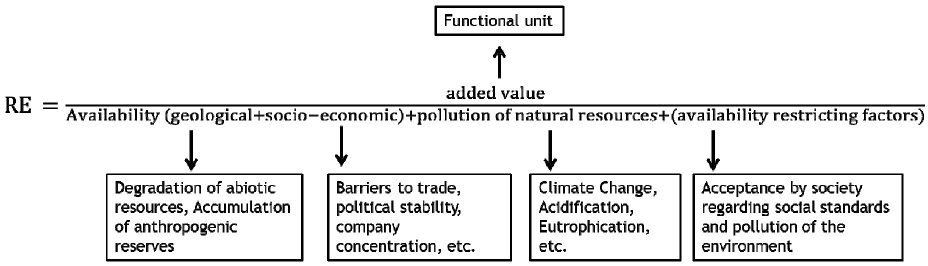

Figure 6: ESSENZ approach to measure resource efficiency

Overall 19 indicators are used within the method to measure resource efficiency. To assess the degradation of abiotic resources and in addition to capture anthropogenic resource stocks the anthropogenic stock extended abiotic depletion potential (AADP) is used(Schneider et al. 2015b). For case studies where values for AADP are missing the Abiotic Depletion Potential (ADP) (Oers et al. 2002)can be used instead. However, the user has to be aware that ADP provides different information than AADP – thus the interpretation has to be adapted accordingly.

The (socio-) economic dimension is reflected with eleven indicators (see Table 1):

Aspect	Description	Indicator
Static lifetime	Depletion time with current production technologies	Reserve-to-annual-production ratio
Company concentration mine production	Concentration of mine production in certain companies	HHI—index is calculated by squaring the market share of each company or country with regard to the production or reserves (Rhoades 1993)
Country concentration reserves	Reserve concentration in certain countries	
Country concentration mine production	Concentration of mine production in certain countries	
Feasibility of raw material mining	Feasibility of mining projects due to legislative framework conditions	Policy Potential Index (Cervantes et al. 2013)
Governance stability of mine production countries	Stability of governance in producing countries	World Governance Indicators (The World Bank Group 2013)

Aspect	Description	Indicator
Demand growth	Increase of demand (past)	Percentage of annual growth based on past developments
Companion metal fraction	Appearance as companion metal withinhost metal ore bodies	Percentage of production as companion metal (Erdmann et al. 2011)
Recycling	Recycled content of a resource	Percentage of recycled content (Fischer-Kowalski et al. 2011)
Barriers to trade	Raw materials underlying trade barriers	Enabling Trade Index (Hanouz et al. 2014)
Price variation	Volatility of raw materials	Volatility (Bundesanstalt für Geowissenschaften und Rohstoffe 2014)

Table 1: Socio-economic aspects and indicators used within ESSENZ

Environmental damages are evaluated with the CML life cycle impact assessment methods (Guinée et al. 2002)for the subjectsclimate change, acidification, eutrophication, ozone depletion and smog. Even though the categories land use and biodiversity are of upmost important for resources they have to be excluded in the approach due to lack of sufficiently applicable methods (Finkbeiner et al. 2014).Furthermore, two indicators expressing customer acceptance are included in order to consider availability restrictions which result from customer intolerance to the violation of certain standards. Both, the non-compliance of social standards e.g. child labor as well as the pollution of the environment e.g. eco toxicity due to mining operations can lead to a boycott from consumers.

Due to the existing challenges of implementing SLCA into practice, the social dimension is only reflected by this one indicator. Thus, a comprehensive assessment of social aspects over the life cycle is missing within the presented approach. However, the proposed indicator should be seen as a defaultvalue and users are encouraged to determine additional social aspects regarding the supply chain of their product under assessment.

In order to illustrate the implementation of this approach, a case study for smartphones is presented. Inventory data were obtained based on Buchert et al. (2012) and Müller (2013).

Results

As the approach is still under development and data for determining the presented indicators have to be collected, only a glimpse of the results can be presented here. A comprehensive evaluation of smartphones with the developed ESSENZ method will be presented during the CILCA 2015, Lima, Peru.

Resource efficiency of materials differs when economic and environmental aspects are taken into account in addition to a geologic assessment. For smartphones gold is the most geological scarce metal as its worldwide availability is low. However, considering socio-economic indicatorsplatinum and rare earth metals are of most importance for several of the introduced aspects e.g. feasibility of raw material mining. Environmental damages are mostly caused by gold installed in the smartphone, except for the potential ozone layer depletion, where cobalt is the hotspot. Regarding customeracceptance the social indicator is dominated by cobalt, predominantly mined in Congo.The environmentalacceptance of the product could be decreasedby rare earth and gold which both have high toxicity impacts.

Conclusions

The proposed method supports a more realistic assessment of resource efficiency as supply risks and environmental impacts associated with resources can be identified. However, as the developed method is relatively new it needs to be applied in more case studies to validate its robustness. For the socio-economic and social aspects calculated indicator results for metals and fossil energy carriers are provided in form of an excel table.

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Integrated method to assess resource use in the context of sustainable development (ESSENZ +)

(Vanessa Bach, Martin Henßler, Markus Berger, Klaus Ruhland, Laura Schneider and Matthias Finkbeiner)

Evaluating the use of resources in line with the requirements of sustainable development has become more and more important in the last decades due to the increasing relevance and decreasing availability of resources - especially of metals and fossil raw materials - for industrial and technological development. Thus, the efficient use of resources as part of a circular economy is a commonly mentioned strategy to sustainably manage the earth's limited resources.

The introduced approach (ESSENZ +) was developed in cooperation with the six European companies Daimler AG, Evonik, Knauer, ThyssenKrupp, German Copper Institute (Deutsches Kupferinstitut) and Siemens. It addresses the sustainable use of resource within four dimensions: physical and socio-economic availability as well as social and environmental implications.

Within the category physical availability limitations to the availability of abiotic resources are evaluated due to abiotic resource depletion and materials lacking accumulation in the anthroposphere. Furthermore, socio-economic aspects constraining the access to abiotic resources due to structural conditions of the market as well as disrupting structures within society are take into account. Overall 10 categories with related characterization factors (which are based on the Distance-to-Target approach) are considered for the quantification of the following socio-economic constraints: concentration of reserves, concentration of production, company concentration, political instability, demand growth, trade barriers, price fluctuations, feasibility of exploration projects, mining capacity and recycling.

The ESSENZ + method extends the original ESSENZ method by providing more comprehensive approaches to determine social and environmental implications of resource use. To quantify social implication three indexes (small-scale mining which often correlates with forced labor and violence, human rights violations and geopolitical risk e. g. due to armed conflicts) are established.

To assess environmental impacts two approaches are recommended. First, potential impacts of the considered product are assessed by carrying out an LCA. Second, a screening indicator is established to determine the state of the ecosystem where the resource is mined. Therefore, the regional biodiversity based on the Terrestrial Ecoregions Indicators by World Wildlife Fund as well as the capability of a country to safeguard conservation areas based on the Environmental Performance Index are taken into account.

The applicability of ESSENZ+ is demonstrated in a case study of a Plug-In Hybrid compared to a conventional engine. While the Plug-In Hybrid shows lower impacts in most of the environmental categories including climate change, it uses more materials, which have a high physical and socio-economic scarcity. Companies can use these results to analyze potential constraints in their supply chain in more detail and to develop mitigation options such as material specific sourcing strategies.

By providing characterization factors for 36 metals and four fossil raw materials for all four dimensions, ESSENZ+ is easily applicable and thus supports the comprehensive assessment of resource use addressing major challenges related to the secure and sustainable supply of raw materials.

Integrated method to assess resource use in the context of sustainable development (ESSENZ +)

SUSTAINABLE ENGINEERING

Evaluating the use of resources in line with the requirements of sustainable development became more and more important in the last decades due to the increasing relevance and decreasing availability of resources - especially of metals and fossil raw materials - for industrial and technological development. Thus, the efficient and sustainable use of resources as part of a circular economy is a commonly mentioned strategy to sustainably manage the earth's limited resources.

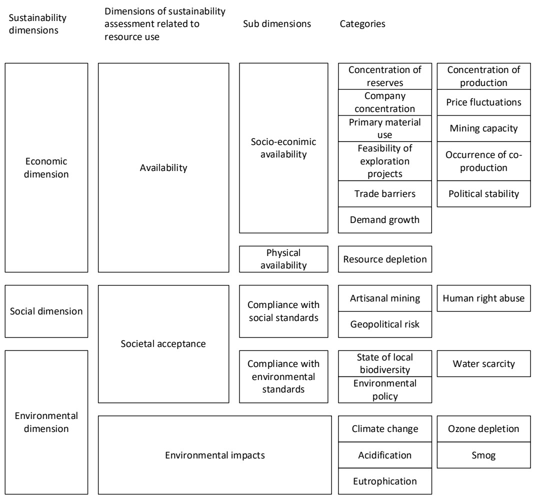
ESSENZ+ methodology

The ESSENZ+ methodology aims at assessing abiotic resources in the context of sustainable development on product level. Overall 23 categories are taken into account in the three dimensions availability, societal acceptance and environmental impacts.

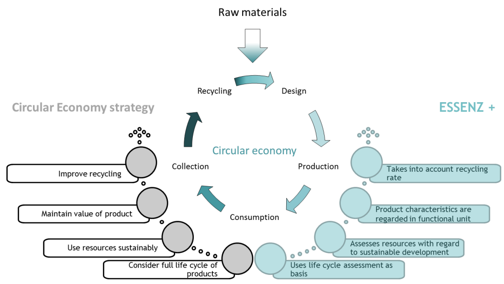
The availability of resources is determined by applying the ADP (baseline approach – crustal content) indicator. For the socio-economic availability the access to resources due to structural conditions of the market as well as disrupting structures within society is taken into account. Characterization factors are established based on the distance-to-target-approach.

Societal acceptance refers to the acceptance of consumers with regard to environmental and social effects arising due to non-compliance with standards (societal norms as well as laws and regulations) over the supply chain. Social implication are identified with regard to small-scale mining, human rights violations and geopolitical risk. The state of the regional biodiversity, water scarcity impacts as well as the capability of a country to safeguard conservation areas are addressed for determining compliance with environmental standards.

Environmental impacts are assess by applying existing LCIA methods.



How ESSENZ+ can support the EU circular economy strategy



The main goal of the European circular economy strategy is to maintain the value of products, materials and resources as long as possible, while minimizing the generation of waste. This goal is expressed in several objectives (e.g. reduce environmental impacts) for which measures are already and will soon be established.

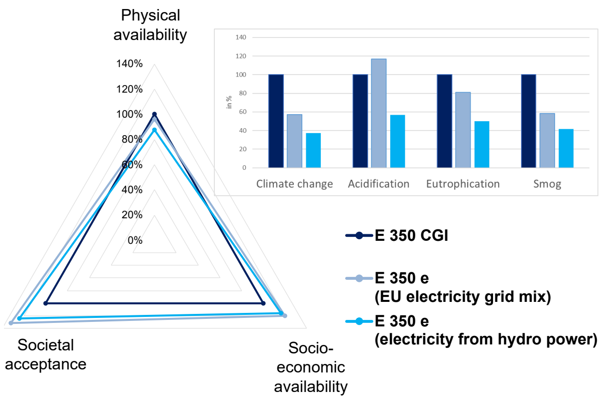
The ESSENZ+ methodology can support this strategy by assessing measures established on product level. Therefore, the use of abiotic resources in the context of sustainable development is evaluated, addressing environmental and social impacts as well as their availability, which constitutes the basis of every economy.

Applying ESSENZ+ in product systems

ESSENZ+ is applied to assess resource use and related impacts of three cars taking into account the entire life cycle:

- E 350 CGI, conventional petrol engine (predecessor)
- E 350 e, plug-in hybrid powered by European electricity grid mix
- E 350 e, plug-in hybrid powered by electricity from hydro power

By assessing resources in line with sustainable development, all important aspects are considered and trade-offs of the systems can be identified. Due to the lower consumption of fossil fuels, the E 350 e demonstrates distinct advantages with regard to environmental aspects (exception is the category acidification, which is higher when using European electricity grid mix) as well as physical availability. The increased demand for resources for the plug-in hybrid shows higher risks with regard to socio-economic availability and societal acceptance.



Conclusions

The ESSENZ+ methodology can be applied to assess product systems in the context of sustainable development as well as in line with the EU circular economy. The results can be used to establish measures and action to enhance the efficient and sustainable use of resources.

To further enhance ESSENZ+ anthropogenic availability of abiotic resources should be taken into account. Further, biotic resources should be included to adequately assess product systems considering different resource types.

Vanessa Bach[1*], Martin Henßler[2], Markus Berger[1], Klaus Ruhland[2], Laura Schneider[1] & Matthias Finkbeiner[1]

[1] Technische Universität Berlin , Chair of Sustainable Enginnering, Phone: +49 (0)30/314-27941 · *vanessa.bach@tu-berlin.de

[2] Daimler AG, Corporate Environmental Protection, RD/RSE, 70546 Stuttgart, HPC G211, Germany