Monetizing LCA Results to Improve Environmental Criticality Assessment of Abiotic Raw Material Use

vorgelegt von

M. Sc.

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an der Fakultät III - Prozesswissenschaften

der Technischen Universität Berlin

zur Erlangung des akademischen Grades

Doktorin der Ingenieurwissenschaften

\- Dr.-Ing. –

genehmigte Dissertation

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Tag der wissenschaftlichen Aussprache: 23. September 2022

Berlin 2022

Acknowledgements

This thesis would not have been possible without the support of many people that motivated and helped me.

I want to thank my supervisor Prof. Matthias Finkbeiner who introduced me to the topic of LCA and inspired me to further explore the topic of monetization in LCA. Matthias, thank you for your availability and your trust during this journey to the PhD.

I am also grateful to my supervisor Prof. Guido Sonnemann for dedicating time and effort to evaluate this thesis.

A special thanks goes to Dr. Vanessa Bach that accompanied me during this entire time. It was her that encouraged me to pursue a PhD while I was still a masters' student. Thank you, Vanessa, for giving me your time, your advice, the ability to grow and for always believing in me as well as your support in letting me learn new things.

I want to thank Dr. Till Bachmann and Dr. Masaharu Motoshita for their contribution and help for the review paper and the expertise they shared with me. I also want to thank my colleague and co-author Marco Muhl who provided the idea for my first criticality paper.

Without my team at the Chair SEE I would not have learned what is more important than intellectual achievement and publishing papers: what it is to work in a team and to always help each other. You are great and really an exception in academia. I was always glad to come to the office, because I knew even in difficult times, I would always find a listener and friendly words, someone to drink a beer with or to have a walk over the Rosa-Luxemburg-Steg together. For the last phase in writing the framework I want to thank Johanna Pohl for her encouragement and good advice and some trips to the Prinzenbad that made this last summer of concluding the PhD much more enjoyable.

My family always fostered critical thinking and education. I am very grateful for your support and your confidence that you have given me during the PhD.

My friends Christina, Zora, Nora, Clem, Nina, Linda, Lucia and Malika were always there for me during this time. Thank you for listening to my complaints and bearing with my unreliability especially in these last months.

A special thanks goes to my trainers and friends at the Muay Thai class at the Schokofabrik in Berlin Kreuzberg. The sport and support kept me going during lock-down times. I would not have been able to motivate myself without you to continue writing this work. Thank you!

Carsten you were the first person that even gave me the idea that academia could be something for me. You were always there for me and endured me even in difficult phases. Without your support and love I would never have been able to write this work that I dedicate to you.

Abstract

Material demand still rises on a global level, which leads to a variety of environmental impacts. This trend is not expected to change in the near future. However, the use of more varied materials is essential for economic development. The environmental impacts of these increasing material applications in quantity and variety do not leave supply risks untouched. On the one hand, the environmental impacts of material use can lead to supply risks, as sourcing countries might not be willing to deteriorate their environment to supply other countries with raw materials. On the other hand, the environmental impacts of material extraction and processing can be in conflict with environmental goals and policies that a country that purchases these materials has. Both of these aspects contribute to environmental criticality, as they render a trade relationship less stable. This thesis aims to improve existing methods to assess the environmental effects of abiotic raw material use that can affect criticality. Therefore, within this thesis a revised criticality assessment was developed that can be applied to a world region (the EU) based on the SCARCE method (previously applied to Germany). Within criticality assessment the weighting and aggregation of indicators is a challenge. Therefore, a new aggregation approach for the environmental dimension of criticality assessment was developed by monetizing the environmental impacts. This modification is building on a review paper that analyzed all currently used monetization methods in Life Cycle Assessment (LCA) on quantitative and qualitative level. Two methods to quantify environmental costs by monetizing LCA results were developed that can be used to measure environmental criticality. One method considered the environmental categories in the method SCARCE. The second method monetized all ReCiPe impact categories that address human health and terrestrial ecosystems. To apply the developed methods, two case studies were conducted to assess environmental costs of i) global material production and ii) the EU's low-carbon development. The damages for the EU's low-carbon development range from €13.1 billion to €74.8 billion, with €38.9 billion as the medium estimate per year. The results showed that especially the material demand of the mobility transition contributes to environmental criticality. For global material production, the environmental costs range from €0.4 trillion/yr (low) to €5 trillion/yr (high). The materials with the largest cost contribution were energy carriers (33%-60%), iron (12%-20%), aluminum (5%-8%) and thus materials that are used for basic functions in the daily life. Both methods show where the environmental impacts lie with country resolution. They can be further used, developed and modified to investigate how material extraction and its environmental impacts affect overall economic activity and human wellbeing, but also how they might affect the stability of a trade relationship. The methods are thus a first step to identify measures that can reduce environmental impacts of raw material extraction and processing with the intention to stabilize trade relationships in the long run.

Keywords: Monetization, LCA, Abiotic raw materials, Resources, Weighting

Zusammenfassung

Die Nachfragen nach Materialien steigt weltweit weiterhin an, was zu einer Vielzahl von Umweltauswirkungen führt. Es ist nicht zu erwarten, dass sich diese Entwicklung in naher Zukunft ändert. Der Einsatz vielfältigerer Materialien ist für die wirtschaftliche Entwicklung unerlässlich. Allerdings beeinflussen die Umweltauswirkungen dieser zunehmenden Materialmenge und Materialvielfalt Versorgungsrisiken. Einerseits können die Umweltauswirkungen der Materialnutzung zu Versorgungsrisiken führen, da Länder, in den die Materialien gewonnen werden, möglicherweise nicht bereit sind, ihre Umwelt zu belasten, um andere Länder mit Rohstoffen zu beliefern. Andererseits können die Umweltauswirkungen der Materialgewinnung und -verarbeitung im Widerspruch zu den Umweltzielen und -richtlinien stehen, die ein Land hat, das diese Materialien erwirbt. Beide Aspekte tragen zur Umweltkritikalität bei, da sie eine Handelsbeziehung weniger stabil machen. Diese Arbeit zielt darauf ab, bestehende Methoden zur ökologischen und ökonomischen Bewertung von abiotischer Rohstoffnutzung, die die Kritikalität beeinflussen können, zu verbessern. Daher wurde im Rahmen dieser Arbeit eine überarbeitete Kritikalitätsbewertung basierend auf der SCARCE-Methode (zuvor angewendet auf Deutschland) entwickelt, die auf eine Weltregion (die EU) angewendet werden kann. Innerhalb der Kritikalitätsbewertung ist die Gewichtung und Aggregation von Indikatoren eine Herausforderung. Daher wurde ein neuer Aggregationsansatz für die Umweltdimension der Kritikalitätsbewertung entwickelt, indem die Umweltauswirkungen monetarisiert werden. Diese Modifikation baut auf einem Review-Artikel auf, der alle derzeit verwendeten Monetarisierungsmethoden in der Ökobilanzierung auf quantitativer und qualitativer Ebene analysiert hat. Es wurden zwei Methoden zur Quantifizierung von Umweltkosten durch Monetarisierung von Ökobilanzergebnissen entwickelt, die zur Messung der Umweltkritikalität verwendet werden können. Eine Methode berücksichtigte die Umweltkategorien in der Methode SCARCE. Die zweite Methode monetarisierte alle ReCiPe-Wirkungskategorien, die sich auf die menschliche Gesundheit und terrestrischen Ökosysteme beziehen. Um die entwickelten Methoden anzuwenden, wurden zwei Fallstudien durchgeführt, um die Umweltkosten i) der globalen Materialproduktion und ii) der Energie- und Verkehrswende der EU zu bewerten. Die Umweltschäden durch die Ressourcennutzung für die Energie— und Verkehrswende in der EU reichen von 13,1 Mrd. € bis 74,8 Mrd. €, wobei die mittlere Schätzung 38,9 Mrd. € pro Jahr beträgt. Die Ergebnisse zeigten, dass insbesondere der materielle Bedarf der Mobilitätswende zur Umweltkritikalität beiträgt. Für die globale Materialproduktion reichen die Umweltkosten von 0,4 Billionen €/Jahr (niedrige Schätzung) bis 5 Billionen €/Jahr (hohe Schätzung). Die Materialien mit dem größten Kostenbeitrag waren fossile Energieträger (33 %-60 %), Eisen (12 %-20 %) und Aluminium (5 % 8 %) und damit Materialien, die für Grundfunktionen im täglichen Leben verwendet werden. Beide Methoden zeigen in welchen Ländern die Umweltauswirkungen auftreten. Sie können weiterverwendet, weiterentwickelt und modifiziert werden, um besser zu verstehen, wie sich die Materialgewinnung und ihre Umweltauswirkungen auf die allgemeine Wirtschaftstätigkeit und das menschliche Wohlergehen auswirken, aber auch, wie sie die Stabilität einer Handelsbeziehung beeinflussen können. Die Methoden sind somit ein erster Schritt, um Maßnahmen zu identifizieren, die die Umweltauswirkungen der Rohstoffgewinnung und -verarbeitung reduzieren können, auch mit dem Ziel, Handelsbeziehungen langfristig zu stabilisieren.

List of Core Publications

Paper 1: Arendt, Rosalie; Bachmann, Till M.; Motoshita, Masaharu; Bach, Vanessa; Finkbeiner, Matthias: Comparison of Different Monetization Methods in LCA: A Review. *Sustainability* **2020,** *12*, 10493. https://doi.org/10.3390/su122410493.

Paper 2: Arendt, Rosalie; Muhl, Marco; Bach, Vanessa; Finkbeiner, Matthias: Criticality Assessment of Abiotic Resource Use for Europe– Application of the SCARCE Method. *Resources Policy* **2020** 67 (August): 101650. https://doi.org/10.1016/j.resourpol.2020.101650.

Paper 3: Arendt, Rosalie; Bach, Vanessa; Finkbeiner, Matthias: The global environmental costs of mining and processing abiotic raw materials and their geographic distribution. *Journal of Cleaner Production* **2022** https://doi.org/10.1016/j.jclepro.2022.132232

Paper 4: Arendt, Rosalie; Bach, Vanessa; Finkbeiner, Matthias: Environmental Costs of Abiotic Resource Demand for the EU's Low Carbon Development. *Resources Conservation and Recycling* **2022** 180 (May): 106057 https://doi.org/10.1016/j.resconrec.2021.106057

List of Abbreviations

Abbreviation	**Stands for**
AADP	Anthropogenic stock extended abiotic depletion potential
ADP	Abiotic depletion potential
AoP	Area of protection
CEENE	Cumulative exergy extraction from the natural environment
CExD	Cumulative exergy demand
CML	Centrum voor Milieuwetenschappen Leiden
DALY	Disability adjusted life year
EDIP	Environmental Development of Industrial Products
EPS	Environmental priorities strategies
ESP	Economic resource scarcity potential
ESVD	Ecosystem valuation database
EU	European Union
GLAM	Global Guidance for Life Cycle Impact Assessment Indicators and Methods
IRP	International Resource Panel
ISO	International Standard Organization
LCA	Life Cycle Assessment
MFA	Material flow analysis
ORI	Ore requirement indicator
SCP	Surplus cost potential
SOP	Surplus ore potential
SED	Solar energy demand
UNEP	United Nations Environment Program
USGS	United States Geological Survey
WTA	Willingness to Accept
WTP	Willingness to Pay

Table of Contents

1 Introduction

Resource extraction and processing is necessary to support economic activity, while it leads to substantial environmental impacts. Additionally, economic short-term availability as well as long term availability of abiotic materials might pose limits to human wellbeing and prosperity. These environmental and economic impacts of material consumption are assessed with a variety of methods.

This introduction starts with the background and motivation of this thesis and elaborates on the development of global resource consumption. Next, existing environmental and economic assessments that have been used to assess environmental and economic impacts of abiotic raw material use and extraction are presented. Subsequently, the gaps and challenges related to this research field are outlined, followed by an outlook on the structure of this thesis.

1.1 Background and Motivation

1.1.1 Global Resource Consumption

Abiotic raw materials are a necessary component of most products and are essential for economic activity in many key sectors (European Commission, 2008; National Science and Technology Council, 2016). The use of metals and fossil fuels is increasing sharply, and this trend is not expected to change (IRP et al., 2019). The development of metal ore and fossil fuel extraction since 1970 is shown in Figure 1, which depicts this increase and shows that the extraction increased more than twofold since the 1970ies.

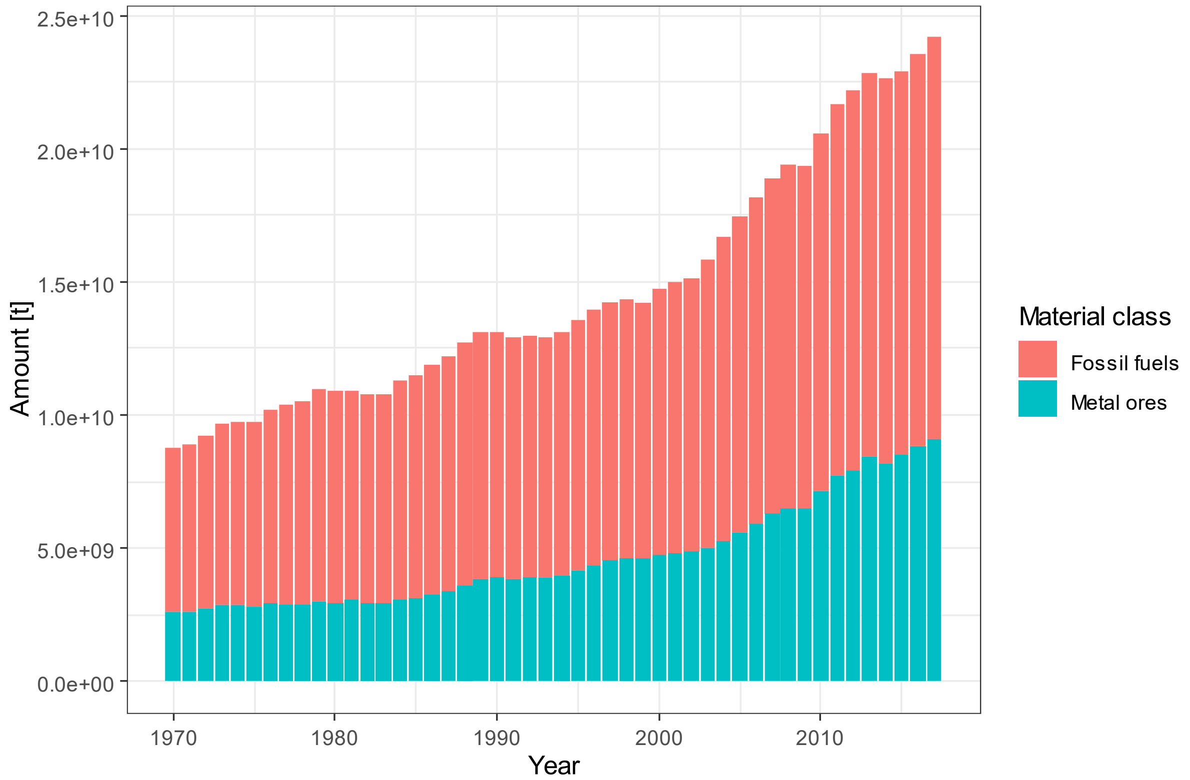

Figure 1: Development of material extraction from the 1970 until today based on United Nations Environment Program International Resource Panel Global Material Flows Database (UN-IRP, 2018)

Additionally, more varied materials are used. In antiquity only up to 7 metals were used in products, while today a smartphone contains nearly all elements of the periodic table (Sykes et al., 2016). These ongoing trends do not only lead to a depletion of a finite stock of raw materials, but to significant impacts on the environment, such as the transformation of land, the use of water and contribution to the climate crisis through logging and fossil energy consumption in mining and processing (IRP et al., 2019).

As not all materials are available in all world regions and also not all products are manufactured in proportional amounts in all countries, these characteristics lead to complex dependency-nets that are shaped by geological distribution of materials and their manufacturing in different countries (Cimprich et al., 2019; Graedel et al., 2015). These dependency-nets are influenced by trade policies and political attitudes of governments (Jackson and Green, 2017) and their export policies (Bach et al., 2016). Thus, the availability and the environmental impacts of materials are shaped by i) geological availability; ii) the global division of production iii) political availability (determined by trade policies and a country's political stability) iv) environmental impacts and v) social impacts (Bach et al., 2017, 2016). These factors become more relevant, when the demand of materials change rapidly, as always more complex combinations of materials and new materials are needed for electric appliances and for the energy and mobility transition.

While Figure 1 underlines the growth of material extraction from a mass perspective, it does not show the environmental impacts of the material extraction, nor the economic vulnerability that the reliance on ever greater material mass causes.

1.1.2 Environmental and Economic Assessment of Abiotic Raw Material Use

Different methods have been applied to assess the environmental and economic effects of material extraction and processing on the environment and the economy. Some key methods are LCA, material flow analysis (MFA), input-output-analysis and criticality assessment. LCA has evolved as a company internal tool to optimize energy use in production (McManus and Taylor, 2015), which is now a standardized environmental management method (ISO, 2006a, 2006b), while criticality assessment of raw materials on national level has its origin in the assessment of the supply risk and vulnerability of materials that are necessary for national defense, while it is building on classical risk assessment in companies (Glöser et al., 2015; Schrijvers et al., 2020). Monetization of environmental impacts has developed due to market failures that lead to the underprovision of goods that are not traded on markets, but are necessary for economic activity none-the-less (Hanley and Spash, 1993). Within this thesis new methods to quantify environmental criticality were developed, which builds on criticality assessment, LCA and monetization, which is why these methods are introduced in more detail in the following subchapters.

These methods have developed regarding their scope until today. In Table 1 the goal of the assessment methods, their scientific foundation and result is outlined.

Table 1: Overview of methods regarding their goal, foundations and results

Method	Goal	Foundations	Result
Monetization of environmental impacts	Translates an environmental impact into an economic unit	Environmental economics	A monetary value per environmental impact
LCA	Assesses the environmental impact of products or services along their supply chain	Energy accounting in companies and natural science/engineering	Environmental impact profile of a product or service
Criticality assessment of raw materials	Vulnerability of countries, companies or products to the supply risk of a certain material and the likelihood of the supply risk of the material	National policy, risk assessment	Specific criticality for different materials that can be used for policy recommendations

As the main focus of this thesis in the improvement of environmental criticality assessment through monetization of LCA results, Table 2 depicts the relation of the three assessment methods to environmental criticality assessment.

Table 2: Outline how environmental impacts affect criticality, as well as measurement and aggregation options; the main areas of research that are touched in this thesis are highlighted in green

How do environmental impacts influence criticality?	Measurement options	Aggregation options
On national level: • Countries stop to export to protect their environment because extraction leads to too high local impacts • Can be in conflict with the importing countries' environmental goals for scope 3 impacts • Environmental groups and civil society criticize trade policies and up-stream impacts of material extraction, which negatively influences the reputation of a country **On company level:** • Can be in conflict with national legislation • Environmental groups and civil society criticize trade policies and up-stream impacts of material extraction, which negatively influences the reputation of a company **On product level:** • Civil society criticize trade policies and up-stream impacts of material extraction, which can lead to consumer boycotts	**Impacts measurable with LCA:** Potential impacts of mining and processing along the supply chain **Impacts not measurable with LCA:** Accidents Environmental policies News coverage	**Aggregations options:** Pairwise comparison Equal weighting Monetization No aggregation

Environmental criticality assessment aims to quantify the supply risk caused by environmental impacts of material extraction (Manhart et al., 2019). Due to the increased number and mass of materials that are extracted, the environmental pressures and thus the environmental criticality is likely to rise. Environmental criticality is relevant at national, company or product level. What criticality assessment is will be outlined in more detail in chapter 1.1.2.1. Some of the environmental impacts that contribute to environmental criticality can be measured with LCA, while for example accidents or news coverage of environmental impacts go beyond LCA indicators. The measurement of environmental impacts of mining in the context of LCA is the topic of chapter 1.1.2.2. Different methods to aggregate indicators in criticality assessment exist. One of them is monetization, which is outlined in chapter 1.1.2.3.

In the following the state of the art of the three assessment methods is described.

1.1.2.1 Criticality Assessment

Criticality assessment is used as an assessment method to measure the dependency on a material or product and likelihood of the unavailability of a material or product. It has its roots in classical risk assessment (Glöser et al., 2015). The scopes of a criticality assessment differ. It can be executed at product, company, country, regional and the global level. The beginning of the identification of critical or sometimes called strategic raw materials, lies with materials that are necessary for war and defense, while the assessment is today more and more focused on economic growth and the sustenance of key industries (Schrijvers et al., 2020). A common definition of critical raw materials are materials that combine two properties. First, they have a high probability of an occurring supply risk. Second, a system (a product, national economy or company) has a high vulnerability to a supply restriction of that material (Cimprich et al., 2019). Criticality has been mostly visualized using a criticality matrix, which has been first introduced by the U.S.A.'s National Research Council (2008). It is depicted schematically in Figure 2.

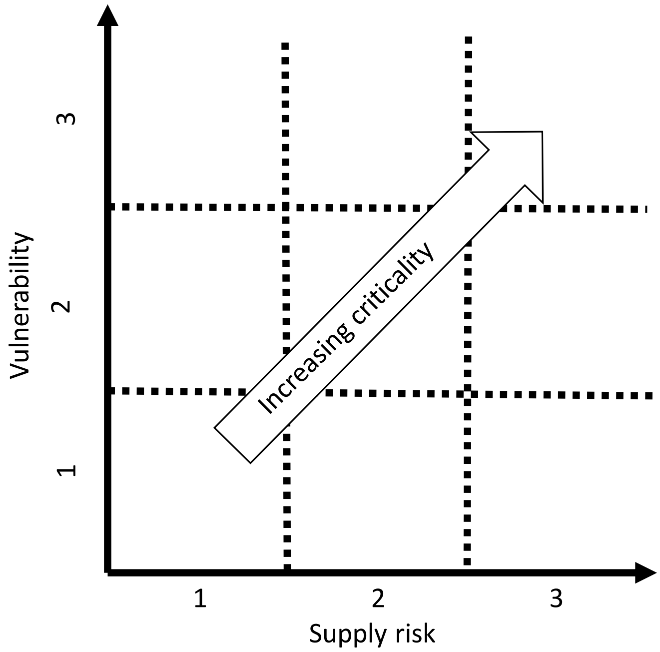

Figure 2: Schematic depiction of a criticality matrix based on National Research Council (2008)

The supply risk and vulnerability are calculated based on a variety of indicators and underlying data sources (Helbig et al., 2016b). A comprehensive review of criticality assessment has been presented by Schrijvers et al. (2020). The first criticality assessments for raw materials that is close to the criticality assessment today, have been developed in 2008 for the U.S.A. (National Research Council, 2008) and United Kingdom (Morley and Eatherley, 2008). The European Union (EU) has started to publish a list of critical raw materials since 2011 (European Commission, 2011). Since then, the EU releases such a list and continually revises its methodology. The EU's assessment is and was only focused on economic parameters. Today most countries have such an assessment method (Schrijvers et al., 2020). But also criticality assessments for products have been developed to assess the short term availability of production factors (Bach et al., 2016; Cimprich et al., 2017; Gemechu et al., 2016; Helbig et al., 2016a).

The environmental and social dimensions of criticality have been taken into account by some methods. The first peer reviewed publication by Graedel et al. (2012) presenting a criticality assessment considered environmental issues as a third axis via LCA indicators to supplement the vulnerability and supply risk assessment. Manhart et al. (2019) developed a method only related to environmental criticality. It included some LCA indicators, but also considered aspects that are beyond the scope of LCA such as probability and severity of environmental hazards. The first method to include all sustainability dimensions in criticality assessment was the ESSENZ method (Bach et al., 2016) at product level and the thereof derived SCARCE method (Bach et al., 2017) at country level that also both draw on LCA. On product level it is assumed that environmental and social criticality might lead to consumer boycotts (Bach et al., 2016). On a country level environmental and social impacts of material use could compromise political goals related to environmental integrity and social justice. While some methods keep environmental and social impacts as a separate dimension for criticality assessment (Bach et al., 2017, 2016; Graedel et al., 2012), others propose that environmental impacts should be integrated into the supply risk dimension (Glöser et al., 2015).

As Schrijvers et al. (2020) outline in their review, one of the main challenges is to aggregate different impacts in a criticality assessment. Several options to weight and aggregate exist. In the ESSENZ and SCARCE method it is generally not recommended to aggregate, but if aggregated results are shown they are aggregated using equal weighting. Glöser et al. (2015) have used pairwise comparison for aggregation. The approach by Vogtländer et al (2019) is relevant for this thesis, as it monetizes the value at risk of criticality assessment, by assessing price fluctuations. They express the determined value at risk in monetary terms.

1.1.2.2. Life Cycle Assessment and its Assessment of Abiotic Resource Use

LCA is an environmental management method that assesses environmental impacts of products and services along their supply chain. It is standardized in the ISO standard 14040/44 (ISO, 2006a, 2006b) and quantifies environmental impacts in relation to a provided function (functional unit). The quantified environmental impacts are systematized in impact categories that cover environmental impacts of products as far as possible (but not exhaustive (Finkbeiner et al., 2014)).

Resource use has been integrated early into LCA, as every product fabrication contains the extraction of some materials. Whole mining industries in a country have been assessed using LCA (Strezov et al., 2021), as well as processes to compare different extraction and manufacturing modes with one another to reduce the environmental impacts of mining (Erkayaoğlu and Demirel, 2016; Simate and Ndlovu, 2014). Several industry associations that are active in mining, publish and collect Life Cycle Inventory data and conduct LCA case studies (Harris and Broadbent, 2019; Vaccari and Tikana, 2017; Wang, 2022).

In order to assess the environmental impact of any product (also mining and processing) an impact assessment that condenses the elementary flows to an understandable unit is needed. Mining and processing of abiotic raw materials is relevant, because the exhaustion of resources for human economic development is conceptualized as one of the main area of protection (AoP) next to human health and ecosystems in LCA (Goedkoop et al., 2008; Goedkoop and Spriensma, 1999; Huijbregts et al., 2017). Impact assessment methods in LCA can be clustered in midpoint and endpoint methods. Midpoint methods refer to a potential environmental impact, while the endpoint assessment measures the potential environmental damage that could be caused by the potential environmental impact to the AoP (Bare et al., 2000). A simplified depiction of environmental releases and resource use, its relation to potential impacts (mid-point) and potential damages (end-point) is shown in Figure 3.

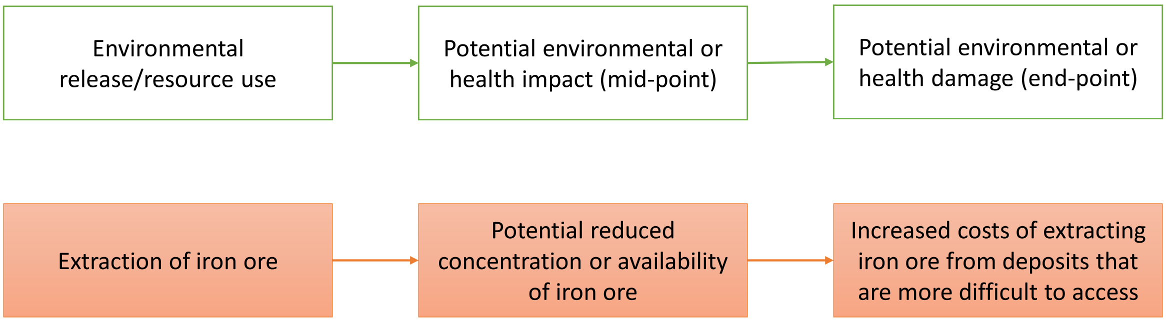

Figure 3: Schematic depiction of mid and endpoint impact assessment (above) and example of mid-point and end-point based assessment for resource use based on the future effort principle (below)

A widely used impact assessment method is ReCiPe (Huijbregts et al., 2017). It is suitable for the European context and was first developed in 2008 (Goedkoop et al., 2008) as a combination of CML 2000 (Guinée et al., 2002) and Eco-indicator 99 (Goedkoop and Spriensma, 1999) and thus a synthesis of mid- and endpoint methods. The damages to the AoPs human health, resources and ecosystems are considered in the end-point assessment. The assessed mid-point indicators are climate change, stratospheric ozone depletion, ionizing radiation, particulate matter formation, photochemical ozone formation, terrestrial acidification, freshwater eutrophication, marine eutrophication, human and ecotoxicity, water use, land use, mineral resource scarcity and fossil resource scarcity.

The assessment of abiotic resources is intensively discussed in the LCA community and a variety of methods exist (Dewulf et al., 2015). The methods can be roughly categorized into depletion methods, supply risk methods, future effort methods and thermodynamic accounting, of which an overview is provided in Figure 4 based on the review by Sonderegger et al. (2020).

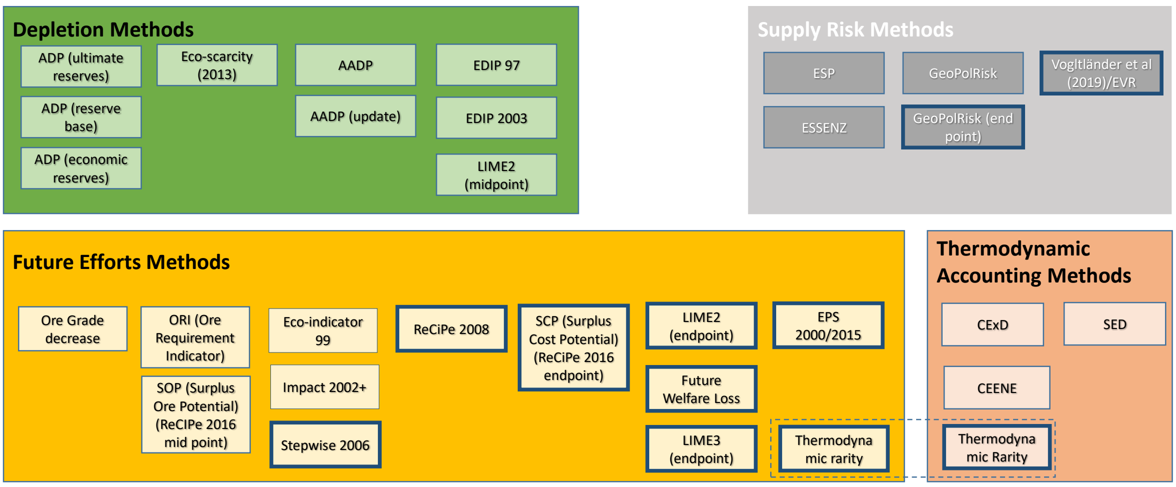

Figure 4: Overview of methods to assess resource use in LCA based on the categorization by Sonderegger et al. (2020); with some own modifications, methods that quantify the impact in monetary terms have a thicker dark blue frame; supply risk methods are greyed out as they were explained in the paragraph before, in relation to product specific criticality assessment

The four different categories of methods are outlined shortly in the following; for a brief introduction of all 27 methods see the reviews by Sonderegger et al. (2020) and Berger et al. (2020). The first methods to assess the impact of resource use are the depletion methods, of which the most used indicator was the Abiotic depletion potential (ADP). It calculates the availability of resources compared to their overall availability in the earth's crust (Guinée and Heijungs, 1995; van Oers et al., 2020; van Oers and Guinée, 2016). This approach was modified through the inclusion of the anthropogenic stock (Schneider et al., 2015, 2011).

Another methodological category is the future effort method that calculated how extraction leads to declining ore grades. At the midpoint level, the declining ore grades or additional quantities of needed ores are assessed. At endpoint level, on the other hand, additional necessary energy is determined, that can be quantified in monetary terms as additional expenditure (Huijbregts et al., 2017; Vieira et al., 2016). For example, Steen (2016, 1999) in the EPS-method, determined how expensive it would be to extract resources from the earth's crust if all known deposits are exhausted.

Another category to quantify the impacts of resource use is thermodynamic accounting. There are different approaches, but all measure the energy used in a product. The cumulative exergy extraction from the natural environment (CEENE) (Dewulf et al., 2007) and the cumulative exergy demand (CExD) (Bösch et al., 2007) method measure how the high the exergy in a product is compared to a "natural" reference state, that was defined by Szargut et al. (1988). The thermodynamic rarity method is classified as a thermodynamic accounting method and future effort method. It quantifies how the depletion of ore bodies reduce the benefits of a high concentration of metals in an ore body. This is done through a comparison to the needed exergy to concentrate materials to a similar degree from a completely dispersed state (Valero and Valero 2015).

The supply risk methods rather focus on the impact that the dependency on a certain material has on the product itself. They quantify whether the provision of the material is safe and how vulnerable a product is to a supply restriction. As product related criticality assessment has already been elaborated on in section 1.1.2.1, this is not repeated here. However, two approaches discussed in LCA have not been explained in that section, which have also not been included in the review by Sonderegger et al. (2020)- namely the endpoint method of the GeoPolRisk method that quantifies the associated economic damage that is caused by supply risks (Santillán-Saldivar et al., 2021b) and the approach by Vogtländer et al. (2019), that quantifies the value at risk.

The future effort method has been questioned by the observation that material prices are not increasing with declining ore grades (Henckens, 2016; Jowitt et al., 2020). This view has been voiced the context of LCA as well (De Nocker and Debacker, 2018).

Further, mining and processing have several environmental impacts that are not directly connected to the depletion of a finite stock, e.g. the use of energy, carbon emissions, acid mine drainage and land use (IRP et al., 2019). Material mining and production generally contributes to several impact categories that an LCA covers (Awuah-Offei and Adekpedjou, 2011) not only the use of materials.

1.1.2.3. Monetization of Environmental Impacts in the Context of LCA

This subchapter depicts the foundations of monetizing environmental impacts in general and in LCA. This subchapter is more detailed than the criticality assessment and LCA subchapter, as the main contribution of this thesis lies with the detailed analysis and the methodological development of monetization methods to aggregate indicators that are used to quantify environmental criticality (as visible in Table 2). In environmental economics positive and negative environmental impacts have been conceptualized as an externality that can be quantified in terms of a monetary value (Pigou, 1920). This applies also to impacts that affect goods that are usually not traded on markets such as species and habitats as well as clean air and water. This approach is normally used to compare economic gains of an activity with its non-marketed losses. Therefore, the environmental impacts of a certain production process become comparable to its economic gains, as both have the same unit: “money”. This theory was first proposed by Pigou to compare marginal costs with marginal benefits of environmental pressures. Based on the relation of marginal costs to marginal benefits, a social optimum tax could be calculated that would correct the “market failure” (Edenhofer et al., 2021; Pigou, 1920). Today monetization of environmental impacts is mostly used in cost-benefit-analysis (Freeman III et al., 2014; Hanley and Spash, 1993). The monetization is applied to either evaluate the costs or benefits of a legislation or project to determine, if it should be executed from a welfare economic perspective or to calculate the optimal cost of an environmental tax that should correct market failures. The most discussed current tax is the carbon tax, whose exact recommended magnitude vary widely. The different estimates of the externality of greenhouse gases (Bachmann, 2020) are the cause of this variation.

The approaches to monetize environmental impacts are varied in LCA as well and are described in detail in a publication of this thesis (Arendt et al., 2020), but also by earlier and later review works (Amadei et al., 2021; Pizzol et al., 2015), as well as articles on the environmental economic foundations (Bachmann, 2019). The core distinctions are shortly outlined here. Approaches such as abatement costs that quantify the costs of a pollution by the costs to abate it, differ from approaches such as damage costs that measure the damage to human health, and ecosystems in monetary terms. Under socially optimal conditions marginal abatement and damage costs will be the same. The relation of costs to benefits and marginal costs to marginal benefits are depicted in Figure 5.

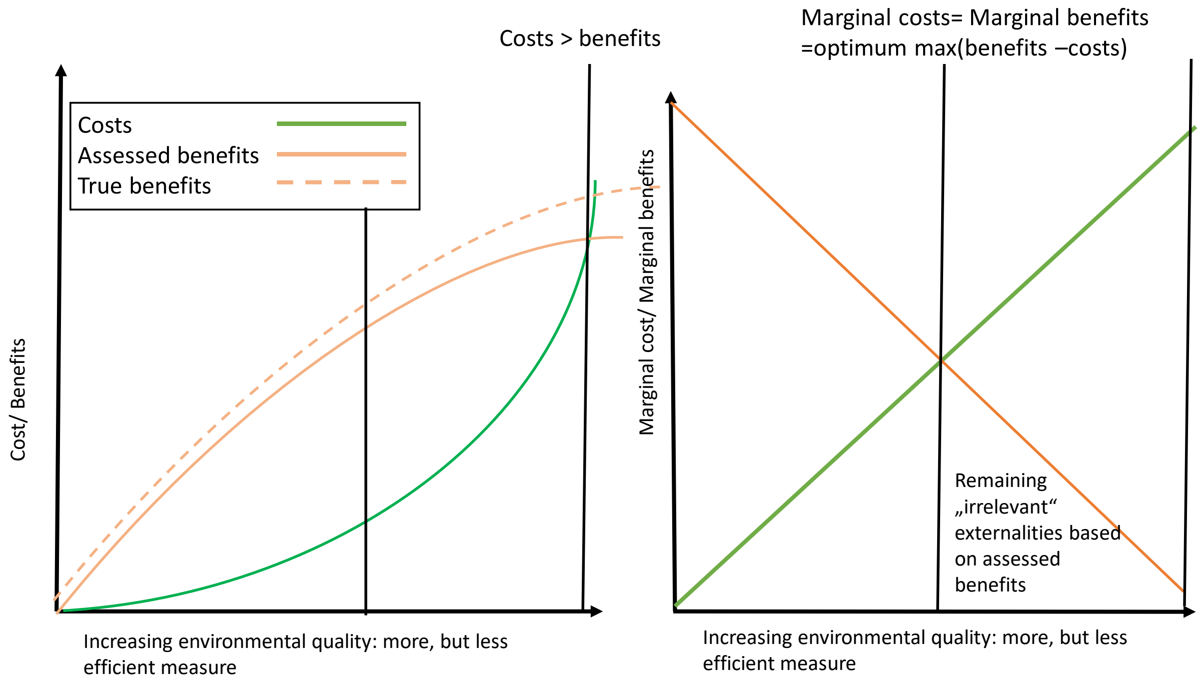

Figure 5: Shows cost and benefits curve (left) and their relation to marginal costs and benefits (right) based on (Bachmann, 2019)

In the figure on the left side, it is visible that the benefits of a measure to reduce an environmental impact clearly outweigh the costs, when the environmental quality increases. This increase and difference declines moving along the x-axis. In environmental economic theory, a measure should be used to such a degree that benefits are maximized. This is the case, when marginal benefits equal marginal costs, which is graphically shown on the right side of Figure 5. The “true benefits” cannot be quantified as all methods to quantify them are only an approximation. The “true benefits” can also be lower and not only higher than the assessed benefits.

The monetary damages of an environmental impact (equal to the benefits of avoiding pollutants) are difficult to determine. The environment and human health, which the damages affect, are not traded on markets. Different ways to express these damages in monetary terms are shown in Figure 6.

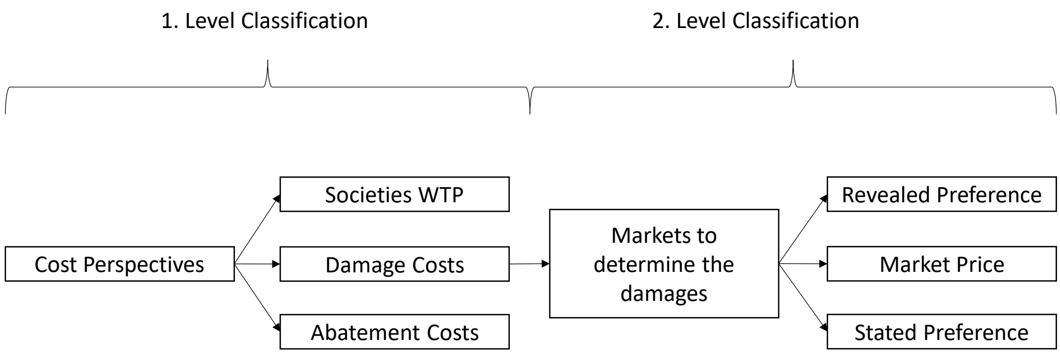

Figure 6: Classification of approaches to monetize non-marketed goods (Arendt et al., 2020)

The fist level classification distinguishes between societies Willingness to Pay (WTP) (determine the damage via ecological taxes that are already introduced), damage costs and abatement costs. For damage costs, additional distinctions are possible. The environmental costs can be determined by the market price of the goods that they provide, by revealed preference methods or stated preference methods (survey based) as shown in Figure 6. Based on the method, a different share of the total economic value is covered. The different values assessed from total economic value perspective are shown in Figure 7.

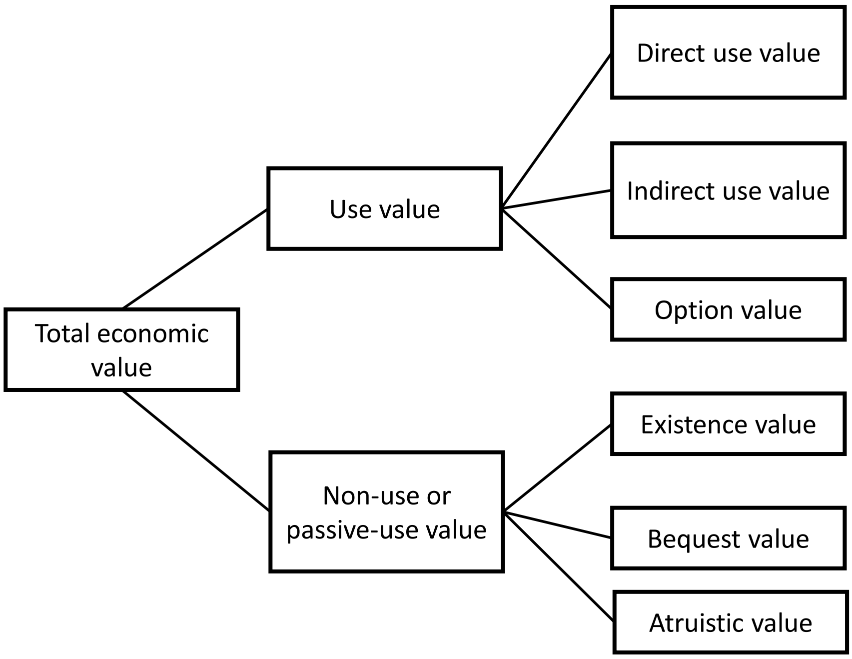

Figure 7: Different kinds of values based on (Bachmann, 2019)

The theory of total economic value assumes that values beyond direct use value from private purchase and consumption exist. In the following these other types of values are explained. Indirect use value is, for example, the value an individual derives from a nice view, because it is not consumed directly. The option value is the value that is obtained from having the option to do something, even though the option is not used. Non-use or passive use values are derived by the pure value from something existing. This value is unaffected by the fact that a person will never see or visit a certain site. The remaining values are the bequest value (leaving something to the next generations) or altruistic value (the enjoyment of giving something to humanity in general). Typically, only the survey-based methods that ask for WTP assess value beyond direct use value.

In LCA, monetization is used to weight environmental impacts of one impact category against another, to decide which impact category has a higher importance (ISO, 2006b; Pizzol et al., 2017). Weighting is an optional step in an LCA that is not allowed if a single score is created and products are compared and results published. While it is difficult to compare the divers impacts of one product and to prioritize impact categories, there is not the possibility to have one universal weighting set, as weighting is always based on value choices (ISO, 2006b) and values are diverse amongst humans. As there is no universal weighting set and as explained before, different options to monetize exists, it is a question whether the approach to monetize might affect the prioritization of the impact categories.

To illustrate ongoing debates and different methods to monetize, they are briefly introduced for the impact category climate change. The monetization of climate change impacts is very developed in environmental economics. For climate change two decisions that are based on values i) the discount factor as well as ii) the impact of environmental damages on inequality have a high influence on the estimated environmental costs of emitting one ton of carbon (Adler et al., 2017; Adler and Treich, 2015; Stern, 2006). The marginal damages of one ton CO_2 have diverging recommended values. The highest estimate without discounting adds up to \$10 000/t CO_2 (Archer et al., 2020). Recently the costs of carbon have been assigned to different countries (Ricke et al., 2018), so that it becomes possible to identify, which countries suffer most and least from the emission of one ton of carbon. All of these costs are dependent on the taken socio-economic pathways and also on different value parameters that underline how monetization is inherently political and determined by normative choices.

The use of abiotic resources was covered in environmental economics mostly with a shadow price that was tackled through the Hotelling's Rule (Hotelling, 1931). It is based on the fact that the market discount rate is usually higher than the social discount rate, resulting in an exhaustion of resources that is faster than the social optimum. This approach has been used to derive a shadow price that corrects the market price with a factor to obtain the socially optimal price (Huppertz et al., 2019).

Following this theoretical introduction, the derived gaps and challenges are outlined and how the work in this thesis is related to them.

1.2 Gaps and Challenges

As shown in the previous sections, abiotic resource extraction and processing and its environmental impacts, as well as criticality have been a concern for a long time now and significant efforts have been made to better assess these impacts. However, some gaps and challenges remain that are outlined in the following paragraphs. An overview of them is given in Table 3.

Table 3: Overview of identified gaps and challenges related to methodological issues (left) and empirical challenges (right) gaps and challenges that are addressed in this thesis are black, those that are not addresses are grey

<table>
<tr><th colspan="2">Methodological Gaps and Challenges:</th><th>Application and Data Gaps and Challenges</th></tr>
<tr><td>Criticality Assessment:
• How to deal with free trade zones?
• How to quantify social impacts?
• How does certification affect criticality?
• How does recycling affect criticality?
• Integrate supply risk of energy input for extraction of materials</td><td rowspan="2">LCA indicators in criticality assessment, monetization and mining:
• How to quantify marginal and non-marginal impacts?
• Integration of up-to-date research in environmental economics (ecosystem services)
• How to account for geographical shifts of impacts?
• How do environmental impacts affect equity?</td><td>Data:
• Not regionalized enough inventories of mining processes
• Trade trace back model of materials and tracking of stockpiling
• Quantitative social data for mining</td></tr>
<tr><td>Monetization:
• How to quantify marginal and non-marginal impacts?
• Monetize with WTA</td><td>Empirical testing, verification and validation
• How do predicted and occurring supply risks compare?
• How do predicted environmental hotspots in mining compare with conditions on site?</td></tr>
<tr><td colspan="2">Weighting and aggregation:
• How can indicators in criticality be aggregated?
• How to weight impacts in LCA monetarily?
• How to prioritize material use reduction to decrease global environmental pressures?</td><td>Combination of LCA and monetization with other methods to enable optimization or increase knowledge of environmental impact patterns:
• With MFA
• With Integrated assessment models
• With energy system models</td></tr>
</table>

The outline of the gaps and challenges starts with the methodological gaps and challenges in criticality assessment, followed by the gaps related to LCA indicators to measure environmental criticality monetarily and then in weighting and aggregation in general. Sometimes a methodological gap is connected to a data gap, if this is the case the connection is described directly.

A challenge in criticality assessment is, that supply risks often affect several countries, if these countries are in a free trade zone. Therefore, countries with similar trade restrictions should be assessed together. This has not been done by the EU, e.g., the fact that Norway extracts oil and gas is not lowering the EU's criticality for oil and gas, even though it is in a free trade zone with Norway.

An additional methodological gap is that the social dimension of criticality assessment, is currently lacking enough quantitative, material-specific data to be assessed quantitatively or monetarily. So, a damage-based assessment for social impacts in criticality assessment is currently not developed, due to the fact that neither an impact pathway nor the necessary quantitative data are available.

An additional gap is that in most criticality assessments the criticality of recycled materials is included in very different ways. Some methods include an indicator for recycling, primary material input or impact of recycling on the supply chain that reduces the vulnerability (Bach et al., 2017; Deloitte Sustainability et al., 2017; Santillán-Saldivar et al., 2021a), while others strive to calculate an own criticality score for secondary material (Pelzeter et al., 2022). A comparison and additional development of this research is currently missing. Additionally, it would be of interest to analyze how recycling can lower criticality, especially environmental and social criticality and environmental costs, which is mostly not done due to a lack of data.

Furthermore, the criticality of input materials to extract and process a material, is not considered in criticality assessment for example that provisioning of steel is dependent on hard coal and thus the criticality of steel is also affected by the criticality of coal. This analysis would depend on the necessary fossil energy quantity or other material inputs that are needed for mining and processing.

Now the methodological gaps and challenges related to LCA indicators to measure environmental criticality monetarily are outlined:

Generally, LCA is mostly used to assess products, however if LCA is applied to bigger systems, for example global material flows, this is a challenge for a given impact assessment. This debate and methodological challenge has been addressed by Boulay et al. (2019) and Forin et al. (2020), as well as by Steen (2006). Steen (2006) provide monetized average environmental impacts. It would be preferable to have marginal and average impact assessment methods to choose accordingly, depending on the size of the analyzed system. The size of the analyzed system will deliver the information, whether the environmental impact can be considered marginal or not. Average and marginal characterization factors have been provided by Boulay et al. (2019), but only for the impact category water use. This challenge is also relevant in the context of monetization, as marginal and average environmental costs vary significantly.

Furthermore, the current monetization methods used in LCA do not represent the state of the art in environmental economics. For example, the calculation of ecosystem services has not yet been implemented in monetization in LCA.

An additional shortcoming that is relevant for monetization and LCA is the impact of environmental impacts on equity. As equity weighting is common practice in welfare economics (Anthoff et al., 2009; ISO, 2019), monetization factors for environmental impacts depend on who is actually affected by the damage. This is based on the assumption that the monetization factor would rise if an impact on human health (e.g., particulate emissions) would affect someone that is already exposed to severe health damages. This is due to the law of diminishing marginal utility (that a good is valued more strongly the less we have of it and the other way round (Gossen, 1854)). The inclusion of this effect in monetization and LCA would require much more granular data regarding the affected population of a given environmental impact.

Furthermore, the environmental impacts of criticality assessment (quantified by LCA), but also environmental damages in monetary terms are often not explored as geographically distributed. Thus, geographical shifts of impacts are neglected. While often a certain change in material production might reduce the impact of one impact category overall, the impact might be shifted to another region. This leads to an increase of environmental impact in a before unaffected area. The geographical location of an impact provides information on the affected population’s wealth. If the geographical distribution of impacts is available, it will be easier to include the environmental impacts on equity.

Now the challenges regarding monetization and weighting are outlined:

Monetization in LCA is mostly related to weighting. Weighting is applied, when different assessment indicators are combined. The decision how to weight -and thus, which material dependencies should be reduced with priority to reduce environmental impacts or supply risks- poses a severe challenge. The problem of weighting and aggregation is similar in LCA and criticality assessment. This is due to the fact that many factors influence supply risks and vulnerability as well as environmental impacts and it is no easy task to identify which factor is how decisive.

Monetization is applied in LCA since 1992 and a review of monetization approaches has been provided (Pizzol et al., 2015). However in recent years additional approaches (Bruyn et al., 2018; De Nocker and Debacker, 2018; Murakami et al., 2018; Trucost, 2015) have been developed that were not covered by the review. The variety of available factors can be confusing for practitioners. Additionally, there is no structured overview where the variety of factors comes from and what influences the magnitude of the results.

Moreover, monetization in environmental economics is used in a different context than in LCA, e.g., to calculate externalities or environmental costs. This possibility to assess externalities of consumed products is underexplored in LCA.

In the next paragraph the empirical and data gaps and challenges are described:

Generally, the inventory data sets regarding mining and processing of abiotic raw materials have low granularity, for example it is not possible to easily compare the environmental impacts from different mine sites for most materials, as the sphera (Sphera Solutions Inc., 2021) and ecoinvent (Ecoinvent, 2021) database often only provide one or two datasets per material (particularly for specialty metals).

Another challenge is that, even if such data was available, it is very difficult to identify where a purchased material was mined and processed. For a more detailed criticality assessment trade trace back models would be needed. Such a model should identify where materials are stockpiled and where only transit trade occurs, which might be achieved by a global material specific MFA.

An additional data gap is the lack of quantitative and industry specific social data, for which on-site visits in mines and processing facilities are necessary.

For criticality assessment it would be beneficial to compare retrospectively how predicted supply risks in criticality assessments compare to actual supply risks. This would help to validate the predictive quality of criticality assessment and could identify which indicators are most suitable to predict an occurring supply risk. Such an assessment is missing for identified environmental hotspots in criticality assessment as well. Site-specific studies related to the materials that were identified as posing the greatest environmental risks might close this gap.

Additionally -and this is related to the quantification of marginal and non-marginal damage quantification in LCA- it is underexplored to combine LCA with other assessment methods beyond criticality assessment, namely MFA (De Meester et al., 2019; Ismail and Hanafiah, 2021), integrated assessment models (Baumstark et al., 2021; Rauner et al., 2020) and energy system models (Algunaibet et al., 2019). Some of these combinations have already been explored, but to the author's knowledge, without considering the severe difficulty that non-marginal assessment of LCA poses and how it affects the impact assessments and results. The integration of monetized LCA results would allow these methods, if they are set up to optimize, to use single objective optimization that considers several environmental impacts.

To summarize the following gaps and challenges are addressed, but not fully resolved in this thesis:

- Lacking overview and analysis of currently used monetization methods
    - Quantitative and qualitative overview
- Criticality assessment
    - Exploring different approaches to weighting in environmental criticality assessment through monetary weighting of environmental impacts
    - Developing a weighting method for environmental criticality assessment that helps to prioritize materials
    - Assessing a free trade zone for its supply risk
- Geographical distribution of impacts
    - Mapping environmental impacts of resources
- Application of monetization beyond weighting-exploring the potential of monetization
    - First combination of LCA indicators and the ecosystem service valuation database
- Identify how the choice of the aggregation methods change results of LCA and criticality assessment

The following gaps are not addressed in this thesis:

- Weighting method to aggregate all dimensions of criticality assessment
- Quantitative assessment of social impacts in criticality assessment
- A thorough cause effect chain of environmental impacts measured with LCA and their impact on the provisioning of ecosystem services
- Marginal and non-marginal impact assessment methods in LCA and monetization
- Combine LCA with MFA and other assessment methods
- Explore how increased recycling rates affects criticality and environmental costs

In order to close some of the described gaps and challenges this thesis is dedicated to the following areas of research: (1) analyzing the properties of currently used monetization method in LCA (2) exploring further areas for the application of monetized LCA results beyond weighting (3) to use monetized LCA results to trace the environmental impacts of abiotic raw material production and their geographical distribution as a new environmental criticality assessment.

1.3 Structure of Thesis

Followed by this introduction and motivation of the thesis (chapter 1), the research goals and targets are formulated (chapter 2). The results chapter (chapter 3) contains the core publication of this thesis, and is followed by a discussion (chapter 4) of the associated uncertainties, the relation of the presented research to methodological trends, as well as strengths and weaknesses of the developed approaches and future research. The thesis closes with a short conclusion (chapter 5). The structure is visualized in Figure 8.

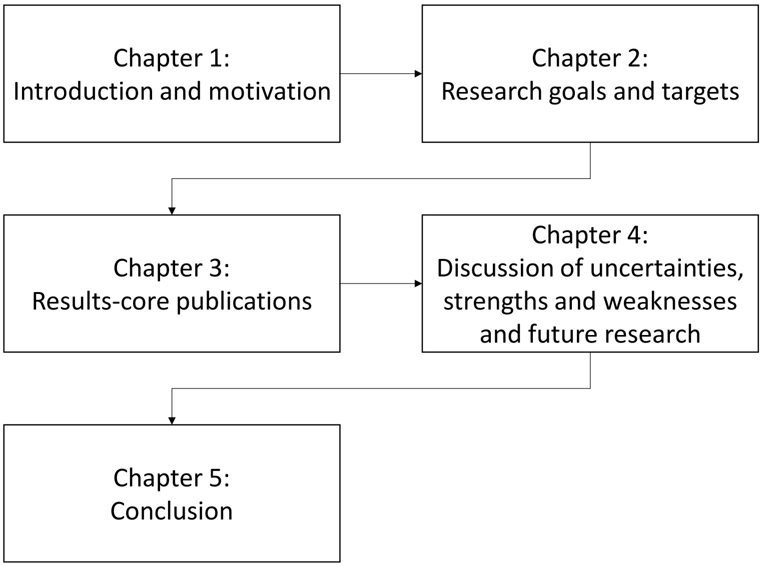

Figure 8: Overview of the structure of the thesis

This thesis aims to improve the environmental criticality assessment of abiotic raw material use. It builds on three assessment methods: two have been used to assess environmental and economic impacts of material use, namely criticality assessment and LCA and one that has been used to assess the impact of environmental impacts on human welfare: monetization of environmental impacts. The overall aim of the thesis it to improve the environmental and criticality assessment of abiotic raw material use by monetizing LCA results. This will be further specified in the next chapter that elaborates on the thesis' goals and research questions.

2 Goal and Research Targets

This chapter contains the thesis goals. First, the overarching goal is described followed by the core research questions (2.1). Then the relation from the research questions to the publications (2.2) is illustrated. Subsequently the research questions with their respective targets and the connection of the publications are detailed (2.3).

2.1 Overarching Goal and Research Questions

The overarching goal of the thesis is to improve the environmental criticality assessment of abiotic resource use by monetizing environmental impacts, which are measured by LCA. Therefore, this thesis aims to develop assessment methods to better measure and evaluate the environmental damages of abiotic raw material use in monetary terms that can affect the stability of a trade relationship between countries. The overarching research question of this thesis is: How can the monetization of LCA results improve the environmental criticality assessment of abiotic raw material use?

This thesis is focusing on three different assessment methods, which have been described in the introduction: criticality assessment, LCA and monetization of environmental impacts.

The following research questions specify the thesis' overarching goal.

1. Research question: Which monetization methods are used in LCA and how do they compare qualitatively and quantitatively?

This research question is analyzed in Paper 1 that reviews currently applied monetization methods in LCA. Additionally, the uncertainty and varying possible monetization factors are addressed and applied in Paper 3 and 4.

2. Research Question: What could be possible applications of monetization in LCA beyond weighting of impact categories?

This research question is answered in Paper 3 and 4. The monetized LCA results can be set into relation of overall economic activity to identify how high the magnitude of the environmental damages is in relation to the economic benefits. Another application builds on Paper 2 and 4: the integration of monetized LCA results into environmental criticality assessment. In Paper 2 environmental hotspots are identified by LCA indicators. The same impact category indicators that were considered in Paper 2 were monetized in Paper 4, which constituted an alternative environmental criticality assessment.

3. Research Question: How can monetized LCA results be used to identify which abiotic raw materials dominate the environmental impacts of material production and their geographical distribution?

This research question was answered by Paper 3 and 4 that applied monetization to the environmental impacts of abiotic raw materials in two case studies. Paper 3 quantified the environmental impacts of global material production and which countries are most affected. While Paper 4 assessed the environmental impacts of materials that are needed for the EU's low carbon development and can complement an economic criticality assessment with the environmental dimension. The determined impacts were absolute and allowed for a ranking of the materials. Additionally, it was possible to assign the costs to the affected countries.

2.2 Publications of Thesis

The core publications of this thesis are listed in this chapter. How they answer the research questions, which were formulated in section 2.1 is addressed in chapter 2.3.

Paper 1: Arendt, Rosalie; Bachmann, Till M.; Motoshita, Masaharu; Bach, Vanessa; Finkbeiner, Matthias: Comparison of Different Monetization Methods in LCA: A Review. *Sustainability* **2020,** *12*, 10493. https://doi.org/10.3390/su122410493.

Paper 2: Arendt, Rosalie; Muhl, Marco; Bach, Vanessa; Finkbeiner, Matthias: Criticality Assessment of Abiotic Resource Use for Europe– Application of the SCARCE Method. *Resources Policy* **2020** 67 (August): 101650. https://doi.org/10.1016/j.resourpol.2020.101650.

Paper 3: Arendt, Rosalie; Bach, Vanessa; Finkbeiner, Matthias: The global environmental costs of mining and processing abiotic raw materials and their geographic distribution. *Journal of Cleaner Production* **2022** https://doi.org/10.1016/j.jclepro.2022.132232

Paper 4: Arendt, Rosalie; Bach, Vanessa; Finkbeiner, Matthias: Environmental Costs of Abiotic Resource Demand for the EU's Low Carbon Development. *Resources Conservation and Recycling* **2022** 180 (May): 106057 https://doi.org/10.1016/j.resconrec.2021.106057

2.3 Targets and Connection of Papers

The fulfillment of the thesis goals through method development and performance of case studies will be specified in this subchapter. To do so, the outlined research questions from section 2.1 are assigned to the respective research targets. It is illustrated how these connect to the publications of this thesis.

1. Research Question: Which monetization methods are used in LCA and how do they compare qualitatively and quantitatively?

Targets:

a) Identify the main methods of monetization in LCA

b) Determine whether the choice of the monetization method in LCA affects the prioritization of impact categories

c) Quantify how the different methods determine the yearly environmental costs of an average EU citizen in absolute terms

2. Research Question: What could be possible applications of monetization in LCA beyond weighting between impact categories?

Targets:

a) Identify possible applications of monetization in LCA

b) Develop methods that realize potential new applications (as identified in 2a) of monetization in LCA beyond weighting

3. Research Question: How can the monetized LCA results be used to identify environmental hotspots of abiotic raw material production and their geographical distribution?

Targets:

a) Calculate the environmental costs of global material production

b) Calculate the monetary damages induced by environmental impacts of material demand of the EU's low carbon development with LCA and monetization

The scope of the different Papers, which methods they use and how they connect is shown in Figure 9. The assessment of environmental criticality with LCA indicators and its aggregation through monetization draws on the scheme in Table 2 in the introduction.

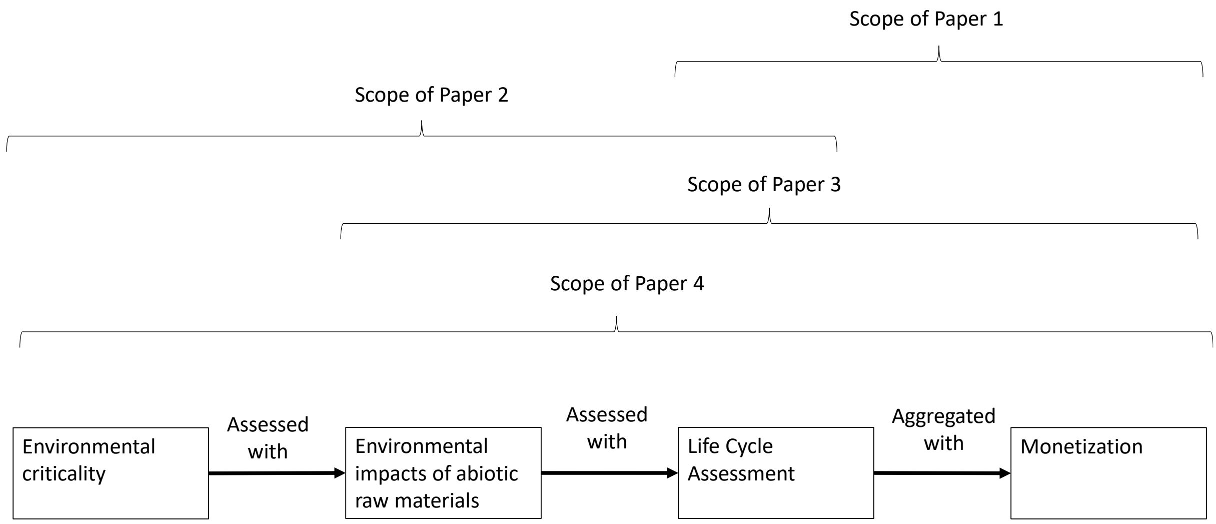

Figure 9: Method and topics of the thesis and how the different Papers cover these topics

Figure 9 shows how the Papers are connected and which assessment methods they cover, but also how they connect to the different targets. Paper 1 only focused on LCA and monetization of impacts. It derived the results to reach targets 1a- 1c, by i) identifying the main methods related to monetization in LCA, ii) determining whether the choice of the monetization method in LCA affects the prioritization of impact categories and iii) quantifying how the different methods determine the yearly environmental costs of an average EU citizen in absolute terms.

Research question 2, which explores additional options to use monetization in LCA is related to Paper 2-4. Target 2a) aims to explore additional applications of monetization in LCA beyond weighting- this is realized in Paper 3 and Paper 4. Paper 3 identified the applicability of monetized LCA results for cost-benefit-analysis and Paper 4 for environmental criticality assessment. Both papers also propose methods that realize these potential new applications and thus fulfill target 2b). Especially reaching target 2a and 2b drew on the experiences from Paper 2 as well.

Research target 3a) was answered by Paper 3 that determined the environmental costs of global material production. Target 3b) was answered by Paper 4, which quantified environmental damages of the EU's material demand for its low-carbon development. Both targets were reached by the application of the derived monetization methods.

3 Results

This section presents the publications of this thesis. Each paper is accompanied by an introducing paragraph that outlines its connection to the research questions and targets.

3.1 Comparison of Different Monetization Methods in LCA: A Review

The results are contained in the following publication:

Arendt, Rosalie; Bachmann, Till M.; Motoshita, Masaharu; Bach, Vanessa; Finkbeiner, Matthias: Comparison of Different Monetization Methods in LCA: A Review. *Sustainability* **2020,** *12*, 10493. https://doi.org/10.3390/su122410493.

The publication is mostly connected to research question 1. To answer this question, all currently used monetization methods in LCA were identified and analyzed from a qualitative and quantitative perspective (target 1a). The qualitative criteria are the following: used cost perspective and the type of market used when assessing damages, the included AoPs, the use of equity weighting, the discount rate, whether marginal or non-marginal impacts are valued and the handling of uncertainty. The quantitative assessment was performed by converting all monetary units into the same currency and then by converting to the same impact category indicator. Target b of research question 1 was to identify, which impact categories were prioritized, while target c of research question 1 was the assessment of the yearly external costs of an EU citizen, which has been calculated for the different methods in the paper. It was possible to identify that the prioritization of impact categories varies: Stepwise and Ecovalue assign over 50% of the per capita damages to climate change, while EPS and LIME 3 assign around 50% to mineral and fossil resource use, when assessing the yearly external costs of an EU citizen. Based on the results we concluded that practitioners should choose monetization methods with care and potentially apply varying methods to assess the robustness of their results. The results of this review could be used for research question 2 and 3 as a foundation to identify possible applications and to develop own methods.

sustainability

MDPI

Review

Comparison of Different Monetization Methods in LCA: A Review

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Received: 15 November 2020; Accepted: 13 December 2020; Published: 15 December 2020

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Abstract: Different LCA methods based on monetization of environmental impacts are available. Therefore, relevant monetization methods, namely Ecovalue12, Stepwise2006, LIME3, Ecotax, EVR, EPS, the Environmental Prices Handbook, Trucost and the MMG-Method were compared quantitatively and qualitatively, yielding results for 18 impact categories. Monetary factors for the same impact category range mostly between two orders of magnitude for the assessed methods, with some exceptions (e.g., mineral resources with five orders of magnitude). Among the qualitative criteria, per capita income, and thus the geographical reference, has the biggest influence on the obtained monetary factors. When the monetization methods were applied to the domestic yearly environmental damages of an average EU citizen, their monetary values ranged between 7941.13 €/capita (Ecotax) and 224.06 €/capita (LIME3). The prioritization of impact categories varies: Stepwise and Ecovalue assign over 50% of the per capita damages to climate change, while EPS and LIME3 assign around 50% to mineral and fossil resource use. Choices regarding the geographical reference, the Areas of Protection included, cost perspectives and the approach to discounting strongly affect the magnitude of the monetary factors. Therefore, practitioners should choose monetization methods with care and potentially apply varying methods to assess the robustness of their results.

Keywords: monetization; monetary valuation; LCA; weighting; environmental valuation

1. Introduction

Human activities lead to manifold impacts on the environment. Without a unique metric, it is difficult to decide which of the impacts are most severe and thus need to be prioritized to reduce overall environmental impact. Monetization of environmental impacts is one solution to this problem. The monetization of environmental impacts is the conversion of environmental impacts caused by releases of environmentally harmful substances or the use of natural resources to monetary units.

Today, monetization of environmental impacts is mostly used in such contexts as (environmentally extended or social) cost-benefit-analysis (CBA) [1].

While monetization can also facilitate the creation of markets for so-far non-marketed goods (e.g., for emission permits or payments for ecosystem services), this article focuses on determining economic value of impacts caused by environmental releases, leading to associated costs to society. Distinguishing between these kind of market prices of goods and economic values of impacts is important [2,3] and is sometimes confused (e.g., when scholars just use the emission permit price of CO_2 to determine the associated damages of the emission as in [4,5]). Leaving aside this theoretically inconsistent approach, different ways to determine costs for the reduction or exhaustion of natural goods (such as biodiversity or biotic and abiotic natural resources) are possible, notably the marginal damages of an emission to nature or as a second-best approach the marginal costs of reaching a political target (e.g., the Paris agreement) [6]. When determining marginal damage costs, a range of different approaches exist to convert biophysical flows into monetary units namely:

- The market price approach,
- The revealed preference approach,
- The stated preference approach.

Which approach is used depends on whether the flows are measured through their connection to a marketed good (market price approach), to their connection to a surrogate good (revealed preference) or whether the Willingness to Pay (WTP) is measured through surveys (e.g., contingent valuation) [6].

Monetizing life cycle assessment (LCA) results is one form of weighting in LCA [7] that enables the handling of trade-offs between impact categories. As Life Cycle Assessment (LCA) is one of the recommended management tools to quantify environmental impacts along the life cycle, there are movements to connect LCA with CBA [8] or to derive a single score through monetized LCA results as an index for ecological performance [9]. The advantage of monetized environmental impacts is that they can overcome the problem of trade-offs between the many impact categories an LCA usually analyzes. However, since it usually delivers a single score, it currently only conforms to the ISO standard (14040/44) [10,11] if it is used for internal communication in an organization. Since 1992, approaches to monetize LCA results have been developed, among which the first method was the Environmental Priority Strategies (EPS) [12]. Since then, many other methods have evolved, including updates of existing methods, such as the LIME method in Japan that has been updated three times [13–15], or the method by Vogtländer et al. [16] (Environmental-Costs/Value-Ratio- hereafter EVR) that has been updated several times. Thus, the questions arise: how do these methods compare, and does it matter which method is applied?

Up to now, six peer reviewed reviews address monetization in LCA, which differ in their scope and core conclusions. The first review by Finnveden [17] evaluated the EPS method and compared it to an approach developed in the first funding period of the External Costs of Energy (ExternE) project. Several calculation errors in the EPS method were found. The next review also by Finnveden et al. [18] concluded that if LCA results are monetized, all impact categories should be monetized using the same cost perspective (defined as monetization approaches such at abatement costs, damage costs or societal WTP). Based on this assumption, the Ecotax approach was developed that monetizes all impacts via environmental taxes. The next review by Ferreira et al. [19] compared EVR, Stepwise and Ecovalue08 quantitatively, identifying Ecovalue08 and EVR as having the lowest and highest monetary factors, respectively. However, they did not provide an in-depth analysis on the reasons why the results differ, but highlighted that there is research needed to compare the underlying mechanisms in more detail. The most comprehensive review up to date was published by Pizzol et al. [20]. It classified the different methods according to their monetization approach and their impact categories. It developed a scoring system to evaluate the different monetization approaches. They concluded that LIME2 and Stepwise are the most suitable methods because they provide explicit weighting between areas of protection (AoP) that are congruent to the impact endpoints in environmental LCA. They also perform a small quantitative comparison of different LCA monetization methods on a per-impact category basis, but with limited coverage of impact categories.

Two more recent reviews focused on the monetization of the impact category climate change [21] and monetization in LCA in the building sector [22]. Durão et al. [22] state that only mid-point monetization is compatible with EPDs, because they provide monetary factors that are compatible with the required CML-midpoint categories that must be included in an EPD. Dong et al. [21] identified that the emission values for global warming have similar values but still there is a spread of one to two orders of magnitude for the monetary valuation of CO_2. They identify which damages and endpoints are integrated into the damage assessment, which was also done in this review but for more impact categories.

While also being as comprehensive as Pizzol et al. [20], the novelty of this review is that a link between the monetary damage factors and the valuation of the respective AoP is established when the documentation allows. Through this analysis, it is possible to obtain an overview, which AoPs are linked to which impact categories and how the respective monetary valuation of the AoP is connected to the monetary factor for the emission. Further, we integrate the new LIME update, the new EPS update, the Trucost method, the MMG update and the EVR update, which have not been included in any review up to date.

Inspired by the review of different monetary values for GHG by Bachmann [3], monetary factors from the analyzed methods for the different impact category indicators are confronted, after adjusting them to allow for direct comparison. The comparison is done quantitatively and qualitatively and evaluates which kind of damages are included in the different monetization methods. Further, we analyze the distribution of the monetary values per impact category. Thus, the aim of this review is to perform a detailed quantitative and qualitative comparison of the currently used monetization methods in LCA on a general and on a per impact category basis, which has not been done in any review so far. Further, we establish a link between qualitative and quantitative differences of the applied methods. Based on this approach, the aim of the paper is to:

- Provide an overview of existing and relevant monetization methods in LCA;
- Determine criteria which influence the magnitude of the costs and why;
- Assess how the different monetization methods value and prioritize environmental damages of an average EU citizen;
- Identify overarching weaknesses within the impact categories;
- Outline overarching challenges to establish a roadmap according to which monetization in LCA can develop towards a consensus.

The paper is structured as follows: in the upcoming section, the qualitative and quantitative evaluation methods are described. Then the results are presented. First, the general qualitative results (Section 3.1) and then the quantitative results (Section 3.2) are displayed. The results chapter ends with Section 3.3 that integrates the qualitative and the quantitative findings per impact category. In Sections 4 and 5, the results are discussed and a conclusion is drawn.

2. Materials and Methods

In order to assess the different methods and their monetary units, four steps were taken (see Figure 1).

First, a key word search was performed in Scopus and Web of Science. Fifty articles were extracted that contain the words "LCA" AND "monetization" OR "LCA" AND "monetary valuation" published after 2013. In this year the last comprehensive literature review was closed [20].

In the second step, relevant papers describing the monetization methods were identified by screening all the extracted abstracts and papers. In those papers, more references that are relevant were mentioned so that 100 studies were studied in total. We classified the studies into case studies (43 papers), methodological publications (38 papers) and discussion/review papers (19 papers). Based on the discussion and review papers, an evaluation scheme for the qualitative assessment was developed. Nine relevant monetization methods were identified that fulfilled the following three criteria:

- They have an associated peer-reviewed case study released after 2012;
- They are the latest method update of the respective method (e.g., because LIME 3 was included, LIME2 and LIME were not);
- They have a strong connection to LCA and provide monetary factor(s) per impact category.

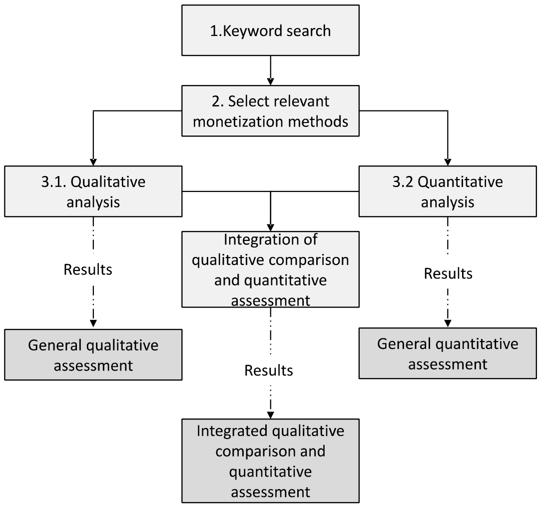


Figure 1. Steps to perform the review of the relevant monetization methods in LCA including linked results (displayed in a darker shade of grey).

Only one of the identified methods (Trucost) is a pure weighting method; the other methods have a coupled life cycle impact assessment and weighting. In the third step, the quantitative and qualitative comparison was performed. It was limited to those impact categories for which at least two of the nine monetization methods provide monetary values. The qualitative comparison was based on criteria deemed significant (see Section 2.1 for the specific criteria). They were adopted from review and discussion papers like [3,20,21], but were also based on environmental economic foundations [6,23].

The final step is the integration of the qualitative comparison and the quantitative assessment. The question that was guiding us in this final step is: can the quantitative differences be explained by the qualitative differences?

2.1. Qualitative Comparison

The criteria for the qualitative comparison are:

- The cost perspective and the type of market used when assessing damages (see Figure 2),
- The included AoPs,
- The use of equity weighting,
- The used discount rate,
- Whether marginal or non-marginal impacts are valued,
- The handling of uncertainty.

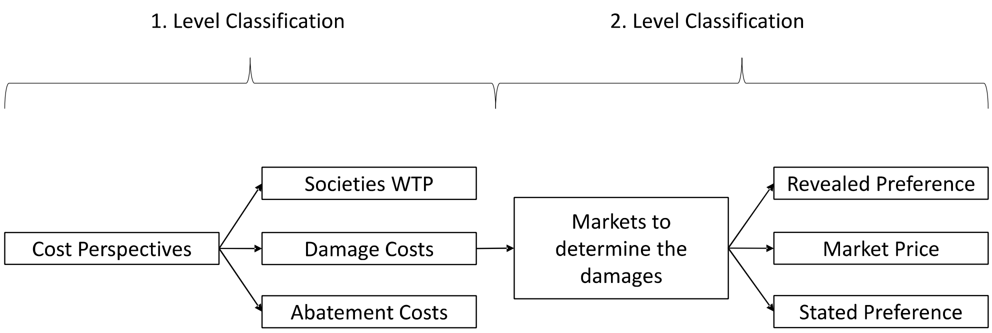


Figure 2. Cost perspectives and markets to determine damages that were used to analyze the monetization methods.

These aspects are assessed per method and per impact category. In the following, these criteria are explained in more detail.

Cost perspectives: The cost perspectives were classified as specified in Bachmann [6], i.e., damage costs based on individual's WTP or proxies thereof (here referred to as societies WTP), and abatement costs (tier 1). Damage costs are further distinguished according to the kind of market on which the monetary values are determined. The damage-oriented cost perspectives can be further classified into the market price approach, revealed preference approach or stated preference approach (tier 2). The classification of the cost perspectives can be seen in Figure 2.

Another important distinction usually made in environmental economics is the difference between willingness to pay (WTP) and willingness to accept (WTA). The WTA is up to seven times higher [24] than the WTP, due to the income effect [23,24], bad-deal aversion [25], the endowment effect or study design. While the difference between WTP and WTA has been studied intensively, there is a strong tendency in environmental economics to use WTP instead of WTA [23], especially because WTA often yields protest votes and higher values. Therefore, we pay specific attention to whether WTP or WTA is determined by the monetary valuation methods.

Moreover, all monetary valuation methods (irrespective of WTP and WTA) have been criticized for being wealth sensitive (obtaining higher results for wealthier people). This is an aspect that we analyze through (i) the covered geographical scope and (ii) the use (or not) of equity weighting.

AoPs: The different impact categories in LCA have a link to an AoP. For ReCiPe, for example, these AoPs are human health, resource scarcity and ecosystem quality [26]. LIME 3, by contrast, distinguishes four AoPs: human health, social assets, biodiversity and primary production [27]. Further AoPs distinguished include the built environment or labor productivity [28]. The methods establish different links and therefore include different damages: for example, some include the cause-effect chain from POCP (Photochemical Ozone Creation) to material corrosion and some do not. For the purpose of this study, we distinguish the following AoPs: human health, agricultural production, ecosystems, resources (abiotic and biotic resource production), working capacity, buildings and materials and human wellbeing (by which we mean the enjoyment of scenic beauty and positive aspects of enjoying an unpolluted environment beyond health). Following the links between impact categories and AoPs from ReCiPe (figure 1.1 in the ReCiPe report) and the Environmental Prices Handbook (figure 5 and table 25 in the Environmental Prices Handbook) [26,29], we determine which AoPs are included in which impact category of those methods that have the damage cost perspective. If a link is outlined in the Environmental Prices Handbook, it does not necessarily mean that it is quantified separately in its monetization (e.g., the Environmental Prices Handbook connects climate change impacts to human health ecosystems and resources, but uses the abatement costs to derive its monetary factor for climate change). Further, we compare how the AoPs for human health and ecosystems are monetized.

Equity weighting: Equity weighting is the practice in which environmental prices are corrected for their wealth sensitivity [30]. Depending on the used factors, environmental damages that make poorer people suffer can be scaled in a way that they are as valuable (or in extreme cases even more valuable) than damages to richer people. Generally, it is an ethical decision whether equity weighting is used or not. We assess it as a criterion, because it has an impact on the results, since methods with equity weighting will yield higher results.

Geographical scope: As explained earlier, monetary valuation of environmental damages is often wealth sensitive. In addition, a certain emission may be particularly damaging in a certain area (e.g., since an area is especially species rich and thus more highly valued). For these reasons, the geographical scope is analyzed.

Discounting: Discounting is used in economics to convert future costs into current value. Hellweg et al. [31] discussed discounting in relation to LCA and concluded that LCA impacts should not be discounted. Discounting of values associated with non-marketed goods is a topic which has always been controversial (see particularly the Nordhaus/Stern debate [32,33]). Note that cutting-off impacts that occur after a certain point in the future (e.g., 100 years from now) is also a kind of discounting [3,31]. Without taking a position in this debate, we recognize that a higher discount rate will yield lower results and vice versa. Therefore, we obtain the discount rate to have another criterion for possible differences in the assessment results.

Marginal/non-marginal analysis: Usually, in LCA, potential environmental impacts are assessed due to a small change in emissions (marginal). This is because LCA is usually used to assess environmental impacts of products that only bring about a small change. Since LCA is used more and more to assess bigger objects such as cities [34,35], territories [36] and organizations [37], its impact modelling and associated characterization factors need to be adapted [38,39]. The various impact assessment methods assessed in this review approach this topic differently. The chosen reference point has a significant impact on the assessed damages, which is also influenced by the choice between consequential and attributional LCA. Some methods just take the average damages of all emissions (in a certain impact category) globally; they determine the damages by average damages following this equation:

$$\text{Average Damages} = \text{AD} = \text{TD}/\text{q}, \tag{1}$$

where TD stands for total damages, and q for quantity of the respective emission (adapted from [40]). Others assess the damages of a small emission change (in that impact category), the mathematical description of marginal damage is (adapted from [40]):

$$\text{Marginal Damages} = \text{MD} = \text{Change in TD}/\text{Change in q}, \tag{2}$$

For some impacts such as climate change or water use, the marginal damages of the environmental impact increase as the background or reference environmental impact level rises. As a result, the marginal damages of those impact categories will be higher than average damages in our analysis. This can be seen in the rising damages for the next unit of CO_2 emissions for nearly all integrated assessment models as temperature increases [41], or through the conclusion that the CO_2-price should rise, because its marginal damages also rise with increasing emissions. Revesz et al. [41] highlight this effect for species richness when they conclude that most integrated assessment models underestimate the damages to ecosystems, since the value of the next species lost rises when species become scarcer as they go extinct. For other impact categories the cause-effect relationship is different and site-dependent, e.g., acidification.

Uncertainty: We analyze whether the methods provide uncertainty factors and which methods they use to determine uncertainty.

2.2. *Quantitative Assessment*

For the quantitative assessment all monetary units are converted to 2019 €. The conversion was performed according to the guideline of ISO 14008 [42]. First, the monetary values are inflated by the Consumer Price index, and then they are transferred to € by Purchasing Power Parities, where necessary. Some methods do not provide the reference year for the monetary units. Then we assumed that the €-unit was from the respective publication year. The following equation was used to correct for inflation based on [43].

$$X_t = X_b \times CPI_t/CPI_b \tag{3}$$

where X_t stands for the currency in the target year, X_b for the currency in the base year and CPI for the Consumer Price Index in the respective years t and b. The values were taken from different statistical offices [44–46].

After the correction for inflation, we converted currencies in the following manner:

$$\text{currency value}_{\text{target}} = \text{currency value}_{\text{original}}/\text{PPP}_{\text{original}} \times \text{PPP}_{\text{target}} \tag{4}$$

while PPP stands for Purchasing Power Parity. $PPP_{original}$ is the PPP value of the original currency and PPP_{target} is the PPP of the target currency (in our case €). PPP values were taken from OECD [47].

To compare all methods quantitatively, they need to relate to the same impact category indicator and unit. For some impact categories (e.g., climate change), all monetization methods use the same impact category indicator (CO_2-e), whereas for other impact categories these impact category indicators differ. Thus, the impact category indicators were converted to the same unit according to Owsianiak et al. [48] and Dreyer et al. [49]. First, a unit conversion factor (UCF) to convert unit a to unit b is determined:

$$UCF_{a \to b} = 1/CF_{a \to b} \tag{5}$$

Then the monetary factors are converted to unit b by the following equation:

$$MF_b = MF_a \times UCF_{a \to b} \tag{6}$$

MF_b stands for the monetary damage per unit of b, MF_a for the monetary damage per unit of a. $CF_{a \to b}$ stands for the characterization factor for the substance a that characterizes the substance in terms of b (i.e., how much b is as harmful as one unit of a, thus b is the reference impact category indicator). $CF_{a \to b}$ has the unit b/a. $UCF_{a \to b}$ stands for the unit conversion factor from the impact category indicator a to b. The applied CFs and derived UCFs are available in Table S1 in the supplementary material. For LIME3, we had to extract the impact category specific results. We did this using ReCiPe impact assessment for all impact categories except for mineral resources (CML) and water (Ecoscarcity) [50].

Where possible, ReCiPe [26] (Hierarchist perspective) conversion factors were used. For some impact categories other approaches were needed (e.g., USEtox [51], Impact 2002+ [52], methods from the ILCD recommendations for acidification [53] or TRACI [54]), because certain flows were not characterized in ReCiPe. For the impact categories mineral resources, fossil resources and toxicity, a different approach was taken. For mineral resources only antimony was assessed, because several methods did not conduct a characterization step, but just monetized at inventory level. To convert the values of the fossil energy carriers from mass units to energy units, the mass units were multiplied with the heating value from [55]. For the comparison of human toxicity units and for freshwater ecotoxicity, USEtox was used. Moreover, some methods did not provide information to which media the toxic substance was released, so assumptions were necessary. Further, only some impact assessment methods distinguished between cancer and non-cancer effects. In order to reach some comparative results anyway, the following approach was taken: first, the monetary damage per emission were converted to CTUh (Comparative toxic units) cancer or non-cancer. The characterization factors used were USEtox characterization factors and are given in CTUh per kg of emission. These were converted to DALYs (Disability Adjusted Life Years) to obtain a comparable € value per DALY, applying the conversion factors by Huijbregts et al. [56] (one CTUh non-cancer equals to 2.7 DALYs and one CTUh cancer equals to 11.5 DALYs). This yielded the monetary value per DALY. For the methods that provided values for CTUh-cancer and CTUh non-cancer separately, we assessed those values separately.

As a next step, the distribution of the values for the different impact categories were compared. To compare all impact categories with each other, we divided the obtained values per impact category by the mean of the impact category of all analyzed methods and displayed the distribution in a boxplot. To avoid the fact that methods which provide various values have a stronger weight in the mean of the impact category, we calculated an average monetary value for each method and derived the mean based on those average values (e.g., only the central estimate of the Ecovalue method was considered).

Then, we created a ranking of the quantitative results across impact categories (i.e., we assessed how many times a certain method had the highest, second highest value etc.). This enabled us to compare our obtained quantitative results with the formulated hypothesis in the general part to where we expect higher or lower values. This was done using the following formula:

$$\text{Score for method}_x = (\textstyle\sum \text{rank of impact category}_y \times \text{rank})/(\text{number of impact categories covered}) \tag{7}$$

Finally, we multiplied the obtained values with European normalization factors for 2010 [57], to see whether the methods obtained similar weighting between midpoints and to determine the extent to which the obtained monetary values differ across impact categories. For mineral and fossil resource use we took global values from [58] and assumed corresponding emissions per capita, as the European values only provided one value for mineral and fossil resources. Further, land use, terrestrial and marine ecotoxicity had to be excluded, because the European normalization did not fit to the impact category indicators we used for the unit conversion.

2.3. Integration of Qualitative Comparison and Quantitative Assessment Per Mid-Point Impact Category

As a final step, the quantitative and qualitative results were compared in detail. We identified the reasons why the values are different and connected our qualitative finding with our quantitative results. The integration of these results has a per impact category focus.

3. Results

In this chapter, the results are presented. Firstly, we describe the results of the qualitative assessment (Section 3.1) and the quantitative assessment (Section 3.2) per monetization method. To interpret the quantitative results, we partially draw on the results from the qualitative assessment. Then the results of the integration of the qualitative comparison are generated and the quantitative assessment is performed on the per impact category basis (Section 3.3).

3.1. Qualitative Criteria Based Assessment

Based on the established criteria in the method section we classify the assessed methods. An overview of all qualitative results is delivered in Table 1. This result supplements the specific analysis of all impact categories and how they are assessed. In the remaining part of this section, the results for all criteria for the different methods are outlined.

Cost perspective: Most methods use the damage costs as their cost perspective, whereas the EVR uses abatement costs and Ecotax uses societies' WTP. There are some exceptions to this, for example in the impact category global warming most methods use abatement costs. Even though seven methods mostly use damage costs, they use different approaches and techniques to determine the damages. The damages were determined through the market price approach, through the revealed preference approach (with techniques such as the travel cost method) or through the stated preference approach (with techniques such as choice experiments or contingent valuation). Find more details on the specific impact categories in the integration of the qualitative comparison and the quantitative assessment (Section 3.3). Another finding is that all methods that determine damage costs through contingent valuation use the WTP approach, but no method uses WTA.

AoPs: In the following, we will present the underlying monetary values that the different methods use for the AoPs human health and biodiversity/ecosystems. The final impact score depends on the applied impact assessment and on the monetized AoPs. Only the methods that use damage costs and monetize damages to certain AoPs are displayed in the following for the AoPs human health and biodiversity (Table 2).

Table 1. Overview of methods based on the criteria presented in the method part.

Method	Cost Perspective	AoPs [1]	Equity Weighting	Geographical Scope	Discounting	Marginal/non-Marginal	Uncertainty	Associated Publications
Ecovalue12	Damage costs/stated preference and market price	Divers for different impact categories, partially no documentation	Not clearly documented	Sweden	Unclear	Marginal	Provides min and max values	[59–63]
Stepwise 2006	Damage costs/Ability to pay	Human health biodiversity resources	Yes	Global	Unclear	Marginal	Discussed qualitatively	[64,65]
LIME3	Damage costs/stated preference	Human health, social assets (natural resources), terrestrial ecosystems, NPP [1]	No	Global with country resolution (G20 countries)	No	Marginal except for climate change	No	[13,27,66–69]
Ecotax 2006	Societies' willingness to pay	Not applicable	Not applicable- as it is not connected to an AoP	Sweden	Not applicable	Marginal	Provides min and max values	[18]
EVR (version 1.6)	Abatement costs	Not applicable	Yes but only applicable for human toxicity	Europe Netherlands [2]	Not applicable	marginal		[16,70,71]
EPS	Damage costs/mostly market price and revealed preference	Human health, bio productivity, biodiversity, abiotic resources, water, labor productivity	Yes, every human welfare loss is treated as if they were an OECD citizen	Global	0%	average	Provides uncertainty factors	[28]

Table 1. *Cont.*

Method	Cost Perspective	AoPs [1]	Equity Weighting	Geographical Scope	Discounting	Marginal/non-Marginal	Uncertainty	Associated Publications
Environmental Prices	Damage costs and abatement costs	Human health, ecosystems, buildings and materials, resource availability, wellbeing	No (but use only one DALY [1] for all European countries)	Europe	3%	Marginal	High, low and central value (but not for LCA [1] weighting factors)	[29]
MMG-Method	Damage costs abatement costs restoration costs	Human health-, biodiversity, agricultural production, resources	Not explicitly treated-	Europe, Flanders [2], global [2]	3%	CO_2, POCP [1], Water -marginal, for the other impacts unspecified	Provides a high middle and low estimate	[72–75]
Trucost	WTP [1] through ecosystem services (market price) or stated preference	Human health, ecosystem services based on de Groot [76], abiotic resources	Yes, DALYS [1] for all people are weighted equally	Global	Yes, but rate is unclear (for human health it is 3% as it is based on [77])	Eutrophication, abiotic resources acidification, smog, toxicity,: marginal, land use: average:	Qualitative description of uncertainty and limitations	[78]

[1.] Abbreviations are DALY: disability adjusted life year, AoP: area of protection, POCP: photochemical ozone creation, LCA: Life Cycle Assessment, WTP: willingness to pay, NPP: net-primary production [2] These geographical scopes are not considered in the evaluation.

Table 2. Underlying values for methods that monetize the AoPs biodiversity and human health.

Method	AoP Human Health	AoP Biodiversity
MMG	53,363.5 €$_{2012}$/DALY obtained from [79]	Based on NEEDS/restoration costs [80] value provided is: 46 €/PDF/kg 1.4-DCB-e, as the source is the same as the low value of environmental Prices it should be also 0.024 €$_{2015}$/PDF/m^2/yr
EPS	50,000 €$_{2015}$YOLL (Years of life lost)	NEX (normalized extinction of species) 56 billion € per year [81]
Trucost	Based on New Energy Externalities Development for Sustainability (NEEDS) project [77] Corrected for global average income (value not disclosed)	Through ecosystem services and Net Primary Production (NPP) (the biodiversity will actually measure the same as NPP if they are correlated); with data from de Groot et al. [76] (value not disclosed)
Ecovalue	Not explicit at least for acidification and eutrophication. They rather include wellbeing	Not explicit at least for acidification and eutrophication. They rather include wellbeing
Stepwise	Ability to pay 74,000 €$_{2003}$/QALY	1400 €/BAHY- as One BAHY is equal to 10,000 PDF/m^2/yr = 0.14 €$_{2003}$/PDF/m^2/yr
LIME 3	23,000 USD$_{2013}$ /DALY	4,100,000,000 USD$_{2013}$/EINES
Environmental prices	55,000 €$_{2015}$/DALY Mortality: 50,000 €$_{2015}$ to 110,000 €$_{2015}$ Morbidity 50,000 €$_{2015}$ to 100,000 €$_{2015}$	High: 0.649 €$_{2015}$/PDF/m^2/yr (based on high estimate of [82]) Central: 0.083 €$_{2015}$/PDF/m^2/yr (based on medium estimate of [82]) Low: 0.024 €$_{2015}$/PDF/m^2/yr (based on [80])

For the AoP human health three methods (Environmental Prices, Trucost, and the MMG method) orient their values based on the valuation in the NEEDS project [77,79,83] and thus are based on contingent valuation. The other values are in a similar order of magnitude, but have been obtained by different approaches. The EPS method uses the value of 50,000 € based on the average foregone income by an OECD citizen. Its approach is similar to AoP valuation in Stepwise, where the budget constraint (the amount of income of a US citizen in a year) was used to value a life year. The lowest obtained value is supplied by the LIME3 method that has been obtained by a choice experiment in G20 countries where people were asked how much additional tax they would be willing to pay to reduce DALYS, species extinction or net primary production (NPP). The Ecovalue method does not explicitly value human health and ecosystems, at least not for eutrophication and acidification [60]. For the impact category human toxicity the reference of the Ahlroth paper (the Espreme project [84]) is no longer accessible and could therefore not be compared here.

For the AoP ecosystems and biodiversity the studies by Ott et al. [80] (based on restoration costs which is a technique of the market price approach) and Kuik et al. [82] (based on the estimated ecosystem services) were used by two methods: the MMG method and the Environmental Prices. The MMG method used the study by Ott et al. [80], which is also used for the low estimate of Environmental Prices [29]. The EPS system uses another source: the study by McCarthy et al. [81] that determined the financial need to meet biodiversity targets. It can be interpreted as a kind of restoration cost approach. Trucost applies a correlation between species diversity and net primary production and bases its valuation on ecosystem services and values provided by [76], thus reducing the amounts of AoPs covered (biodiversity and NPP are treated as one AoP- as opposed to e.g., LIME3). This is in line with the approach by Costanza et al. [85], who estimated the value of the world's ecosystem based on ecosystem services (that they found to be worth ~twice annual global GDP per year as a minimum estimate or 33 trillion annually). For Trucost, we cannot compare the values since no AoP values are disclosed. The only comparable values are those from Kuik et al., Ott et al. and Stepwise as they have the same units or conversion factors are provided. The Stepwise values are based on the assumption that developed countries spend 2% of their GDP on conservation [64]. Based on this expenditure, Weidema et al. [64] estimate a certain value per species and a certain species density from which they derive their BAHY value. The other biodiversity units are very difficult to transform and compare, so that we cannot say which of them assign a relatively high or low value to biodiversity.

The information on the covered AoPs per impact category are displayed in the supplementary material (Table S2). The impact categories mineral resources, fossil resources and the toxicities were excluded, as the covered AoPs were only resources, human health and ecosystems, respectively.

Equity weighting: Regarding the question of equity weighting, all methods that apply a damage cost approach use equal factors for the value of human health, apart from the LIME3 method. This means health damages of richer people are weighted equally even if their WTP exceeds the currently used factor.

Geographical Scope: We have two methods that have Sweden as their geographical reference (Ecovalue12 and EcoTax) and four with a global focus (LIME3, STEPWISE, EPS and Trucost) and three with focus on Europe (Environmental Prices, MMG and EVR). It is not clear whether those factors should be applied to products purchased in the respective country, or whether regionalized monetization values are necessary for products that are produced in a wide range of geographical regions. The Ecotax method does not deliver guidance on this, while in the Ecovalue08 publication [59] it is stated that the values derived for eutrophication and acidification are only applicable for Sweden and need to be adjusted to be used for emissions in other geographical regions. The Environmental Prices handbook states clearly that its values are to be used for an average emission source in Europe, thus they cannot be used for the global supply chain. The same is true for the MMG method. The EVR method recommends using its monetary values for European emission sources, as different abatement costs arise in countries with different technological development levels.

Discounting: For the assessed methods, we found discount rates between 0% and 3%. Some methods do not state whether they use a discount rate (Stepwise, Ecovalue12). Trucost uses a discount rate, but does not disclose which it chooses. Environmental Prices and MMG use a discount rate of 3%, while EPS uses a lower discount rate of 0%. The authors of EPS do not disclose whether this is a social discount rate or the pure rate of time preference. If the 0% are applied to the social discount rate, this could in fact mean a negative discount rate (which is also discussed in environmental economics as future generations may be poorer [86]). The EVR and Ecotax do not apply a discount rate because they do not assess damages that occur at different points in time. Interestingly, none of the methods use a discount rate of over 3% as proposed by Nordhaus in the DICE model (4.25% [87]).

Marginal/non-marginal: Most assessed methods use marginal costs—in line with the usual assessment in LCA. However, LIME3 uses average damages for CO_2. An exception to this rule is the EPS. The EPS always uses average damages—it calculates the total damages of an emission flow and divides it by the total emissions.

Uncertainty: The approaches to handle uncertainty vary in the different methods. The MMG method, Ecovalue12, and the Ecotax method provide min-max estimates. The MMG method uses different cost perspectives for this. Stepwise and Trucosts only discuss uncertainty and limitations of their values qualitatively. The EPS system provides uncertainty factors to its values. Environmental prices deliver low, medium and high values, but these values are not delivered for LCA weighting factors. LIME3 does not provide an analysis of uncertainty.

3.2. Quantitative Assessment and Comparison of Different Methods

In Table S3 in the supplementary material, the different monetary values of the analyzed methods for the 18 impact categories that have been determined by the quantitative comparison are displayed. They can also be used as guidance in determining which impact category is covered by which method. In Figure 3, we show the quartile distribution of the different impact categories.

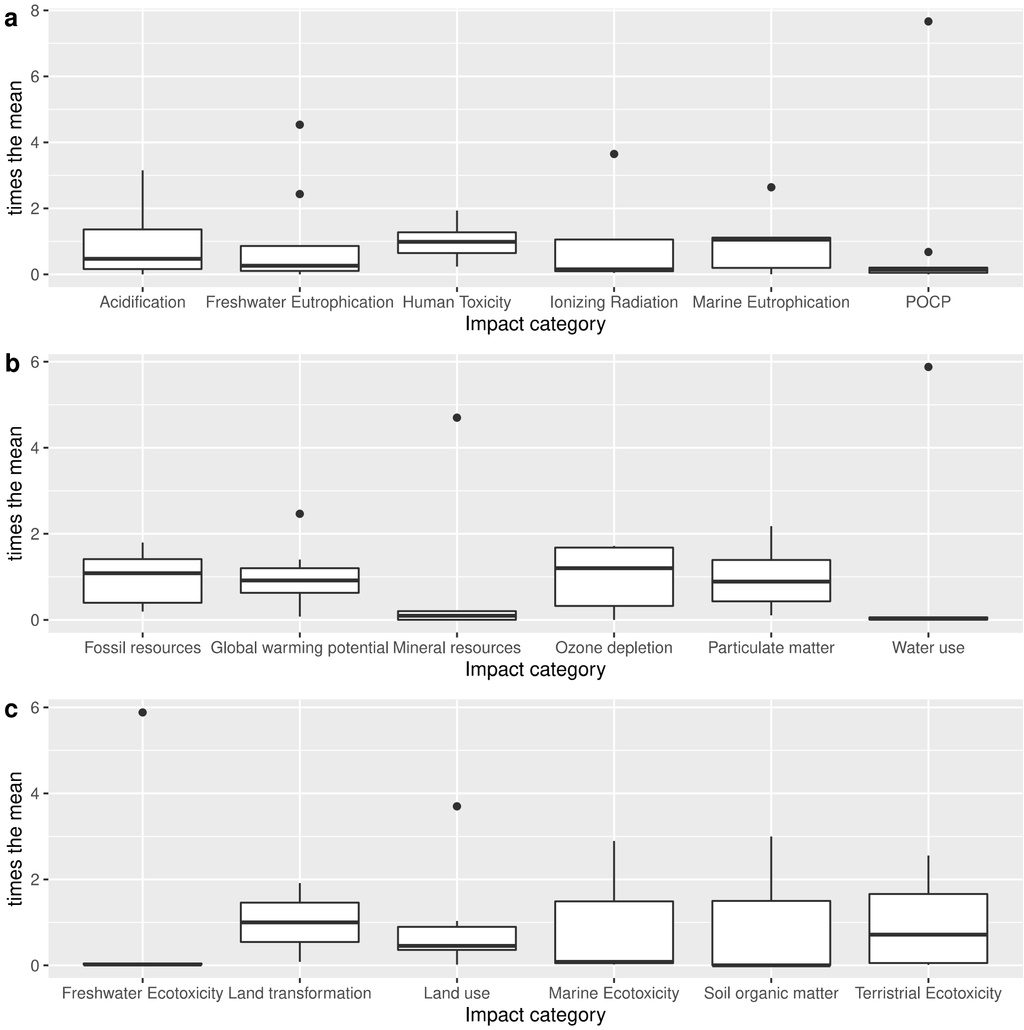


Figure 3. Distribution and variation of the different monetary values per impact category displayed in a boxplot: for the impact categories acidification, freshwater eutrophication, human toxicity, ionizing radiation, marine eutrophication and POCP (**a**); for the impact categories fossil resources, global warming potential, mineral resources, ozone depletion, particulate matter and water use (**b**); for the impact categories freshwater ecotoxicity, land transformation, land use, marine ecotoxicity, soil organic matter, terrestrial ecotoxicity (**c**). The dots are outliers, and the thick central lines represent the median, while the boxes and lines show the quartile distribution.

To be able to show the distribution of all values in one graph, we have divided all values by their mean. The values for human toxicity are nearly forming a normal distribution around its median, while eutrophication shows some outliers with a lot higher values and a fat tailed distribution to the right. Impact categories, where the underlying impact assessment is similar, such as photochemical oxidation, global warming, ozone depletion or methods with similar underlying data such as human toxicity or particulate matter, have a more even distribution, while nearly all impact categories linked to ecosystems show very high spreads in their values (freshwater ecotoxicity & terrestrial ecotoxicity, freshwater eutrophication, acidification). The impact categories that have a closer link to human health such as human toxicity and particulate matter show a normal distribution. To determine whether the methods with the damage costs perspectives converge, a brief analysis showed that these distributions only become slightly narrower if we exclude EVR and Ecotax and are still not-normally distributed (see Figure S1 in the Supplementary Material).

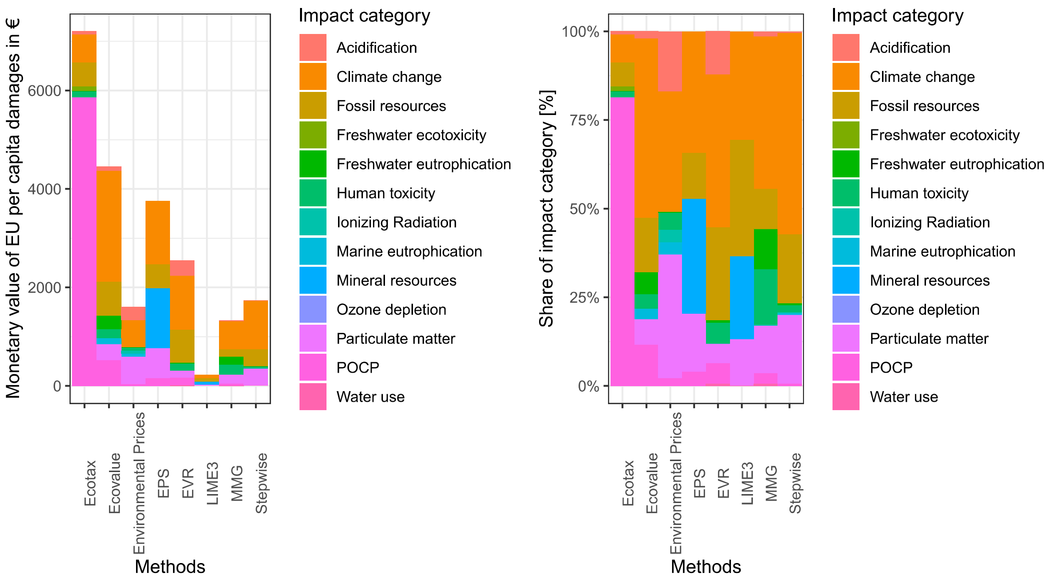
Now the results of the overall ranking are discussed. They show which method delivered the highest and lowest result across all impact categories (as described in Equation (7)). The ranks are not to be interpreted normatively; they just underline which methods delivered comparatively the highest and lowest monetary factors per environmental impact. The results are shown in Table 3.

Table 3. Methods and associated ranks.

Method	Rank
Ecovalue	1
Ecotax	2
Environmental Prices	3
EVR	4
MMG	5
Trucost	6
EPS	7
Stepwise	8
LIME3	9

Based on the ranking of the inter impact category comparison, the Ecovalue method had the highest values, followed by the Ecotax method, the Environmental Prices, EVR, the MMG method, and finally Trucost, EPS, Stepwise and LIME3. This means that the Ecovalue method had on average the highest values across all impact categories and LIME3 the lowest. The interpretation of the magnitude of the values are compared on a per impact category basis in Section 3.3. However, when we consider the broader picture, it seems that the geographical scope of the methods plays an important role: both approaches that cover Sweden (Ecovalue and Ecotax) have the highest value, while the three following methods (Environmental Price, EVR and MMG) have European coverage. The methods with the lowest values (Trucost, EPS, LIME3 and Stepwise) have a global scope. Interestingly, a slightly higher value for human health as provided by Stepwise does not play a significant role in increasing the results, apart from those impact categories that are strictly connected to human health such as particulate matter. Another interesting finding is that the methods that assess abatement costs do not show strictly higher or lower values than the other methods.

In the final step of the quantitative assessment, we multiplied the obtained monetary values of the emissions with the emissions and resource demands of an average EU citizen. The results are displayed in Figure 4.



(**a**) (**b**)

Figure 4. Monetary damages per capita and year (average EU citizen) obtained by the different LCA monetization methods in € (**a**) and the respective share of the total value in [%] of the different methods (**b**).

It is visible that the monetary damages associated with per capita emissions show a wide range. The Ecotax method obtains a value of nearly 8000 € of damages associated with average EU per capita domestic emissions and resource demands in 2010. The total per capita damages yielded by the LIME3 method were 224 €. In line with the observation in Ecotax and Ecovalue yielded the highest results. Discounting may also play a role as the obtained monetary value of Environmental Prices and MMG, that use a 3% discount rate, are the 6th and the 7th lowest, respectively. EPS and Stepwise have a higher value for the damages per capita, as impact categories in which they provide higher values (e.g., climate change and fossil resources) constitute a large share of the overall damages.

However, not only the associated damages of the total damages per capita vary: also, the shares and the weighting in between the impact categories differ, as can be seen on the right side of Figure 4. For the Ecotax method over 80% of its damages occur due to POCP emissions. Climate change is rated consistently quite high over all methods (with 7% as the minimum for the Ecotax method). Freshwater eutrophication, marine eutrophication, ionizing radiation, water use and acidification are rated quite low (under 10% with an exception for the MMG method for freshwater eutrophication (11%) and acidification (17% for Environmental prices and 12% for EVR)). In addition, the valuation for human toxicity and particulate matter varies between 1% (Stepwise) to 16% (MMG) and 5% (EVR) to 35% (Environmental Prices), respectively. For some values, the obtained weighting is very alike though: for ozone depletion all methods provide a weight of under 1% and for freshwater ecotoxicity under 2%.

Impact categories that have a strong link to biodiversity such as eutrophication, acidification and ecotoxicity have only very low shares of the monetized damages. However, other impact categories with a strong link to biodiversity such as land use and marine and terrestrial ecotoxicity had to be excluded because of limited data availability or compatibility with the impact category indicators. An inclusion could change this observation.

3.3. Integrated Qualitative Comparison and Quantitative Assessment

In the following, the results for the individual mid-point impact categories for which more than two methods provide a quantitative assessment are presented. The qualitative and quantitative results per impact category are discussed together. The values for the method Trucost had to be excluded as the associated values are not published.

3.3.1. Climate Change

Eight methods provide monetary values for climate change. The lowest value is given by LIME3 (under 0.75 ct per kg or about 7.5 euro per ton). The highest value is provided by Ecovalue maximum estimate which is 50 ct per kg of CO_2-e, resulting in an emission damage of 500 € per ton. The values are displayed graphically in Figure 5. Some methods (Ecovalue and MMG) provide min and max values so that several values per method are visualized.

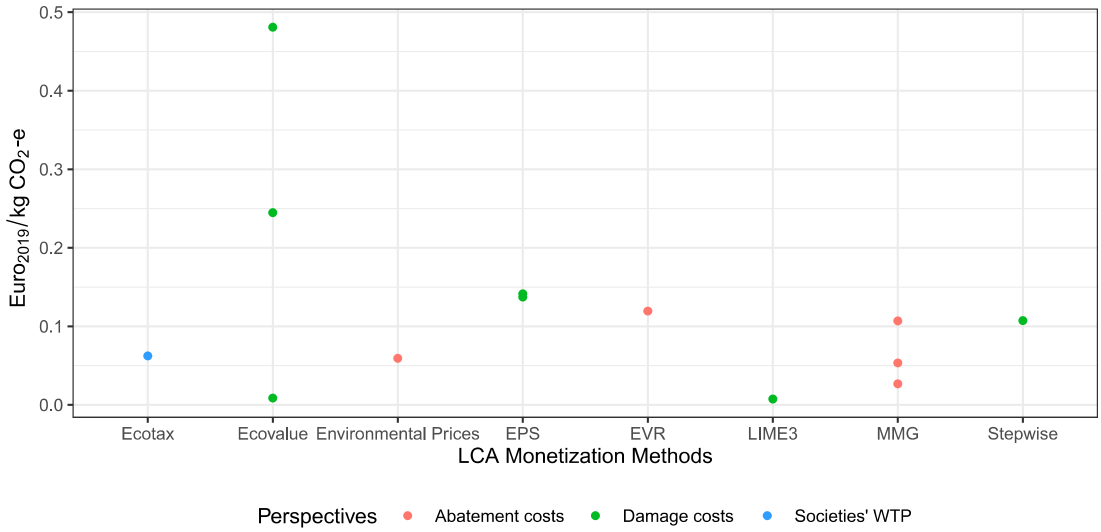


Figure 5. Monetary values for the impact category climate change.

The values spread over two orders of magnitude, which is in line with the results by Dong et al. [21].

Ecovalue08 used the values by the Stern Review and the FUND model [33,88]. It is not clear which value Finnveden and Noring [62] use in the Ecovalue12 update as they state several references ([89–92]) but not a clear calculation method. As these are all publications on integrated assessment models (partially the FUND model and the DICE model), we assume therefore that it contains damages to agriculture, human health and ecosystems. Overall, the four methods Ecovalue, EPS, LIME3 and Stepwise use damage cost and integrate damages to/on human health and biodiversity.

The lowest values are derived from LIME3 and the low estimate of the Ecovalue method (under 10 €/ton). The fact that the value for LIME3 is so low is not surprising as it has relatively low AoP valuation for human health (see Table 2). EPS covers mainly damages that are otherwise associated to integrated assessment models such as increased heat stress. The reduction of working capacity consists more than half its value (as stated by Dong et al. [21]) and is responsible for its very high value.

For the three methods using the abatement cost approach (MMG, Environmental Prices and EVR), the results do not differ significantly. The MMG method mainly uses abatement costs but also other cost calculation approaches. Environmental Prices uses abatement costs. EVR uses abatement costs to reach the Paris Agreement, e.g., needed investment in offshore wind parks; this is maybe why its abatement costs are slightly higher than the other methods.

One aspect that should be noted is that the MMG method uses different CO_2-prices for its buildings depending on when they are emitted in the building's life cycle. This is not in line with the ISO 14067 [93], which does not allow to account emissions with different emission times to be accounted for differently. This is challenging to combine with monetized CO_2-prices in integrated assessment models, as these are usually time differentiated and use a discount rate.

3.3.2. Acidification

Seven of the assessed methods published damage values for acidification and their quantitative comparison are displayed in Figure 6.

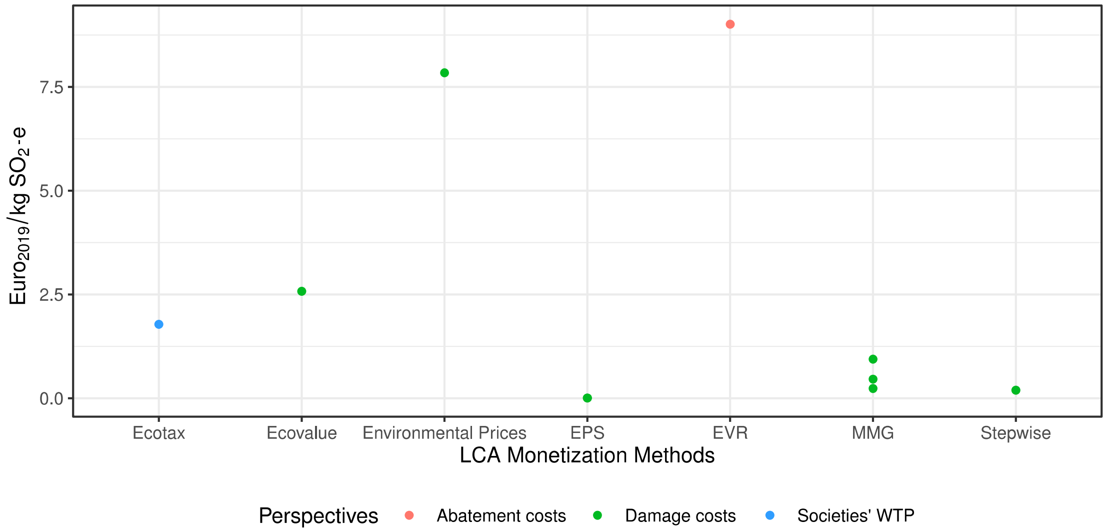


Figure 6. Monetized values for the impact category acidification.

With values between 0.01 € per kg SO_2-e (EPS) and 9 €/kg SO_2-e (EVR) we have again a spread of two orders of magnitude. For EVR, the costs are the costs associated to ultra-low Sulphur content diesel, which cannot be compared to the damage costs. It is noteworthy that in the impact category acidification the abatement costs seem to be higher than the damage costs. The Ecotax method considers taxes on Sulphur content in fuels, which would be too weak a financial incentive to implement the technology suggested by the EVR as it costs more than three times the tax.

Four methods in the impact category acidification use the damage cost approach. Three of these have values under 1 € /kg SO_2-e (EPS, MMG and Stepwise), while the Environmental prices seems to be an outlier with a higher value.

EPS includes all damage endpoints for acidification (species, corrosion and ecosystems), but still has the lowest value. The Stepwise method only includes damages to ecosystems. This explains why its impact is so low [65]. For the EPS, the low values can be partially explained by average, not marginal, damages. The Ecovalue monetary value is higher than the value for Stepwise, MMG and EPS. This is not surprising: their geographical scope is Sweden, so that it is likely that the damages are valued higher due to the higher income of Swedish citizens. Since the values are derived by contingent valuation, they also include wellbeing.

The Environmental Prices value is presumably that high because it provides a higher estimate of biodiversity and ecosystem services (because its values are based on Kuik et al. [82], and not on Ott et al. [80]). Further, the LCA weighting factor is equal to the medium estimate in the handbook that also includes building damages which are based on the NEEDS project [83]. The building damages for Environmental Prices exceed the entire damages of Stepwise and EPS (0.6 € per kg SO_2 only for buildings [29]). It should be noted that the values provided for SO_2 and its associated damages vary by an order of magnitude in the Environmental Prices Handbook (compare Tables 1, 3, and 33 in the Environmental Prices Handbook [29]).

3.3.3. Ozone Depletion

Six methods assess the monetary damages of ozone depletion. For ozone depletion, the results have a spread of one order of magnitude if we exclude the zero value. EVR has the value of zero, while three of the other methods have a value of just over 100 € per kg of CFC-11 (see Figure 7). We can see in the plot that the values are very similar, apart from the MMG method low and central estimate, EPS and the EVR method. That is because the damages are quite high, while the abatement costs are very low. The lowest values are yielded for the abatement costs, since the EVR method concludes that it does not cost anything to prevent CFC-11 emissions. For the damage costs, Stepwise and Environmental Prices have the highest values.

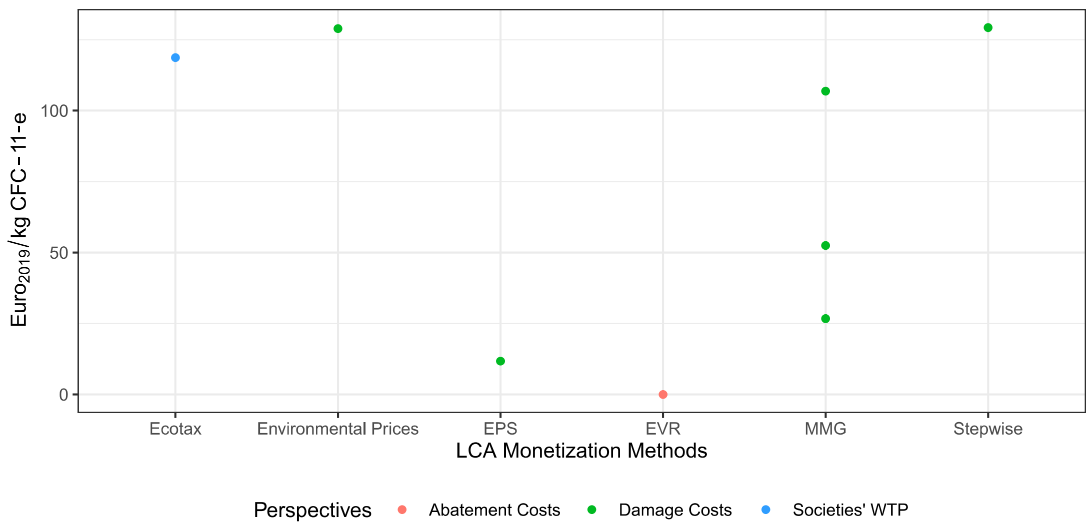


Figure 7. Monetary values for the impact category stratospheric ozone depletion.

For the value obtained by Environmental Prices, damages to human health and ecosystems are included. For the EPS, the impact of the pure ozone depletion value is 11.21 €/kg. EPS only assesses health damages here (skin cancer and sight problems), and no damages to ecosystems. Stepwise includes human health impacts and impact on working capacity [65]. The MMG method considers health damages, damages related to agricultural damages and material damages based on the old shadow prices Handbook (2010) and the ReCiPe version of 2008 [94]. Stepwise had slightly higher associated human health values and includes working capacity. This can explain the higher values of Stepwise. However, the EPS has a high associated value for human health, but yields the lowest value for stratospheric ozone depletion, which is inconsistent. One reason for this inconsistency could be due to marginal and non-marginal impacts; because the EPS provides non-marginal values and all other methods use marginal analysis. In addition, the EPS does not include ecosystem nor working capacity damages. Another aspect that influences the magnitude of the results is the time perspective. As the time perspective of ozone depletion is long, the values are sensitive to the underlying reference scenarios.

3.3.4. POCP

Eight methods assess the environmental impacts of POCP (see Figure 8).

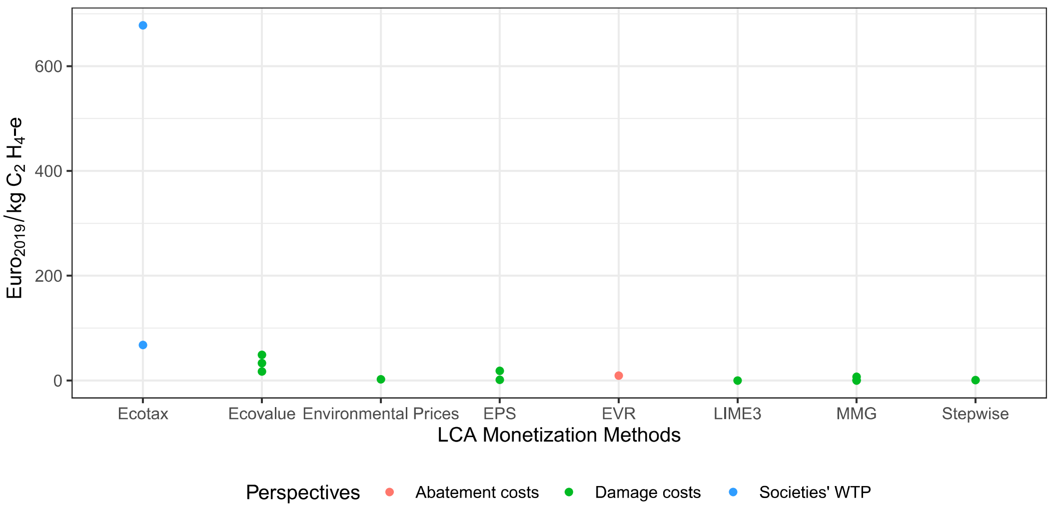


Figure 8. Monetary values for the impact category photochemical oxidation in ethylene equivalents.

The range of the POCP values is high. We have an outlier (which can also be seen in the respective boxplot in Figure 3) with the value provided by the maximum value of Ecotax (763 €/kg), potentially attributable to its method and its geographical scope. The lowest values are provided by the low estimate of the MMG method, which provides a value of 0 €. The basis of the Ecovalue data, which has the highest value after the Ecotax method, is difficult to trace back: Finnveden and Noring [62] deliver no new sources, while the source given by Ahlroth and Finnveden [59] (that published the previous Ecovalue model), the Methodex Project website, is no longer accessible nor existent on the internet [95]. However, Ahlroth and Finnveden [59] state that the Methodex project includes POCP damages on human health (valued through contingent valuation) and agricultural yield losses that are measured through market price. The third highest value is provided by the EPS which includes crop loss, wood loss, asthma and increased COPD cases, but its value is just over 1 €/kg of C_2H_4-eq. The Environmental Prices approach only includes chronic health impacts of POCP and is even lower than the values delivered by the EPS. Stepwise has the second lowest values even though it includes damages to four endpoints and has a very high underlying value of human health. EVR also provides low values since avoiding POCP emissions seems relatively cheap. The MMG method has higher values than Stepwise and Environmental Prices, but lower values than EVR and EPS. It bases its valuation on the ExternE project and on the program EcoSense, therefore accounting for public health mainly, but also for crop damages [96]. Details of how values are corrected for inflation etc. are not provided. LIME3 has the lowest obtained value. It only includes damages to human health, and the respective valuation of the AoP human health is relatively low compared to the valuation of the other studies. Overall, the results are not completely explainable related to AoP valuation. Different impact assessment models, the applied unit conversion factors (as the different monetary values were provided as NOx-e, NOVOC-e, C_2H_2-e and had to be converted with unit conversion factors documented in Table S1 in the supplementary material) and the mechanism of POCP creation that is geographically very varied can be elements explaining the observed differences.

3.3.5. Eutrophication

The impact category eutrophication is covered by seven of the analyzed methods, while three methods (Ecovalue, Environmental Prices, EPS) also address marine eutrophication explicitly, while others do not distinguish between the two and only assess aquatic eutrophication (like Stepwise). Since the distinction is not always available, differences in the results can also occur due to different scopes of the impact categories. The results range from 64 €/kg PO_4-e (upper bound MMG-method) to just over 1 ct (EPS) (as visible in Figure 9).

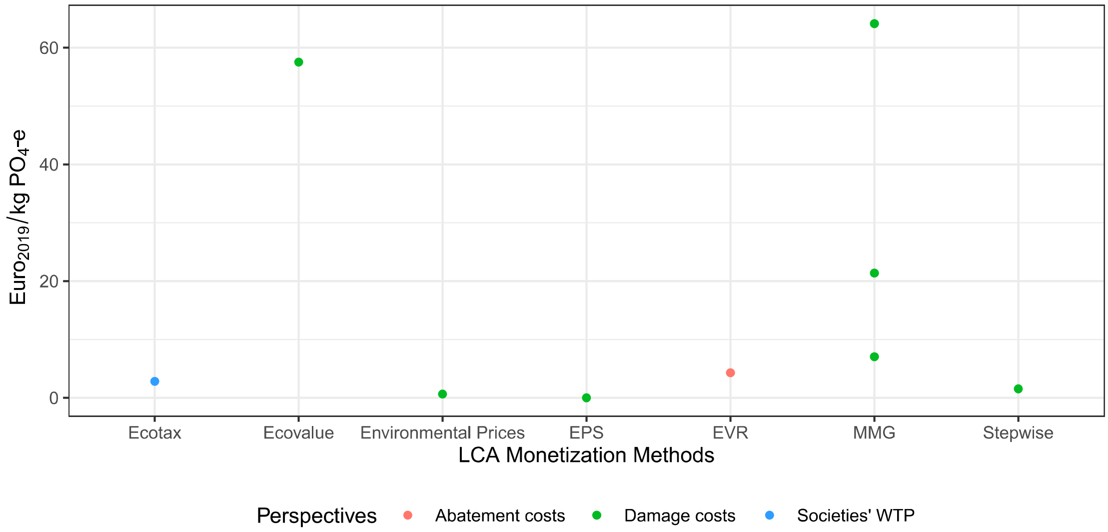


Figure 9. Environmental costs of phosphate-equivalent emissions according to different monetization methods.

The MMG method's highest value is the highest estimate. One reason for the high value of the Ecovalue method can be its development context: Swedish peoples' WTP was estimated to reduce eutrophication. The values from Finnveden and Noring [62] were not updated and are based on contingent valuation studies and travel cost methods from Ahlroth [60]. Since the GDP per capita in Sweden is higher than in other countries, it is also logical that the WTP is higher, since the WTP is wealth sensitive. Already Ahlroth et al. [59] noted that the Ecovalue values are a lot higher than the Ecotax values (that are based on taxes on nitrogen in fertilizers). They concluded that the current environmental tax on eutrophic substances is too low and actually lower than what people in Sweden would be willing to pay. The Ecovalue study also includes the recreational value a lake delivers if it is not eutrophic (and therefore includes wellbeing beyond health). The inclusion of agricultural and ecosystem damages is not explicit, because the assessment in the Ecovalue method does not ask for the reduced biotic production nor for a link from emission to species but for WTP for an increased sight depth. However, stating that species richness is included in this assessment would imply letting the survey participants guess the cause-effect chain between sight depth and species richness.

The MMG method provided values for Eutrophication (damage costs and prevention costs) [75]. The overall range of the results is very high, especially the damage costs that are based on a willingness to pay study to reduce eutrophication in the Baltic sea [97]. Therefore, the values relate to marine eutrophication rather than to freshwater eutrophication. Interestingly, both methods (MMG and Ecovalue) that include contingent valuation deliver the highest values.

The lowest values are yielded by the EPS, which analyses the average impact based on global phosphorous flows [98] and their impacts on species richness. Additionally, the impact on agricultural productivity is covered. Environmental Prices provided the second lowest value. Its assessment only includes effects on biodiversity (PDF) based on ReCiPe.

Stepwise obtained a higher value than Environmental Prices; it also includes only damages to ecosystems and is based on the impact assessment of EDIP 2003. Stepwise values are presumably higher than Environmental Prices because its underlying valuation of biodiversity is also higher (see Table 2).

Only three methods assess the impact category marine eutrophication explicitly. The Environmental Prices method assesses the values based on ReCiPe 2013 and its according PDF with the method by Kuik et al. [82] (transferred its approach to water). The monetary values in €/kg of N-e are displayed in Figure 10.

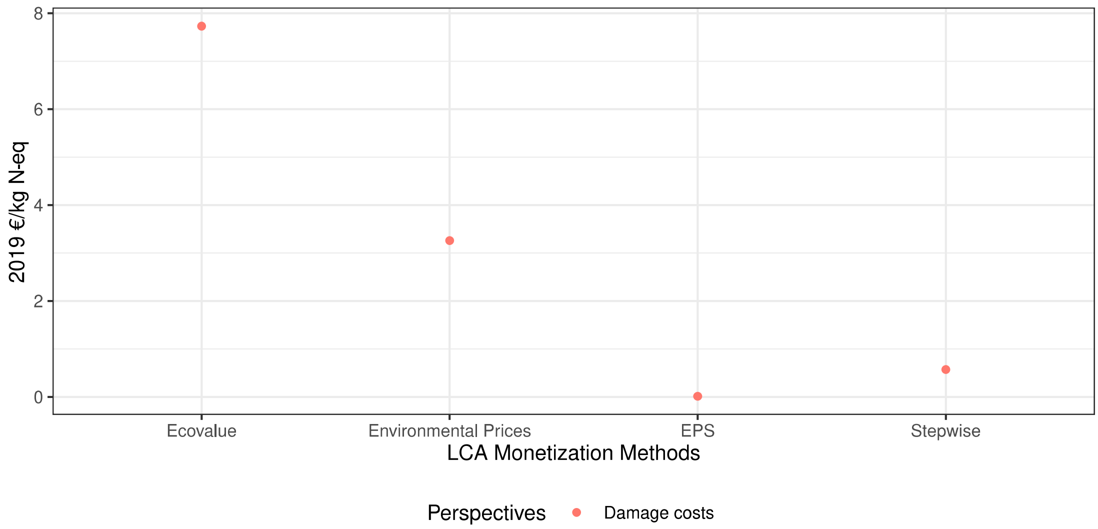


Figure 10. Monetized values for the impact category marine eutrophication.

Ecovalue assesses the value based on a contingent valuation study and travel cost method based on the same approach as above. Again, the high GDP in Sweden might explain the high values for the Ecovalue method. Stepwise includes, as mentioned above, only the damages to ecosystems and has a quite low estimation of the associated values (as no distinct value for marine eutrophication is provided, only a value for generic aquatic eutrophication). However, Environmental Prices included only damages to ecosystems, but yielded a higher value than Stepwise, which is in conflict with its lower biodiversity valuation compared to Stepwise. The EPS provides a value for marine eutrophication impacts and considers damages to agricultural production as well as ecosystems, and provides the lowest value as for freshwater eutrophication.

3.3.6. Particulate Matter

Seven methods cover the impact category particulate matter. The highest value is provided by EPS (160 € per kg $PM_{2.5}$-e) and the lowest value by LIME3, which is just over 7.7 € per kg $PM_{2.5}$-e. Therefore, the results spread again over two orders of magnitude. As particulate matter has mostly impacts on human health, and all methods that use damage cost have similar valuation of human health, it is surprising that the values are so different. The values are displayed in Figure 11.

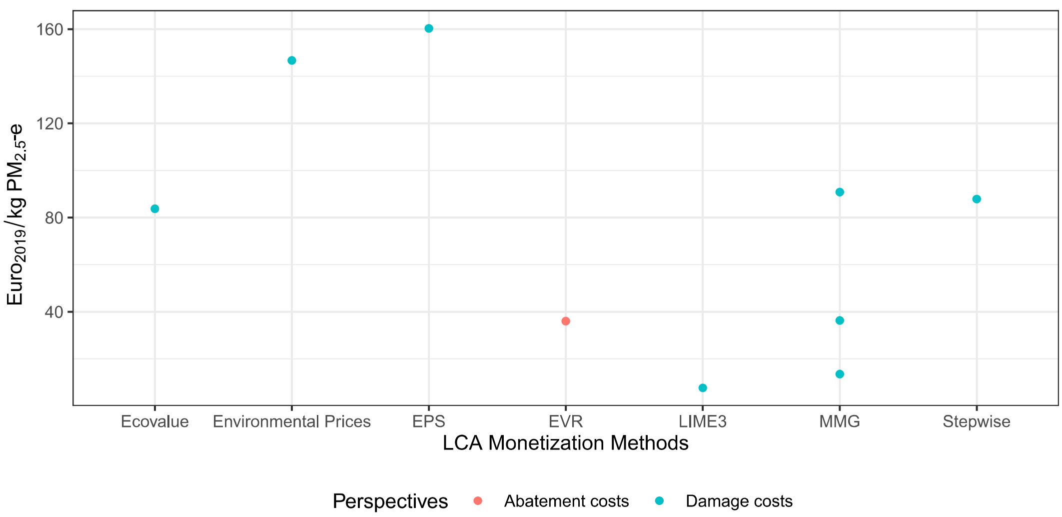


Figure 11. Monetary values for the impact category particulate matter in $PM_{2.5}$-e.

It cannot be precluded that the conversion factor is responsible for the magnitude of the Environmental Prices value: we used TRACI conversion factors [54] (as documented in Table S1) because ReCiPe did not provide a characterization of PM_{10}-e (which is the impact assessment indicator of Environmental prices, but all impact assessment methods should have the unit $PM_{2.5}$-e). The CFs used by TRACI suggest that the impacts of $PM_{2.5}$ are about four times as harming as PM_{10}. While the method by van Zelm [99] supports this order of magnitude, Gronlund et al. [100] suggest that the impact of $PM_{2.5}$-e is less than twice as harming as PM_{10}-e. In addition, the value for EPS is very high, which is in line with its value for human health. The same is true for the Stepwise values, which are lower despite including reduced working capacity. The EVR value is low, suggesting that it is cheaper to abate particulate emissions than to endure their associated damages. Ecovalue's slightly higher value might be due to the Swedish geographical reference area, with a relatively higher GDP. In addition, here the values are converted from PM_{10}-e. The background data of the Ecovalue method are those used by the Shadow Prices Handbook [101] before the Environmental Prices update. It only includes damages to human health. If PM_{10} was assumed to be as harmful as $PM_{2.5}$, then Stepwise would have the second highest value, because the values by Environmental Prices and Ecovalue would be lower than the Stepwise value. Environmental prices also included material damages of particulate matter. The differences between the methods cannot be deduced only by the different valuation of human health. Rather it is reasonable to assume that they are caused by different impact assessment and geographical reference, as particulate emission's damages highly depend on the exposed individuals and therefore on transport models and population density.

3.3.7. Ionizing Radiation

Four methods cover the impact category ionizing radiation: Environmental Prices, Stepwise, EPS and the MMG method (displayed in Figure 12).

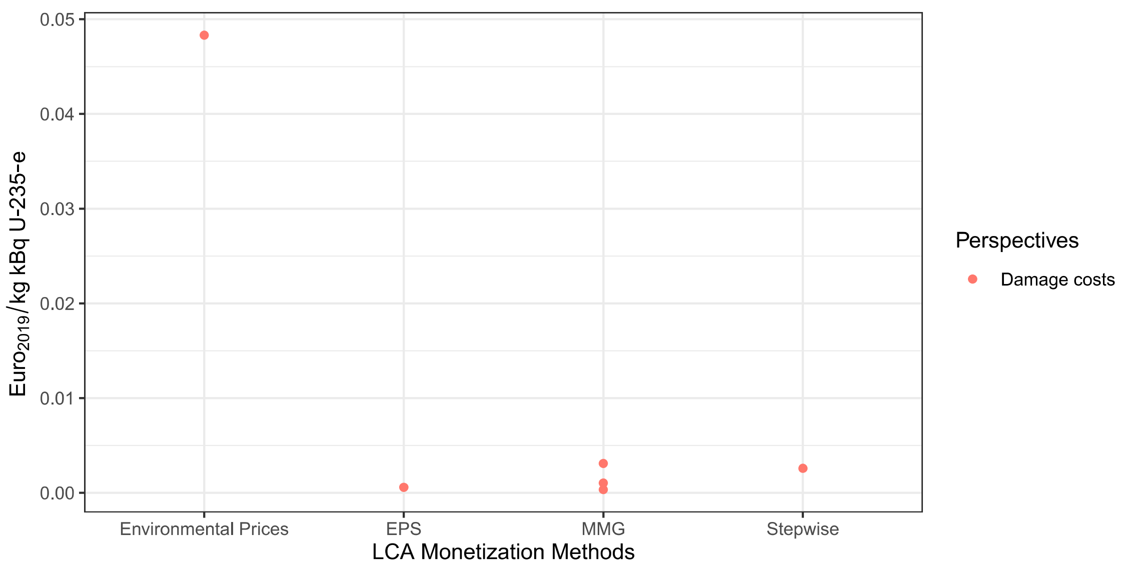


Figure 12. Monetary values of the impact category ionizing radiation.

The provided results per kg kBq U235-eq are very different. The max-value MMG method is 20 times lower than the value for Environmental Prices. This difference is not really explainable as both methods are based on ReCiPe; The MMG assessment is based on ReCiPe 2008 [94] and Environmental Prices on the 2013 ReCiPe update [102]. All assessed methods use the damage cost approach and consider human health damages only. Stepwise uses the Impact 2002+ impact assessment method and EPS uses data based on cancer incidences based on its former version. As all methods have a similar magnitude in their human health valuation, the high spread of the results is not explainable by our qualitative criteria. Owsianiak et al. [48] found large differences in the impact assessment of ReCiPe and Impact 2002+ for the impact category ionizing radiation. While this can explain the difference between the Stepwise and Environmental Prices methods, it does not help understanding the difference between the Environmental Prices and MMG methods.

3.3.8. Mineral Resources

Four method delivered monetary values for mineral resources, but they are very challenging to compare as many different approaches were taken. The external costs per unit of Sb-e (antimony equivalents) range from just under 20,000 €/kg to 0 € from the MMG low estimate which assumes that all external costs are already included in the market price (see Figure 13).

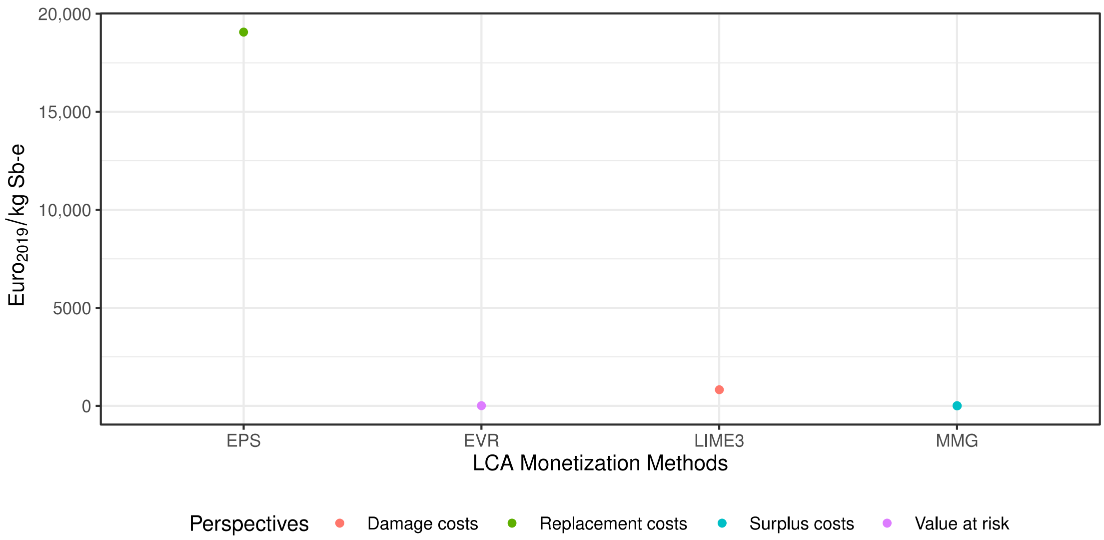


Figure 13. Quantitative comparison of the impact category mineral resources.

Environmental Prices discusses the challenge of resource scarcity but does not provide quantitative weighting factors for LCA. It states that its approach is based on the Hotelling rule [103]. Overall, the method's approach to monetize scarcity does not fit well in the scope of our cost perspective classification from Section 3.1, apart from LIME3 that has been obtained by stated preference studies and therefore is classified as determining damage costs.

Even though the EPS value seems to be an outlier compared to the other methods, the LIME method also provides a value of over 800 €/kg of Sb-e, while the value for EVR and the MMG high estimate are around 10 €/kg Sb-e. Therefore, the results are spread across five orders of magnitude (excluding the zero value for MMG low) and have so far the widest range.

EPS takes the replacement cost approach (so how expensive will it be to obtain a material if no further ore of it is provided), while EVR works with a quantification of value at risk/supply risk. The MMG method with the lowest value used the ReCiPe approach that is also based on surplus costs.

The high associated values for EPS are underlined by the fact that for the EPS scheme the AoP resources seems to be the most important one [28]. The EPS method assigns a high value to the external costs of ores and minerals. These values dominate all versions of EPS.

LIME3 calculates external costs of mineral resources through land use change associated with mining, which is again a different approach compared to the other presented methods.

The MMG values (0–6.65 €/kg Sb-e) are similar to those from EVR (8.1 €/kg Sb-e). As all the methods have very different approaches, it is not surprising that their monetary values also differ.

3.3.9. Fossil Resources

For fossil resources, seven methods provide monetary factors. The values were scaled to the unit €/MJ. All methods obtain a value of under 3 ct per Mega joule (see Figure 14).

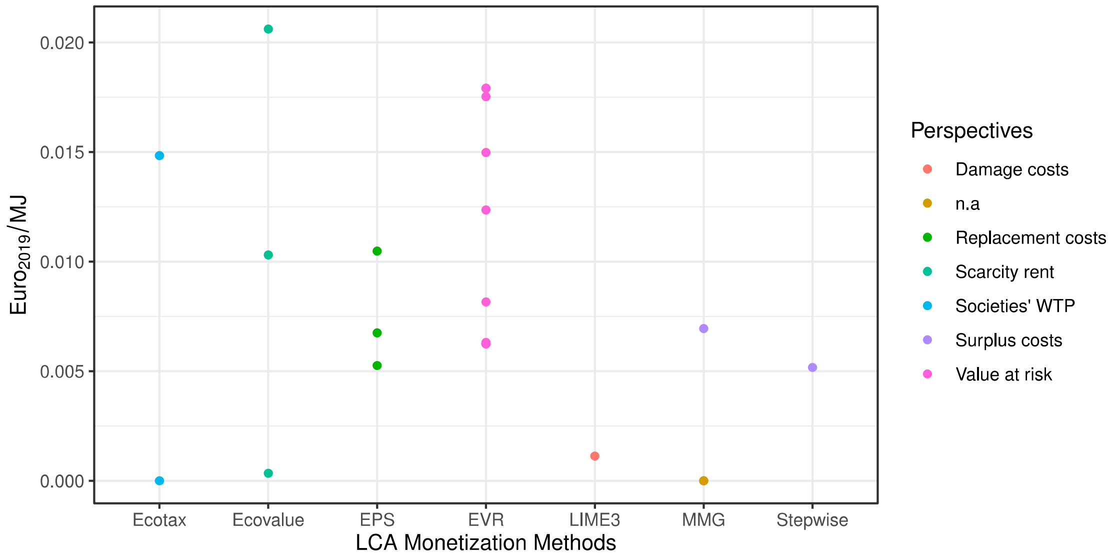


Figure 14. Monetary values associated with the use of one MJ of fossil energy for the impact category fossil resources.

As for the cost perspectives for mineral resources, the taken approaches are difficult to classify with the classification that we have used so far. While the Ecotax method is based on taxes for mineral resource use, which is low, the Ecovalue approach is based on resource rent data.

For the Stepwise method, it is difficult to determine where the values come from. The value used here is from Pizzol et al. [20], while it seems that in Weidema et al. [65] the external cost of European fossil fuels with Stepwise is zero. Therefore, the approach is unclear. It is based on the approach by IMPACT 2002+ [52], that is based on the Eco-indicators surplus cost approach, which is close to the approach used in ReCiPe.

The MMG method assumes, for the central and low value, that resource values are internalized in prices and therefore applies a value of zero. The high value includes the approach in the Ecoindicator 99, which is the surplus costs approach, and military costs to secure resources. However, EPS assesses how costly it would be to produce the same amount of energy from renewable resources (converting wood to oil by Fisher-Tropsch-process or charcoal and biogas production), while EVR has a method to determine value of risk. Both assessments are not provided by values per MJ, but by value per mass of different energy carriers. This is why both methods deliver a variety of values that were obtained by a conversion through caloric values of different fuels (see Table S1 for more information). LIME3 uses the same approach for mineral and fossil resources: it calculates the overexploited economic values by current resource production as an economic externality of mineral and fossil resources use based on the El Serafy's use cost approach.

Because of those very different approaches, (none is used twice apart from the MMG high estimate and Stepwise), it is surprising how narrow the distribution of the values is (even though the results range again between two orders of magnitude if the zeros are left out).

3.3.10. Water

Five different methods include the impact water use. The costs range from costs under one cent per cubic meter to over 30 € per cubic meter (see Figure 15) and thus range over three orders of magnitude.

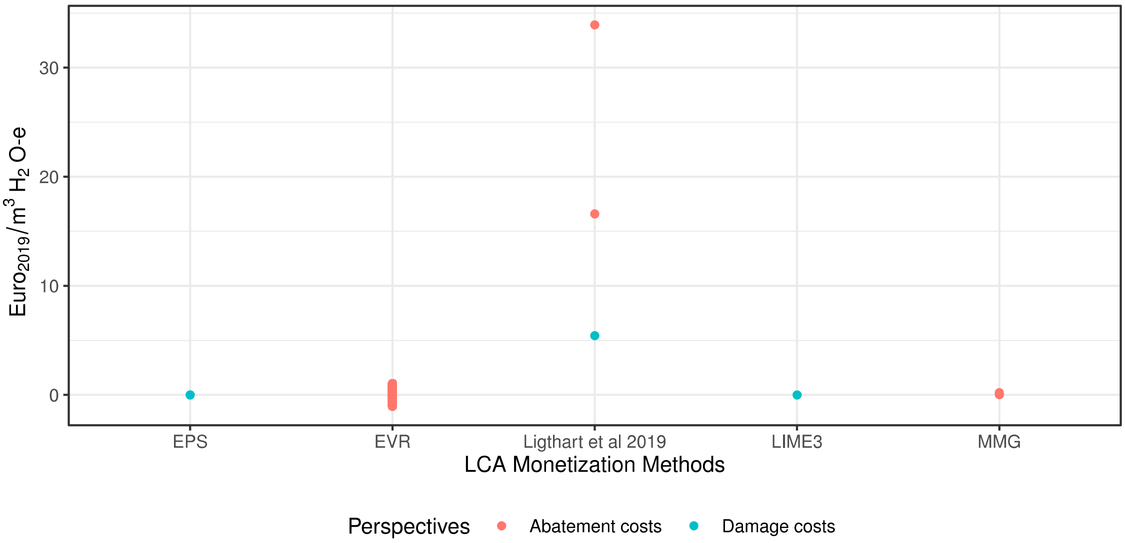


Figure 15. Monetary values associated with the use of one cubic meter water for the impact category water use.

The method by Ligthart and van Harmelen [104] determines marginal abatement costs and marginal damage costs of one cubic meter of H_2O-e based on the impact assessment from Frischknecht et al. [50]. It is included because it is the newest study for shadow costs of water and has an explicit connection to LCA. The abatement costs are determined through the cost of technologies that reduce water use (two different values were provided in the abstract and in the full text- both were considered). The damage costs are based on forgone increased revenues due to irrigation of agricultural products (market price approach). For the EPS the obtained values for irrigation and drinking water are based on costs of drinking water and the assumption that irrigation water costs half the amount of drinking water and therefore they follow the market price approach. The EVR method assessed how to win water back by reverse Osmosis (1 €/m^3) multiplied by the baseline water stress according to Gassert et al. [105]. For some regions, it delivers negative values, which we were not able to understand. While the MMG method is based on the Swiss ecoscarcity method as well, the MMG method applied replacement costs, by evaluating technologies that are applied to win water back in countries with a certain amount of water stress. The abatement costs in the Ligthart method are a lot more expensive than the abatement costs proposed by the EVR or the MMG method. The LIME3 value is based on Motoshita et al. [66,106] and includes damages to human health. The values are close to the MMG central estimate that is based on abatement costs. As the damage costs of the Ligthart method and LIME3 cover damages that are completely different in their nature (damages to agricultural goods opposed to damages to human health through malnutrition and infectious diseases), the different values are in line with the expectations.

3.3.11. Land Use, Land Transformation and Soil Organic Matter

Six methods assessed the impact category land use, while only two assessed land transformation and two assessed soil organic matter. The values for land use are displayed in Figure 16.

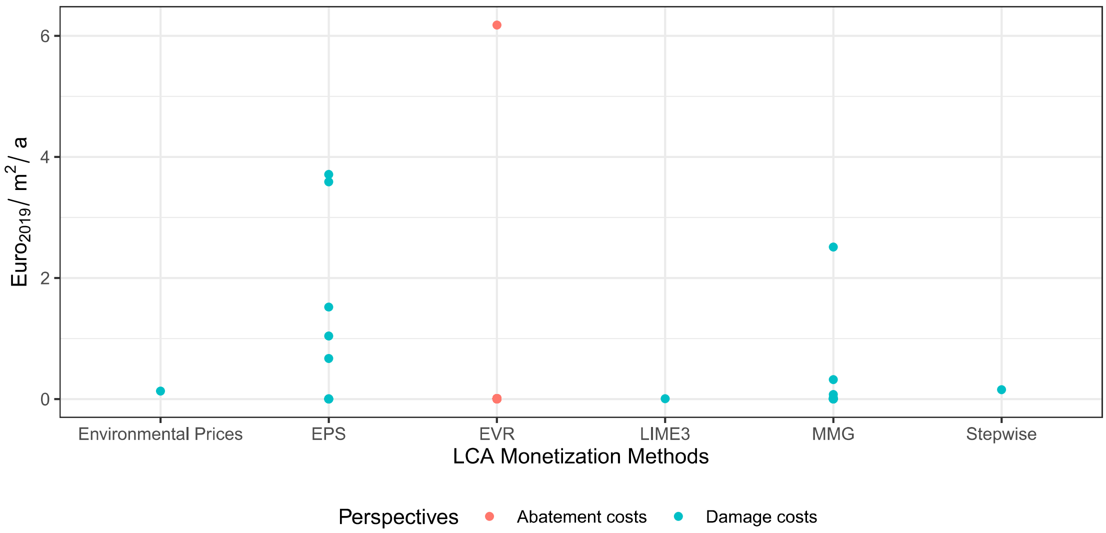


Figure 16. Monetary values associated with the use of one m^2 of land for a year for the impact category land use.

The values for land use range between 6 €/m^2/a (heterogeneous agriculture) from EVR to 12 ct for m^2/a for Stepwise. However, the values also contain different land use types. We assume that the 6 €/m^2/a are a mistake in EVR, because there is no reason why heterogeneous agriculture should have higher eco costs than an artificial area. The calculations in EVR are based on land use for agricultural products from Europe and consider the species richness and the biodiversity factor. EVR does not provide land occupation data for industrial or urban use. For EPS the values for urban and industrial use are the highest (just under 4 €/m^2/a), while the agricultural values are an order of magnitude lower than the agricultural values determined by the EVR. The EPS assesses the impact on biodiversity, but also agricultural damages (if an area is an urban area the fact that no wood and vegetables can be produced on it are included as damages- opportunity costs). The EPS also considers the impact on the reduced drinking water renewal rate. The dominant factor of the EPS though are the effect of the urban heat island on working capacity that constitute over 3 €/m^2/a. For agricultural land use, the values are very low (0.5 ct) and only quantify biodiversity impacts.

Environmental Prices mainly assess impact on biodiversity. Its provided values are higher than the ones provided by EPS for agricultural use. However, it does not publish the associated environmental costs for all kinds of land use and thus provides the same value for urban and agricultural land use.

The MMG method values the impact on biodiversity based on lost ecosystem services for urban and industrial areas, for agricultural areas they use abatement costs, based on a study by Tucker, that is not listed in the methods references. Stepwise calculates damages to biodiversity and obtains a monetary value similar to the Environmental Prices method.

LIME3 considers impacts of land transformation on biodiversity and NPP and of land occupation on NPP. Due to the data availability, only land occupation impacts are assessed in this study. As most other methods do not consider impacts on NPP, the value is difficult to compare. The value and impact determined by LIME3 to primary production seems lower compared to the impacts on biodiversity in the other methods.

For land use transformation, we only have two methods that deliver data (Environmental Prices does not deliver them as a weighting factor in LCA but for application to other studies. It is 3.26 €$_{2015}$/m^2 and assumes a 50-year time horizon of use and a 3% discount rate). The values are displayed in Figure 17.

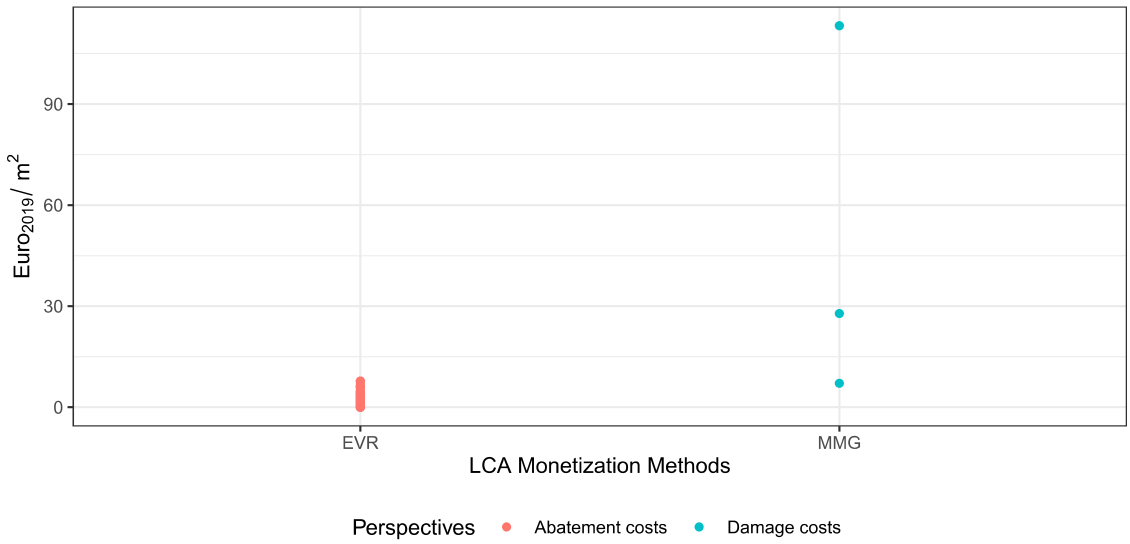


Figure 17. Associated monetary values for the impact category land transformation.

For the MMG method a value from the TEEB study was adapted that is based on land use change in the tropical rainforest and its annual biodiversity losses. They discounted the losses at 3%. It remains unclear whether these values have been directly transferred to Europe. As the tropical rainforest has a very high biodiversity and bio productivity, this baseline is likely to have increased the values. For EVR the approach and the values for transformation and occupation are the same.

Only two methods provide values for soil organic matter (the MMG method and the method by Ligthart and van Hamelen). The values are both oriented towards the midpoint category that is connected to land use [107,108]. The values are shown in Figure 18.

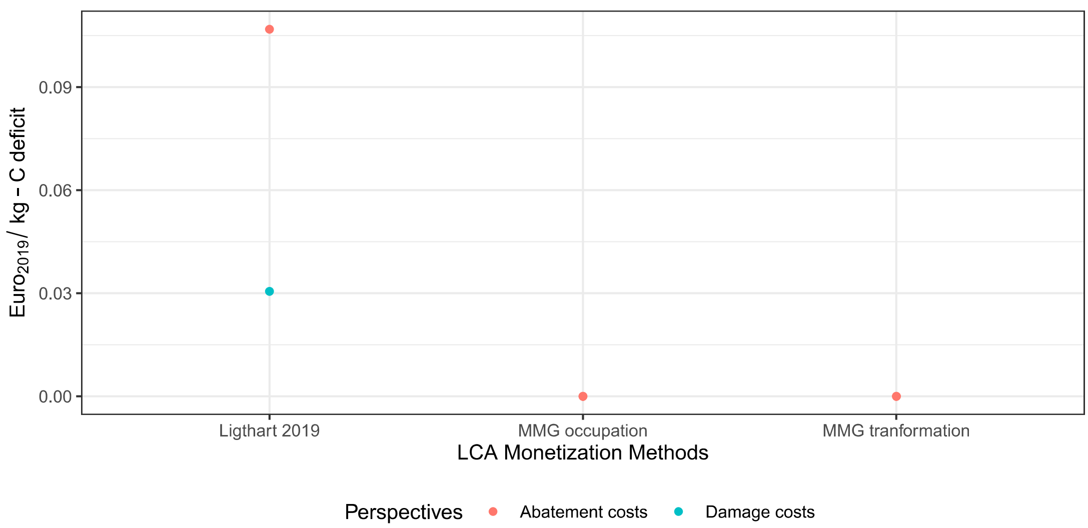


Figure 18. Monetary values for the impact category soil organic matter that is connected to land use.

The MMG method, that delivers lower values, is based on abatement costs and considered the technologies that would be realized if a carbon tax of 100 \$/t CO_2 was implemented, e.g., zero tillage. Based on this, they assume costs of 0.00000034 € per kg C. We do not really understand what they calculated here and how they obtained this very low value. If we assumed 100 \$/t CO_2 (corresponding to 0.1 \$/kg CO_2) the associated value for storing carbon in the ground would be 0.275 \$/kg of C. With this value the MMG value would be much higher than the values from Ligthart and van Hamelen [104].

Further, it is unclear to us why different values for soil organic matter for transformation and occupation are delivered.

The study by Ligthart and van Hamelen [104] determined damage and abatement costs of soil organic matter loss. The damage costs only include economic damages to agricultural productivity (yield loss). The abatement costs are based on avoided costs of fertilizer.

3.3.12. Human Toxicity

Six methods provided monetary values for human toxicity. Their results are visualized in Figure 19.

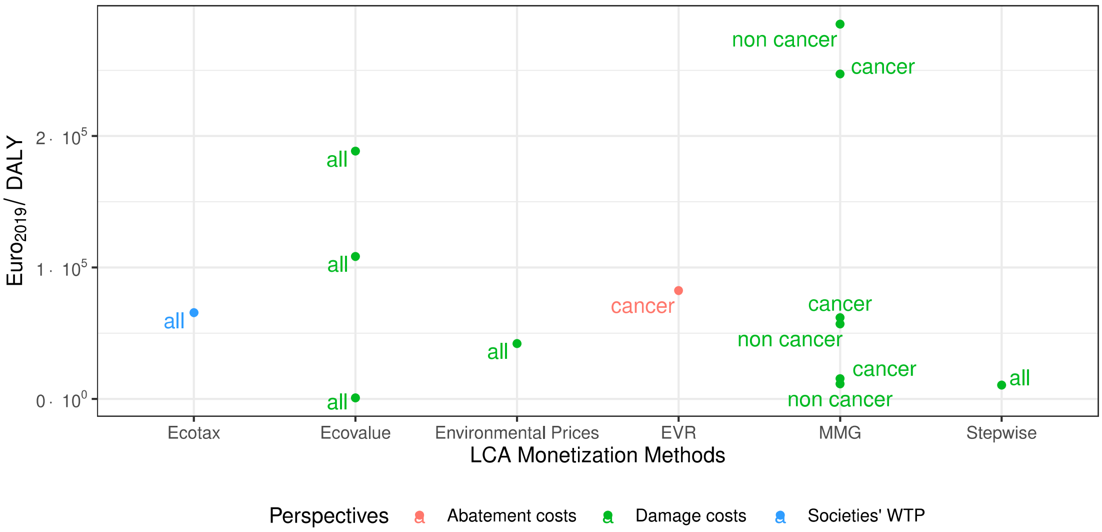


Figure 19. Monetary values per DALY values, cancerous effects and non-cancerous effects are delivered separately by some methods, if these are considered together values are labelled as "all".

As we converted the impact of toxic substances to DALY with the USEtox model, it was expected that the €/DALY values would be similar to the value that has been assigned to human health for VOLY based on the NEEDS WTP studies. The values range around these data with the highest value for the MMG high estimate for non-cancer (with 285,156.68 €/DALY) and the lowest value for Ecovalue's minimum estimate with 771 €/DALY.

MMG includes DALY losses, and for cancer also the costs of cancer treatment and the associated productivity loss [109]. The values for non-cancer have a larger range, because CTUh for non-cancer effects are more uncertain [110].

Another noteworthy observation is that the values for Stepwise are rather low, even though Stepwise has a high valuation of €/DALY (as shown in Table 2). In addition, the cause-effect chain of Stepwise is not based on USEtox but on Impact 2002+. If we use (instead of USEtox characterization factors) first the conversion factor by Pizzol [20] to transfer Vinylchloride (the impact assessment indicator for Stepwise) to convert to 1,4DCB-e to air (0.0057 kg C_2H_3Cl/kg DCB to air) we yield very low values for Stepwise per DALY (~800 €/DALY). This seems much too low. Therefore, these differences must be due to different characterization factors and impact assessment modelling in the toxicity assessments of Impact 2002+ and USEtox, because the valuation of a human life in all assessed methods is in a similar range. Further Stepwise includes damages to resource productivity of humans due to work (Table 13.1 in reference [65]), which all other methods do not include, which is in contrast with the low values obtained here.

For Environmental Prices the valuation is very close to the NEEDS approximation (~42,000 €/DALY), even though it is based on ReCiPe toxicity assessment which seems to not yield very different results compared to USEtox in our assessment. Another rather high value is from EVR (80,000 €) which based on kidney hospital treatment and is an upper bound [111]. The calculation from the emissions back to the DALY yield the same result in our assessment, because our calculations are based on USEtox and the calculations of the EVR are based on USEtox as well. Ecovalue has the highest medium value. According to Finnveden and Noring the value is based on the Shadow Prices Handbook [101], which is also based on NEEDs WTP for life expectancy. If they were income adjusted by Sweden's GDP, the high value would be partially explainable since the Swedish GDP is higher than the EU's average, but the exact transfer is not documented. For the Ecotax method the tax is based on cadmium content in fertilizer, which is very high and yields a value of 65.000 € which is close to Swedish GDP per capita.

If we look at the average values and the variance (see Figure 3), we can see that the data is close to normally distributed and we do not have any outliers. Further research is needed to compare the differences in the underlying toxicity assessments and their effect on the different DALY values.

3.3.13. Terrestrial Ecotoxicity

Three methods provide data on terrestrial ecotoxicity, namely Ecotax, Environmental Prices and Stepwise. The values are displayed in Figure 20.

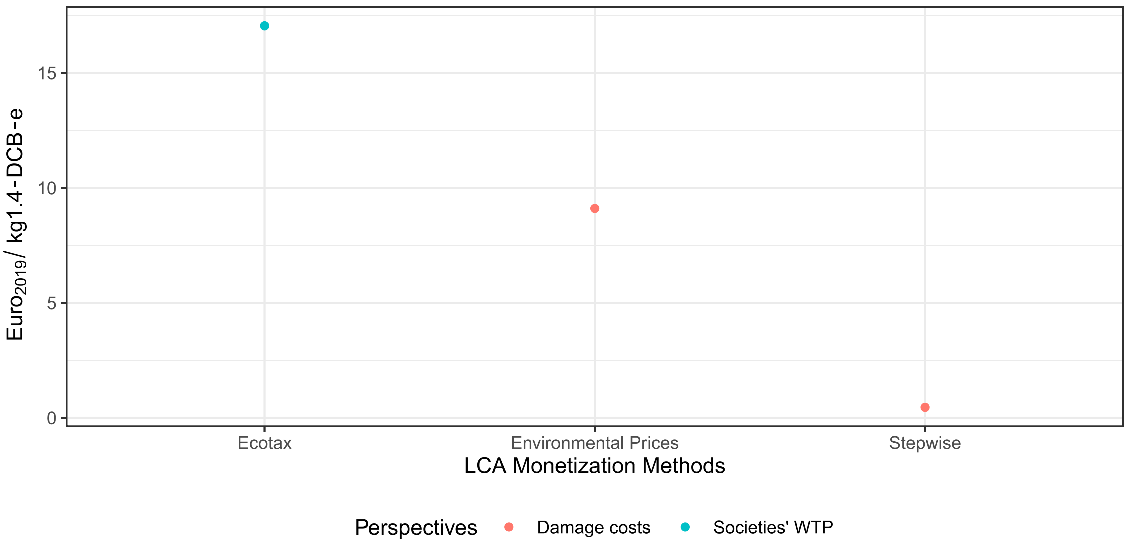


Figure 20. Monetary values for the impact category terrestrial ecotoxicity provided in 1,4DCB equivalents emitted to industrial soil.

Ecotax has the highest value; it is based on a Swedish tax on exceeding cadmium values in fertilizer. For Environmental Prices ecotoxicity impacts are assessed through ReCiPe 2013 [102] and then monetized by the values provide by Kuik et al. [82], that are higher than most other ecosystem values. It is not clear why Stepwise is so much lower than Environmental Prices, even though its biodiversity valuation delivers higher values than the Environmental Prices medium estimate (see Table 2). The differences must therefore be due to different impact assessment: while Environmental Prices uses ReCiPe, Stepwise uses Impact 2002+ as its impact assessment method.

3.3.14. Freshwater Ecotoxicity

The impact category freshwater ecotoxicity is assessed by five methods (see Figure 21).

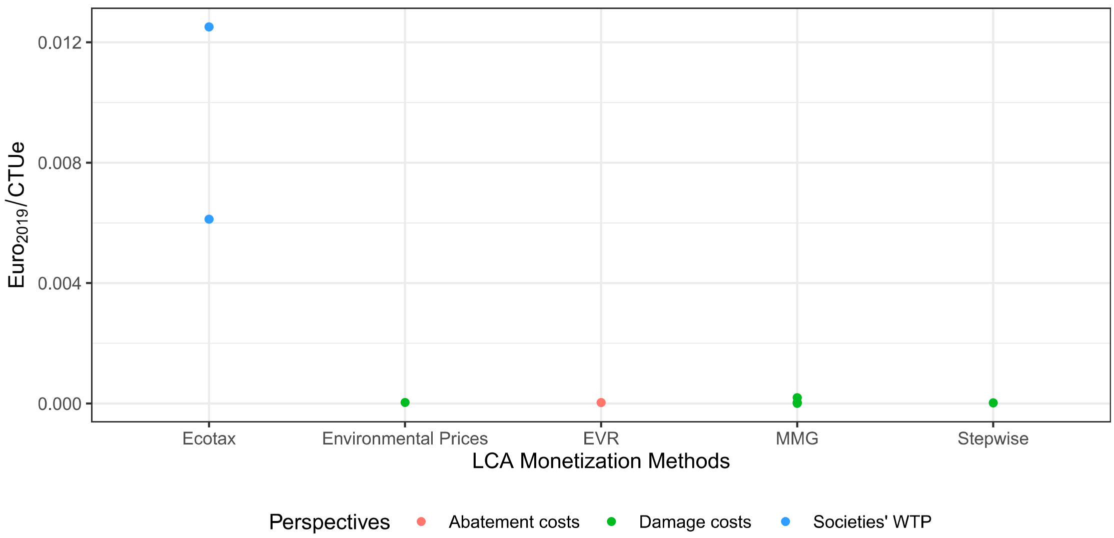


Figure 21. Monetary values associated with freshwater ecotoxicity provided in Comparative toxic Units for ecosystems (CTUe).

Also for this toxicity category, the highest value is provided by the Ecotax method based on toluene emissions to freshwater. EVR costs are based on wastewater treatment to remove toxic substances (lead substance copper). For Environmental Prices, the model is based on ReCiPe and on the species valuation of Kuik et al. [82]. The MMG method is based on USEtox and on the biodiversity valuation of Ott et al. [80]. After Ecotax, the MMG method shows the highest value (high and central estimate). This is surprising because the ecosystem valuation of Ott et al. [80] is lower than the one by Kuik et al. [82]. Accordingly, we would have expected a higher value for Environmental Prices, but apparently the differing impact assessments play a role here. Stepwise values are very low, again presumably due to the different impact assessment used, as for terrestrial ecotoxicity.

3.3.15. Marine Ecotoxicity

For marine ecotoxicity, three methods provide values, namely Ecotax, Ecovalue and Environmental Prices (see Figure 22).

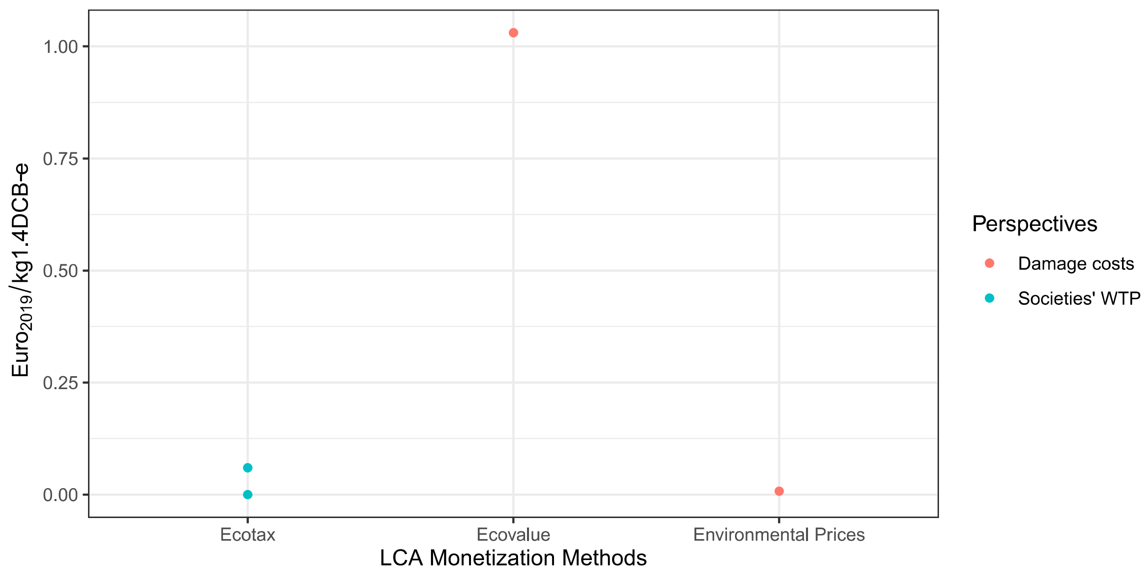


Figure 22. Monetary values for the impact category marine ecotoxicity provided in 1,4 DCB-e to sea water.

It is worth noticing that the damage costs for Ecovalue are much higher than the societies' WTP. It is also remarkable that the Ecovalue method and Ecotax both assess impacts in Sweden. The Ecotax method bases its calculation on taxes based on toxic substances in pesticides (copper), which is apparently a lot lower than the associated damages of such an emission. For Environmental Prices the impact modelling is based on ReCiPe 2013 between individualist and hierarchist perspectives and the economic valuation is based on the data by Kuik et al. [82]. For the Ecovalue method, the values are based on a study by Noring et al. [112] on tributyline pollution, but no documentation could be found how the tributylene pollution was transferred to 1,4-DCB equivalents. Even though Noring et al. [112] excluded very high answers, the obtained values are the highest. The value is based on contingent valuation and WTP that was linked to reaching a certain policy target that would increase the environmental status of several snails and other species. Because of the nature of the assessment in Ecovalue (specific pollutant, geographically discrete), it is not surprising that the value is higher.

Only Environmental Prices provides data that have a link to ecosystems, but they also state that their values have limited validation, especially if site-specific studies should be performed.

What is noteworthy is that for freshwater and terrestrial ecotoxicity the values for the Ecotax method deliver the highest values, while it provides lower values for marine ecotoxicity. This inconsistency in the Ecotax method occurs presumably because the Ecotax method has no underlying link to any AoP.

4. Discussion

Within the quantitative comparison, the most important uncertainties are unrelated to the adjustment of the monetary values for inflation and currency. The process of harmonizing the different impact category (methods and) indicators prior to comparing the weighted results involves the most severe uncertainties (i.e., the unit conversion factors documented in Table S1). The same uncertainty applies to the conversion of the EU normalization factors per capita. Time did not allow us to analyze the influence of different unit conversion factors, that are based on various impact assessment methods, on the results. The conversion factors have a big impact if some methods do not link impacts clearly to an AoP such as the Ecotax method, or if they do not convert environmental releases or extraction of resources to an impact category indicator (as in the EPS or EVR for mineral resources). These methods often monetized impacts at the inventory level. The comparison of the impacts for some impact categories such as the toxicity impacts and the use of mineral resources should be interpreted with great care: As some values that we compared here do not have a common impact assessment, a comparison by substance or by material would be necessary to obtain a more meaningful result. This would allow one to compare how the impact of different substances and materials are valued in more detail and not only for 1,4-DCB-e or Sb-e for the toxicity and mineral resources respectively. Therefore, the monetary factors in Table S3 in the supplementary materials for these impact categories should not be used as weighting factors, as the monetization methods themselves partially do not use an impact assessment but monetize at inventory level (e.g., the EPS or EVR), or use different underlying impact assessments (e.g., USEtox or ReCiPe).

For many impact categories, the main influencing factor could be identified (i.e., differences in the impact assessment, the evaluated AoP or its valuation). For others, however, this was not possible. For the monetization of the impact category freshwater ecotoxicity regarding the MMG method and Environmental prices, for example, the valuation of the AoPs was known and suggested a higher value for Environmental prices for biodiversity, but the obtained values for freshwater ecotoxicity were still higher for the MMG method. Within the scope of this research, it was not possible to separate those two influencing factors (the impact assessment and the AoP valuation). A more in-depth analysis of individual impact categories that explicitly compares different impact assessment methods and monetization of the associated environmental impact would be needed.

Within the qualitative assessment, the geographical scope has been shown to considerably change the magnitude of the results. Compared to the other qualitative characteristics the values were highest if the reference region was rich. This aspect was even more dominant than the chosen discount rate or, if applied, equity weighting (even though choosing an LCA monetary valuation method derived in a rich geographical region to a poorer region can also be perceived as a kind of equity weighting). The relevance of the geographical scope underscores that there is a trade-off between universal applicability and global monetization methods on the one hand, and a site dependent and much more meaningful result on the other. As impact assessment methods are becoming more regionalized, the monetary valuation of associated impacts should also be region specific, to deliver more meaningful results. To calculate the influence of the geographical scope, benefit transfer methods can be used as applied in Ahlroth et al. [63]. But these models need to be applied with caution as valuation of non-marketed goods is often culturally diverse beyond income [113]. Therefore, practitioners that want to monetize their LCA results should pay attention to where the lion's share of the environmental burden occurs and should choose the monetization methods accordingly.

The inclusion of different AoPs varies for the different methods. In environmental economics it is common practice to include damages to crops and working productivity and human wellbeing but these are not universally accepted AoPs in environmental LCA. The differing included AoPs for nearly all impact categories from ReCiPe [26] and Environmental Prices [29], but also for other methods underscore this problem. This aspect can be highlighted regarding the impact category water: are its damages covered sufficiently by damages to ecosystems and induced malnutrition, or should forgone income of farmers due to reduced yields be included as well? Therefore, a further research objective is to clarify the relation of environmental LCA to welfare and environmental economics.

Because many methods do not establish a link to all damaged AoPs (e.g., the link from water use to ecosystems), it can safely be stated that most monetary damage factors merely represent a lower bound. One of the biggest challenges is to monetize biodiversity and ecosystem services. The link between ecosystem services and LCA is mainly established by the Trucost method. A comparison of its biodiversity valuation with that of other methods is precluded because the underlying calculations are not disclosed. Apart from the Trucost method, only a limited number of studies were used to value biodiversity [76,80–82]. This again underlines the need to better integrate research on ecosystem services and environmental economics with LCA. Currently, all other monetization methods in LCA studies assessing biodiversity only include effects on limited amounts of species (vascular plants or birds) as the studies by the NEEDS project [80,82], the study for EPS [81] and the study in LIME3 [69].

Even if the uncertainties in the unit conversion factors and the different geographical scopes were negligible, we doubt that the values would converge significantly. In the last step of the quantitative comparison, it became visible that the monetization methods prioritize different impact categories. The weight assigned to the different impact categories when they were applied to environmental damages of an average EU citizen varied for many impact categories: climate change (7% to 58%), mineral resources (0% to 32%), particulate matter (5% to 35%) and POCP (0 to 82%). Therefore, it is expected that the choice of the monetization method leads to different recommendations regarding product optimization or consumption choices (see Figure 4). If a practitioner wanted to pay specific attention to global warming and assigning ~50% of total human damages seemed right to them, the Ecovalue or Stepwise might be the right choice. If another practitioner was particularly concerned with mineral and fossil resources, LIME3 or EPS would be more in line with their worldview. Additionally, practitioners could apply several monetization methods to their case study in order to verify whether results, such as the superiority of one product over another (in terms of their environmental LCA results) or an identified significant process, are sensitive to the applied monetization method. They should also pay attention to the covered impact categories: a monetization method that does not supply monetary values for land use and energy should not be applied to a study that compares bioenergy with fossil energy, as it will undervalue the impact of bioenergy.

The calculated environmental damages of the damage of an average EU citizen should be interpreted with care: these factors only include domestic environmental impacts and no emissions embedded in trade. If these were included, the damages would rise. The calculation was mainly performed for illustrative purposes to obtain information how the monetization methods weight in between midpoints.

We derive from our results that the monetary weighting of LCA results will benefit from further research and discussion in the field, potentially leading to results that are more meaningful and accepted by a wide scientific community. The information for monetary valuation in the Annex C of the ISO standard for monetary valuation [42], especially Table 2, should be considered by method developers in their choices which underlying valuation studies to choose to value AoPs. Further, standards for good practice in the field of stated preference studies in general and choice modelling in particular [114] should be considered by the LCA monetization community (e.g., attributes of one particular good are traded rather than different goods among themselves). Moreover, advances in participatory valuation in the field of ecosystem services [115] might be interesting for LCA. Furthermore, WTA studies could be used for monetization in LCA to see how the magnitude of the valuation would change if the property rights of the welfare loss associated with the emissions and resources use was assigned to the entity whose welfare is reduced. If the monetary values obtained by WTP studies are used to estimate the amount that the entity whose welfare is reduced by the pollution or resources use should be compensated for, in order to be equally well as before the caused environmental harm, the use of WTP will result in a loss [116]. For that purpose WTA, values would be more adequate.

Availability of information was an issue in the evaluation of some methods. While the documentation of the calculations for the EPS method, EVR, MMG and Environmental Prices was thorough and clear, this was different for the other methods. The extreme end of this was the Trucost method, whose naming is definitely misleading as we are far from a "Trucost" in monetizing LCA results, that did not disclose any calculations. To enable a constant improvement of methods, results and calculations should be documented in detail.

5. Conclusions

This study provided an overview of currently applied monetization methods in LCA. Values from nine methods covering 18 impact categories were compared qualitatively and quantitatively. The quantitative results show a non-normal distribution of the obtained damage factors for the different impact categories. Additionally, the methods emphasize the various impact categories quite differently, showing different preference structures. The most influential criterion was the geographical reference area (i.e., the richer the reference area, the higher the results). Choosing the discount rate and equity weighting turned out to be less important. Overall, we can state that current monetization methods in LCA use a wide variety of monetary valuation approaches. Therefore, varying monetary damage values are obtained. Practitioners should especially pay attention to the coherence of the underlying reference region of monetization methods and their case study. Method developers, in turn, should concentrate on the quality of the valuation studies from which they derive their monetary values. For some impact categories, mainly those concerned with impacts on human health, the monetary values converge more than for others. This is attributed to the fact that valuing the AoP human health is more developed than valuing biodiversity and resources. One of the major identified weaknesses is the valuation of biodiversity, which is currently mainly oriented to valuation of vascular plants in LCA. Monetized LCIA results related to biodiversity and resources show wide ranges and a non-normal distribution. Further consensus is needed on which kind of damages should be included in the AoPs. For example, there is no consensus on whether reduced labor productivity or damages to assets such as buildings should be included. Further, no monetization method for LCA is available that uses WTA.

Supplementary Materials: The following are available online at http://www.mdpi.com/2071-1050/12/24/10493/s1, Table S1 Impact category indicators that were used for comparison, CFs and UCFs used for unit conversion to convert all impact categories to the same units, Table S2. Overview of established links between methods (per impact category) and AoPs, Table S3. Monetary values per impact category for all assessed methods, Figure S1. Distribution of monetary values per impact category without Ecotax and EVR

Author Contributions: Conceptualization, R.A., V.B., T.M.B., M.F.; methodology, R.A., V.B., T.M.B., M.F.; validation, R.A., V.B., T.M.B., M.F.; formal analysis, R.A..; investigation, R.A., T.M.B., M.M.; data curation, R.A., T.M.B., M.M.; writing—original draft preparation, R.A.; writing—review and editing, V.B., T.M.B., M.M., M.F., R.A.; visualization, R.A.; supervision, T.M.B., V.B., M.F.; funding acquisition, M.F., V.B. All authors have read and agreed to the published version of the manuscript.

Funding: This work was partly funded by the Bundesministerium für Bildung und Forschung, project "R2Q:RessourcenPlan im Quartier" (funding number: 033W102E). We would like to thank the Bundesministerium für Bildung und Forschung for its financial support. We acknowledge support by the Open Access Publication Fund of TU Berlin.

Acknowledgments: We want to thank Yasmine Emara for her methodological support regarding the toxicity unit conversion.

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

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3.2 Criticality Assessment of Abiotic Resource Use for Europe– Application of the SCARCE Method

The results describe the content of the following paper:

Arendt, Rosalie; Muhl, Marco; Bach, Vanessa; Finkbeiner, Matthias: Criticality Assessment of Abiotic Resource Use for Europe– Application of the SCARCE Method. *Resources Policy* **2020** 67 (August): 101650. https://doi.org/10.1016/j.resourpol.2020.101650.

The paper contains an analysis of the criticality of abiotic raw materials for Europe. For this purpose, the SCARCE method is applied. It assesses all sustainability dimensions, while the economic assessment is equal to an economic criticality assessment (assessing vulnerability and probability of a supply risk). The environmental dimension is measured with LCA indicators. The methodology of the SCARCE method has been modified to be applied to a world region instead of a country. It has several advantages compared to other criticality assessment methods, as it assesses more categories. Based on this paper, a better understanding of environmental criticality assessment was derived, which builds the foundation to answer research question 2, especially target 2a) and 3b). This paper is embedded in this thesis as it helped to comprehend the underlying environmental criticality assessment and its weaknesses. Especially the aggregation approach could be identified as a weakness. This allowed to see the advantages of a monetized approach which is connected to research question 2 and 3.

Resources Policy 67 (2020) 101650

Contents lists available at ScienceDirect

Resources Policy

journal homepage: http://www.elsevier.com/locate/resourpol

Criticality assessment of abiotic resource use for Europe– application of the SCARCE method

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A R T I C L E I N F O

Keywords:
Resource use
Criticality
Sustainable development
Supply risk
ESSENZ
SCARCE

A B S T R A C T :

Due to current consumption patterns and increasing product complexity, the use of abiotic resources has been rising and has led to supply risk (criticality) challenges in many countries and regions including Europe. The SCARCE method, originally developed to assess criticality in Germany, includes several criticality determinants that are missing from the existing European method by Pennington et al. (2017). Specifically it i) considers additional supply risk and vulnerability categories like price fluctuations, long term availability and importance in future technologies ii) takes a sustainability perspective by including environmental and social aspects of resource use iii) enables the comparison of the European supply risk with the global supply risk. Therefore, we have applied the SCARCE method to perform a criticality assessment of European resource use considering eleven supply risk categories (e.g. trade barriers and political stability) and six vulnerability categories (e.g. economic importance and substitutability) for 42 materials (including metals, metalloids and fossil fuels).

In our assessment, the most critical materials for Europe are petroleum oils, gallium, rare earths and phosphorus, because of their high supply risk impacts due to high primary material use, high trade barriers and low political stability in mining countries as well as vulnerability impacts (due to their high economic importance, high utilization in future technologies and low substitutability). The three materials with the worst social performance (considering small scale mining, human rights abuse and geopolitical risk) are tantalum, cobalt and tin (e.g. because of high production share in small scale mining for tin), while the materials with the worst environmental performance (considering greenhouse gas emissions, water scarcity and sensitivity of the local biodiversity) are gold, platinum and niobium (e.g. because of a high amount of associated greenhouse gas emissions for gold). Our findings show that the European supply risk does not differ significantly from the global supply risk, but some assessment categories show different tendencies. Compared to the global production, the mining capacity of the countries that are exporting to Europe is lower, because the European import mix is often dominated by one country only while the global production is more diverse. Further, countries that are currently exporting to Europe have higher political stability than the countries that dominate the global production, which indicates that Europe might have to develop new trade relations with politically unstable countries to meet its domestic material demand. Overall, our assessment results are in line with the finding of the previously conducted study on critical raw materials by the European Union, but provide some additional insights by considering social and environmental impacts.

1. Introduction

Increased production of resources is attributable to growing overall consumption and increasing product complexity. These effects are rendered even more extreme by continuing global industrialization and technological development that lead to even higher and more varied demand of resources (IRP, 2017; Schneider et al., 2016). The pollution of the natural environment including air, water and soil has risen as well (UN Environment, 2019). It is predicted that the global resource use will more than double by 2050 (IRP, 2017). This could lead to increasing environmental pollution in the coming years.

As resources are key components of every society to sustain production of goods and services for current and future generations and are the basis for all relevant ecosystem services, resource use has to be designed in such a way that it is in line with sustainable development (Ali et al., 2017; Bleischwitz et al., 2018). This concept is reflected in

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https://doi.org/10.1016/j.resourpol.2020.101650
Received 4 December 2019; Received in revised form 3 March 2020; Accepted 4 March 2020
Available online 20 March 2020

many international (e.g. the Sustainable Development Goals (UN General Assembly, 2015)) as well as national and regional strategies (e.g. Europe's Roadmap to a resource efficient Europe (European Commission, 2011)). A sustainable use of resources entails the availability of resources for current and future generations (supply risks), the capability of countries to successfully overcome challenges to resource availability (vulnerability) as well as taking into account environmental and social concerns during extraction, processing and use of resources (Fischer-Kowalski et al., 2011; Hertwich et al., 2010).

In recent years, many methods have been published to determine aspects of resource use related to sustainable development, which considerably improved the assessment of resource use. They are addressing the micro (product), meso (company) and macro (country) level (an overview of existing methods for all three levels is given by Schrijvers et al. (2019)). However, several methods only consider a limited amount of supply risk and vulnerability aspects and therefore are not able to comprehensively reflect all challenges related to resource criticality of countries (Bach et al., 2017). This is especially true for the official methodology of the EU on critical raw materials (which will be called EU method in the following). It was first initiated by the European Commission (2008) and updated recently (Pennington et al., 2017), but it neglects important aspects in criticality assessment. The approach determines criticality based solely on economic importance (to reflect vulnerability) and selected supply risk aspects (e.g. supply mix, import reliance and political stability). Further, the European criticality assessment neither includes social nor environmental effects of resource extraction.

Another method for evaluating criticality on country level is the SCARCE method (approach to enhance the assessment of critical resource use on country level), which was developed by Bach et al. (2017). This method will be the basis for the assessment of criticality assessment of abiotic resource use for Europe in this paper. The main advantages of the SCARCE method compared to the EU method are that it i) considers additional relevant categories in the criticality dimension (e.g. considering long term availability, price fluctuations, and importance in future technologies, for more details see publication by Bach et al. (2017) as well as section 1 of the supplementary materials) ii) allows to compare the European availability constraints with world-wide constraints iii) takes social and environmental impacts into account and thus addresses all dimensions of sustainability. Most criticality assessments focus only on economic criticality, even though social and environmental aspects of mining are very severe and can limit societal acceptance for certain resources (Garcia et al., 2017; Hanna et al., 2016).

Further, within this paper, the SCARCE method is applied to all countries that are currently part of the European single market (EU-28, Norway, Iceland and Switzerland (Liechtenstein was excluded due to data availability constraints)). Since goods, including material resources, can be traded freely in the European single market, all countries that are part of this market face similar availability constraints.

The goal of this paper is to provide a more comprehensive view on Europe's raw material criticality. Next to additional relevant categories, also additional countries (i.e. Norway, Switzerland and Iceland) are analyzed.

In the method section it is explained how the SCARCE method was applied to Europe step by step. Further, the data sources for the calculation of the different indicator values are described. For an introduction of the SCARCE method and its structure and the formulas of the indicator calculation see the supplementary material (section 1). The results chapter begins with the generic display of the results in the criticality matrix followed by a comparison of the results obtained by the SCARCE method with the results of the study based on the EU method by Deloitte Sustainability et al. (2017). Then the results of the sub dimensions of criticality (*supply risk* and *vulnerability*) are outlined. The results section ends with the description of the analysis of the dimension *societal acceptance* that is structured in the subchapters *compliance with environmental standards* and *compliance with social standards*.

2. Method

The method section describes how the SCARCE method, originally developed for the application to Germany, was applied to Europe. Details on the SCARCE method and its structure are provided in the supplementary material (section 1). The application of SCARCE to Europe has six steps which are displayed in Fig. 1. They lead from the data collection via the determination of the criticality matrix to the comparison of European values with global values. In the following, the six steps of this methodological approach are described in detail.

Step 1: check of applicability and data availability

Within step one, we checked the applicability of the SCARCE method to Europe by examining the data availability. In Figure S 1 in the supplementary material, the sub dimensions and categories of the SCARCE method are displayed. The categories that we neglected are marked in grey. We excluded the category *availability of purchasing strategies* because the data/information on such strategies were not available and could therefore not be applied to Europe. We also excluded the category *company concentration* because the data is six years old and cannot be updated since the SNL database (S&P Global Market Intelligence, 2019) does not provide the information that was used to calculate the company concentration for all materials anymore (the data source that was used before is thus non-existant).

Step 2: the data collection

As a second step, the data was collected. Altogether, the data was gathered for 42 abiotic minerals (four fossil energy carriers, five

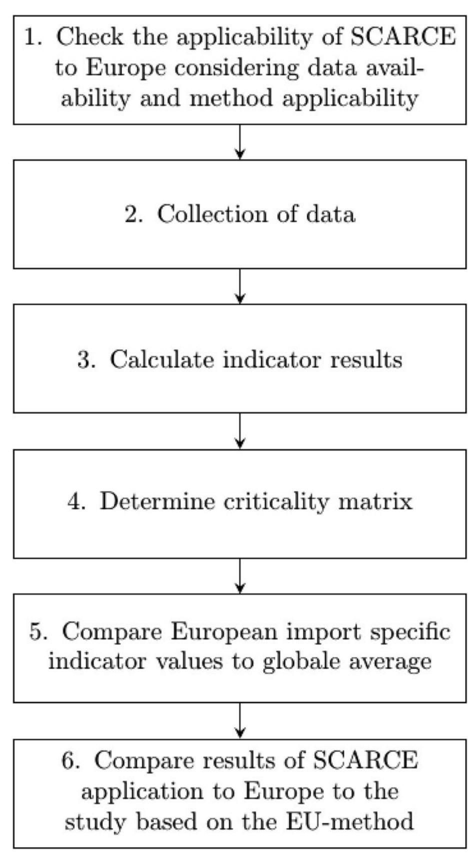

Fig. 1. The method procedure for the application of SCARCE to Europe showing all six taken steps.

metalloids, two non-metals and 31 metals). To determine the material import in tons for the respective resource, the imported amount of the resources to Europe was extracted from UN comtrade, an international trade database, with the reference year 2015 for the EU-28, Norway, Switzerland and Iceland (UN Statistics Division, 2016). An exception was made for gallium, niobium, indium and germanium because comtrade did not provide the import data for these resources individually, but in an aggregated form. Therefore, the data for gallium, niobium, indium and germanium was taken from comext with reference year 2015 and for EU-28 only (Eurostat and European Commission, 2016). Generally, comtrade was preferred, because it provides mass estimates for the imports, whereas this is not true for all trade flows in comext, which sometimes only provides financial trade flows. Because the SCARCE method is mass based, mass trade flows were preferred over financial trade flows. The detailed procedure how the import mix was determined can be found in the supplementary materials (section 1.1).

For the other categories, the underlying data for the indicator calculation were updated if the new data was provided by the publishing organizations, e.g. World Bank. Table 1 displays all chosen indicators and data sources for calculation for the respective categories.

Step 3: calculate indicator and sub dimension results

After we collected the necessary data, we calculated the indicator results for all applicable categories as well as the final sub dimension results. For all the categories of the sub dimension supply risk a distance to target approach (Müller-Wenk and Ahbe, 1990) is applied (indicators are set in relation to a target, and if they are below a certain threshold, they are set to zero. Details can be found in Bach et al. (2017, 2016)). For all considered sub dimensions (supply risk, vulnerability, compliance with social standards and environmental standards), the indicators were scaled (between zero and one and thus relative to one another). For the supply risk categories, the scaling occurred after application of the distance to target approach. To obtain the final sub dimension result, all scaled indicators are summed up using equal weighting to be able to plot the materials in the criticality matrix. The exact guidance on how all indicators are calculated (including equations) and which data is used is provided in the supplementary material section one (1.2 for supply risk, 1.3 for vulnerability and 1.4 for societal acceptance).

Step 4: determine criticality matrix

Within the tradition of visualization of criticality (Buijs et al., 2012; Chapman et al., 2013; Graedel et al., 2012; National Research Council, 2008), we plotted our results in a criticality matrix, with vulnerability as x-axis and supply risk as y-axis. The criticality lines that represent low, medium and high criticality are adapted from Glöser et al. (2015). In order to show the results of the social, environmental, and economic dimension in one matrix, we highlighted materials which exhibited hotspots within the social assessment as squares and within the environmental assessment as triangles. The hotspots are the top five worst performing materials in the sub dimensions compliance with environmental and social standards, respectively. In the result chapter, we also show the results for each of the four sub dimensions individually, because the criticality matrix aggregates 23 indicators. To provide a better understanding of the results, we analyze some results of resources with the highest and lowest values in more detail as well. Since we have used relative weighting and scaled our indicators, we calculated the criticality levels relative to one another. As the sub dimension results of supply risk and vulnerability are based on relative weighting (as explained in section 1.2 and 1.3 in the supplementary material), also the criticality is a relative measure.

Step 5: compare Europe's import-based categories to global category results

Table 1
Categories, indicators and data sources for calculation used in the application of SCARCE to Europe; categories that are import-based are highlighted with a star (*). Import-based means that they are based on actual European sourcing. Only these categories are compared with global supply risk (step 5).For the world's supply risk the import-based indicators are not calculated with imports but with the global production data. This is further specified in section 1.1 of the supplementary material.

Category	Indicator	Data source for calculation
Sub dimension supply risk		
Country concentration of reserves (global production)	HHI (Rhoades, 1993)	U.S. Geological Survey (2017)
Country concentration of production (import-based)*	HHI (Rhoades, 1993)	UN Statistics Division (2016)
Feasibility of exploration projects (import-based)*	Policy Potential Index (Wilson et al., 2013)	UN Statistics Division (2016) Jackson and Green (2017)
Trade barriers (import-based)*	Enable Trade Index (Lawrence et al., 2008)	UN Statistics Division (2016) Geiger et al. (2016)
Demand growth (import-based)*	Historic demand growth of the last five years	U. S. Geological Survey (2017) British Geological Survey (2019)
Mining capacity (import-based)*	Reserves divided by annual production (static reach)	UN Statistics Division (2016)
Political stability (import-based)*	Worldwide Governance Indicators (Kaufmann et al., 2011)	UN Statistics Division (2016) WGI values by Kaufmann and Kraay (2015)
Price fluctuations (global value)	Volatility Index (Federal Institute for Geoscience and Natural Resources, 2014)	Federal Institute for Geoscience and Natural Resources (2018)
Occurrence of co-production (global value)	Semi quantitative measure whether the resource is sourced as the main metal, or partly/only as a companion metal	Angerer et al. (2009)
Primary material use (global value)	Recycled input rates (Graedel et al., 2011)	Graedel et al. (2011)
Abiotic resource depletion (global value)	Abiotic depletion potential (Guinée et al., 1993) measures the resource availability based on crustal content (ultimate resources), whereas the mining capacity relates to currently available reserves	van Oers and Guinée (2016)
Sub dimension vulnerability		
Economic importance (Europe specific)	Value added for sector where the material is applied (Knašytė et al., 2012), taking into account recycled input	Sector specific value added was taken from Eurostat (2018) Sector specific application rates are based on Deloitte Sustainability et al. (2017) and primary material input based on indicator primary material use from supply risk
Domestic required demand (Europe specific)	Imported amount [t]	UN Statistics Division (2016)
Share of global production (Europe specific)	Imported amount divided by world production	Imported amount: UN Statistics Division (2016) World production : U.S. Geological Survey (2017)
Dependency on imports (Europe specific)	1- (European production/imported amount)	European Production: U.S. Geological Survey (2017) Imported amount: : UN Statistics Division (2016)
Substitutability (global value)	Substitutability of material considering	Graedel et al. (2015b) U.S. Geological Survey (2018a)

(continued on next page)

Table 1 (*continued*)

Category	Indicator	Data source for calculation
	performance decrease and price increase	
Utilization in future technologies (global value)	Necessity of material for future technologies, that are crucial for economic competitiveness	Marscheider-Weidemann et al. (2016), Erdmann and Behrendt (2011) Grandell et al. (2016) U.S. Geological Survey (2018b)
Sub dimension compliance with social standards		
Small scale mining (import-based)*	Share of imports in artisanal small scale mining	Mainly Brower et al. (1979) and Godoy (1985) but further information was obtained from (Burki, 2019; Bustamante et al., 2016; Crispin, 2003; Hilson, 2002; Lacerda, 2003; Lu and Lora-Wainwright, 2014; Mutemeri and Petersen, 2002; Shen and Gunson, 2006; Siegel and Veiga, 2009; Veiga et al., 2006)
Geopolitical risk (import-based)*	Voice and accountability and no violence from WGI (Kaufmann et al., 2011) and armed conflicts (Global peace index (The Institute for Economics and Peace, 2008))	WGI: Kaufmann and Kraay (2015) Armed conflicts: The Institute for Economics and Peace (2015)
Human right abuse (import-based)*	Child labor, forced labor, overall torture	Benoît-Norris et al. (2012), Cingranelli and Richards (2010)
Sub dimension compliance with environmental standards		
Sensitivity of local biodiversity (import-based)*	Sensitivity of local biodiversity based on Winter et al. (2018)	WWF (2012)
Water scarcity (import-based)*	Water depletion index (WDI) (Berger et al., 2014)	Berger et al. (2014)
Climate change (global value)	ReCiPe end-point for climate change (Huijbregts et al., 2017)	Values for materials if possible from GaBi (Thinkstep, 2016), if not available from ecoinvent (Ecoinvent, 2016)

As a further step, we compare Europe's supply risk and the compliance with environmental and social standards to the global performance for all import-based indicators (all indicators that are based on the data from actual European sourcing are called import-based, the global indicator they are compared with is based on global production(see equation (1)).

$$\Delta_{i,c} = indicator\ result_{Europe,import,i,c} - indicator\ result_{global,i,c} \tag{1}$$

The difference (Δ) is determined by subtracting the global indicator results from the import-based results for the considered resource i within each category c. If the result is bigger than zero, the supply risk is higher for the imports into Europe. If it is zero, the performance in the category is equal. If it is below zero, the import-mix performs worse in the respective category compared to the global risk. We show the values for each resource and discuss why the indicator values differ, and how the European import mix influences the indicator values compared to the global supply. The comparison was not performed for the sub dimension vulnerability because it does not depend on the supplying countries (the import mix), but on the European economic activity.

Step 6: compare SCARCE method and results to the EU method

As a final step, we discussed the results of our study in relation to the study performed by Deloitte et al. (2017) (in the following referred to as study based on the EU method) that determined critical raw materials for the EU to check our results for plausibility. We discuss how their indicator choices differ from ours and why the obtained results differ (results are displayed in supplementary material section 3 regarding the differences in the method and section 4 of the supplementary material focuses on the differences in the study's results). In order to underline the differences, we display the materials that were assessed as critical by the EU method in the SCARCE criticality matrix as stars, and the materials that we considered but that the EU study neglected as "*x*"es in Fig. 3. The materials that the EU method assessed as non-critical are displayed in diamond shape.

3. Results

In this chapter, the results regarding the criticality of European abiotic resource use are presented. In the first subchapter, we show the criticality matrix which contains all other results in aggregated form (3.1). Firstly, we compare our obtained results with the results of the study of the EU method (Deloitte Sustainability et al., 2017) (section 3.1.1). Next, the sub dimensions of economic criticality, the supply risk, and vulnerability results are discussed in more detail (see subchapter (3.1.2 and 3.1.3)). Finally, the results of the dimension societal acceptance (3.2.) are presented that consist of the environmental (section 3.2.1) and social analysis (3.2.2). Within each subchapter we also compare the import-based results for Europe to results based on the global production.

3.1. European criticality, comparison with EU method

In Fig. 2 the criticality matrix is displayed. The lines are showing the five levels of criticality, which are increasing from one to five, with five being most critical. As displayed in Fig. 2 only one material is in criticality leve five (petroleum oils), one in criticality level four (gallium), three in level three (rare earths, phosphorus and tantalum), two on the border from two to three (bismuth and cobalt), twelve in level two, one between level two and one (niobium) and 23 in level one. Most materials (23) are in level one and are thus considered less critical.

A material has to have a high supply risk and a high vulnerability to be critical (see upper right corner). Materials that are in the lower right corner are materials that have a high vulnerability but a low supply risk. This means that a supply risk of materials like platinum is unlikely, but that the impact of a supply restriction is quite severe. Materials in the upper left corner like bismuth have high supply risk, but only moderate vulnerability. This means that the supply risk of bismuth is high, but that its effects on the economy are not as severe as e.g. a supply constraint of platinum.

Materials that show a hotspot in the environmental assessment have a triangle shape while materials that have a hotspot in the social assessment are displayed as squares. Petroleum oils are the only resource in the criticality level five. They have a high supply risk, since they are imported from countries with low political stability, high trade barriers and low feasibility of exploration projects like Russia (34.5%), Saudi Arabia (9.3%), Iraq (9.3%) and Algeria (5.6%). The high supply risk is also due to its high primary material use, because petroleum oils cannot be recovered or recycled in their main applications (plastic recycling is irrelevant, because only 4–6% of petroleum oils are used for plastic (PlasticsEurope, 2018) and recycled plastics hold only 12% of the global plastic market (Geyer et al., 2017), so that the recycled input of total petroleum oils demand is only 0.48%–0.72% and therefore negligible). Europe's vulnerability to a supply constraint of petroleum oils is high because the high economic importance of petroleum oils (through the provision of energy for different sectors), high dependency on imports (only Norway produces petroleum oils, and not enough to fulfill the domestic demand), its high price fluctuations and because of its high domestically required demand (it has the highest imported amount in tonnes to Europe from all materials). Gallium has the second highest criticality and is the only resource in level four mainly due to its very high demand growth (because of its application in semiconductors in information technology as gallium arsenide (Eheliyagoda et al., 2019)). Moreover, it is mainly co-mined in the processing of bauxite. The

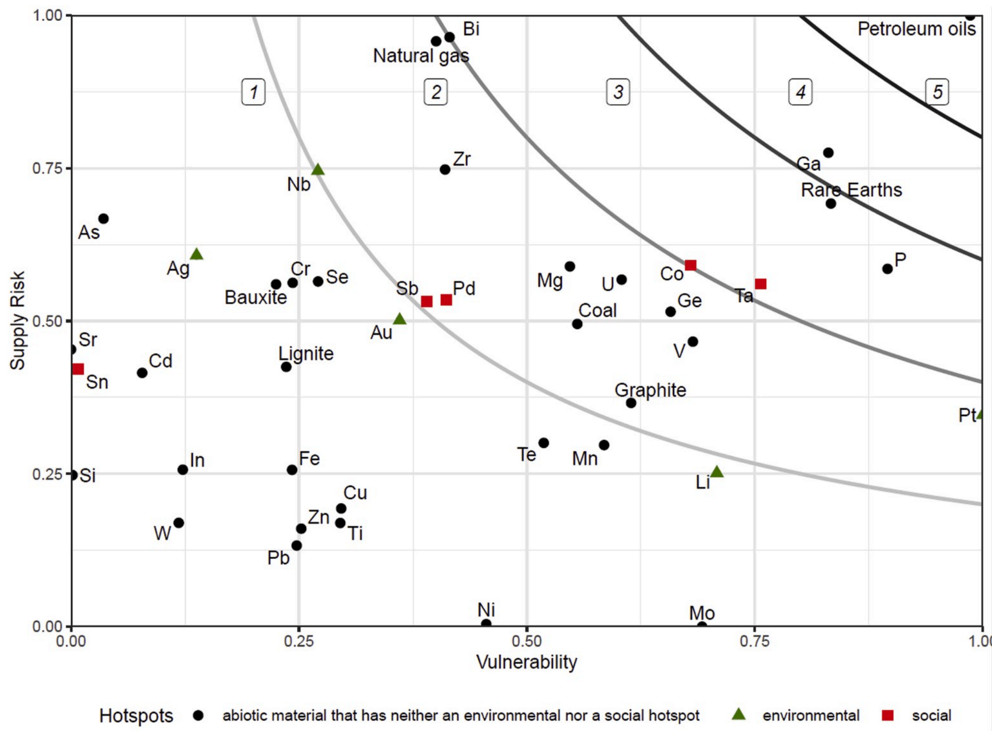

Fig. 2. Criticality matrix for European imports; The criticality lines that separate the criticality levels and are based on the following equation: *Supply Risk*·*Vulnerability* $\leq \{$ 0.2→*Criticality level* 1; *lowest criticality*; 0.4→*Criticality level* 2; 0.6→*Criticality level* 3; 0.8→*Criticality level* 4; 1→*Criticality level* 5; *highest Criticality* ; materials that have a triangle shape have an environmental hotspot, because they are the top five worst performing materials in our environmental assessment, while the squares are the top five worst performing materials in our social assessment.

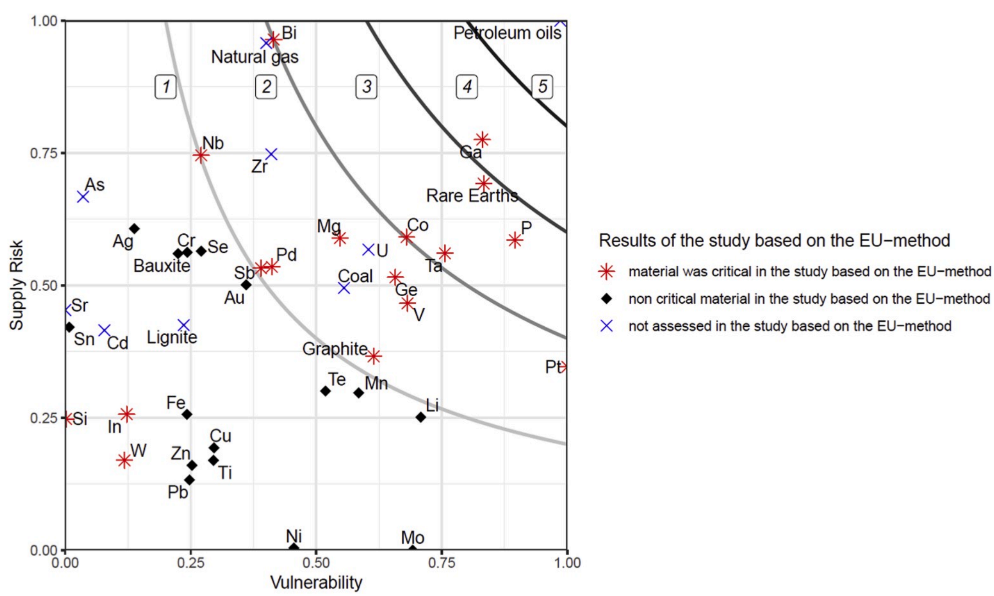

Fig. 3. SCARCE criticality matrix, materials that were assessed as critical by the study based on the EU method (Deloitte Sustainability et al., 2017) are stars, materials not assessed by the study based on the EU method are marked as an "x", while materials that were assessed as non-critical by the study based on the EU method have a diamond shape.

vulnerability score of gallium is also high, because half of its annual world production is imported to Europe and gallium has high importance in future technologies.

The results show that the environmental and social sub dimension are independent of the economic sub dimensions. Apart from platinum, all other environmental hotspots have low criticality levels. Social hotspots do not show a correlation with vulnerability neither, but have a medium supply risk. Tin is the only resource that has a social hotspot and is in the lowest criticality level, while all others are in level two and three.

3.1.1. Comparison to the results of the EU method

In this subchapter, our approach is compared to the study based on the EU method. For a detailed methodological comparison see section 3 of the supplementary material. Next, the differences in results are addressed. Due to the methodological differences outlined in the supplementary material, a quantitative comparison of the outcomes of both studies is not feasible. Thus, we visualized the outcomes by highlighting the EU critical raw materials (star shapes) in the SCARCE criticality matrix (see Fig. 3). The materials that were not assessed in the study based on the EU method are marked as an "x", while the materials that the study based on the EU method assessed as non-critical have a diamond shape.

It can be seen that only tungsten, indium and silicon, which were classified as critical in the study based on the EU method, show a criticality level of one for SCARCE. Therefore, there is a high congruence of the results. Gallium shows a low criticality in the study based on the EU method but a high one in SCARCE. For a material by material analysis of the different results, see the supplementary information (section 4.).

3.1.2. Supply risk

In this subsection, the results of the sub dimension supply risk with its categories *concentration of reserves and production, feasibility of exploration projects, trade barriers, demand growth, mining capacity, political stability, price fluctuations, occurrence of co-production, primary material use* and *abiotic resource depletion* are presented. Fig. 4 presents an overview of the indicator value for each category and all the assessed resources.

Bismuth has the second highest supply risk because of trade barriers, high production concentration, low political stability in importing countries, high occurrence of co-production and very low recycling rates. 94% of current European bismuth supply comes from China, which influences the results strongly as the production concentration is high and China has high trade barriers and low political stability.

Molybdenum has the lowest value because it does not show a supply risk in many of the assessed categories (seven out of twelve). Apart from its rather high price fluctuations, no indicator values are above 0.2. This is also because Europe imports 50% of molybdenum from the US which is not associated with supply constraints. However, this might change considering the current US trade policy (Elms and Sriganesh, 2017).

Next, Europe's supply risk is compared to the supply risk of the global production. This is done for all categories whose calculation was based on European import data namely the categories *concentration of production, feasibility of exploration projects, mining capacity, political stability* and *trade barriers*. The results can be seen in Fig. 5. The difference (Δ) is largest on the top and the bottom. The top values have a better sub dimension result for the European import mix, while the bottom results have a better sub dimension result for the global mix.

In the following, some resources and their different indicator scores are analyzed in more detail in order to show how the European import mix differs from the global production. The materials that are interpreted more comprehensively are bauxite, silver and tungsten, since they show extreme performance differences in Fig. 5. The difference between the European import mix-based results and the global production-based results for natural gas and molybdenum are not discussed here in detail because reasons for the high difference were already discussed when addressing the European supply risk and the

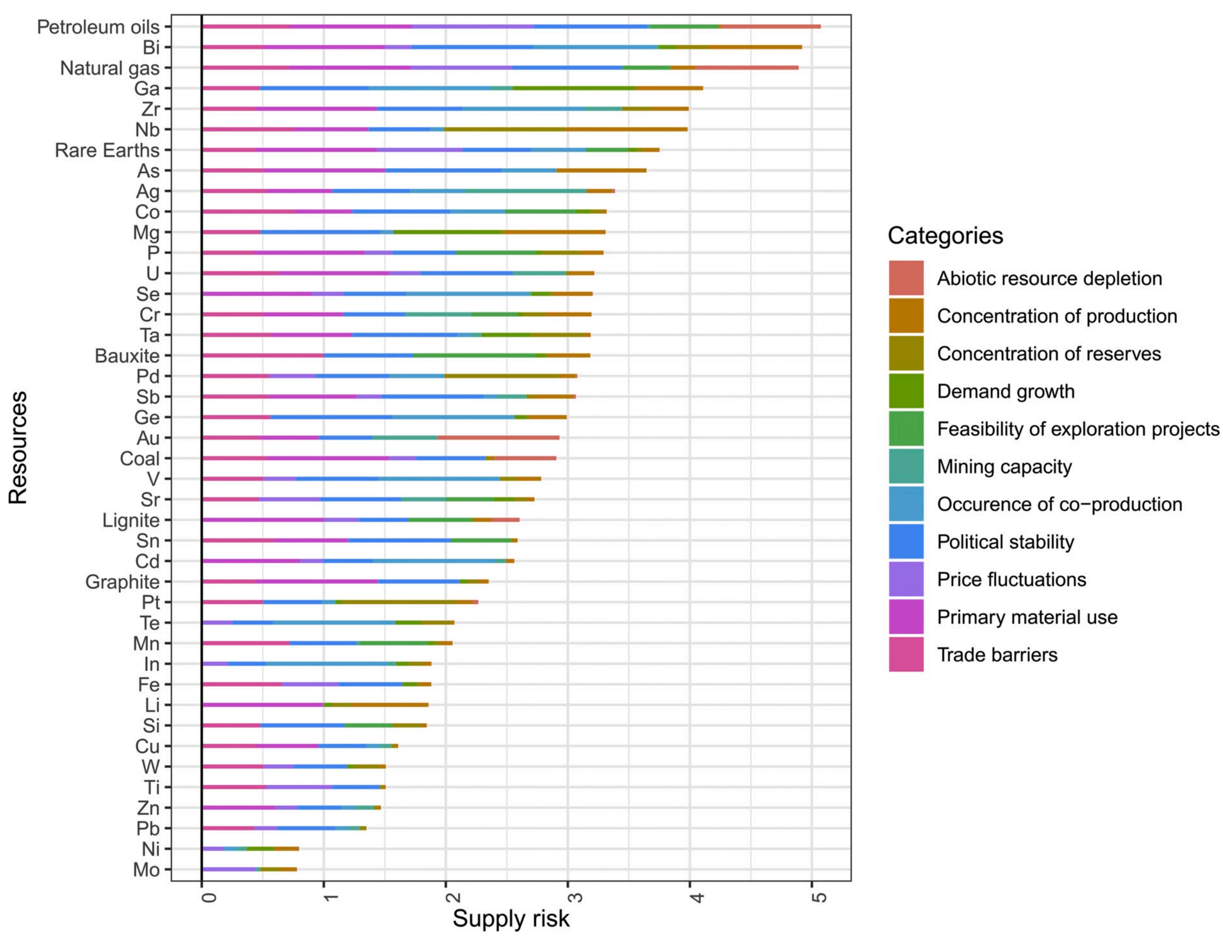

Fig. 4. Results for the sub dimension supply risk for all resources; since all indicators are scaled between zero and one and eleven indicators are assessed, the highest obtainable value would be eleven if one material had the highest value for all categories. Petroleum oils has the highest value with five and Molybdenum has the lowest value with under one.

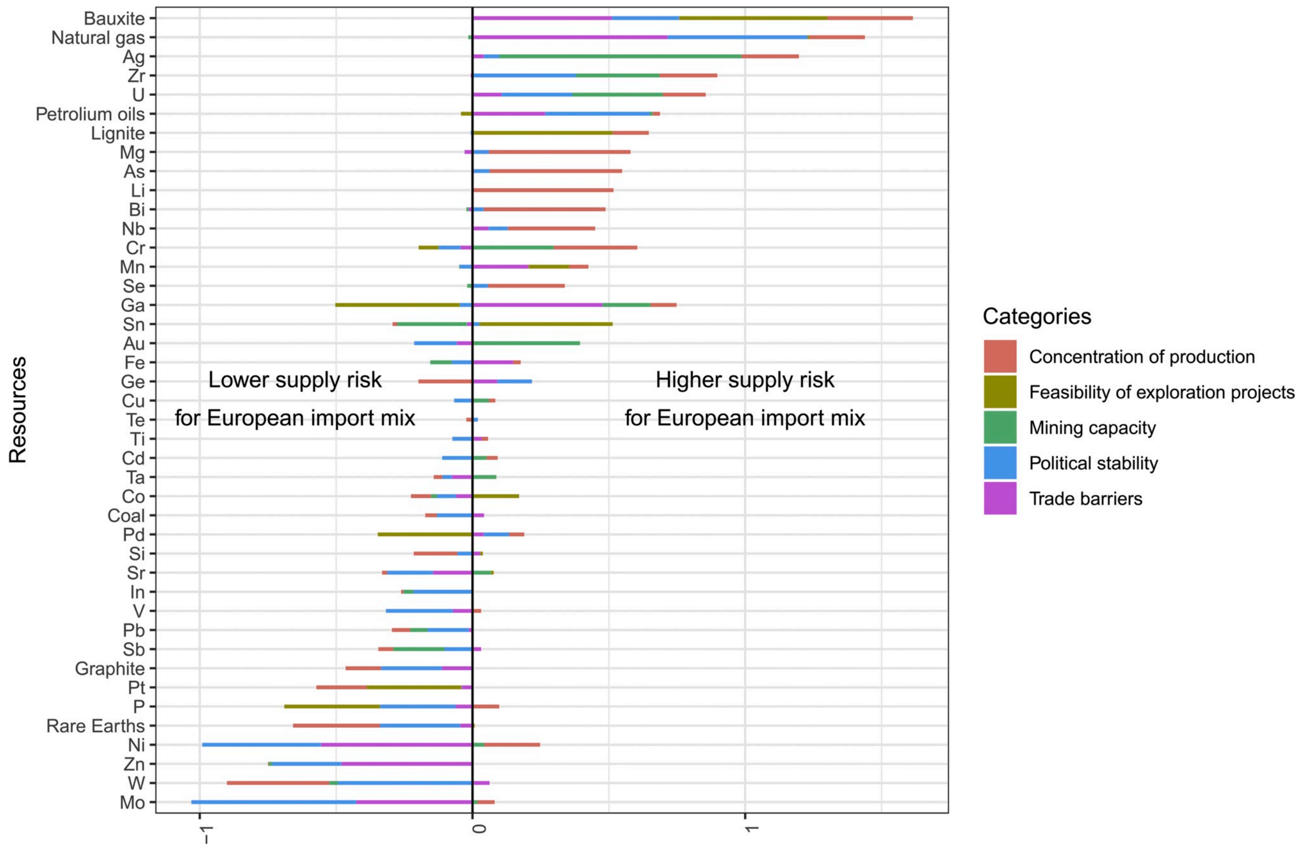

Fig. 5. Aggregated difference of the supply risk comparing the European import mix and the global production.

supply risk categories comparison (for molybdenum); and for the vulnerability assessment and the comparison of Europe and the global production in the compliance with environmental standards (for natural gas).

Bauxite imports into Europe have higher supply risk than the global average because of higher trade barriers, lower political stability and higher concentration of production. These supply risks occur because Europe imports 75% of its aluminum ores and concentrates from Guinea, which only represents 8.12% of world bauxite production. Europe imports only 3% of its aluminum ores and concentrates from Australia, which produces 27.61% of world's aluminium and has higher political stability and lower trade barriers compared to Guinea.

For silver, Europe's concentration of production is higher, because Europe imports 65% from Mexico, which is also the leading producer globally, but only holds 20.32% of the world production. This is also the reason for the significantly lower mining capacity compared to the global average. Mexico only has 6.5% of world reserves, which indicates that Europe might have to find an additional supplier in the future. It should be noted that the biggest silver reserves after Peru are located in Poland (15% of world reserves) and thus within Europe, which limits Europe's vulnerability to a supply constraint.

European tungsten imports have a lower supply risk compared to the global supply. Most European imports come from Canada and not from its global main producer China (representing 80% of global tungsten production), which has lower political stability. Europe's import portfolio is more diverse, and no importing country has a share over 35% (Canada). Therefore, the European tungsten import mix has lower supply risk due to political instability and high production concentrations. Results for the comparison of import-based categories (with the focus on the overall category performance for all resources not on a resource specific interpretation) to the global supply risk can be found in the supplementary materials (see section 2).

3.1.3. *Vulnerability*

In the following, we present the results of the sub dimension vulnerability with its categories *domestically required demand, share of global production, dependency on imports, economic importance, substitutability* and *utilization in future technologies*. The results are displayed in Fig. 6. It is visible that 32 materials cannot be supplied to Europe's economy by domestic mine production, because they have a value of close to one for the category *dependency on imports* (see section 1.3 in the supplementary material and Table S 3 for the equation of the indicator calculation). Nine of these resources (bismuth, tantalum, niobium, molybdenum, phosphorus, antimony, rare earths, vanadium and zirconium) are not mined in Europe at all.

Next, we will present the category results for the resources platinum and strontium in more detail because they have the highest and lowest results, respectively.

Platinum has the highest value for vulnerability because Europe imported close to 100% of the current world production, it is nearly not produced in Europe and is also very important for future technologies like fuel-cells and synthetic fuels. The imported masses for gold and platinum might be influenced by the fact that gold and platinum are used for investment (Graedel et al., 2015b) and are thus traded without having been mined physically before. Thus, they are imported as capital investment and not for the consumptive use in a certain product.

Overall, strontium has the lowest vulnerability score. It is only moderately important for the economy and future technologies but not easily substitutable. It has the lowest indicator value because Europe does not import a high amount of strontium and the domestic production (mainly in Spain) exceeds the imported amount by far. The categories *dependency on imports, share of global production* and *domestically required demand* have thus a low indicator score, resulting in an overall low value for the sub dimension.

It might surprise some readers that Europe is more vulnerable to

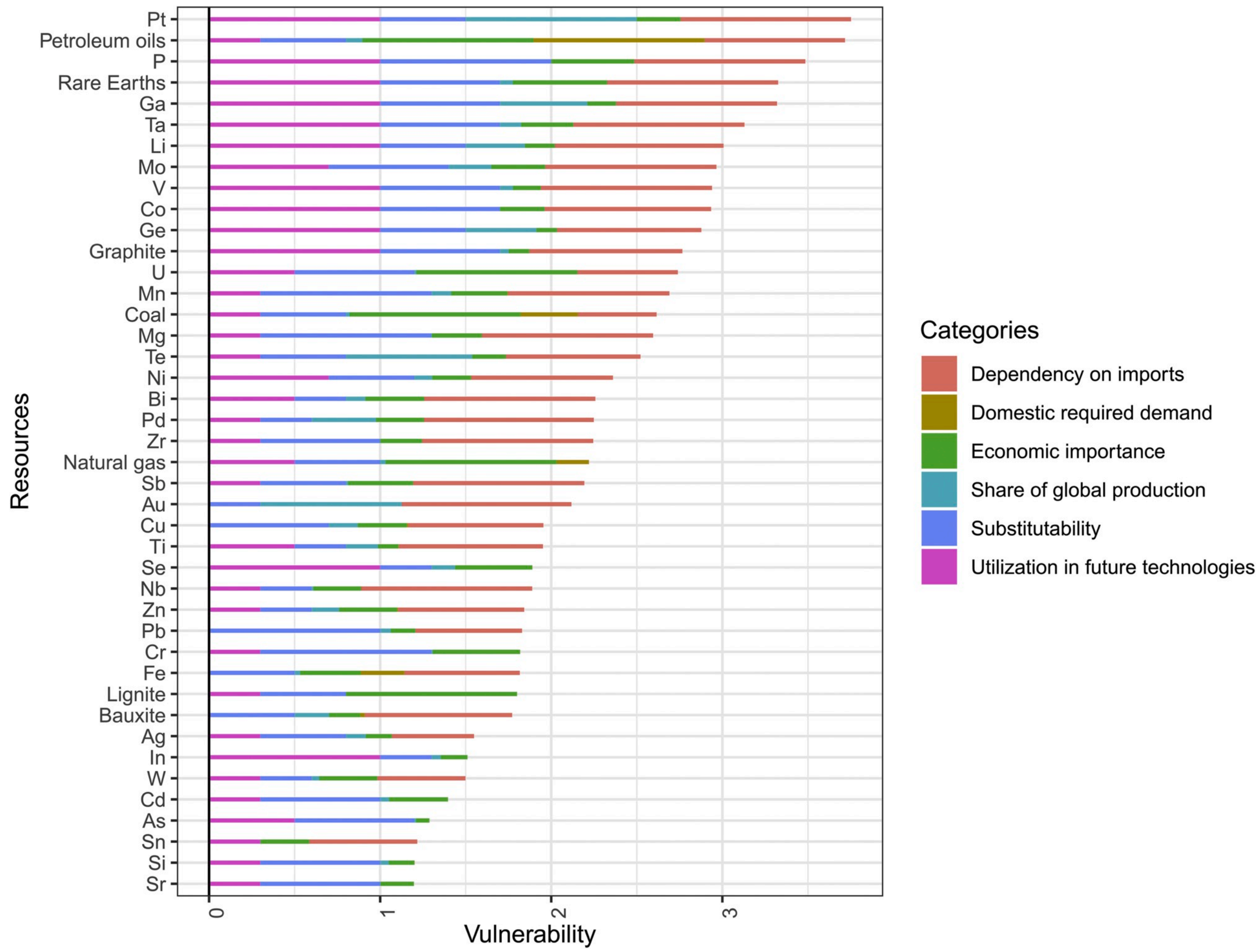

Fig. 6. Vulnerability of Europe to supply constraints; since all indicators are scaled between zero and one and six indicators are assessed, the highest obtainable value would be six if one material had the highest value for all categories. Platinum has the highest value with over 3.5 and strontium has the lowest value with just over one.

supply restrictions for petroleum oils than to natural gas. Their supply risk is in the same order of magnitude, while the vulnerability of Europe for supply constraints regarding natural gas is a lot lower in our assessment. This is because i) natural gas has a lower imported mass and thus a lot lower value for domestically required demand ii) the dependency on imports is quite high for petroleum oils because Norway is the only producing country in Europe. It only produces 2.2% of the world production while Europe imports around 10% of the global production. For natural gas we see a different picture: Europe produces 6% of the global production (Norway, United Kingdom and the Netherlands) and imports only around 4% of the global production. Thus, the dependency on imports is higher for petroleum oils than for natural gas.

3.2. *Societal acceptance*

In the following, the assessment results of the dimension societal acceptance is presented, divided into the sub dimensions compliance with environmental and social standards.

3.2.1. *Compliance with environmental standards*

Subsequently we outline the results of the sub dimension compliance with environmental standards with its categories *water scarcity*, *sensitivity of local biodiversity* and *climate change*.

It is visible in Fig. 7 that the results for *climate change* are very high for some materials (e.g. precious metals and arsenic) and very low for the rest. This can be explained by two aspects: i) economic allocation in LCA databases, which were used as a basis to determine the results (Berger et al., 2019) ii) the emitted greenhouse gas emission of mining and processing these materials are relatively high (Kahhat et al., 2019). However, the impacts are attributable to the very high spread of the indicator. The greenhouse gas emissions that are attributable to the production processes of the materials have a much higher spread than the water scarcity of the different countries. Next, the results for the resources with a high impact (gold, niobium) and some with low impacts (tantalum) are explained in more detail.

Gold has the highest risk to be not compliant with environmental standards. Its production has the highest greenhouse gas emissions and a rather high value for *sensitivity of local biodiversity* but only a moderate value for *water scarcity*. This is due to its mining in South America (Argentina, Chile, Peru), but also in the USA and Canada, which have sensitive biodiversity but only moderate water scarcity.

Niobium has the third highest risk due to its high value for water scarcity. Europe mostly imports it from Brazil, which has very water scarce regions. The major niobium mine in Araxa, though, is not within the respective regions (Dolganova et al., 2019), which indicates that the results would change significantly if local water scarcity instead of country averages would be used.

Tantalum has a low indicator value because it is mostly imported from Rwanda (29.5%), Democratic Republic of the Congo (DRC) (25.17%) and China (25.16%). DRC has the biggest surface water reserves in Africa (Partow, 2011) and also Rwanda has a low WDI score

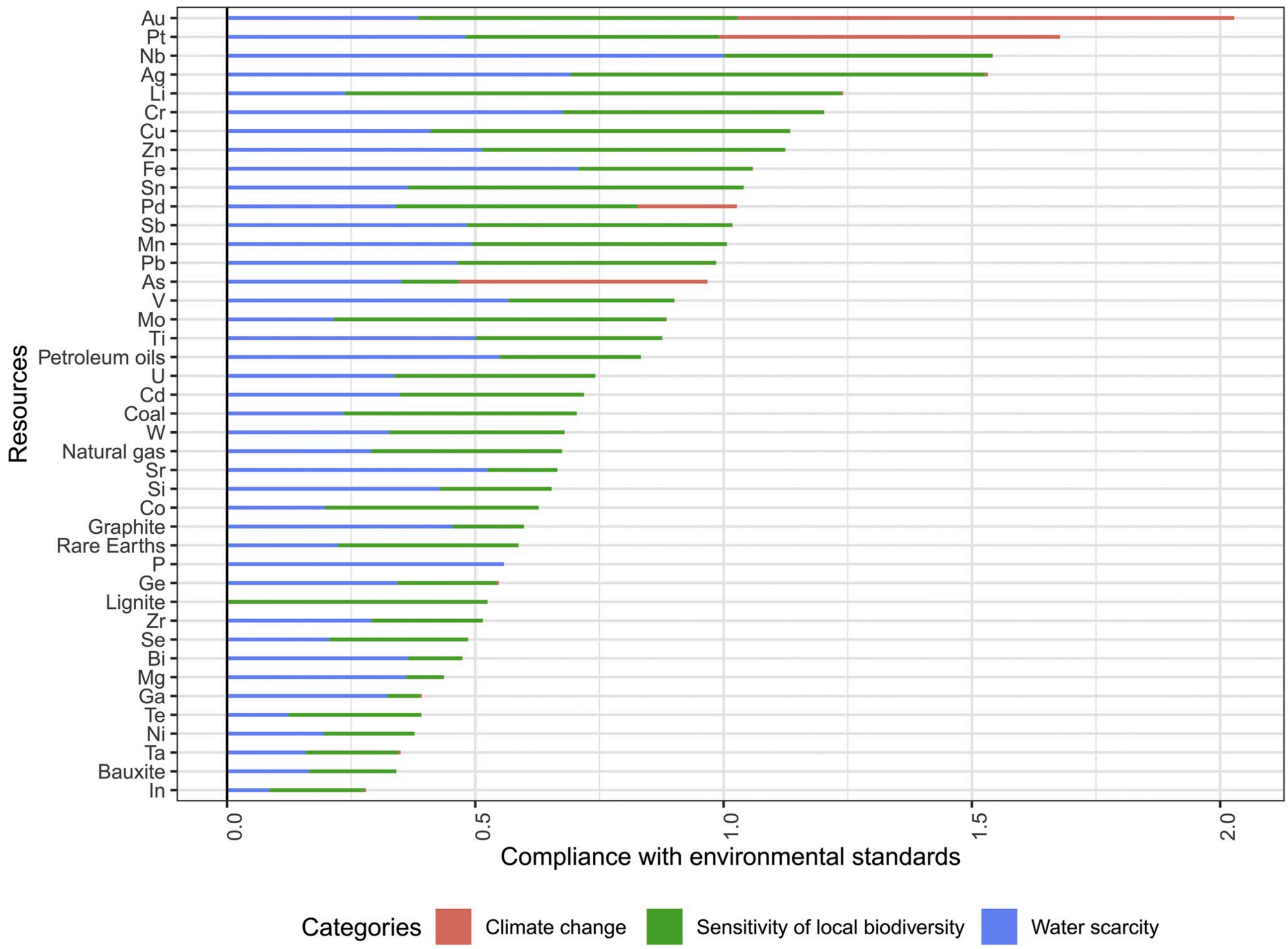

Fig. 7. Compliance with environmental standards for European import mix; since all indicators are scaled between zero and one and three indicators are assessed, the highest obtainable value would be three if one material had the highest value for all categories. Gold has the highest value just over two and indium has the lowest value with under 0.5.

since it has large water reserves and high rainfall. Both have been assessed to only have moderately sensitive biodiversity. China has high water scarcity, but because of its low import-share the overall value is low.

In Fig. 8 the comparison of the European import mix and the global production mix for the sub dimension *compliance with environmental standards* is displayed. The category climate change is not shown because it is not import-based and therefore not Europe-specific.

Next, we present some examples and further analyze the indicator difference for antimony, bauxite and nickel since they show extreme differences.

Antimony is the substance where Europe performs worse compared to the global average, because it is imported mostly from Turkey and Bolivia, which are both water scarce and have sensitive biodiversity. Globally, antimony is sourced more from Russia, China and Tajikistan, which are less water scarce and have less sensitive biodiversity.

For bauxite, Europe outperforms the global production mix in both categories because globally very biodiversity-rich countries like Malaysia and Australia dominate the supply, while European supply is dominated by Guinea, which is ranked with very small water scarcity and not very sensitive biodiversity.

For nickel, Europe's sub dimension result shows the best result compared to the global result. Europe receives its nickel mainly from Canada (65%), which only produces 10% of the global production and is neither particularly water scarce nor has very sensitive biodiversity. The countries that dominate the global supply and produce all around 10% of the global production have high biodiversity (Philippines, New Caledonia) or high water scarcity (Australia).

Next to the extreme values, we could also identify some resources that show trade-offs between biodiversity and water scarcity. Many materials in the European import mix have a better water scarcity value but worse biodiversity impacts (uranium, lignite, rare earths, manganese, molybdenum, copper, lithium). This is due to the more sensitive biodiversity of the importing countries, e.g. lignite is imported into Europe from Indonesia, that has very sensitive biodiversity, but close to no water scarcity. The global production mix is dominated by Australia, China and Bulgaria, which have higher water scarcity, but less sensitive biodiversity.

3.2.2. Compliance with social standards

In this subchapter, we present the results of the sub dimension compliance with social standards with its categories *small scale mining, human rights abuse* and *geopolitical risk* (see Fig. 9).

In the following, materials that exhibited high values in our social assessment, namely tantalum and cobalt, are presented, and the results for lithium, which has a low score, are discussed.

The material with the highest value in the category compliance with social standards is tantalum. Europe imports it mostly from Rwanda, DRC, Brazil, China and Burundi (in that order). Rwanda, DRC and China have very high values in the category *human rights abuse* and *geopolitical risk*. The DRC has ongoing violent ethnic conflicts (Stearns and Vogel, 2015). China and DRC belong to the ten bottom countries of the CIRI, because they have high prevalence of human rights abuse.

Cobalt has the second highest risk of non-compliance. It has

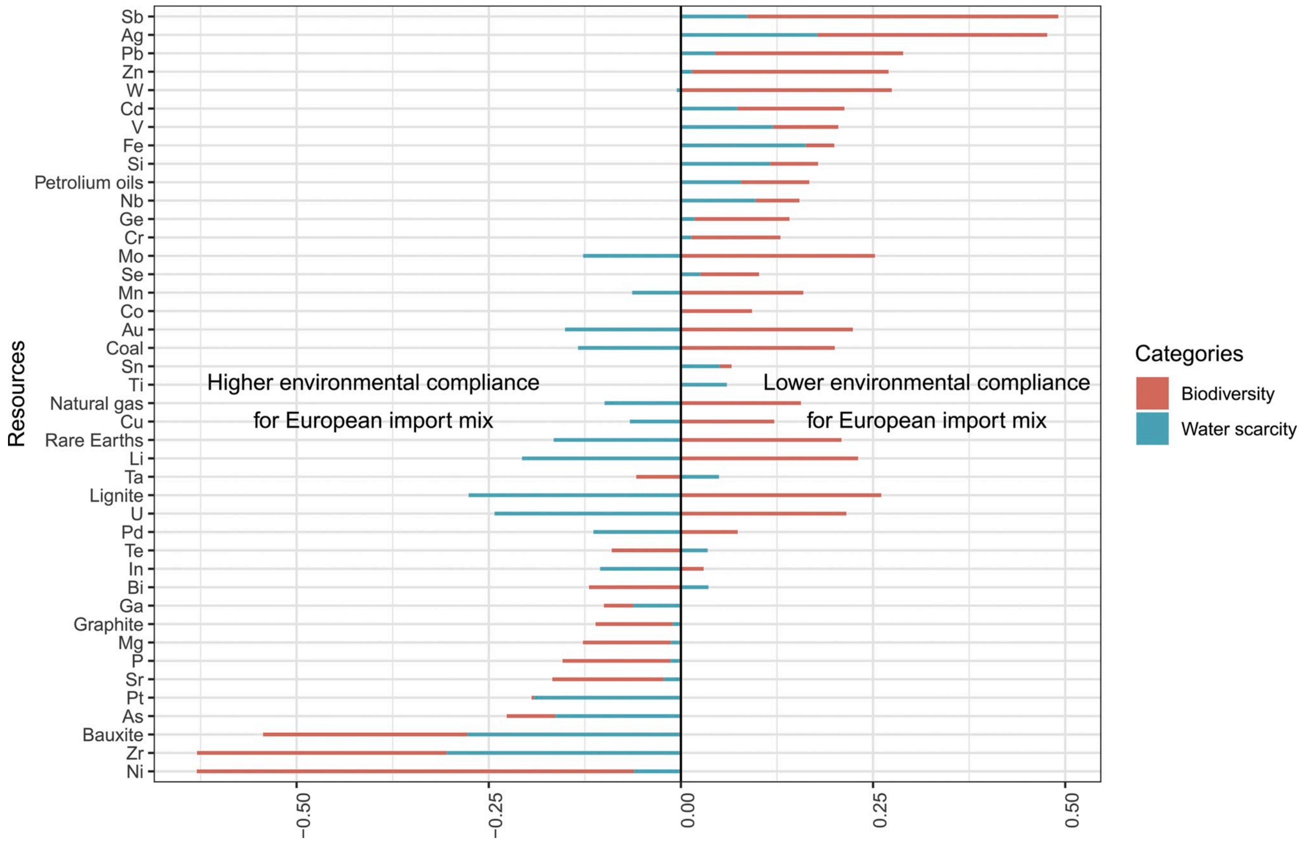

Fig. 8. Aggregated difference of the compliance with environmental standards comparing the European import mix and the global production.

prevalence *of small scale mining*, a high value for *geopolitical risk* and the highest value for *human rights abuse*. The cobalt import mix is dominated by DRC and Russia, while some of the imports also come from the US and Canada. The high value for *small scale mining* is mainly due to its import from DRC. Even though industrial mine production is rising in the DRC, most minerals are won in small scale mining in Congo (BGR, 2019). The *geopolitical risk* value is influenced by Russia and the DRC that have high prevalence of armed conflicts.

Lithium has the second lowest value. It is mostly imported from Chile, which is not associated with *small scale mining* and *human rights abuse* nor *geopolitical risk* in our assessment. The biggest producers are SQM and Albemarle Corp which are industrial producers (Sherwood, 2019).

Next, the comparison between Europe's performance and the global production mix is carried out for the sub dimension compliance with social standards. The results are visualized in Fig. 10.

Subsequently, differences of the category results are discussed for natural gas, uranium and antimony, since they show extreme differences.

Natural gas and uranium have a lower value globally than the European imports in compliance with social standards (60% from Russia, which has high *geopolitical risk* and *human rights abuse*). The main producer of natural gas is the USA (25.5%) that has lower *geopolitical risk* and lower *human rights abuse* than Russia (17.8% of global supply). The main uranium supplier worldwide is Kazakhstan (39.34%) and Canada (22%). Canada and Kazakhstan have lower *geopolitical risk* and *human rights abuse* than Russia. The results for compliance with social standards for Europe would thus be better if it reduced its imports from Russia.

Europe imports 72% of its antimony from Turkey, which has low prevalence of *small scale mining.* On a global level, 77% are supplied by China that has higher prevalence of *small scale mining* than Turkey for antimony. However, Turkey has higher geopolitical risk than China, which is why the European import has higher geopolitical risk than the global supply. However, the overall sub dimension result is better for the European import mix.

4. Discussion

Generally, a criticality assessment is a screening approach to identify a country's or region's dependence on certain materials. Thus, existing assessments including SCARCE can only show that a material is more or less critical in comparison to another material, but cannot predict the severity nor the probability of an occurring supply disruption in absolute terms. Moreover, the criticality is only related to raw materials (ores and concentrates) that are imported into Europe for further processing. It does not consider the dependency on intermediate products or manufactured goods that are imported into Europe and that can be crucial for European economic activity (the materials contained in smartphones produced in China and used in Europe are not considered, therefore no real life-cycle perspective is used). Additionally, our assessment showed the criticality of Europe in the past (reference year 2015) and cannot project future development of demand at this point, but the method could be used for the assessment of development scenarios. Next to the relative nature of the criticality assessment and the other previously stated limitations, one of the biggest challenges is the used data. The applied global production and reserves data rely mainly on the data from the U.S. Geological Survey. All errors that are contained in their supplied data is also reflected in our results. The supplied data is sometimes incomplete, because often information on the US economy are not disclosed in order to protect domestic economic activity. Further, the imported masses from comtrade and comext are based on trade value and therefore the mass flows we used in our assessment are only estimates. Illegal trade was not accounted for. The used indicators (e.g. ETI, WGI, PPI, etc.) also contain some uncertainty that is reproduced (and maybe even multiplied) in our assessment as well. Further, the determination of the import mix and the related assumptions can lead to

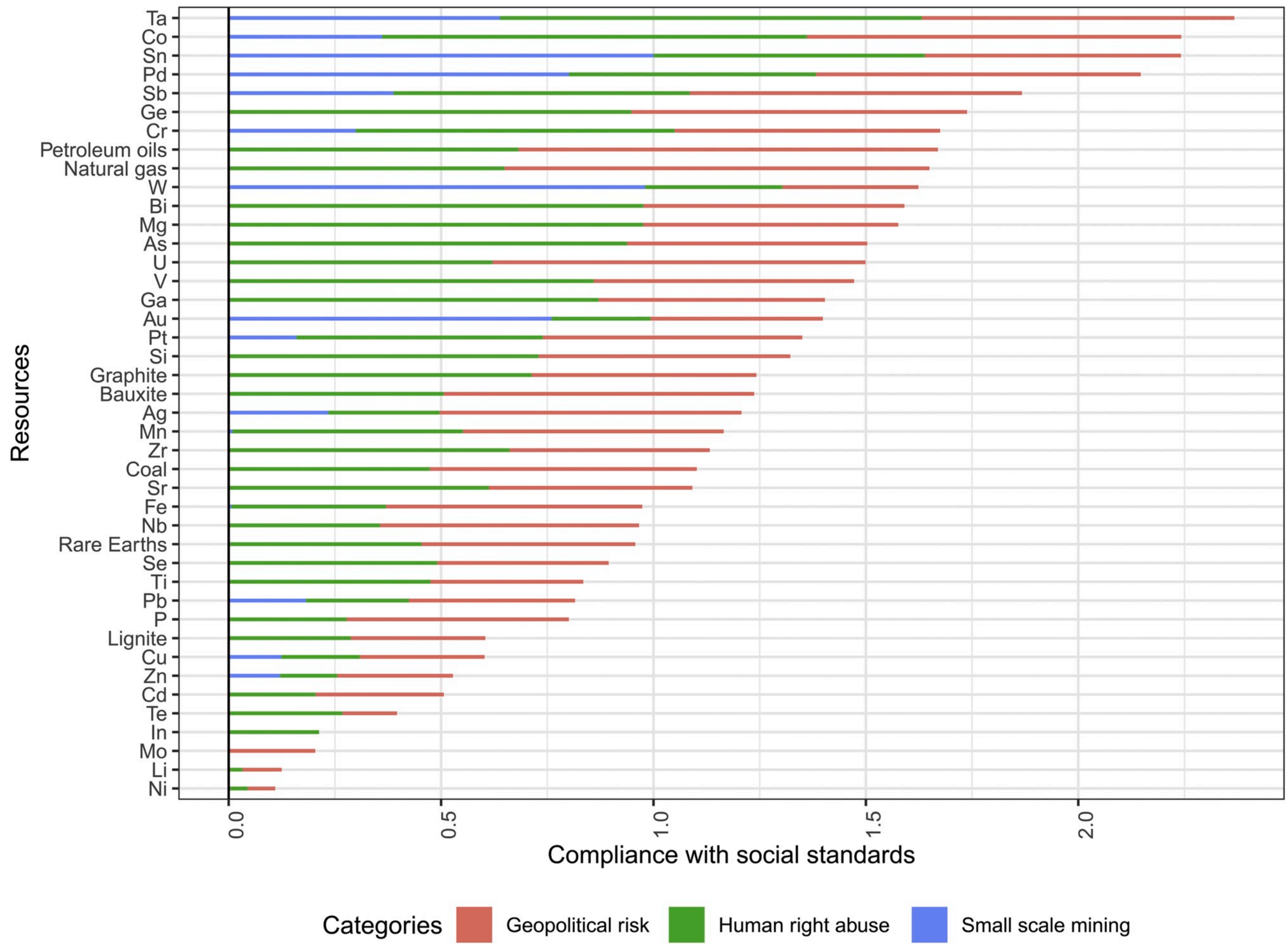

Fig. 9. Compliance with social standards for European import mix; since all indicators are scaled between zero and one and three indicators are assessed, the highest obtainable value would be three if one material had the highest value for all categories. Tantalum has the highest value of over two and nickel has the lowest value with under 0.5.

inaccurate results.

The analysis of the sub dimension *compliance with environmental standards* contains some shortcomings that are discussed with more detail in the following. For the category *climate change* we had two options: to account for the materials with the highest emissions per kilogram (relative emissions) or to account for the materials with the highest emissions of the total imports (absolute emissions). We wanted to highlight that not only mass matters. Therefore, we decided to show that different materials have different climate impacts per kilogram (the relative emissions). Also Graedel et al. (2015a) account for the environmental impacts per kilogram and came to the same conclusion, namely that most precious metals have a high environmental impact per kilogram. It is important to point out, though, that if we had chosen to use the total GHG emissions for the imported amounts, the results would look significantly different. Iron/steel and aluminium would probably be at the top because of their high domestic demand (Allwood and Cullen, 2012).

The categories water scarcity and sensitivity of local biodiversity are not production-specific but only country-specific. This leads to the result that two materials that are produced in the same country would have the same water scarcity value independent of the water demand or pollution that is required for their resource specific production process. This is why niobium, gold and silver have nearly the same indicator value in the category water scarcity, even though their production has different water demands. Comprehensive regionalized water footprint data covering all 42 resources is currently not available. However, this aspect should be included in a future update as soon as the necessary data is available.

With regard to determining environmental aspects, we only focused on the mining stage in our assessment and not the processing of the materials and the environmental impacts associated with it. A good example for this is alumina production in Guinea: in our assessment bauxite has the second best value for compliance with environmental standards (see Fig. 7), but Knierzinger (2018) outlined that alumina production in Guinea has significant environmental impacts. The by-product of alumina production (red mud) has been associated with direct discharge to rivers and groundwater pollution that made groundwater inconsumable (Knierzinger, 2018). Another example is tantalum (mined in Rwanda and DRC), which has a good value in its compliance with environmental standards. However, its producing countries are associated with small scale mining that can sometimes lead to severe environmental effects (Macháček, 2019).

In addition, the SCARCE method does not consider relevant environmental aspects like toxicity. Toxicity aspects have been included in other criticality assessments, namely the approach by Manhart et al. (2019). However, this approach contains shortcomings because it relies on semi-quantitative data only. As the publication did not disclose the results of all materials, the data could not be transferred to our assessment. Some other relevant environmental aspects addressed in the approach by Manhart et al. (2019) are also currently not applied in the SCARCE method, like consideration of co-winning with radioactive substances or the use of auxiliary substances. Although Manhart et al. (2019) include these aspects, they only provide semi-quantitative data of heterogeneous quality.

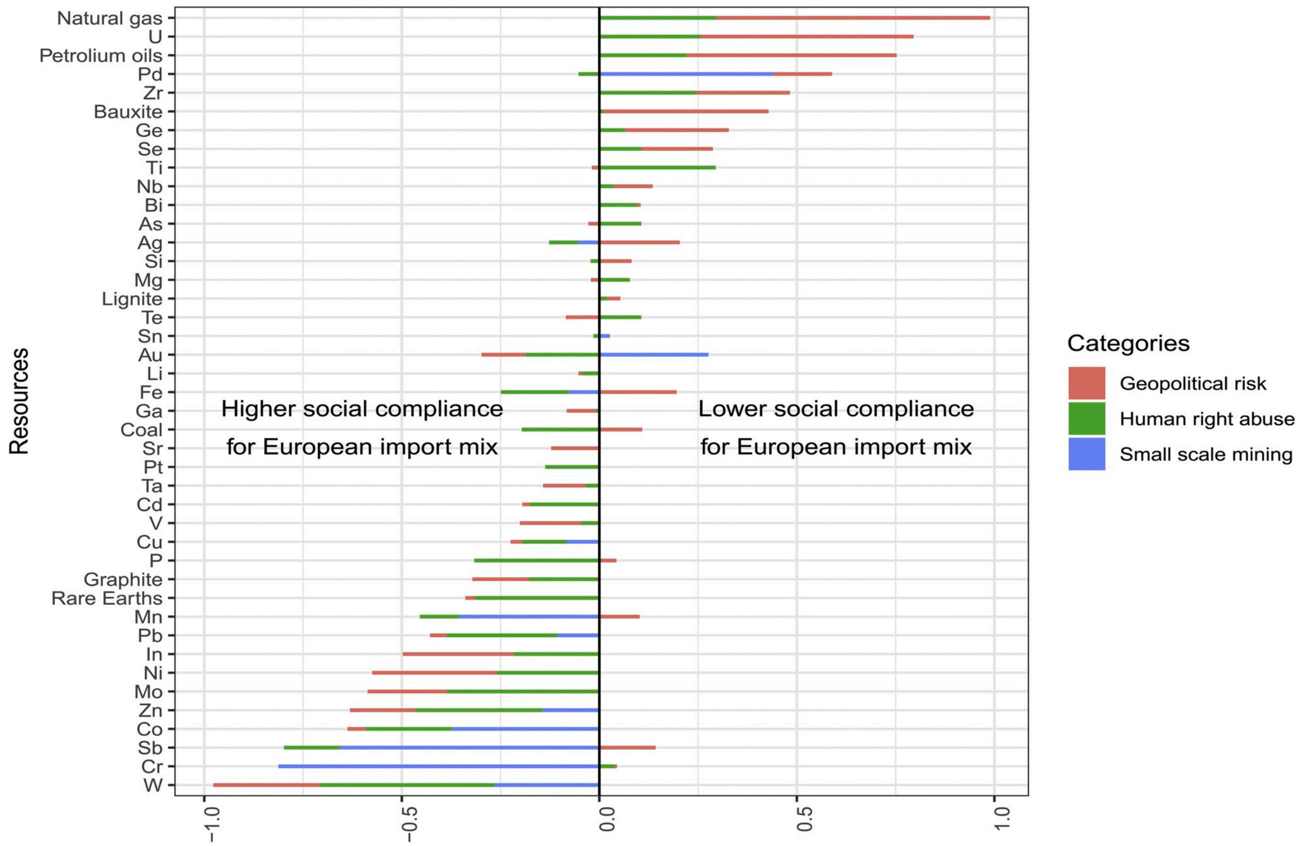

Fig. 10. Aggregated difference of the sub dimension compliance with social standards comparing the European import mix and the global production.

The sub dimension compliance with social standards relies on international statistics and indicators. The data on small scale mining are only estimates because no data on the exact share of small scale mining exists. Further, our assessment does not consider how the local community can be affected by mining. For example, Agusdinata et al. (2018) state that lithium mining in Chile is related to multiple environmental issues, which coheres with our findings (Li has the 5th highest score in risk of non-compliance with environmental standards), but also social risks to the local communities and indigenous exist (Li has the second lowest score of compliance with social standards), which is not considered in our assessment.

5. Conclusion

By applying the SCARCE method, we determined the criticality of 42 raw materials for Europe. Our assessment results are in line with the findings of the previously conducted study on critical raw materials by the European Union, but provide some additional insights: i) we determined the criticality for fossil fuels as well (which is important if resources for new energy technologies are considered to be critical because it should be possible to compare their criticality to the criticality of fossil fuels) ii) we integrated environmental and social hotspot analysis in our assessment iii) we applied a broader geographical scope by including Switzerland, Norway and Iceland, since all countries that are part of the European single market share similar availability constraints iv) we compared European criticality to the global criticality (apart from vulnerability). The most critical materials for the European economy are petroleum oils, rare earths and gallium. Our findings show that the European supply risks do not differ significantly from the global supply risks. There are exceptions for some materials though, e.g. bauxite or natural gas that have higher supply risk compared to the global supply. These are resources that are mainly imported from politically unstable countries like Russia or Guinea. Our findings show that the results of the environmental and social analysis do not correlate with the economic results. Therefore, it is important to consider them in addition to supply risk and vulnerability, as well as separately from these dimensions. Thus, these aspects should be included by the EU during the next method update.

Declaration of competing interestCOI

None.

CRediT authorship contribution statement

Rosalie Arendt: Writing - original draft, Data curation, Formal analysis, Visualization, Investigation, Methodology, Conceptualization, Writing - review & editing. **Marco Muhl:** Writing - original draft, Conceptualization, Project administration, Methodology, Writing - review & editing. **Vanessa Bach:** Supervision, Writing - original draft, Writing - review & editing, Project administration, Methodology, Conceptualization. **Matthias Finkbeiner:** Funding acquisition, Supervision, Writing - review & editing, Writing - original draft, Conceptualization.

Acknowledgements

The work presented here was prepared in the context of the Research Council of Norway project "Environmental assessment of electric vehicle deployment pathways for Europe" (project number: 255199/E20) and the Bundesministerium für Bildung und Forschung project "R2Q: RessourcenPlan im Quartier" (funding number: 033W102E). We would like to thank the Research Council of Norway and the Bundesministerium für Bildung und Forschung for their financial support. Moreover, the authors thank Natalia Mikosch, Joseph Poligkeit and Luis Rott for supporting the research of the underlying data. We thank two anonymous reviewers for their careful reading of our manuscript and their insightful comments and suggestions.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.resourpol.2020.101650.

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3.3 The global environmental costs of mining and processing abiotic raw materials and their geographic distribution

In this section we present the publication:

Arendt, Rosalie; Bach, Vanessa; Finkbeiner, Matthias: The global environmental costs of mining and processing abiotic raw materials and their geographic distribution. *Journal of Cleaner Production* **2022** https://doi.org/10.1016/j.jclepro.2022.132232

This publication assesses the global environmental costs of global material production. It does so by combining LCA and monetization. The approach delivers an answer to research targets 2 a and b as it assesses, which materials cause the highest environmental costs on a global level. It assesses the impacts of global material production, with material and country resolution, which answers research question 3a). All impact categories from the ReCiPe impact assessment method, excluding impacts to freshwater and marine ecosystems were considered. This method and results could be used to complement criticality assessment. Furthermore, a high and low approach to monetize was used to quantify the environmental impacts, which underlines the ranges and uncertainties associated with such an assessment, which contributes to research question 1.

Journal of Cleaner Production 361 (2022) 132232

Contents lists available at ScienceDirect

Journal of Cleaner Production

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

The global environmental costs of mining and processing abiotic raw materials and their geographic distribution

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ARTICLE INFO

Handling Editor: Zhen Leng

Keywords:
External costs
LCA
Metals
Minerals
Extraction
Sustainability

ABSTRACT

The global demand for abiotic resources is rising, and continuous extraction has adverse effects on the environment. In this study, we quantify the total global environmental costs by monetizing the life cycle assessment results of mining and processing 38 abiotic materials. Results are shown in terms of material and affected country. We assess impacts covered by the ReCiPe impact assessment method, excluding impacts on freshwater and marine ecosystems and some crucial local and cultural impacts. Furthermore, we compare the environmental costs with the mining GDP for each country. The total environmental costs range from €0.4 to €5 trillion annually. We find that the highest absolute environmental costs can be attributed to the greenhouse gases, particulate matter emissions and acidification caused by coal and steel specifically. The countries with the highest absolute costs are China and India, because China extracts and processes large amounts of materials, while India carries a large share of the climate damages that global material production causes. The countries that have the most beneficial ratio of environmental costs per GDP are countries that process materials, such as Japan and Germany (steelmaking), and countries that mostly extract oil, as Algeria, Azerbaijan and Nigeria, because they largely externalize environmental costs of upstream and downstream processes. For materials that have a worse relationship between economic gains and environmental impact, or that have high down-stream impacts such as fossil energy carriers, circular economy, material efficiency and substitution strategies should be prioritized to reduce global environmental degradation.

1. Introduction

Material extraction, in particular that of metals and fossil fuels, is growing on a global scale—an acceleration that has been identified since the beginning of the 21st century (Krausmann et al., 2018). This acceleration is associated with growing environmental impacts (IRP et al., 2019). However, the extraction of materials is seen as a possible contributor to sustainable development by industry associations such as the International Council on Mining & Metals (2020). Thus, mining leads on the one hand to substantial environmental degradation, while providing employment and necessary materials for economic development on the other hand. This has been underlined by Monteiro et al. (2019) in showing that mining leads simultaneously to the improvement of some economic Sustainable Development Goals (SDGs) and the deterioration of other social and environmental SDGs. However, the view that resource extraction contributes to sustainable development is controversial. Some researchers connect the pattern of resource extraction with the "resource curse" as well as imperialism and intentional underdevelopment, especially on the African continent (Bond, 2006; Davis and Tilton, 2005). Therefore, they see the risk of unequal exchange in the expansion of the mining of materials (Bonds and Downey, 2012), which dispossesses poorer populations in the Global South. Other researchers found positive effects of the mining sectors on non-mining economic activities, e.g. growth of the non-mining GDP (Ericsson and Löf, 2019; Medina, 2021). Next to such macro-economic and national perspectives, many adverse social impacts can be linked to mineral resource extraction despite possible positive economic effects. These impacts mostly hit the local communities that live close to the mine sites (Conde, 2017; Que et al., 2018).

Nevertheless, none of the aforementioned studies have compared the environmental impacts of mining on the economy with possible economic gains on a quantitative level. Such a comparison could help to identify whether the effects of the mining sector on the economy might be material-specific. For example, for some materials the economic benefits of extraction and processing could exceed the environmental costs, and vice versa. Additionally, the degree of industrialization could

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https://doi.org/10.1016/j.jclepro.2022.132232
Received 23 December 2021; Received in revised form 26 April 2022; Accepted 11 May 2022
Available online 17 May 2022
0959-6526/

have an influence on the environmental and economic impacts of material extraction.

Life Cycle Assessment (LCA) is an environmental management tool that assesses the environmental impacts of products and services along their life cycle (ISO, 2006a; 2006b), and has been applied to assess the environmental impacts of resource extraction. Recently the tool is used more and more to assess the large-scale environmental impacts of production and consumption patterns in society. This has been done through coupling LCA with material flow analysis (De Meester et al., 2019; Ismail and Hanafiah, 2021; Turner et al., 2016), applying LCA to cities (Cremer et al., 2020), the mining sector in a given country (Strezov et al., 2021) and organizations (Martínez-Blanco and Finkbeiner, 2017). The environmental impacts that are quantified in an LCA study are grouped into categories that are measured by impact assessment methods. So far, the application of LCA to the impact of resource use has been heavily debated (Berger et al., 2020; Sonderegger et al., 2020). Depending on the research questions, the assessment of a finite stock or the short-term availability of resources for production should be in focus. However, the environmental impacts of land use change, and environmental releases caused by extraction should be always accounted for in an LCA context.

A common LCA impact assessment in the EU is ReCiPe, which assesses the impacts with respect to mid- and endpoints (Goedkoop et al., 2008; Huijbregts et al., 2017). While the midpoints are linked to potential environmental impacts and are more scientifically sound, the endpoints allow for a more damage-based assessment through the translation of potential environmental impacts into threats to human health or ecosystems. Even though the endpoints of human health, species loss per year and resource use can communicate damages, they cannot be compared to each other, as they have different units, and cannot be related to economic gains that are caused by products or sectors. Monetization is an approach to assessing the impacts of pollutants on economic welfare by expressing them in monetary terms. Different options for how to monetize impacts using LCA exist, such as damage costs or abatement costs (Amadei et al., 2021; Arendt et al., 2020; Pizzol et al., 2015).

With this research, we aim to determine the negative environmental impacts of mining and processing materials on a global level using LCA. The impacts are assigned to the countries where these materials are mined and processed. Country-specific production processes from LCA databases were used if available, while mostly global average data had to be used to quantify the environmental impact. To obtain metrics that can be compared with other economic indicators, the LCA results are monetized. To reach this aim we will:

- Assess the environmental costs of the global production of 38 abiotic resources on a country level by using LCA coupled with monetization
- Compare the calculated environmental costs with employment in the mining sector and mining's contribution to the GDP
- Identify which materials and countries have a high or low ratio of employment or GDP contribution by mining and processing to environmental costs

The paper is structured as follows: following this section, the methods of determining the costs and the economic indicators are described (see section 2). Then the results are shown (section 3) and discussed (section 4). The paper closes with a short concluding section (section 5).

2. Method

To assess the environmental costs of global abiotic resource extraction and production, six steps were taken that are explained in the following and visualized in Fig. 1.

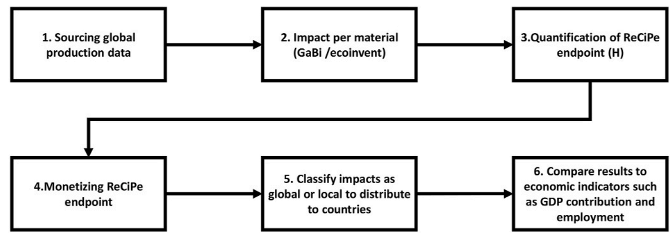

Fig. 1. Six steps to relate environmental costs of resource extraction and production to economic effects.

Step 1. First, we sourced the global production data of all 38 analyzed materials from the United States Geological Survey (USGS) (United States Geological Survey, 2016) and British Geological Survey (BGS) (British Geological Survey, 2019). The materials contained all major metals and metalloids as well as fossil energy carriers. Building materials were excluded. This step delivered the information on how much material is produced in each country.

Step 2. In the next step, we extracted data on the environmental impacts of these materials from the GaBi database (Sphera Solutions Inc., 2021), and if not available there from the ecoinvent database (Ecoinvent, 2021). The underlying processes are documented in the supplementary materials (section 2). Region-specific processes were taken if available from GaBi or ecoinvent; otherwise global averages or processes from other countries had to be used. Generally, we followed a cradle to gate approach and aimed to assess the impacts of delivering a certain mass of material. For the fossil energy carriers, the combustion was excluded to allow for a cost-benefit perspective from the extracting countries' point of view. The LCA impact assessment covers some impacts like acidification, but neglects others such as impacts on the view or on noise and cultural value. An overview of which impacts are covered or neglected is given in supplementary material section 1 based on scientific literature related to the topic (Finkbeiner et al., 2014; Huijbregts et al., 2017; ISO, 2006a; Manhart et al., 2019).

Step 3. Subsequently, for all materials the ReCiPe (H) endpoint values were obtained per kg of material. This was done to determine the damage of the material production to the endpoint and the areas of protection (AoPs) "ecosystems" (measured in species*year/kg), "human health" (measured in DALY/kg) and "resources" (measured in $/kg). Generally, we followed a cradle to gate approach. Which part of the processing is included is documented in the supplementary materials (section 2) as well as the system boundaries of mining and processing for a cradle to gate analysis (Fig. S1); for a detailed explanation of the system boundaries, see the documentation of the supplied processes (Table S1).

Step 4. For the endpoints, monetization factors were derived. The sources for the monetization approaches are displayed in Table 1. The approaches represent a damage-based assessment. They cover all the ReCiPe AoPs and seek to identify which of the impact categories are the most significant. To be able to show a range and underline the uncertainty that is associated with monetization, we used a high and low

Table 1
Monetization of the three areas of protection.

Area of protection	High	Low
Ecosystems [species*yr]	Average value from Kuik et al. (2008) as in Bruyn et al. (2018)	Median value from Kuik et al. (2008)
Human health [DALY]	European Commission (2009)	Annual average global income similar to Steen (2016) and Weidema (2009)
Resources [$]	Already monetized based on Huijbregts et al. (2017)	Assume no externality due to surplus costs following Jowitt et al. (2020)

monetization factor for each respective endpoint.

The high disability adjusted life year (DALY) valuations are based on the European impact mitigation regulation (European Commission, 2009), and the low value is derived from the average per capita income globally as a proxy for willingness/ability to pay, following Steen (2016) and Weidema (2009). For the species valuation the high and low values are obtained from the average and median valuation of Kuik et al. (2008), respectively. The transformation of the biodiversity unit potential disappeared fraction of species (PDF)/m^2yr is described in Arendt et al. (2022), through the division of the value per square meter by the global species density. For abiotic resources, we choose the “future effort” method for the high estimate, which calculates extra costs that will have to be paid due to declining ore grades. However, Jowitt et al. (2020) show that such surplus costs have still never occurred, as technological improvement has offset the impact of declining ore grades, and that the relative cost per kilogram of material has declined. This is why the low estimate assumes no external costs for the effects on the AoP “resources”. The surplus cost approach can be interpreted as a damage cost approach, as it quantifies costs that future generations will have to pay to obtain the same materials that we can obtain today at a lower price. The high and low monetization approaches do not differ significantly from a methodological perspective apart from the monetization of metal and fossil resource extraction, which is assumed to be zero for the low estimate. For biodiversity and human health, the approach is the same but places a higher value on the AoPs, for example biodiversity for the high estimate uses the mean value, which was significantly higher than the median in the meta study by Kuik et al. (2008). A similar debate has been carried out regarding human health monetization—whether the income of the highest income group should be taken as a reference (as has been done by Weidema (2009)) or corrected with the income of the different countries (as has been done by Rauner et al. (2020)). In this study we followed the recommendation of Weidema (2009) to value the loss of a DALY equally globally, but once with the proposed relatively high DALY value of an average EU citizen and once with the value of global average income.

Overall, two scenarios—one high and one low scenario, from which six additional scenarios can be derived by combining the low and high monetization of the three AoPs in all possible ways, thus eight scenarios in total—are derived. All species*yr values are scaled according to the Ecoregion factor of the countries, as described in Arendt et al. (2022). However, for climate change global values are applied, as it causes damage globally. For ecosystems, only the impacts related to terrestrial ecosystems were considered, as the values by Kuik et al. (2008) refer to terrestrial ecosystems. All money metrics are converted to Euro with purchasing power parities and inflated to 2018 values with the consumer price index. To put the derived costs into perspective, the yielded underlying costs for the low valuation of one ton CO_2 result in a CO_2-price of €20 per ton CO_2 and with the high valuation €163 per ton CO_2.

Step 5. As we aim to compare the negative environmental effects with economic effects (such as GDP and employment) on a country level, the environmental impacts of extraction need to be assigned to countries. Therefore, impacts are classified as local or global. This classification is simplified by assuming all impacts to be local impacts apart from global warming. Therefore, trans-boundary effects of air, soil and water pollution are neglected. Furthermore, the impact category stratospheric ozone depletion was excluded, as no reasonable distribution to countries could be identified and the impact is overall small compared to the other impacts (0.002% of the total costs). Moreover, metal extraction and fossil resource extraction are also assumed to cause local costs carried by the producing countries, as the additional costs of extraction will reduce resource rents. However, in Arendt et al. (2022) we chose a different approach. We assigned the damages calculated with the Hotelling rule to countries that use the respective materials as inputs. An ideal approach would distribute the costs from declining resource rents to producing countries and the damages of forgone production benefits to countries that use the materials as inputs for further economic activity, but such an approach does not yet exist. Regarding resources, it is assumed that the negative impacts of additional extraction costs are carried by the countries themselves in the high estimate, for in the low estimate they are zero, as shown in Table 1.

The cost per material is calculated according to equation (1):

$$c_m[€] = \sum_i^n m_m[kg] \cdot \frac{a_i}{m_m[kg]} \cdot x\left[\frac{€}{a_i}\right] \tag{1}$$

where c_m stands for the costs of material m, m_m stands for the global production of material m in kilograms. m_m is multiplied by all environmental impacts a_i, which are then multiplied by the respective monetization factor x that quantifies the external costs per impact a_i. This is done for all considered impact categories. Then we obtain the total environmental costs caused by material m in all considered impact categories.

To calculate the impacts per country, equation (2) is applied:

$$c_c[€] = \sum_m^n m_{m,c}[kg] \cdot \sum_i^n \frac{a_i}{m_m[kg]} \cdot x\left[\frac{€}{a_i}\right] \tag{2}$$

where c_c [€] stands for the costs in country c, $m_{m,c}$ stands for the amount of material m produced in country c, which is then summed over all materials that are produced in that country with their respective emission intensities a_i. The emissions are then multiplied by the respective monetization factor x'. For climate change, a different equation has to be used, because CO_2 causes global damages and the regions are affected differently. The ReCiPe method does not deliver geographically discreet damage factors (Huijbregts et al., 2017). As the DALY distribution also occurs due to floods and storms and the respective areas will also be hit particularly hard in terms of biodiversity loss, we distributed in the following way:

We assume that human health and species damages by climate change are proportional to economic damages, and distribute the calculated damages based on the country-level social costs of carbon determined by Ricke et al. (2018). Of course, it would be advantageous to distribute the costs according to biodiversity damages and human health damages only, but country distributions for these impacts are not available to our knowledge. To calculate their costs, the total externalities of carbon emissions are calculated following equation (3):

$$c_{co2,\ total}[€] = \sum_m^n m_m[kg] \cdot \frac{a_i[kg\ CO_2 - eq.]}{m_m[kg]} \cdot \left(b\left[\frac{DALY}{kg\ CO_2 - eq.}\right] \cdot c\left[\frac{€}{DALY}\right] + d\left[\frac{species*yr}{kg\ CO_2 - eq.}\right] \cdot e\left[\frac{€}{species*yr}\right]\right) \tag{3}$$

where $c_{co2,total}$ stands for the total costs that occurred due to CO_2 emissions, m_m stands for the mass of material m, a_i for the emission intensity of material m, b for the DALY intensity per CO_2-equivalent, c for the monetary factor per DALY, d for the species*yr factor per carbon emission and e for the monetization factor for damages to species.

The total costs, as calculated above, are distributed to countries based on the data by Ricke et al. (2018) following equation (4):

$$c_{co2,\ c}[€] = r_c\left[\frac{€}{tCO_2}\right] \cdot \left(r_{tot}\left[\frac{€}{tCO_2}\right]\right)^{-1} c_{co2,\ total}[€] \tag{4}$$

Where $c_{co2,c}$ stands for the cost of carbon for country c that is caused by the global emissions caused by mining and processing, r_c stands for the country-specific costs of carbon for country c for one ton of carbon, which are obtained from Ricke et al. (2018), and r_{tot} for the global costs of carbon for one ton of carbon. r_c is divided by r_{tot} and thus derives a dimensionless percent share that quantifies the country-specific share of global damages for the emission of one ton of carbon. This percent share is then multiplied by the total costs of carbon from the global material production obtained from equation (3) $c_{co2,total}$. We did this for all countries to distribute the impacts to the different countries.

The costs per country are calculated twice, once with CO_2 impacts and once without, to identify which share of damages only occur due to local mining (domestic costs), but also to identify the relation to the total damages (damage costs). All costs are set in relation to the population estimate of the respective countries to obtain a damage score per capita.

Step 6. As a final step, the environmental costs are set in relation to the share of GDP that is generated by mining. For minerals we took the total production value (International Council on Mining & Metals, 2020), while for oil and gas only mineral rents were available (World Bank Group, 2021a, 2021b). Mineral rents are the difference between production costs of a mineral at world market price and the production costs in a certain country that are below the world market price (World Bank, 2021). As the environmental impacts of processing (e.g. steelmaking and aluminum production) were considered as well, the GDP generated by basic metal manufacture, e.g. steel making or production of aluminum ingots, is included and based on the UNIDO database (UNIDO, 2020). The ratios were calculated the following way for country c:

$$\frac{GDPcont_c}{tc_c} = \frac{TPV_c + OR_c + NGR_c + BMGDP_c}{tc_c} \tag{5}$$

where $GDPcont_c$ stands for the GDP contribution of mining for country c, tc_c stands for the total environmental costs in country c, TPV_c stands for total production value of minerals and metals, OR_c stands for the oil rents of country c, NGR_c for the natural gas rents of country c and $BMGDP_c$ stand for the GDP contribution of basic metal manufacturing for country c. Thus, the GDP contribution of oil rents, natural gas rents, total production value of minerals and coal mining as well as the GDP contribution made by the manufacture of basic metals are summed and set in relation to the environmental costs, where tc_c can stand for total domestic costs (excluding CO_2-damages) or damage costs (including CO_2-damages).

For employment in the mining sector, only a limited number of countries supply data, so that the ratio of external costs to employment can only be calculated for such countries. Artisanal small-scale mining was based on (Hilson, 2016). For other countries, diverse sources were used (Argentina; Instituto Nacional de Estadística y Censos (Argentina), 2021; Bocangel, 2001; Bureau of Economic Analysis (U.S.A.), 2020; CEIC Data, 2021; Commonwealth of Australia, 2021); Consejo minero, 2021; EMIS, 2019; Eurostat, 2021; Finnish Minerals Group, 2021; Gálvez Delgado et al., 2020; Garside, 2020; Gunson and Jian, 2001; Hampel-Milagrosa and Siba, 2019; IndustriALL, 2019; Instituto Brasileiro de Mineração, 2021; Matthysen, 2015; Minerals Council South Africa, 2020; Ministère de l'Economie et des Finances-Direction générale du Trésor Francais, 2020; Philippine Statistic Authority, 2018; Puri-Mirza, 2021; Statistics Botswana, 2016; Statistik Austria, 2021; TDRA, 2021; The Economic Times, 2019). The numbers and sources are supplied in supplementary materials section 3.

3. Results

The presentation of the results is structured in the following way: We start with the cost distribution per material (section 3.1) and identify the dominating impact categories. Then the costs are distributed to countries and shown per capita (section 3.2). Subsequently, the distribution of costs for the most relevant impact categories in terms of their cost shares will be outlined. The remaining impact categories are shown and interpreted in the supplementary material section 4.1. Lastly, the relation between the environmental costs caused and the GDP and employment generated are outlined (section 3.3). Within the paper, only the low estimate is shown for a minimum estimate. The high estimate is presented in section 4.2 of the supplementary materials, but ranges will be addressed in the paper.

3.1. Cost distribution per material

The overall environmental costs generated by global resource extraction, in the low estimate of the outlined approach, are €0.4 trillion per year. Compared to €80 trillion yearly world GDP, this represents 0.5%. The maximum estimate is €5 trillion per year, thus more than ten times larger and representing 6.4% of global GDP. Fig. 2 shows the mass share of the different materials (left), the cost distribution to materials (middle) and impacts (right) for the low estimates of the different materials. The respective figure for the high estimate is provided in section 4.2 of the supplementary materials (Fig. S17).

On the left side of Fig. 2, it is visible that fossil energy carriers dominate the total material mass. Coal is the material that is extracted in the highest amount (44%) of the total mass, followed by crude oil (25%), natural gas (14%) and iron (8%). The remaining materials that represent more than 1% of the total mass are lignite (6%) and phosphorous (2%). While mass is a factor in determining whether a material has a high impact, there is only a small overlap between the materials with the highest produced mass and respective costs as shown in Fig. 3, where we visualize the costs related to the produced mass.

Focusing on the cost distribution (Fig. 2 in the middle), only seven materials cause more than 5% of the total costs, while 31 materials cause below 5% and 26 below 1% of the costs. The materials with the highest absolute costs are iron (19%), coal (16%), magnesium (11%), manganese (11%), natural gas (10%), aluminum (8%), crude oil (8%) and chromium (5%). Thus, magnesium, manganese, chromium and aluminum represent a high share of the costs even though they have only a low share of the mass (see Fig. 3). Whereas lignite has a high share of produced mass (close to the amount of iron), it has relatively low environmental impacts. However, this assessment only includes the mining stage for lignite, not the stage in which the highest impacts of fossil fuels are caused: combustion. The impacts of combustion were intentionally not assessed, as they are only carried by the extracting countries if they are extracted for self consumption. With these system boundaries it becomes assessable, why it is economically preferable from an individual countries' point of view to continue extraction, even if domestic environmental costs are included, while the environmental costs of combustion can be externalized to other countries. Some materials have close to the same mass, e.g. silver, gold, titanium and copper, but have a variation regarding the caused costs, as shown in Fig. 3.

When we assess which impact categories are dominant (Fig. 2 on the right), it becomes apparent that the highest impact is climate change (64% of costs). 43% of the climate change impacts are impacts on ecosystems, and only 21% are damages to human health. For climate change, the materials contributing most to environmental damage are iron (23%), coal (18%), magnesium (13%), crude oil (10%), aluminum (8%) and manganese (7%). The second most important impact is particulate matter, which represents 14% of costs. For particulate matter, manganese production (18%), aluminum (14%), iron (14%) and coal (11%) play a significant role. For the particulate emissions of manganese production, the processing and the respective energy demand have a big influence (Davourie et al., 2017; Farjana et al., 2019). Particulate matter is followed by terrestrial acidification impacts (10%), where manganese (22%), iron (13%), aluminum (11%) and coal (10%) represent the largest contributors. These impacts are largely caused by acid mine drainage, which is caused by the presence of pyrite in mining deposits that are exposed to water (Simate and Ndlovu, 2014). The fact that the largest impact is caused by manganese is in line with Farjana et al. (2019), who identify acidification as a significant impact category for manganese. However, acid mine drainage is mostly associated with coal mining, which only represents 10% of the costs here, but the largest share from the other materials is likely to be induced by the upstream emissions from coal extraction for energy production.

In the AoP "resources", the surplus costs of metal and fuel extraction

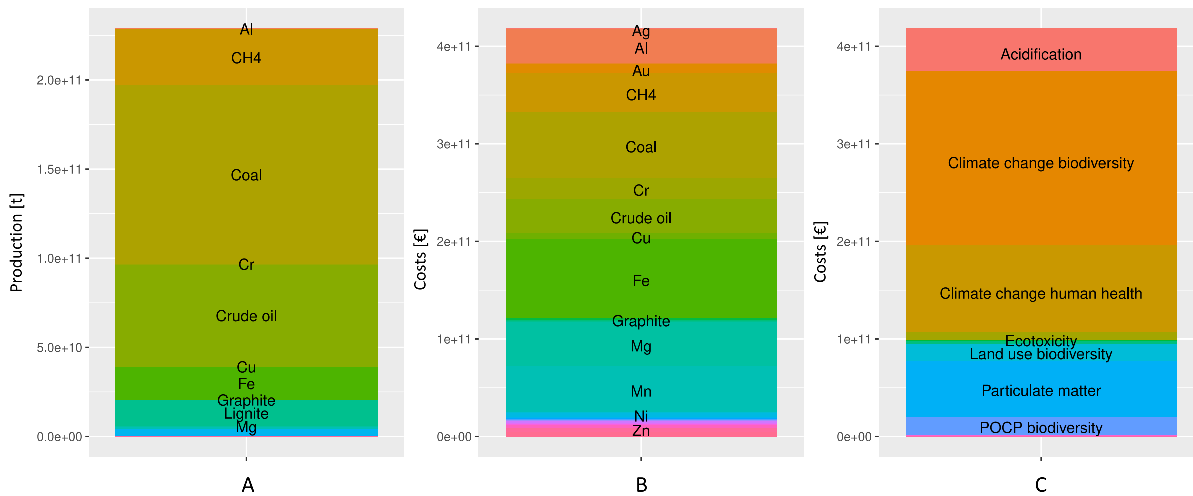

Fig. 2. Mass share of materials (left, A), cost share of materials (middle, B), share of impacts (right, C) prices in euro, low estimate.

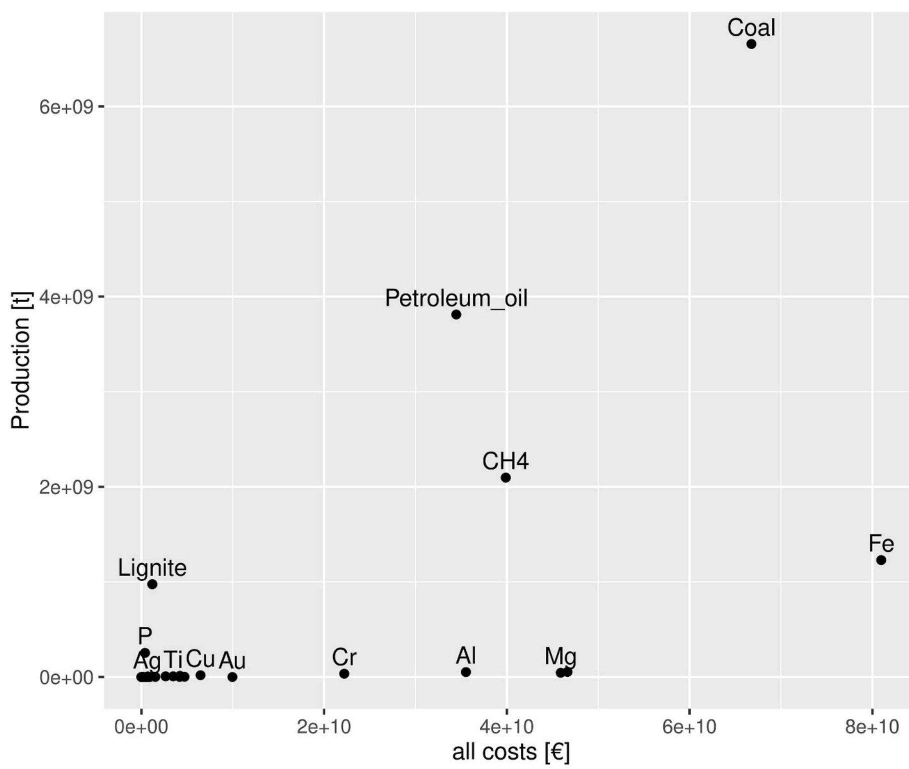

Fig. 3. Relation of environmental costs to mass for the analyzed materials, low estimate.

are assumed to be zero in the low estimate, as explained in the method section (Step 4). When the approach of ReCiPe is applied (as is the case for the high estimate), the fossil fuel surplus extraction costs are very high and dominate the results (with about 2.3 trillion externalities alone and a high share of fossil energy carriers) (see Fig. S 17 in the supplementary materials).

For the total costs of the high estimate, crude oil has the largest share, followed by natural gas and coal. The costs for the fossil energy carriers are dominated by fossil fuel-related future effort costs (as determined by ReCiPe) and represent 43% of the total costs. Aluminum and iron have moderate costs that are mostly related to their climate change impacts. All other costs remain negligible (see Fig. S17 in the supplementary materials). The remaining impact categories will be outlined and described in the supplementary materials (section 4.1).

3.2. Geographical distribution of the costs

In the following, the geographical distribution of the costs is analyzed in more detail. The domestic costs are defined as the costs without climate damages, while the damage costs are defined as all costs including climate damages. Fig. 4 shows the domestic caused costs (all costs apart from climate change) (A) and the domestic costs per capita (B). C displays the damage costs that countries carry (including climate costs they have not caused themselves), and D shows these costs per capita.

The highest domestic costs occur in China (Fig. 4 A) and are caused by aluminum, steel and coal production, which cause acidification and particulate emissions. The high costs of South Africa are caused by manganese and chromium mining and production, while for Australia

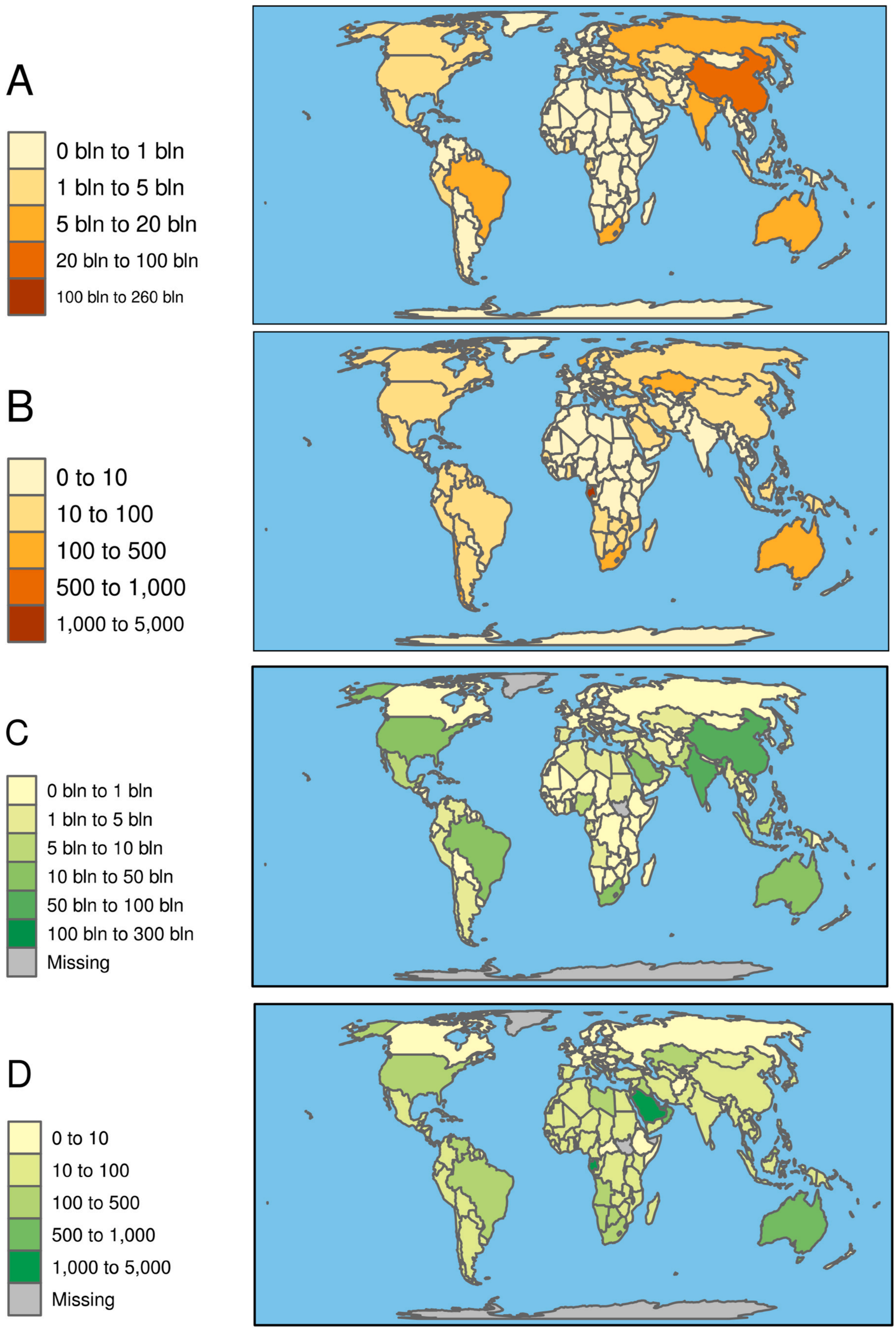

Fig. 4. Domestic costs (A), domestic costs per capita (B), damage costs (C) and damage per capita (D); all numbers are given in [€], low estimate.

manganese, zirconium and coal play a role.

For Russia the impacts are dominated by fossil energy carriers, natural gas, coal and crude oil. For the countries with the highest domestic impacts (China, South Africa, Australia, Russia, India and Brazil), we supply plots that show the share of each material in causing the domestic costs in the supplementary material section 4.1 (Figs. S11-S16). The processing of iron and steel causes Japan and Germany's high impact. However, their mining-related impacts are low because they import ores. For India, the costs are dominated by coal mining and steel production. India produces many different materials that also contribute to the domestic costs. For the damage costs, climate change shifts the picture of the affected countries. Northern Europe and Canada remain relatively unaffected (and the social costs of carbon distribution by Ricke et al. (2018) assume they have economic gains due to climate change). If the costs are compared to relative GDP per capita and one takes into account the law of diminishing marginal utility, African, Asian and Central American countries are most affected by the current production pattern of raw materials because they have high amounts of production and are particularly influenced by climate change. Therefore, they have higher damage costs (C in Fig. 4). In the high estimate, the costs increase significantly. For materials that are produced in China, iron, coal and magnesium cause the highest costs, followed by the U.S.A.'s fossil resource production. These cost streams are dominated by future effort and climate change costs (shown in Fig. S17 in the supplementary material).

In Fig. 5, we show the distribution of the origin of CO_2-emission costs (where they are emitted and not where they cause damages), particulate matter costs and impacts on terrestrial acidification and photochemical ozone formation potential (POCP) damages to ecosystems.

The highest contributing greenhouse gas emissions are all related to activities in China. These are the production of iron/steel, coal, magnesium and aluminum. The high impacts can be related to the fact that China is currently building up a material stock of infrastructure and buildings, which is largely saturated in Western Europe and North America (IRP et al., 2019). The next highest contributors are natural gas extraction from the U.S.A. and Russia, followed by manganese mining and production in South Africa.

For particulate matter, the impacts are similarly distributed compared to climate change; however, the largest impact comes from aluminum produced in China, not steel production, as aluminum has a higher emission intensity per mass unit. Manganese has high particulate matter emissions in South Africa and Australia. These impacts are likely a result of the high energy demand of aluminium and manganese production.

For terrestrial acidification, aluminum from China is also the largest contributor, followed by Chinese iron/steel production. Manganese and chromium mining and production have the largest contribution in South Africa. Australian manganese is also relevant. The acidification impacts most likely occur due to high energy demands that steel and manganese production have. These are met by coal, which causes acid mine drainage and has acidifying emissions in its combustion stage.

The impact of POCP on ecosystems is mostly caused by coal mining (22%), followed by manganese (15%), aluminum and magnesium (both 9%), which is most likely due to energy demands and associated NO_X emissions from energy production, but might also be induced by non-methane volatile organic compounds emitted from transport processes. The four materials that cause the highest costs in China are iron, coal, magnesium and aluminum. The costs for South Africa are caused by manganese and chromium and for Brazil by niobium and manganese.

3.3. Environmental costs related to economic indicators

A comparison of these environmental impacts with economic gains earned by the extraction and processing of the assessed materials yields Fig. 6. The line in both images separates the countries that have a ratio above one from the countries below one. All countries above the line have a ratio above one, which means that costs exceed benefits.

It is visible that for many countries the ratio of environmental costs to economic gains is lower than one, meaning that the mining sector delivers larger economic gains than losses. Thus, the mining sector delivers higher income or contribution to the increase of the capital stock than environmental costs. For some countries, though, this relation is less advantageous, e.g. Rwanda, Afghanistan and Gabon. However, it could be that some of these countries underreport benefits. The relation is the highest for industrial producing countries that also process materials, such as Germany and Japan (steelmaking), but also for oil-producing countries such as Algeria and Azerbaijan. This picture might change however, if the impacts from upstream processes from iron and steel production and the downstream-processes from oil extraction were be included in the assessment. Thus, there is the probability that this beneficial ratio is the result of the externaization of particularely harmful life cycle phases of the materials' use.

Rwanda has the highest ratio of costs to GDP, and the main material that is produced in Rwanda is tantalum. It could be that most of these materials are sourced by small-scale mining in the informal sector and are thus not reported in terms of GDP gains. The low estimate of the environmental costs exceed the economic gains measured in GDP only for the following countries: Rwanda, Gabon, Madagascar and Afghanistan.

Thus, for the majority of countries it is beneficial from a cost-benefit perspective to continue their mining and extracting activities, if only domestic damages are considered. When the damages of climate change are included, 20 countries' GDP losses are higher than the benefits gained due to mining activities (Fig. 6 right panel). These countries belong to Africa, Central and South America and South East Asia and are severely impacted by climate change. Countries that generate the largest costs (China, Brazil, India, and Russia) have a ratio below one and thus have higher benefits from their mining activities than costs. Countries that also process materials and manufacture them into products, such as Japan and Korea, have the most beneficial ratio. In addition, oil-producing countries such as Algeria and Azerbaijan have quite a good ratio, as the environmental impacts of fossil-based materials do not occur during mining and processing but rather in the combustion/use stage, which was excluded in this assessment.

When we use the high cost estimate, 20 countries exceed one in the relation of domestic environmental costs to GDP, including such major mining countries as Australia and Brazil (see Fig. S19 in the supplementary material). In addition, some European countries are included. In the case of the Netherlands, for example, this can be explained by the future effort costs (ReCiPe) it incurs for the extraction of natural gas in the North Sea.

Based on the visualization in Fig. 6, it becomes apparent that for countries where the overall share of economic activity in mining is low but where large amounts are still produced, the ratio of environmental impact is often particularly disadvantageous (e.g. Rwanda or Afghanistan).

In Fig. 7, we show the damages per job generated in the mining industry for the low estimate.

The damages per job give an indication of the environmental costs for each employment position in mining. To see how they compare with the overall impact on society, they can be roughly compared with the average wage in that country. They can also give an indication of how desirable the protection of such employment positions is from a welfare perspective. In Fig. 7 it is visible that for most countries the damages exceed 500€ per work position, apart from some African countries. The underlying employment numbers for most African countries are estimated based on statistics on small-scale mining and contain diamond mining and the sourcing of colored gemstone as well, which have not been considered for the calculation of environmental costs, and thus the environmental costs per job are likely underestimated. With this analysis it becomes clear that the positive cost-benefit ratio analyzed in Fig. 6 will not only come from wages, but will also largely contribute to the

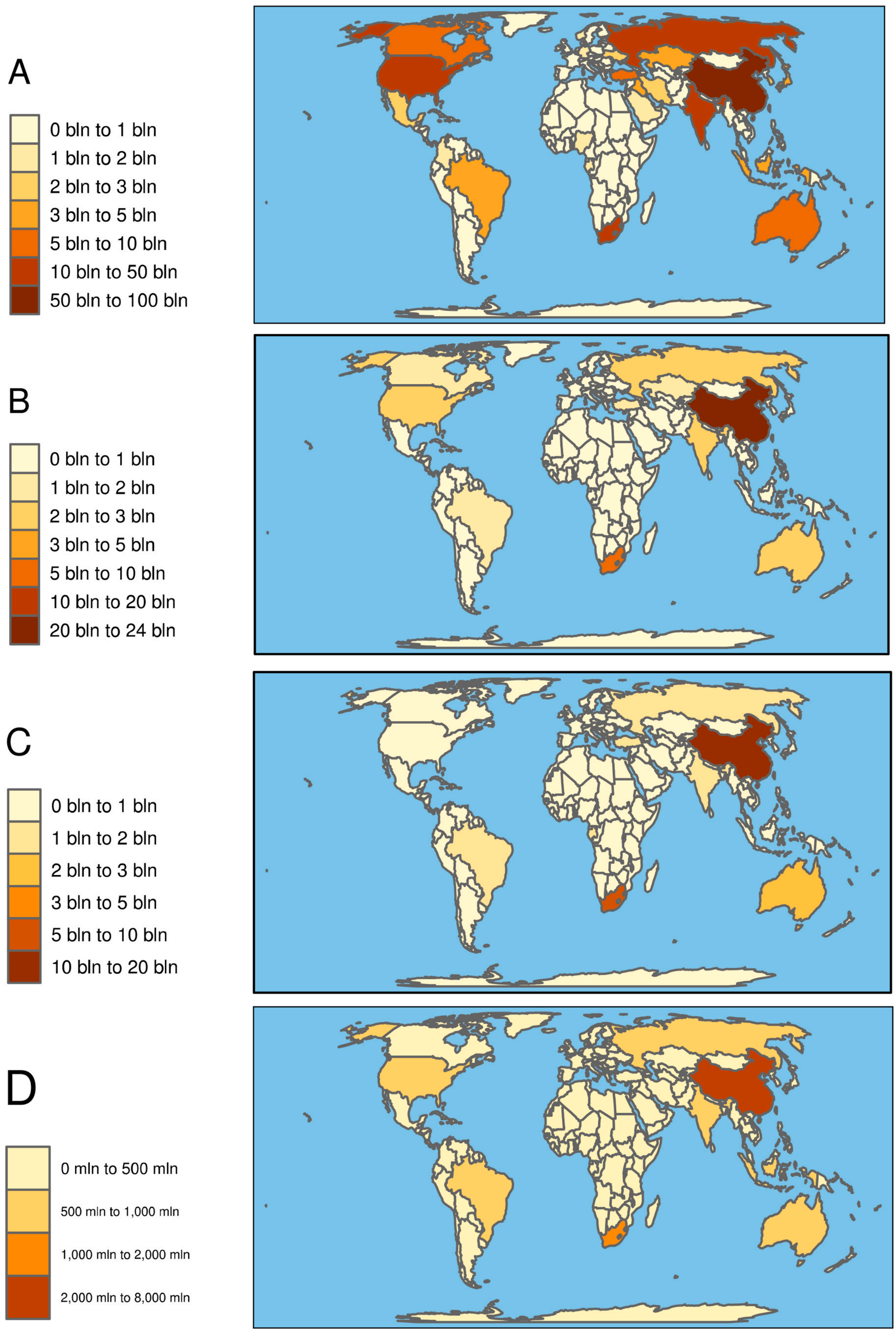

Fig. 5. Impact distribution for climate change (cost origin (A)), particulate matter (B), terrestrial acidification (C) and POCP damages to ecosystems (D); all numbers refer to [€], low estimate.

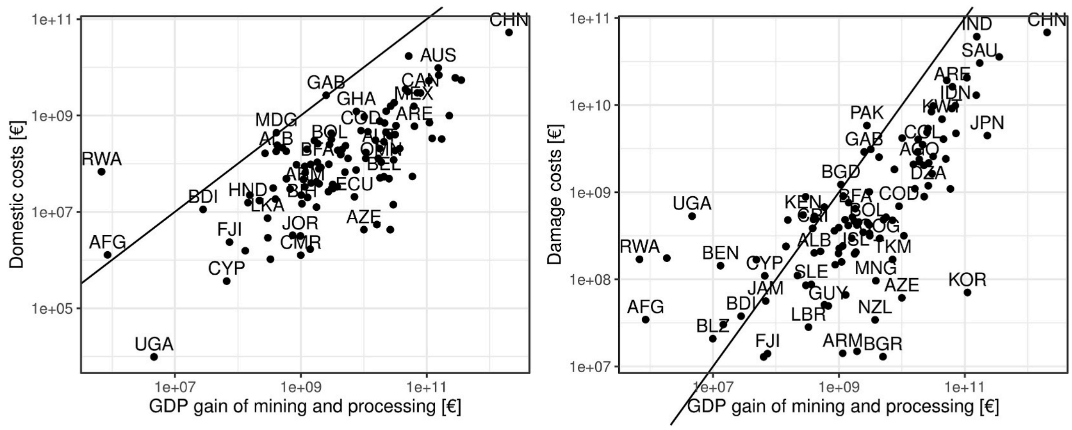

Fig. 6. GDP gain vs domestic costs (left), GDP gain vs damage costs (right), low estimate, double logarithmic scale.

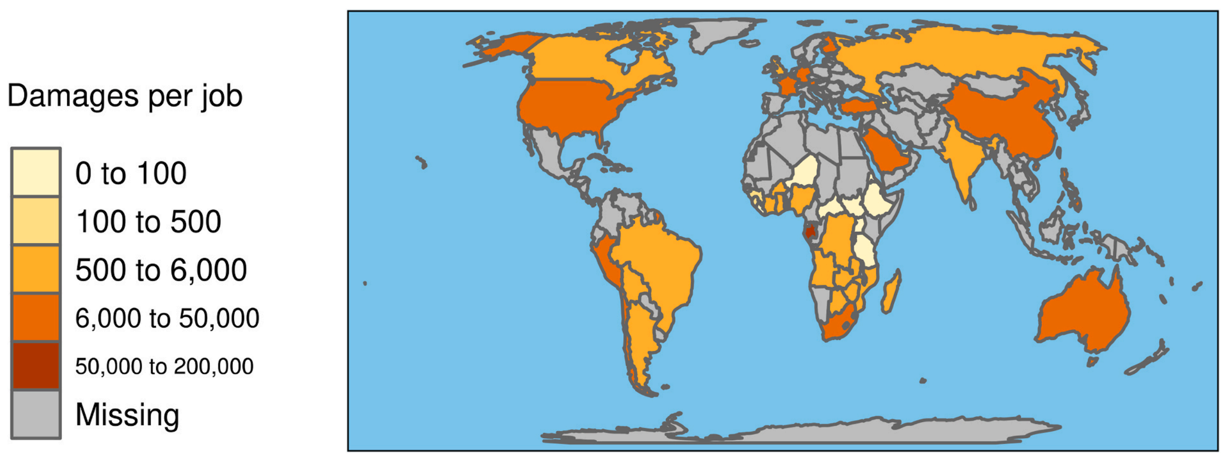

Fig. 7. Environmental costs per job in [€], low estimate.

increase of the capital stock.

4. Discussion

In the following, the methods and results are discussed. There are several shortcomings with regard to the underlying data sources. First, the global production data are obtained from USGS. Next to the fact that USGS data are politically influenced (Mobbs, 2005), they also do not deliver endless granularity in the sense that they have cut-off criteria for how many countries' material production is reported. This leads to an underestimation of global production and underlying impacts. Additionally, we were not able to differentiate between the mining and processing of all materials, as the LCA databases did not supply enough process steps in between mining and processing, and such a differentiation is only made for a few materials by the USGS (e.g. iron ore and bauxite). The result is that such a differentiation could not be made. Another shortcoming is how environmental impacts are assigned to the materials. Often, metals are won as co-products, and the allocation in LCA databases is mostly done according to economic value, which can change over time. Additionally, the environmental impacts are distinct in different countries and due to changing ores and a varied energy supply. Such a differentiation was not possible for all materials, as the granularity of the data does not allow for it. Another shortcoming is that the databases we used are inconsistent, as ecoinvent often provides higher values, especially in the impact categories of toxicity (Pauer et al., 2020). Thus, the chosen database affects the results significantly, and it is not clear which underlying database has higher quality, as ecoinvent is more transparent, while GaBi provides more up-to-date industry data. Future research could focus on a single material that has been identified as significant (e.g. iron or aluminum), trace the different processing steps that could be identified by a material flow analysis and couple them with more granular LCA data in order to map the geographical distribution of the mining and processing with a higher level of certainty and resolution. An example of developing a new method and applying it to a site-specific case study was provided on coupling emergy analysis and LCA for different gold mining techniques in Columbia (Cano Londoño et al., 2019).

Regarding the monetization, many challenges remain as well. Several factors, which influence the chosen monetization method depend on value choices (Arendt et al., 2020; Bachmann, 2019), as the monetization of environmental impacts draws on welfare economics and economics is a social science. The main shortcoming is that we used European values to monetize impacts on species, as no global values were available (high and low estimate). Impacts on marine and freshwater species have been neglected as no monetization was available, which again leads to an underestimation of the results. Moreover, impacts that occur due to accidents (e.g. fatalities while working in the mines, major oil spills or breaking dams as well as floods and storms induced by climate change) are not considered and thus again lead to an underestimation of overall damages from the mining and processing of materials. For the high human health valuation, we also chose the recommended value for European income and for the low valuation of average global income. This approach comes close to the so-called "human capital" approach that has been criticized on many grounds (Tan, 2014). The monetization approach in ReCiPe for resource depletion assumes that ore grades and the concentration of ore bodies and fossil reserves decline, which leads to increased effort and energy demands. This has in fact never happened—production costs have declined, as efficiency has increased and prices for materials are declining (Jowitt et al., 2020). Thus, the monetization regarding these aspects, which dominate our assessment for the high estimate, might be overestimated.

The results in this study have been monetized to compare them with other economic indicators such as GDP generation due to mining and processing. However, e.g. Sandel argues that markets or monetization not only provide information but also transmit value (Sandel, 2012) and assume that everything can be interchanged. Therefore, some people resist the monetization of health gains or species diversity, as it never reveals anything about these goods as such, neglecting intrinsic value and rather showing how the society values them. Additionally, the monetization applied here assumes a weak sustainability perspective, as it assumes the exchangeability of natural, human and financial capital, which is debated in the context of LCA (Steen, 2006).

For the economic indicators that the monetized results are compared with, several limitations occur. For natural gas and oil, only resource rent and not total production value (which includes income in the sectors) is taken into account. For employment, the data is incomplete and comes from very diverse sources, which might lead to large inconsistencies. Additionally, it is not clear whether the data includes work in quarries to mine building materials that have not been included in this study and thus would reduce the ratio of environmental costs to employment. In addition, the employment numbers contain some inconsistencies, as there are no harmonized international sources on employment in the mining and processing of materials.

The calculated cost-benefit ratios in section 3.3. do not allow for a conclusion regarding the working conditions and the benefits for the local population. Additionally, the study does not show how the economic gains and damages are distributed and e.g. what effects they have on inequality, which is very relevant from a welfare economics point of view. If, for example, the resource rents are distributed to stockholders that do not pay taxes and the damages affect the most vulnerable, the welfare impact of the income might also be negative, even if the cost-benefit ratio is positive. To identify whether this is the case, field research and socio-economic analysis is needed on a site-specific basis. These kinds of insights are also needed to derive tailored policy measures to reduce the environmental costs and avoid imposing them on those that are least able to bear them. Furthermore, most mining companies are Australian and Canadian, which are countries with comparatively high income and which benefit from extraction with environmental impacts on much less wealthy countries. However, if the mining is owned by the respective country itself, and the resource rents are reinvested into finance public infrastructure such as education and health care, the impacts might also be positive.

The GDP gain that is not caused by wages will largely benefit richer individuals. Richer individuals will have a comparatively smaller welfare gain due to the law of diminishing marginal utility (Gossen, 1854). However, this contribution will also depend on the ownership of the mining companies, e.g. whether they are companies traded on the stock market or whether they are owned by the state and how the state uses the obtained revenue.

Further shortcomings of the study are that the interaction between ecology and economy has been neglected, meaning that we did not take an ecosystem service approach that highlights the importance of intact ecosystems to economic activities and human health. One aspect is that mining might be in conflict with ecotourism, as it deteriorates views and the enjoyment of nature, therefore our estimation undervalues the protection of nature.

Nevertheless, our research provided many valuable insights into the environmental impacts of mining and processing worldwide. We identified which countries benefit, from a cost-benefit perspective, from the continuation of local mining activities, which is valuable if a game theoretical analysis of fading out the extraction of fossil resources is aimed for. The approach we developed can be used to analyze different large-scale developments in changing patterns of resource extraction. Moreover, it shows that under the given production conditions, the biggest impacts still stem from the materials that are used in large amounts such as iron/steel and aluminum, for which energy use is decisive. Many of the climate change impacts (as quantified here, representing over half of the impacts for the low-estimate scenario) could be reduced if the use of fossil energy were shifted to renewables. The biggest costs are carried by China, which China mostly bears for exporting goods to other countries (Yang et al., 2020). To reduce the costs (as calculated here), a more detailed analysis of how the materials are used and how they can be used more efficiently needs to be developed, possibly building on work that has been carried out in this direction (Allwood and Cullen, 2012).

5. Conclusion

The conclusion of this study is that mining has varying effects. On the one hand, it diminishes the concentration of ore bodies and deposits and transfers them to the technical environment in the form of stocks or to the biosphere in the form of CO_2 or other emissions. The extraction of the 38 abiotic resources assessed in this paper leads to substantial damages: from €0.4 to €5 trillion per year, as determined with LCA coupled with environmental costs. On a country level, in comparing the environmental costs to the GDP contribution of extraction and processing, our analysis shows very diverse pictures. The countries with the most beneficial ratio are those that manufacture basic metals (such as Japan, Germany or South Korea) but also countries that produce crude oil (such as Algeria and Azerbaijan). This is likely due to the fact that they externalize the environmental costs of relevant life cycle stages of material processing and use (ore extraction for steel or combustion for oil) to other countries. The countries with the least beneficial ratio are Rwanda and Afghanistan, possibly due to much artisanal economic activity that has not been reported in international statistics.

The most relevant impacts for the low estimate are related to climate change, particulate matter and acidification. The results change completely if the high estimate is taken, in which case future efforts to extract fossil energy carriers exceed all other impacts. Furthermore, environmental impacts are not the consequence of a single material that could easily be phased out. The largest climate change impacts derive from materials that are mined and produced in big amounts and that are part of daily life: iron, aluminum and coal. The approach developed here supports the analysis of the welfare impacts on national economies and can be refined for more small-scale analysis.

CRediT authorship contribution statement

Rosalie Arendt: Writing – original draft, Conceptualization, Data curation, Formal analysis, Visualization. **Vanessa Bach:** Project administration, Writing – review & editing, Supervision. **Matthias Finkbeiner:** Project administration, Writing – review & editing, Supervision.

Declaration of competing interest

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

Acknowledgements

This work was funded by the Bundesministerium für Bildung und Forschung, project “R2Q: RessourcenPlan im Quartier” (funding number: 033W102E). We would like to thank the Bundesministerium für Bildung und Forschung for its financial support. Additionally, we thank three anonymous reviewers for their helpful feedback as well as Charlie

Zaharoff for professional English proofreading, which greatly improved the style of this article.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2022.132232.

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3.4 Environmental Costs of Abiotic Resource Demand for the EU's Low Carbon Development

In this section we present the publication:

Arendt, Rosalie; Bach, Vanessa; Finkbeiner, Matthias: Environmental Costs of Abiotic Resource Demand for the EU's Low Carbon Development. *Resources Conservation and Recycling* **2022** 180 (May): 106057 https://doi.org/10.1016/j.resconrec.2021.106057

This publication is tied to research question 1, 2 and 3. It gives an answer to the question 2, by proposing a method that can be applied to assess environmental criticality that is founded on LCA and environmental costs/monetization of life cycle impact assessment results. As this field of application is identified and a method is developed, target 2a) and b) are fulfilled. Damages caused by greenhouse gasses, land use, non-renewable resource depletion, and freshwater consumption were considered. These are equivalent to the environmental SCARCE categories from Paper 2. Additionally, the environmental costs of the EU's low-carbon development were quantified, contributing to target 2b). Then, it was possible to quantify, which materials dominate the costs and which countries are most affected (fulfilling research question 3). For the cost range, only the necessary mass and the discount factors was varied, but in the discussion, the effect of choosing different monetization factors is taken into account, contributing to research question 1.

Resources, Conservation & Recycling 180 (2022) 106057

Contents lists available at ScienceDirect

Resources, Conservation & Recycling

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

Environmental costs of abiotic resource demand for the EU's low-carbon development

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A R T I C L E I N F O

Keyword:
Monetization
Abiotic resources
Carbon neutrality
Critical resources
Green new deal
External costs

A B S T R A C T

Low-carbon development is one of the main goals of the European Green New Deal, but the European Union relies on many raw materials to realize it. This study quantifies the environmental impact of projected abiotic resource demands for the low-carbon development of the EU in 2050 based on lifecycle assessment coupled with environmental costs. We account for damages caused by greenhouse gasses, land use, non-renewable resource depletion, and freshwater consumption. The total environmental costs of the materials needed for low-carbon development in 2050 range from €13.1 billion to €74.8 billion, with €38.9 billion as the medium estimate. These costs seem substantial, but represent only 3.7% of the costs that the EU generates due to its current territorial carbon emission level. Based on our findings, materials for EV batteries cause 45.8% of the costs. The analysis showed that the damages are dominated by associated carbon emissions in mining and processing (47.5%) and abiotic resource depletion (45.1%), mainly induced by increased demand for nickel, iron, copper, and aluminum. Additionally, we were able to trace the geographical distribution of the impacts. Our model assigns the highest absolute environmental costs of the EU's low-carbon development to China, the U.S.A., India, and Saudi Arabia. The highest relative costs compared to GDP are paid by Guinea and Gabon. We conclude that responsible consumption strategies for the assessed materials should be established to enable low-carbon development with minimum environmental costs. This will be facilitated by the approach developed here.

1. Introduction

Mining of raw materials is essential for economic activity but is leading to significant and increasing environmental impacts (IRP et al., 2019). Many specialty metals will experience demand shifts due to the need to transition to a low-carbon economy, and their increased sourcing may lead to adverse effects on the environment (e.g.(Bazilian, 2018)). While studies on certain technologies have already quantified the environmental impacts of transitioning to a low-carbon economy (McManus, 2012), the total amount of materials needed for low-carbon development in all sectors has not been assessed in as much detail (Yavor et al., 2021).

One way to assess adverse environmental effects of demand shifts for the necessary metals would be integrating environmental impacts into the criticality assessment. However, the EU's current approach to criticality assessment measures only the economic importance and supply risk of materials necessary for the energy and mobility transition (Blengini et al., 2020), neglecting the adverse environmental effects caused by mining and processing of the necessary materials. In the foresight study, in which the material demand for low-carbon development was projected for the EU including a criticality analysis (Bobba et al., 2020), no assessment of possible associated carbon emissions or other environmental impacts linked with the sourcing of these materials was carried out. Such an approach risks shifting the burden between lifecycles (emissions shift from use to production phase e.g., in the case of electric vehicles) or to other environmental impacts. For example, the opening of a new mine site might lead to negative environmental effects beyond carbon emissions.

In addition to the European Commission's approach, many studies have focused on the availability and importance of resources for economic development (Graedel et al., 2015), including low-carbon development (Miyamoto et al., 2019), yet research on the corresponding environmental impacts on producing and exporting countries has been less pronounced. Methods that focused on the environmental burdens do so mostly at a product and not at country or regional level. The SCARCE method considers non-renewable resource depletion, water use, biodiversity, and carbon emissions (Arendt et al., 2020b; Bach et al., 2017). However, it does not include absolute damages of total material

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https://doi.org/10.1016/j.resconrec.2021.106057
Received 4 June 2021; Received in revised form 6 October 2021; Accepted 10 November 2021
Available online 7 January 2022

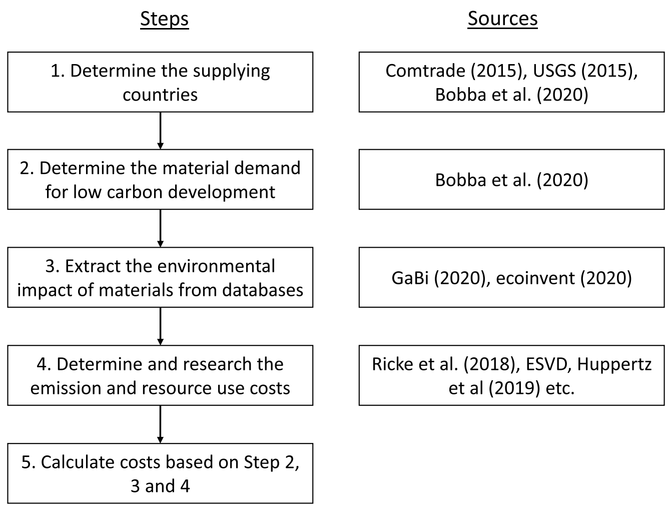

Fig. 1. Steps and respective sources to determine environmental costs of the EU's low-carbon development; ESVD stands for ecosystem services valuation database and USGS for United States Geological Survey.

use, but rather relative damages per kilogram as does the assessment of Graedel et al. (2015).

One way to assess quantitative environmental impacts of products and services is lifecycle assessment (LCA) (ISO, 2006). LCA case studies that have addressed the transition to a low-carbon economy include Hertwich et al. (2015), which quantified the environmental impact of the energy transition with the help of LCA, but only considered impacts of bulk materials and not the depletion of materials such as cobalt, graphite, platinum, and lithium. Additionally, Jochem et al. (2016) quantified external costs of electric vehicles, but neglected the costs of resource depletion, which Rauner et al. (2020) did not account for either in their cost-benefit analysis of the coal exit. Recently, Strezov et al. (2021) performed an LCA for the Australian mining industry and calculated the relation of economic value-added to pollution intensity. However, they did not monetize the pollutants and thus did not account for the impact on overall welfare. Product-based LCA studies of electric vehicles are available (Dolganova et al., 2020), including on a European level (Xu et al., 2020). Furthermore, national case studies for energy transitions exist (Reinert et al., 2020). However, these studies have several shortcomings: They do not trace back where the environmental burden is carried and how high it is in welfare-economic terms. Thus, it is difficult to improve, prioritize and identify possible geographical shifts in ecological burdens induced by the transition to a low-carbon economy.

To assess LCA results in welfare-economic terms, some LCA methods monetize the impacts of abiotic resource depletion (ADP) (Murakami et al., 2018; Ponsioen et al., 2014; Vogtländer et al., 2019). However, they do not always integrate monetized environmental impacts beyond resource depletion. LCA was also used to integrate environmental impacts next to criticality assessment within the ESSENZ method (Bach et al., 2016), but without monetizing them.

This research aims to build on the environmental dimensions of existing criticality assessments, with a focus on the ESSENZ and SCARCE methods, to derive a monetized damage score for resource production and imports of raw materials needed for low-carbon development. Such a damage score could be used to complement economic criticality assessment and evaluate the welfare-economic impact of resource extraction on the provisioning countries. For this purpose, LCA results will be monetized based on an overview of different monetization methods (Arendt et al., 2020a), considering the impact categories climate change, water use, land use, and ADP. We deemed these impacts the most significant, as climate change, ADP and water use were assessed already in ESSENZ and SCARCE and land use is the main driver of biodiversity loss (de Baan et al., 2013). The research also aims to test the assumption pointed out by e.g. Ricke et al. (2018) and Srinivasana et al. (2008) that the environmental costs generated by consumption patterns are largely not carried by those that use the resources. Therefore, we decided to map the environmental costs generated with country resolution to see whether the EU member countries carry only a small fraction of the costs that they cause with their development. Thus, our approach enables us to quantify the welfare-economic effects with country resolution of environmental impacts caused by increased material demands required for technological transitions. We applied this approach to the low-carbon development of the EU and showed the impacts on the provisioning countries' economic activity.

Hence, the goals of this paper are to:

- Assess the total environmental costs of the material demand for transition to a low-carbon economy in the EU.
- Identify which materials cause the highest costs and to which technologies they are linked.
- Show how these costs are distributed between countries.
- Demonstrate how these costs relate to the countries' GDP.

The paper is structured as follows: First, the method section is outlined in five steps. Then the general results are presented following the structure of the papers' goals. Subsequently, the results are discussed and a conclusion is drawn.

2. Method

Five steps were taken in order to estimate the environmental costs of European low-carbon development. They are visualized in Fig. 1 and explained in more detail in the following.

Step 1: The materials relevant for low-carbon development were identified based on the study by Bobba et al. (2020). The countries that exported these materials to the EU were determined by extracting the data from UN Comtrade (UN Statistics Division, 2016). They were also adjusted for transit countries by excluding countries that exported but did not produce (procedure as in (Arendt et al., 2020b; Bach et al., 2017)). Additionally, we added the EU 27 countries that produced materials based on data by USGS (U.S. Geological Survey, 2018) as their production can be used to meet the increased demand. Only countries that exported the raw materials as primary materials, not materials contained in products, were considered. As a result, we obtained a list of countries and their delivered quantities for the EU's demand for all materials that were identified as relevant for low-carbon development by Bobba et al. (2020).

Step 2: The material demands for low-carbon development were obtained from Bobba et al. (2020) considering three 2050 scenarios for low, medium and high demand, respectively. The data represents the material demand in the year 2050 (per annum in 2050, not total demand from today until 2050). Our cost analysis thus refers to the costs in the year 2050 in the case that these scenarios occur. Based on this data, we scaled the amount of resources from the supplying countries in Step 1 linearly (e.g. if the demand doubled, we multiplied the supply determined in Step 1 by a factor of two; therefore assuming that all countries will deliver twice that amount in the future). Moreover, we created a recycling and a non-recycling scenario, basing the primary material use (old scrap ratio) on Talens Peiró et al. (2018) and assuming zero environmental impact for the recycled content. As Bobba et al. (2020) supplied the material demands disaggregated for different technologies (electric motors, EV batteries, PV modules, wind power plants), we are also able to show the cost distribution for different technologies.

Step 3: Based on the environmental dimension in the SCARCE criticality assessment, the environmental impacts of the respective resources were considered for the categories climate change (in t CO_2-e per kg of material based on IPCC (Stocker et al., 2013)), water use (in m^3 of blue water per kg of material, impact assessment based on Motoshita et al. (2018) and Pfister et al. (2009)), and land use (in area*time per kg of material). They were determined using GaBi (Thinkstep, 2020) and ecoinvent data (Ecoinvent, 2020) and applying the impact assessment methods presented in Table S 2 in the Supplementary Material 1. For ADP, the masses derived from Step 2 sufficed as the approach by Huppertz et al. (2019) monetizes and quantifies the impact in one step. The underlying processes for the different materials are provided in Table S 1 in Supplementary Material 1, including the origin of the dataset. Mining, production, and refining of the 21 abiotic resources are considered (cradle to gate). Production and construction/installation of low-carbon infrastructure are not assessed.

Step 4: To monetize the environmental impacts identified in Step 3, environmental cost factors were determined for climate change emissions, water use, land use, and ADP. An overview of the associated impact assessment methods and cost determination is provided in Table S 2 in Supplementary Material 1 and described in more detail in the following paragraphs. The derived cost factors for water and land use are supplied in Supplementary Material 2.

Climate: The values by Ricke et al. (2018), which are already supplied on a country level per ton of emission, were used for carbon emissions. We chose the median estimate, which equals €290/t CO_2. We chose the values by Ricke et al. (2018) as they provide country resolution costs. Previous reviews have shown that the range of costs assigned to carbon emissions is high (Arendt et al., 2020a; Bachmann, 2020; Dong et al., 2019), and the value chosen for this assessment is higher than most values provided.

ADP: Costs of ADP are based on global values provided by Huppertz et al. (2019), which is based on the Hotelling rule (Hotelling, 1931). Costs occur due to an extraction rate of exhaustible resources above the social optimum, which creates costs as compared to a socially optimum depletion pathway. We allocated the cost factors proportional to the GDP contributions of sectors in which the materials were used. The application of a material in certain sectors was derived from the EU's critical raw material study (Deloitte Sustainability et al., 2017), while the global and country outputs of certain sectors were extracted from the UNIDO Database (UNIDO, 2020). The output data is from 2015 as the data for other years was less complete. For the material m_x in country c, we add up the outputs (o) of all sectors i to n that material x is used in, then divide this total by the world output of these sectors as shown in Eq. (1).

$$\frac{\sum_i^n o[€]_{c,i,x}}{\sum_i^n o[€]_{world,i,x}} \cdot p_x\left[\frac{€}{kg}\right] = p_{c,x}[€/kg] \qquad (1)$$

This results in country-specific output shares for each material. The costs per kg of material x, represented by p_x from Huppertz et al. (2019), are distributed per country by multiplying the costs per material with the share of country c so that we obtain a vector of damages for all countries c for the use of material x expressed as $p_{c,x}$.

Land use: For the calculation of lost ecosystem services due to land occupation for mining, we use values from the Ecosystem Services Valuation Database (ESVD) update from 2020 (de Groot et al., 2020a, 2020b). To avoid the risk of double counting, only damages to ecosystems induced by land occupation and not by indirect impacts from climate change on land ecosystems are considered. The ESVD provides only values per square meter on biome level, and the data on land occupation per material from Steps 1 and 3 is available on a country level. Therefore, we first calculate the average costs per square meter occupied for each country c, where a_i is the costs that occur due to the occupation of a square meter of biome i for one year. This is done based on the area weighted value per biome i and its share $g_{i,c}$ in country c obtained from the WWF (2012). The value of occupying a square meter in country c a_c is calculated in Eq. (2).

$$a_c\left[\frac{€}{m^2}\cdot yr\right] = \sum_i^n a_i[€/m^2\cdot yr]\cdot g_{i,c}[\%] \qquad (2)$$

This equation yields the necessary cost factor per square meter for country c a_c. We derived costs per square meter for all countries accordingly.

Water use: For water use, three areas of protection are monetized: impacts on human health in the unit disability-adjusted life year (DALY) based on Motoshita et al. (2018) due to malnutrition, species loss caused by water consumption based on the Water Stress Index (WSI) (Pfister, 2011), and for damages to yields based on the method by Ligthart and van Harmelen (2019). To obtain the damages associated with human health, the DALY per cubic meter water use in country c h_c was multiplied with the monetized DALY value f to obtain the damages per cubic meter for one cubic meter of water use in country c j_c as shown in Eq. (3).

$$j_c[€/m^3] = f[€/DALY]\cdot h_c[DALY/m^3] \qquad (3)$$

This results in country-specific cost factors (for countries c) per cubic meter of blue water for human health.

The endpoint model by Pfister et al. (2009) is used to determine the impact of water use on species. The species year value is monetized with the monetary values by Kuik et al. (2008), of which we chose the high estimate since this method determines European values and biodiversity is lower in Europe than on average (de Groot et al., 2020a). For the biodiversity impact of water use, the values by Kuik et al. (2008) are country-weighted based on the method by Winter et al. (2019, 2018) but recalculating the country-weighted ecoregion factor based on the new ecoregion factor by Lindner et al. (2019). A more detailed outline of this scaling is supplied in the supplementary material.

For the impact on the reduced economic output due to limited water availability, the global price factors by Ligthart and van Harmelen (2019) are used and scaled with the WDI (annual average non-agricultural use (Berger et al., 2018)), assuming that more water-scarce regions have an increased marginal productivity of a cubic meter of water. The global values were scaled with the WDI and changes from the global average of €0.863/m^3 to a range of €0/m^3 to €3.25/m^3, which also result in a country-specific factor of euro per cubic meter.

Step 5: The obtained supply mix from Steps 1 and 2 is multiplied with the environmental impacts from Step 3 and the environmental costs obtained from Step 4. Thus, the material quantity produced in country c $m_{x,c}$ is multiplied with the environmental impact a i_a per mass of material m_x and finally multiplied with the costs per i_a k_a, which results in the environmental costs for country c as shown in Eq. (4).

$$Costs\ for\ Country[€]_{c,x,a} = m_{x,c}[t]\ \cdot i_a/m_x[t] \cdot k_a[€/i_{\ a}] \tag{4}$$

Eq. (4) applies to water and land use, as we assume that these costs are mainly carried in the countries in which they occur. CO_2 emissions and resource use cause global damages. Their costs are determined according to Eq. (5) based on the total mass m_x, the intensity of the environmental impact per mass i_a/m_x, and the country-specific cost factor for that impact. For resource use, the term i_a/m_x is equivalent to 1 as the impact is the used mass.

$$Costs\ for\ Country[€]_{c,x,a} = m_x[t]\ \cdot i_{\ a}/m_x[t] \cdot k_a[€/i_{\ a,c}] \tag{5}$$

The costs for country c caused by the material x represent only the costs associated with the supplied amount of material m_x due to impact i_a. To obtain the total costs, these occurring costs must be totaled over all materials m_x and all impacts i_a according to Eq. (6).

$$Costs\ for\ Country[€]_{c,total} = \sum_{x}^{n} \sum_{a}^{n} m_{x,c}[t] \cdot i_{\ a}/m_x[t] \cdot k_a[€/i_{\ a}] \tag{6}$$

This results in the total costs carried by country c for all materials x to n and all considered environmental impacts a to n. As a final step, all values are transferred to 2018 euros (inflation adjusted with consumer price index and currency conversion with purchasing power parities). As the damages and emissions will occur in 2050, we also applied a discount rate. For the central estimate presented in this paper, we used a discount rate of 0%, as we assume that the damages of CO_2 emissions, abiotic depletion potential, water use, and land use are at minimum likely to increase proportionally to economic growth. In Supplementary Material 1 (Section 6), we provide results with a 1% and 3% discount rate respectively, which results in 17 scenarios (with low, medium, and high material demand, with and without recycling, and with a 0%, 1%, and 3% discount rate). The ADP and CO_2-emission factors are already discounted to net present value. Thus, the cost factors with 0% discount rate have been discounted for climate change and resource depletion but to reflect damages today, not in 2050.The obtained costs are totaled over all countries and visualized with the tmap package (Tennekes, 2018) in RStudio.

However, taking into account the law of diminishing marginal utility (Gossen, 1854), damages in poorer countries (and also more unequal countries in terms of wealth) should be identified as environmental damages will affect utility more strongly. Therefore, country-specific costs are also set in relation to the countries' GDP to see how relevant the damages might be to the overall economic activity and quality of life of the respective human beings and the ecosystem quality. Additionally, to put the range of the costs into perspective, they are set in relation to the environmental costs of the current level of carbon emissions that are also calculated based on Ricke et al. (2018) and the territorial emissions of the European Union in 2018 (EEA, 2021).

3. Results

The results are divided into the following sections: First, the costs associated with the different materials, their environmental impacts, and the associated technologies are outlined (3.1). Then the country distribution (3.2) and the relation of the environmental costs to the GDP of the different countries (3.3) and their link to the different environmental costs are presented. In the results, all costs refer to the medium non-recycling scenario with a 0% discount rate. The results of all other scenarios are visualized in Supplementary Material 1 and their effects are discussed and compared in the Section 3.4. Intermediate results on environmental quantities and all costs per material and country are available in Supplementary Material 2.

3.1. Costs per material and technology

The total damages for the medium no recycling and no discounting scenario are €38.9 billion. However, the total environmental costs due to abiotic resource demand of the technologies are only 3.7% of the annual damages of the EU's current carbon emissions, assuming 3.6 billion tons of annual carbon emissions in 2018 and a carbon price of €290 (same price as for former estimates). Therefore, the benefits of low-carbon development are likely to exceed the costs of the resource demand that it will cause. Nevertheless, we will focus in greater detail on where and how these costs are caused. In Fig. 2 the costs per material are visualized in Panel A. It is apparent that the associated costs are dominated by nickel (amounting to €9.7 billion and thus 24.9% of the total costs), followed by iron (14%), aluminum (13.1%), cobalt (10.94%), copper (10.2%), lithium (7.7%), and graphite (7.5%).

It is observable that the damages of most material costs are dominated by abiotic resource depletion and climate change. However, the impact from land use on ecosystem services exceeds 10% for some materials such as zinc, rare earths, manganese, and graphite. Water use has some impact for lithium, rare earths, and aluminum but always stays below 10%.

Fig. 2 Panel B shows the distribution of the damages according to impacts. The impacts of climate change are €18.5 billion (and thus 47.5% of the total costs). The total damages for ADP are €17.5 billion (45% of total costs). Land use and reduction of ecosystem services cause only approximately €2.1 billion in damages in 2050 (5.4% of total costs), while damages caused by water use are in the range of €790 million (causing 2% of the total costs). The land use costs are mainly generated by graphite (25%) and cobalt (13%), followed by manganese (11%). The highest water use costs are caused by lithium (34%), aluminum (30%), and nickel (15%).

Most of the costs are connected to materials necessary for battery technologies (nickel, graphite, cobalt, lithium). Their costs make up 45.8% (electro-mobility batteries) and 8.4% (renewable energy batteries), respectively. The most dominant material is nickel, which represents 20% of climate impact, 15.3% of water impact, 10% of land impact, and 32% of abiotic depletion costs. The impact cannot be explained by the share of necessary mass, as the demand for nickel is only 12% of the total mass. Looking at other materials emphasizes this even more. The results of cobalt and lithium show that a purely mass-based assessment cannot capture the often disproportionate environmental impacts. Cobalt, for example, represents only 2% of the imported mass but causes 10.94% of the total costs. Lithium also constitutes only 2% of the needed mass but causes 7.7% of total costs. This non-linear relationship is visualized in Figure S 1 in Supplementary Material 1, while the cost shares of the associated materials for different technologies are listed in Table S 3 in Supplementary Material 1.

3.2. Cost distribution per country

Fig. 3 Panel A shows the costs mapped on a country level, while Panel B shows the same costs related to country GDP. In Panel A, our model indicates that China carries a large share of overall costs (€8.2 billion and thus 21% of all costs), followed by the U.S.A. (€4.5 billion), India (€3.9 billion), and Saudi Arabia (€2.2 billion). The EU carries only €2.7 billion of these costs and thus only 7% of the total cost, which indicates that the EU will probably pay only a small part of the environmental costs induced by its low-carbon development. Panel B will be analyzed in more detail in Subheading 3.3.

The absolute damages for the four assessed impacts are shown per impact in Fig. 4 and are analyzed in more detail in the following. The global impacts due to climate change are outlined first followed by

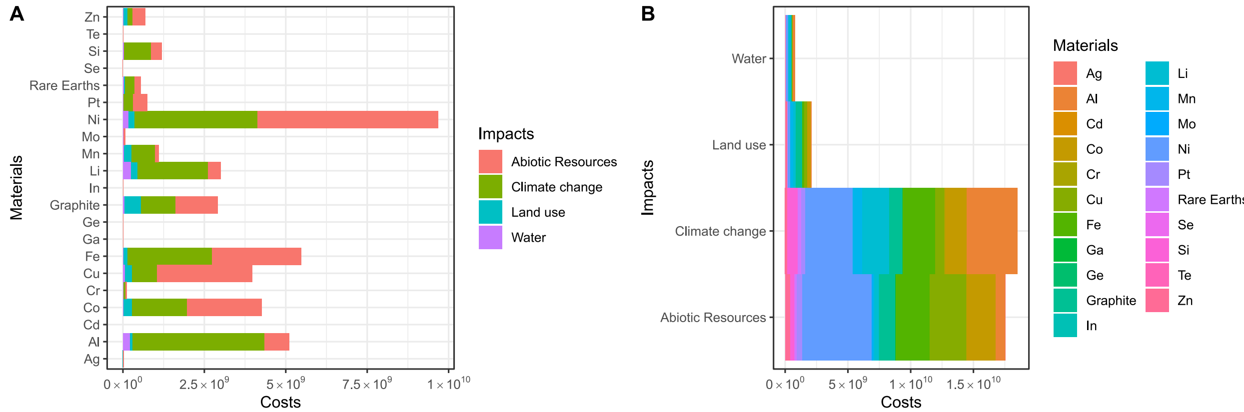

Fig. 2. Costs per material (A), cost per impact (B).

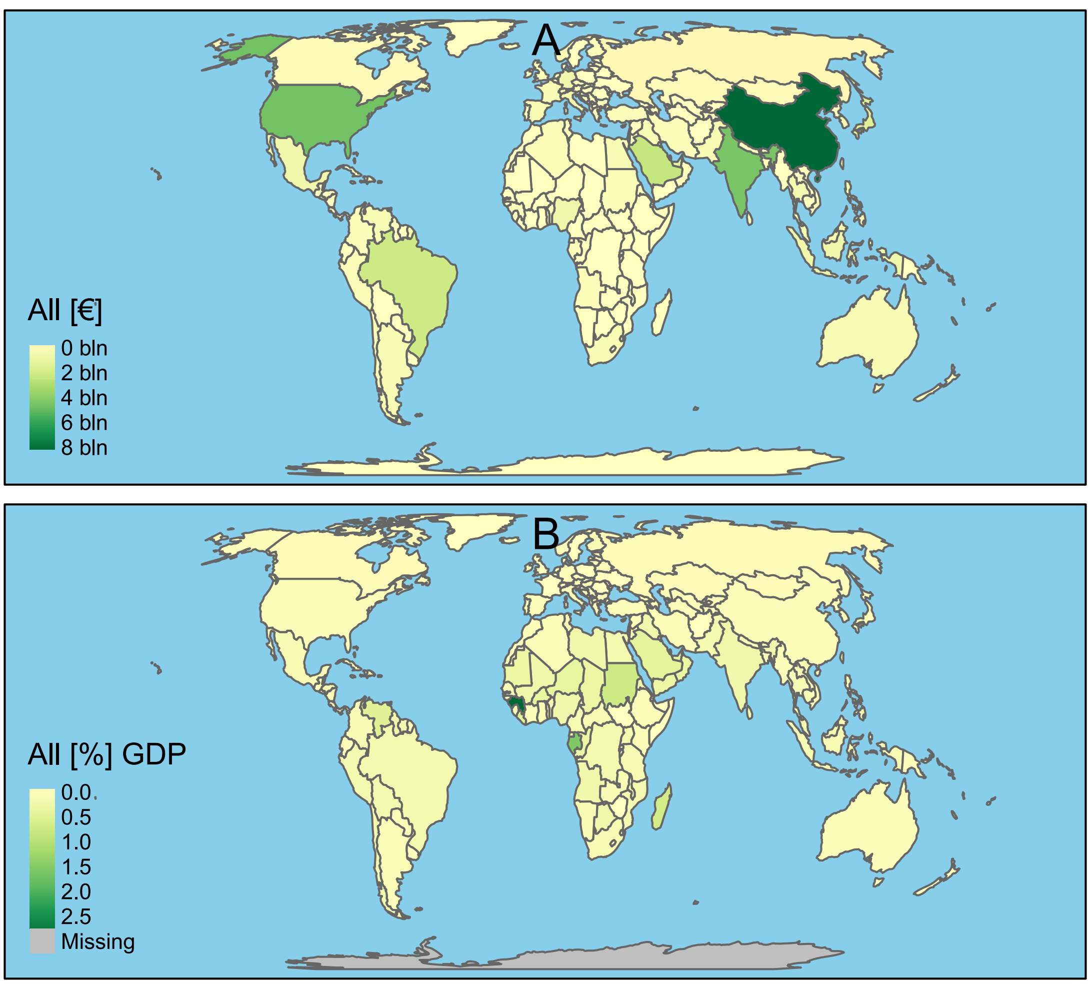

Fig. 3. Total environmental costs including damages due to climate change, water use, land use, and abiotic resource depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] of GDP (B) medium demand, 0% discount rate, no recycling considered.

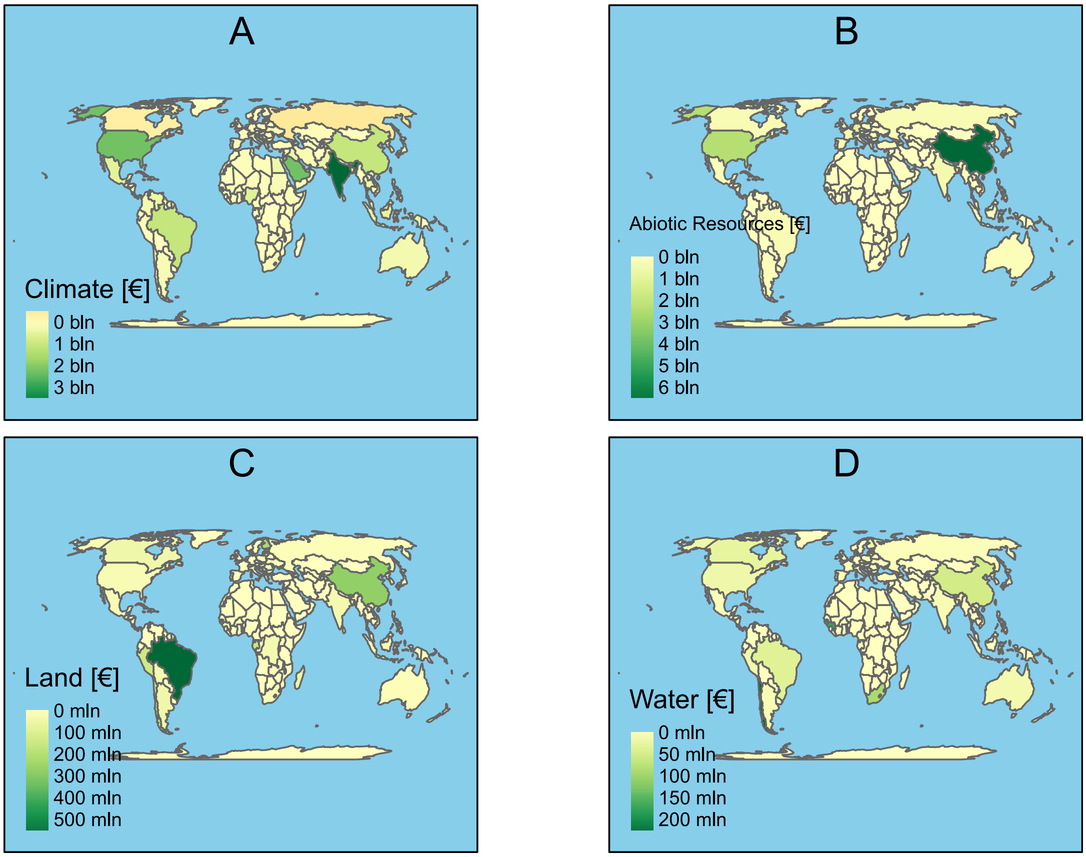

Fig. 4. Environmental costs due to climate change (A), abiotic resource depletion (B), land use (C), and water use (D), medium demand, 0% discount rate, no recycling considered.

abiotic resource depletion, land use, and water use.

For carbon emissions, it is visible that the countries where the carbon emissions occur (see Supplementary Material 1, Figure S 2 Panel A) have only a limited overlap with the countries that carry the highest social costs of carbon (see Fig. 4 Panel A). The most relevant materials that contribute to climate damages are nickel, aluminum, and iron. Based on the supplying countries for the EU, the highest emissions take place in Guinea (aluminum), Finland (cobalt), Brazil (iron, graphite, and aluminum), and Canada (nickel). The damages from these emissions are not concentrated in these countries as visible in Fig. 4 Panel A, but in India, the U.S.A., and Saudi Arabia. Nevertheless, some overlaps occur: Brazil and China cause a large share of emissions (12% and 6% respectively) and both carry just under 6% of the associated costs. The EU has no burden associated with carbon emissions due to its low-carbon development, as Ricke et al. (2018) provide negative values (a benefit) of climate change for most of the EU 27 countries because their temperature is assumed to be below the economic optimum (resulting in a benefit of €589 million).

The ADP costs are distributed to countries in Fig. 4 Panel B. The picture for ADP is similar to that for climate change, as the countries whose material depletion cause the largest damages are not affected by these damages accordingly. This distribution of damages occurs because, the countries that are most dependant on the materials for their economic activity (and have therefore been assigned the largest share of the damages as outlined in the method section Step 4) do not overlap with the countries that extract the largest share of these materials. Nickel, iron, and copper production are responsible for the greatest damages. As visible in Supplementary Material 1 Figure S 2 Panel B, the countries whose production and mining cause the greatest environmental costs are Canada (nickel), followed by Brazil (iron), Finland (cobalt), and Chile (copper). These countries pay only a small fraction of the costs. The associated burdens are carried by the two largest economies (China and U.S.A.), even though their contribution to the costs of depletion is much lower. Other countries with high industrial economic activity such as Japan also have comparatively high damages, because their high industrial output in the transport sector is dependent on certain materials (iron, nickel, copper, and cobalt) and could therefore be sustained longer if materials were depleted at slower rates. The EU's share of the burden is €2.87 billion and thus 16% of overall ADP costs.

Compared to climate and abiotic depletion costs, land use and water costs are more concentrated in the countries that supply the EU with its materials for low-carbon development. The country with the highest costs related to land use is Brazil, which carries about a fourth of the associated land costs (dominated by graphite). Brazil has high shares of rainforest biome, which delivers large amounts of ecosystem services and thus has high land use costs. Mining is already one of the main causes of deforestation in the Amazon today (Sonter et al., 2017), representing about 10% of deforestation. The destruction of the rainforest due to mining could therefore be accelerated due to low-carbon development as many of the key materials are mined in Brazil. The country with the next highest damages is China. It currently supplies 43% of Europe's graphite imports, which has the main impact, but China has a lower value than Brazil as its biomes have lower ecosystem services than Brazil's. The next highest burden is borne by Finland due to cobalt production. However, this only holds true if Finland really supplies Europe with nearly all its increased cobalt demand, which we assume based on the use of linear upscaling. This might influence results significantly, especially for cobalt. The EU currently imports very small amounts only. Finland produced ten times the amount of the current EU imports. Based on linear upscaling, we assume that most of the increased demand can be met by Finnish production. However, if the DRC, the country that currently dominates cobalt supply worldwide and has the

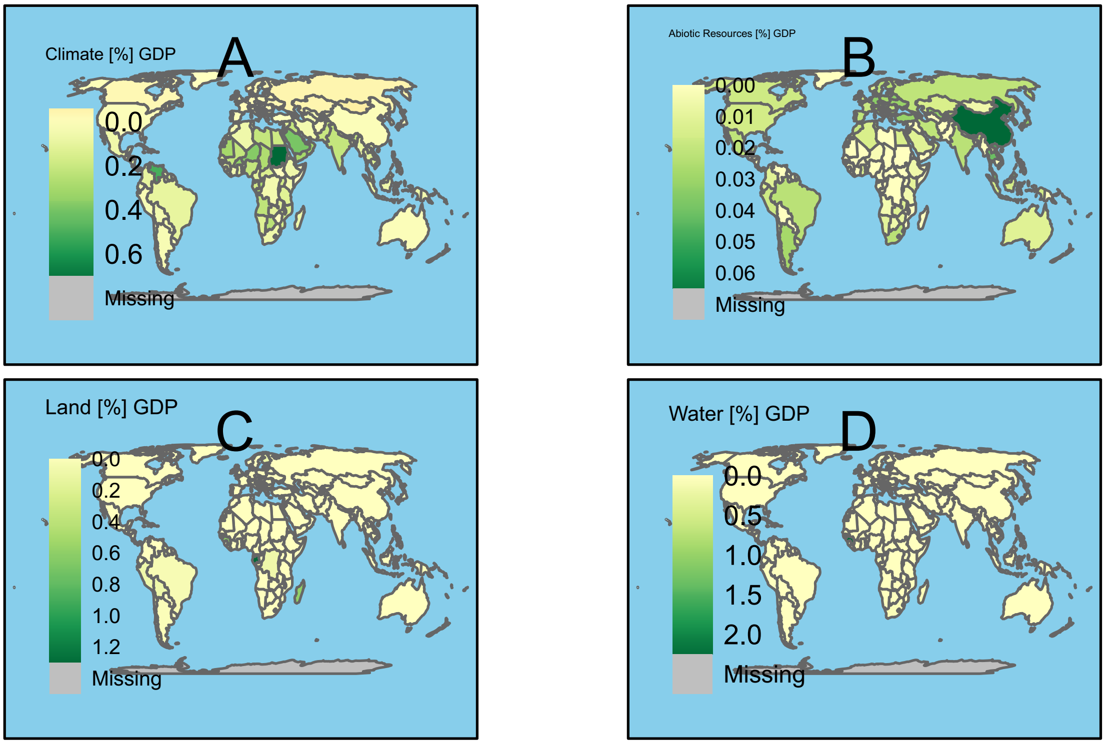

Fig. 5. Damages related to GDP for climate change (A), abiotic resource depletion (B), land use (C), and water use (D), medium demand, 0% discount rate, no recycling considered.

largest deposits, dominated European supply, the costs would increase tenfold (from around €270 million to over €3 billion) and the damages to ecosystem services would increase significantly. This is due to the fact that the DRC has very high ecosystem service value per m^2, as most of it is covered with tropical rainforest biome. The fact that mining causes deforestation in DRC has been already identified as a main challenge (Butsic et al., 2015) and is likely to worsen with increased demand for cobalt. The EU's share of burden for land use is €246 million and thus 11.7% of overall land use damages (mainly from Finnish cobalt mining).

The countries carrying the largest burden related to water use are Chile (30%), Guinea (22%), South Africa (12%), China (7%), and Brazil (5%) (see Fig. 4 Panel D). The severe water scarcity due to mining in Chile has been shown by others (Aitken et al., 2016) and is thus in line with the expectations. For Chile, the damages are mainly due to lithium production. Because the reference data is from 2015 and more lithium has since been produced in Australia, Australia's burden might rise. The Chilean water impacts occur due to economic water damages and species loss as Motoshita et al. (2018) assume no impact of malnutrition induced by water use in Chile. Guinea is an important bauxite mining site, and this activity leads to water impacts on species followed by economic impacts and impacts on human health induced by the EU's rising demand. The EU has environmental costs of only €52 million related to water use, and 7% of water use costs are due to Finnish cobalt mining, Greek nickel mining, and Spanish copper mining.

3.3. Cost distribution per country, related to GDP

If the costs are expressed related to GDP, we see a different picture (Fig. 3 Panel B) compared to the results presented in Subheading 3.2. The countries suffering most damages related to their GDP are Guinea (2.9%), Gabon (1.5%), Madagascar (0.7%), and Sudan (0.7%). While the absolute damages are dominated by global costs related to climate change and abiotic resource depetion, the damages of land and water use related to GDP can be tied to specific local mining. The damages in Guinea due to bauxite mining, in Gabon to manganese, and in Madagascar to graphite, while the effects in Sudan are attributable to the high impact of climate change there. We use the results in Fig. 5 to understand the costs related to GDP and the contribution of the different environmental costs.

Fig. 5 Panel A shows that Sudan, Venezuela, Kuwait, and Saudi Arabia have the highest damages for climate change related to their GDP. None of these countries contributes significantly to the carbon emissions related to the EU's low-carbon development (all below 0.1% of the total emissions). However, the assessed damages never exceed 0.6% of GDP.

When the ADP damages are expressed related to GDP as shown in Panel B of Fig. 5, the damages are even less concentrated compared to the climate change damages, with the highest relative damages being carried by China (0.06% of GDP). Based on the country allocation chosen in the methods of ADP, industrialized countries with a high share in productive sector output would benefit most from a slower resource depletion pathway as their high industrial output depends on the supply of raw materials. As these countries mostly have a very high GDP, the costs are relatively low when compared to the total GDP.

For land and water use, the damages are more concentrated compared to climate change and ADP, even though the environmental costs are much smaller (as they represent 5.4% and 2% of damages respectively). For water use, the impact on Guinea is most severe. For land use, manganese mining in Gabon and bauxite mining in Guinea exceed damages of 1% of GDP. The pressures on Madagascar are induced by graphite mining. The cases in which environmental costs exceed 1% of a country's GDP are always linked to local mining activities, even though the damages from water and land use were small compared to damages from ADP and carbon emissions. As these impacts affect ecosystem service provision and malnutrition, it is probable that the related costs would be borne by low-income and thus less resilient human beings.

3.4. Impact of scenarios

In the following, we will outline how the results change for the seventeen different scenarios. The magnitude of the costs varies from €13.1 billion to €74.8 billion in damages in 2050. The most severe effect has a 3% discount rate that reduces costs occurring in 30 years by 59% and a 1% discount rate that reduces costs by 25%. Additionally, recycling also lowers the associated damages as we assumed zero environmental impact due to recycling. The scenario with the lowest associated costs is thus the low-demand, recycling, and 3% discount rate scenario (see Figure S 43 to Figure S 46 in Supplementary Material 1). The scenario with the highest costs is thus the opposed scenario with high demand, no recycling, and no discounting (displayed in Supplementary Material 1 Figure S 47 to Figure S 50). However, the damages caused by the materials remain relatively consistent, with nickel having the highest values in nearly all scenarios, apart from the high-demand scenario in which aluminum has a higher relative importance. In the recycling scenarios, the relative importance of nickel, copper, and cobalt decreases as these materials have higher recycling rates than the other materials. The distribution of the impact categories—with climate change having the highest damages, followed by ADP, land use, and water use—is consistent in all scenarios.

4. Discussion

In the following, we first address uncertainties in the underlying data, assumptions, and methods, then look at how the results could be used in policy-making, criticality assessment, and further research.

The main weakness of the study is related to the large amount of data and the associated uncertainties. The amounts of materials imported into the EU provided by comtrade are provided as estimates only. Furthermore, we scaled up the current supply mix (imported amounts and domestic production) linearly to meet the demand projected in the foresight study, thus we did not consider exhaustion of deposits and changing trade partners. This limits the results of the study as the supply of resources can change dramatically in a 30-year period. These shifts can be extreme, especially for specialty metals that are produced only in small amounts. For example, the U.S. produced about the same amount of rare earths as China in 1995 (USGS, 1996). Furthermore, the EU foresight study assumes that certain technologies will be adopted. For example, battery technologies could change so that fewer or other materials are needed.

The emission intensities and land occupation intensities per kg of material extracted from the GaBi and ecoinvent databases contain some uncertainties. The databases often provide aggregated factors for mining and processing and do not account for different production processes in different countries. Therefore, the aspect that mining and processing takes place in different countries (e.g., as in aluminum production) is neglected. Moreover, the underlying environmental impacts represent the environmental damages of the production per mass unit today, but production might increase in efficiency in 30 years. The system boundaries to account for environmental impacts differ between the databases (ecoinvent often contains more infrastructure) but could not be harmonized as no database contained all materials. Another limitation of the study is that we include only carbon emissions, ADP, land use, and water consumption as they are considered the most severe for mining. The inclusion of additional impact categories would lead to higher environmental costs. Our assessment is therefore an underestimate. Additional impact categories should be included in upcoming studies.

Moreover, the monetization has significant uncertainties and tends to undervalue impacts on biodiversity (Arendt et al., 2020a). The monetization approaches for ADP and carbon emissions that are linked to economic growth have higher results compared to the impacts estimated through non-market valuation for water and land use, even though relatively high European values were applied to human health and species valuation. The social costs of carbon have a wide range: from €7.6/tCO_2-e ($10 minimum estimate of the interagency working group (IWGSCC, 2013)) to €69,009.50 /tCO_2-e ($100,000 (Archer et al., 2020)). If we applied the lower carbon emission damages, climate change would only represent 2% of the total costs and be only €484 million, while application of the higher factor would increase the results by 370 times. For abiotic resource depletion, a similar picture arises. Some economists argue that stock depletion is not a problem in the future (Jowitt et al., 2020) and thus does not create an externality, while others apply much higher environmental costs than we did (e.g. (Steen, 2016)). As we allocated according to sector output, we account only for producer not for consumer surplus. An additional problem is that we use only marginal values for non-marginal problems, but average environmental impacts are not available for use with any of the impact assessment methods. Additionally, we expressed the damages in relation to GDP to highlight wealth differences between countries, not within countries. Regarding the monetization of impacts on ecosystem services, the approach could be improved, as currently we include only land occupation and assume that the ecosystem services that are provided by the ESVD cannot be delivered if the space is occupied. Damages to ecosystem services from other emission sources are currently not considered, but further research is needed here to link ecosystem services and LCA (Alejandre et al., 2019). The monetization methods applied in this research were used for a damage-based assessment that includes damages to the economy as well as human health and biodiversity for most impacts assessed (as applied for water and land use). For carbon emissions, the country-level social costs of carbon do not contain information on distribution of DALYS and lost species on country-level, only economic damages. Further, intrinsic values cannot be monetized (Sandel, 2012), and monetization is therefore anthropocentric.

In addition to the fact that this research underestimates environmental costs due to the exclusion of several impact categories, it neglects the increased demand for concrete and energy for the infrastructure construction of low-carbon development as it includes only the demand for metals and ferrosilicon, not for bulk materials. Moreover, the use of historic recycling rates for the recycling scenarios leads to overestimation of the costs. The assumption of zero environmental impact for recycling, however, leads to overestimation of the environmental benefit.

Despite the outlined shortcomings, the approach developed here has many possible research applications, namely integration into criticality assessment, supplementing integrated assessment models (following and deepening the approaches already integrated in the REMIND model (Baumstark et al., 2021; Rauner et al., 2020)), as well as coupling environmental cost analysis with material flow analysis. Integration into criticality assessment will allow inclusion of supply disruptions that occur due to protests over environmental degradation caused by mining as well as political reaction to a deteriorating environment or to mining and processing of materials conflicting with national climate targets. Additionally, criticality assessment can be further developed in the direction of sustainability assessment considering both supply disruption and a country's impact on other countries. Coupling environmental costs with material flow analysis and a more realistic estimation of impacts from recycling can provide additional insights into the interaction of material flows with society. Moreover, it will be possible to calculate the relation of economic benefits and challenges posed by mining and processing of materials in a society that is using materials ever more dynamically. In this study we have only assessed the impact of the EU's low-carbon development. It should be further investigated how the results would change if other world regions adapt a similar development pathway. The adaptation of a similar pathway by other world regions is likely to increase the pressure on countries that are currently mining the necessary materials.

Possible policy responses to lower the environmental costs could include development of material-efficiency as well as recycling strategies, especially for the materials for which damages are high and recycling allows for a reduced environmental burden. In our study, we neglected recycling processes that are in the early research stage and currently not widely applied (despite promising approaches for e.g. lithium ion batteries, including lithium and cobalt (Gao et al., 2020)). Thus, recycling could reduce economic criticality (Santillán-Saldivar et al., 2021) as well as the environmental burdens associated with extraction. For the materials for which high local impacts on water availability and ecosystem services have emerged, recycling would also avoid these local burdens. A further reduction of these impacts could be enabled by supply chain management practices such as responsible sourcing, employment of local people, and restoration of the environment.

Additional possibilities to reduce the environmental costs, especially for the electric mobility and energy technologies, could include rethinking current consumption lifestyles. Bobba et al. (2020) project 250 million new electric cars and 40 million fuel cell cars in the EU's vehicle fleet, which would equate to 0.65 cars per person. Increasing urbanization trends and expanding other forms of mobility such as cycling and public transport could reduce the environmental costs of the mobility transition (Byrne et al., 2021).

5. Conclusion

This study showed that materials for the EU's low-carbon development contribute to several environmental pressures such as climate change, ADP, reduction of ecosystem services, and availability of water, but that the environmental costs incurred are much lower than the environmental costs caused by the current GHG emission level of the EU. However, these costs fall disproportionately on the countries that mine and produce materials, in this case China, Brazil, the U.S., and Saudi Arabia in absolute terms and Gabon and Guinea in relation to their GDP. These costs were identified with the newly developed method and should be monitored and if possible reduced through tailored policy measures. Methodologically, the approach developed here has great potential to complement integrated assessment models to account for environmental impacts beyond carbon emissions as well as to be integrated into criticality assessment, as environmental impacts might be a reason to reduce mining activities. On a general basis, the approach can be applied to any metabolic shift that our society is facing (e.g. digitalization or low-growth scenarios) to see how it influences people's welfare using geographical resolution.

Funding

This work was funded by the Bundesministerium für Bildung und Forschun, project "R2Q : RessourcenPlan im Quartier" (funding number: 033W102E). We would like to thank the Bundesministerium für Bildung und Forschung for its financial support.

CRediT authorship contribution statement

Rosalie Arendt: Writing – original draft, Conceptualization, Data curation, Formal analysis, Visualization. **Vanessa Bach:** Project administration, Writing – review & editing, Supervision. **Matthias Finkbeiner:** Project administration, Writing – review & editing, Supervision.

Declaration of Competing Interest

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

Acknowledgements

We thank Lisa Winter for her contribution and discussion related to the monetization of biodiversity and ecosystem services. Additionally, we thank four anonymous reviewers that greatly improved the quality of this manuscript with their thoughtful comments and advice. Furthermore, we are grateful to Rebecca Holmes Peterson for careful proofreading that refined the language and style of this article.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2021.106057.

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4 Discussion and Outlook

This chapter starts with a discussion of the uncertainties in the results (4.1). Then it reflects on the links of the developed methods to current methodological trends (in criticality assessment and in monetization) (4.2). The chapter closes with the strengths and weaknesses of the developed methods (4.3) and possible future research (4.4).

4.1 Discussion of Uncertainties in the Results

The qualitative sources of uncertainty for the performed analysis are depicted first, followed by a quantitative comparison of the developed methods from Paper 2-4. Sources of the underlying uncertainties are the production figures e.g. by the USGS (U.S. Geological Survey, 2018; United States Geological Survey, 2016); and for Paper 2 and 4 the calculation of the import mixes, as no detailed trace back of the materials was performed (i.e. where they are really extracted).

Furthermore, the underlying data for Paper 2 and 4 were fused and manipulated, which leads to some uncertainties. For the trade data it was assumed that countries that did not produce a certain material, but exported it, exported the world production mix. A trade trace back model (to identify where materials that are exported and imported were originally mined and processed) could further explore how this assumption influenced the results. One of the greatest uncertainties of Paper 4 was the linear upscaling of current trade patterns.

The uncertainty of a criticality assessment and also of an LCA are difficult (if at all) verifiable stochastically nor in terms of significance as defined in statistics. The main reason is that most errors are logical not statistic errors and that data is mostly available only as a distinct data point and not as a distribution. For criticality assessment, several factors are interplaying that inhibit a quantitative prediction of an occurring supply risk. Sources of uncertainties are connected to underlying data and the political influence that many of the indicators are prone to, such as the data sourced from the USGS (Mobbs, 2005) and from the World Bank (Fleck and Kilby, 2006; Kilby, 2013).

A source of uncertainty related to Paper 1 is the variation of the conversion factors for the review to yield one reference impact category indicator. A discussion of this will be performed in section 4.3 and can be found in the discussion section of Paper 1. Next to the uncertainties of the underlying quantitative data, the monetization has a significant influence on the results, which could be underlined based on the review in Paper 1.

The range of associated environmental costs can be best illustrated with greenhouse gas emissions. The magnitude of the emissions has comparatively lower uncertainty compared to the other impact categories. However, the environmental costs of one emitted ton of carbon vary widely. Some aspects relate to all environmental interactions, others are carbon dioxide emission specific. These differences have been discussed in the literature (Bachmann, 2020; Dong et al., 2019; Edenhofer et al., 2021; Kornek et al., 2021):

- Socio economic pathway (emission and economic development- climate change specific)
- Equity weighting (relevant for all impacts)
- Discount rate (relevant for impacts with time lag- loss of biodiversity, climate change, resource depletion)
- How non-marketed goods are integrated in the assessment (relevant for all impacts)

In Paper 1, the monetization methods were compared, which yielded a non-normal distribution of most impact categories and mostly a spread across two orders of magnitude. However, two new approaches have been developed in Paper 3 and 4 that have not been compared yet. They will be compared in the following, but also with the results from Paper 2, to see if the monetization influenced the results. An uncertainty assessment has been partially carried out for Paper 3, as a high and low monetization factor was applied and therefore, the damages could be provided in ranges. For Paper 3, the dominating materials changed due to the chosen monetization method. The high estimate was dominated by fossil energy carriers, while the low estimate was dominated by iron.

In this paragraph the damage factors and developed monetization methods from Paper 2, 3 and 4 are compared to each other. As outlined in the discussion of Paper 2, if relative impacts per kg material are quantified, precious metals have the highest impacts. However, when the absolute impacts of the material flows are measured, the bulk materials (iron/steel and fossil energy carriers) have the highest impacts (Paper 3). To assess how the different methods assess the impact per kilogram, we have calculated the derived cost/impacts per kg and how the different methods rank the materials for Paper 2-4. The ranking of the materials is shown in Table 4.

Table 4: Ranking of materials by the different papers for the impacts of the materials per kg in descending order; the materials gold (Au), rare earths and graphite are highlighted in red, blue and green respectively, as their ranking is analyzed in more detail in the upcoming text

Rank	Ranking Paper 3 high	Ranking Paper 3 low	Ranking Paper 4	Ranking Paper 2
1	Au	Au	Au	Au
2	Pd	Pd	Pd	Pt
3	Pt	Pt	Pt	Nb
4	Ta	Ta	Ta	Ag
5	In	In	In	Li
6	Ga	Ga	Ga	Cr
7	Li	Li	Ag	Cu
8	Ag	Ag	Li	Zn
9	U	U	U	Fe
10	Nb	Nb	Co	Sn
11	Co	Sb	Rare Earths	Pd
12	Sb	Co	Mg	Sb
13	Zr	Te	Nb	Mn
14	Te	Zr	Mo	Pb
15	Se	Se	Te	Mo
16	Ni	Ni	Ti	Ti
17	Sr	Mg	Ni	Oil
18	Mg	Sr	Al	U
19	Graphite	Graphite	Sb	Cd
20	Mo	Mn	Zr	Coal
21	Mn	Al	Mn	CH_4
22	Al	Cr	Si	Sr
23	Rare Earths	Ti	Se	Si
24	Ti	Rare Earths	Cu	Co

25	Cr	Si	Cd	Graphite
26	Si	Mo	Cr	Rare Earths
27	Cu	Cu	Zn	P
28	Cd	Cd	Sr	Lignite
29	Zn	Sn	Sn	Zr
30	Sn	Zn	Pb	Se
31	Pb	Pb	Graphite	Mg
32	Fe	Fe	Fe	Ga
33	CH_4	CH_4	CH_4	Te
34	Oil	Coal	Lignite	Ni
35	Coal	Oil	Oil	Ta
36	P	P	P	Al
37	Lignite	Lignite	Coal	In

All methods (from Paper 2, 3, and 4) rank gold consistently as having the highest impact per kilogram. Paper 3 and 4 have very similar results and also the high and low ranking per kilogram is largely consistent with only minor changes. For example, lithium is judged as having a higher impact per kilogram as silver and vice versa. Some materials are weighted very differently, for example rare earths (weighted higher for Paper 3) and graphite (weighted higher for Paper 4). These differences might be explained by the inclusion of additional impact categories such as toxicity. The toxicity impacts of rare earths are very high and decisive in making these materials significant for the assessment in Paper 3. For graphite, land use had a high impact and land use was assigned a higher monetary damage for Paper 4. This was due to the fact that its impacts were coupled to delivering ecosystem services, which was neglected in Paper 3.

For Paper 2, 3 and 4 there are some significant differences. Tantalum has a high rank in Paper 3 and 4 (rank 4) and a low rank for Paper 1 (rank 35). Aluminum is considered low impact for Paper 1 (rank 36), while it has a medium score for Paper 3 and 4 (rank 22 and 23). This is due to the fact that it is extracted in countries with low water scarcity and not so sensitive biodiversity. However, its energy demand combined with the comparatively high environmental impacts per kilogram are higher than the country characteristics from where it is sourced would suggest. Fossil energy carriers get a low score for impacts per kilogram, but already in Figure 1 it is clear that very high masses are extracted. This shift changes, when the total impacts (Paper 3) are assessed especially for the high estimate, as ReCiPe assigns the depletion of fossil energy carriers comparatively high environmental cost (Alvarenga et al., 2016; Rørbech et al., 2014).

The damages per kilogram are plotted for all the materials and the methods introduced in Papers 3 and 4 in Figure 10 (Paper 2 does not apply a monetary unit and can therefore not be plotted in the same figure).

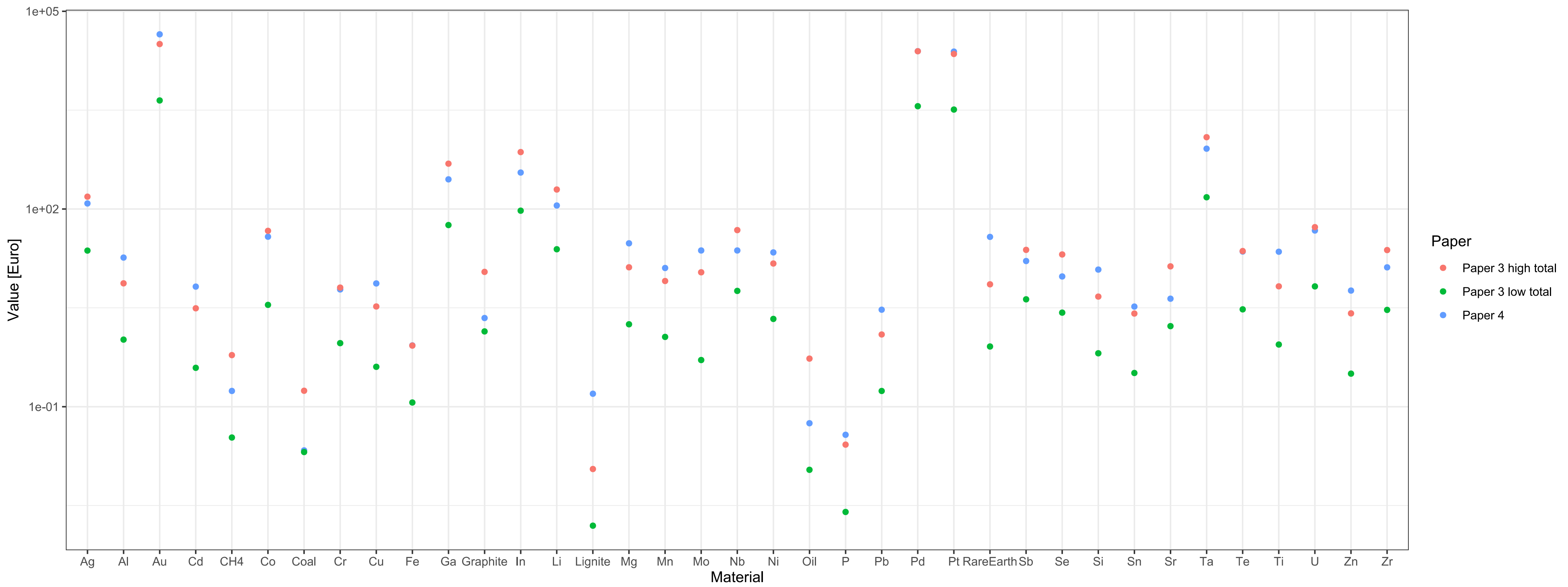

Figure 10: Impact per kilogram of material for the different methods in logarithmic scale

While the ranking for Paper 3 and 4 is very similar, the values per kg as shown in Figure 10 vary widely. The most extreme case is lignite, where the maximum value per kg is equivalent to €1 per kg and the low estimate 0.1 ct and thus 800 times as high as visible in Figure 10. The material with the narrowest distribution is coal, with a factor of only 8.5. It is visible that for every material, the low monetized results of Paper 3 are the lowest, while for the high estimate of Paper 3 and Paper 4 the picture is more divers. This is the case even though eight impact categories are covered in Paper 3, while Paper 4 only covers four impact categories. The environmental costs per material calculated in Paper 4 exceed the costs calculated for the high estimate of Paper 3 for 20 materials. For 17 materials the costs of the high estimate of Paper 3 are higher. Thus, no clear picture emerges. Paper 4 has by far the highest costs especially for the economic damages of water, but also a higher CO_2 price (€290/ton instead of €180/ton for the high estimate of Paper 3). For some materials the high estimate of Paper 3 and the values from Paper 4 are rather close together (cobalt, silver, chromium, platinum, tin, uranium).

One reason is that the future effort method assessing metals for ReCiPe provides lower values than the weighting with the Hotelling's rule as assessed by Huppertz et al. (2019). Additionally, economic damages of water use are neglected in the approach in Paper 3 and also ecosystem services of land use are not quantified. Thus, the fact that more impact categories are covered in Paper 3 sometimes yields lower environmental costs compared to an integration of feedback from environmental impacts on the economy. As outlined in Paper 1, the values do not converge also not for the methods developed here, but these are too few values to derive a judgement about the distribution. As shown in Paper 3, the developed research helps to better understand the ranges and also the choices, perspectives and world views that produce a certain monetized number. However, both approaches are sure to be an underestimation, as Paper 4 covers less impact categories and Paper 3 neglects damages to physical capital and economic activity that was covered e.g., through the social costs of carbon in Paper 3, but also ecosystem services, which have been quantified by the Ecosystem valuation database (ESVD).

In a next step the results of Paper 3 and 4 are applied to the case study that was conducted in the other paper, respectively. With this application not only the derived values per kilogram, but the quantified environmental costs can be compared. Thus, the monetization factors from Paper 3 (previously applied to global production) are applied to the EU's low-carbon development and the monetization factor from Paper 4 (previously applied to low-carbon development) are applied to global material production.

If the cost factors of Paper 3 are used for the case study of the EU's low carbon development, environmental costs of €87 billion per year for the high estimate and €11 billion for the low estimate are quantified. Distribution of the high and low valuation are shown in Figure 11 and Figure 12, respectively.

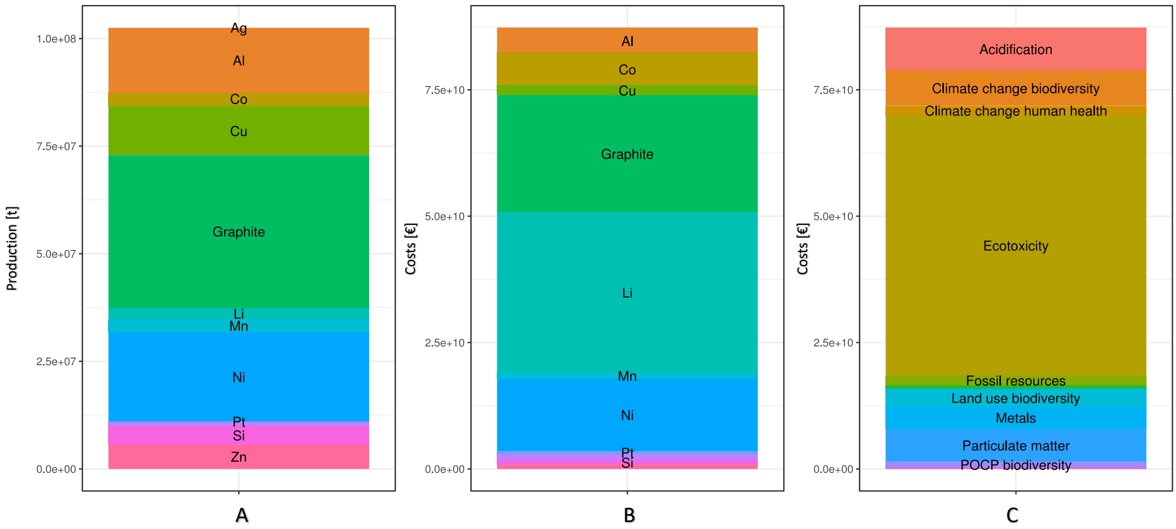

Figure 11: Results of applying monetization factors from Paper 3 to case study of Paper 4 (low-carbon development), A – the distribution of the produced masses to the materials, B the distribution of environmental costs to the materials and C the distribution of the costs to the impact categories; high estimate

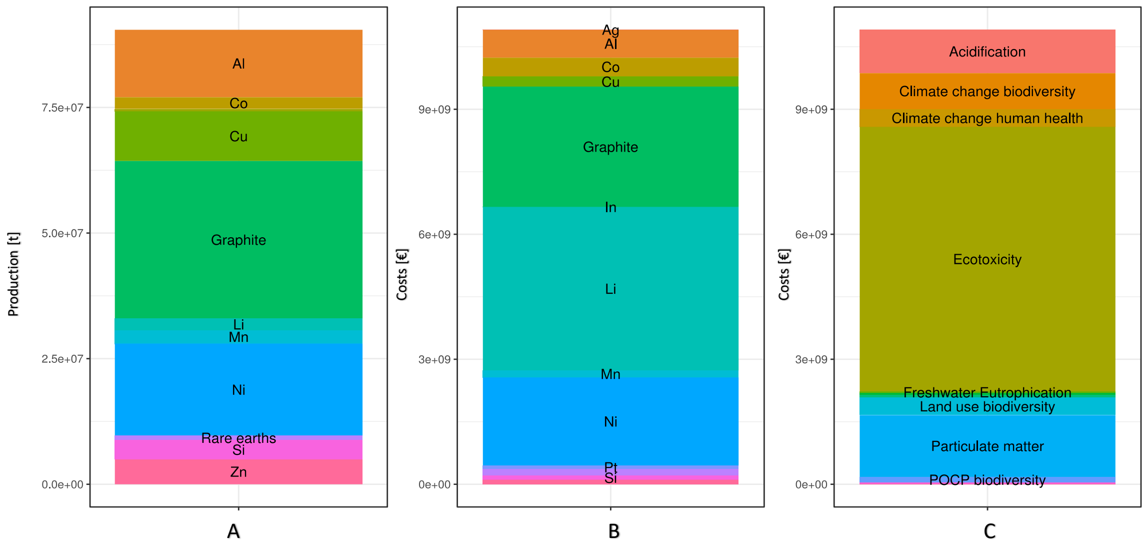

Figure 12: Results of applying monetization factors from Paper 3 to case study of Paper 4 (low-carbon development), A – the distribution of the produced masses to the materials, B the distribution of environmental costs to the materials and C the distribution of the costs to the impact categories; low estimate

The high estimate depicted in Figure 11 is higher than the medium estimate of Paper 4 (€38.9 billion), while the lower estimate is lower (shown in Figure 12). However, the dominant impact category for both estimates is ecotoxicity induced by graphite and lithium. Ecotoxicity needs to be assessed and evaluated with care in LCA as it contains significant uncertainties (ILCD, 2010; Kerkhof et al., 2017; Lehmann et al., 2015). The costs are dominated by lithium and graphite, while nickel has a lower share of the impacts, which was the dominant material in Paper 4. Further even acidification and particulate emissions have higher damages than climate change. It appears that impact categories that were not covered in Paper 4 play a significant role and should be included in further studies on the environmental costs of the energy and mobility transition.

Then we applied the monetization factors of Paper 4 (formerly applied to low-carbon development) to the global production of materials. The results are shown in Figure 13.

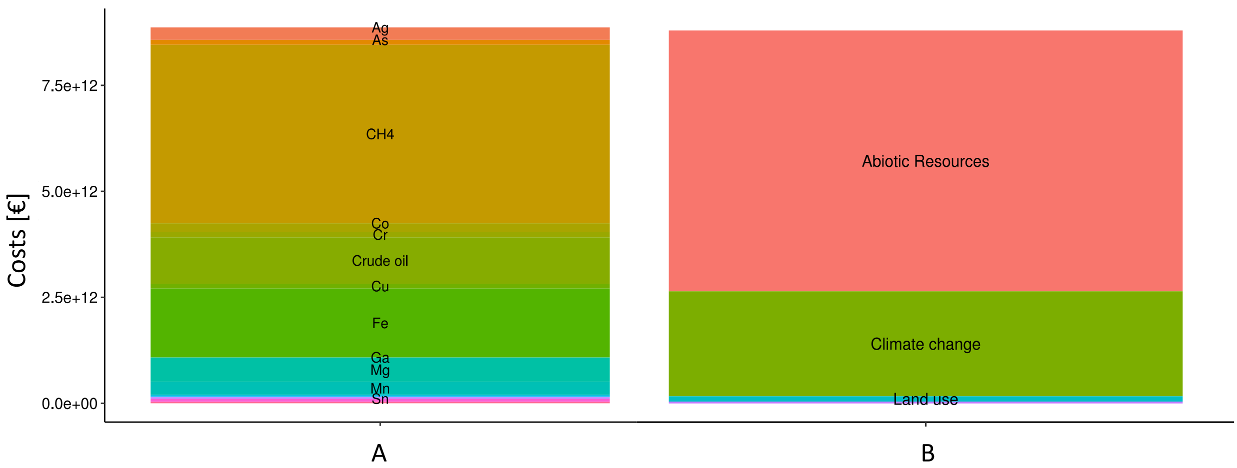

Figure 13: Results from applying the monetization factors from Paper 4 to the case study of Paper 3 (global material production) A: Distribution of environmental costs per material; B: distribution of environmental costs to the impact categories

The environmental costs are dominated by the impact category abiotic resource depletion for the application of the method in Paper 4 (70 % of costs) to the case study of Paper 3; and thus represent a higher share, than for the approach used in Paper 3. The costs calculated with the approach in Paper 4 applied to the global production yield total costs of 2.7 trillion and are thus in between the high and low estimate of the results from Paper 3. The results are to a large extend dominated by fossil energy carriers, especially natural gas. Land use and water use play close to no role. One reason for this might be that the environmental costs from depletion and climate change are global impacts and the local cultural impact of water and land use is not assessed in the methods, even though they might be severe.

To summarize, the outlined uncertainty assessment shows that the derived methods in this thesis share the challenge that was outlined as a result of Paper 1: the choice of the monetization method has a severe influence on the results and can change the prioritization of impact categories. There is also a trade-off between the methods that were developed as a part of Paper 3 and Paper 4: the method in Paper 3 covers more impact categories, while Paper 4 uses more up-to-date methods from environmental economics. However, both methods deliver monetary results in the same range, if the high estimate from Paper 3 is used. The uncertainty assessment highlights, what was recommended already in Paper 1: practitioners should choose their monetization methods carefully and ideally apply varying methods and critically evaluate the obtained results.

4.2 Discussion in Relation to Methodological Trends

In the following, the methods that were developed in this thesis are compared and put into perspective in relation to recent development in the field. First, focusing on trends in monetization in LCA and then followed by a discussion of trends in environmental criticality assessment.

Next to the presented review in this paper, two other reviews on monetization in LCA have been performed: one five years prior (Pizzol et al., 2015) and one year after Paper 1 (Amadei et al., 2021). Additionally, there is a review that focusses exclusively on monetary valuation of greenhouse gas emissions in LCA (Dong et al., 2019). The review by Pizzol et al. (2015) developed a criteria evaluation scheme with the goal to recommend one weighting method and concludes that choice experiments are most suitable to derive a monetary value for the damages to affected AoPs and that it would be best to use the LIME method. They also follow the reasoning by Finnveden et al. (2006) that it is best to use a monetization method that only uses one consistent approach for the valuation for all AoPs.

This reasoning is not in line with common practice in environmental economics. If a market price for a good exists (like e.g. the reduced availability of ores), or a revealed preference method can yield reliable results, these might as well be used in combination with stated preference methods as has been done in the New External Costs of Energy project (Desaigues et al., 2006; Preiss et al., 2008). In Paper 1 of this thesis, no monetization method was recommended, but the wide range of monetization methods and their qualitative characteristics was shown. This was based on the assumption that there are no purely scientific grounds on which market price, revealed preferences or stated preferences can be preferred. Moreover, abatement costs to reach a political target are also considered as a valid option. If a target is determined democratically, the marginal costs to reach it should remain a valid option (ISO, 2019). They can be perceived as a synthesis of the distance-to-target weighting and monetization.

Amadei et al. (2021) do not give a normative recommendation, which monetization method should be used, but rather focus on the compatibility of monetization methods with the environmental footprint method and compare how many coefficients were available. They find a similar variability in their results compared to Paper 1, as also Dong et al. (2019) did.

During the writing of this thesis (2019-2022) a UNEP/SETAC task force on weighting took place under the phase 3 of the Global Guidance for Life Cycle Impact Assessment Indicators and Methods (GLAM). Its goal is to derive a recommendation until autumn 2022 for at least one globally applicable weighting set, which is after this thesis was completed. Recommending at least one weighting method at global level was determined as the overall goal of the task force. In Paper 1 it was identified that the geographical scope is the most decisive parameter for the identification of a weighting method and the magnitude of the results. These differing payment abilities are blurred by a global weighting set, which is the goal of GLAM. Additionally abatement costs were excluded for GLAM, with the line of reasoning that the weighting has to be performed at AoP level. Moreover, these decisions made it more difficult to integrate monetization factors that were already determined in environmental economics, as these often do not monetize at the AoP level. Within the GLAM process a similar normative reasoning was followed compared to the review by Pizzol et al (2015), which resulted in a recommendation of the LIME method. A new round of questionnaires was developed, that should be distributed globally. This process and its results have not been finished, while this thesis was concluded, therefore the magnitude of the results cannot be compared.

However, the LIME 3 (Itsubo et al., 2018; Murakami et al., 2018) method yields very low environmental costs for CO_2 (€7/ton). This would be below most values considered in policy today, which risks that the benefits of reducing pollution will be undervalued. Additionally, damages of pollutants to physical capital are neglected with the AoP approach recommended in GLAM (e.g., damages to infrastructure- especially induced by floods and storms caused by the climate crisis), which leads to an underestimation of the benefits of abatement. The authors of LIME 3 acknowledge this gap (Murakami et al., 2018).

The developed methods from Paper 3 and 4 in this thesis are diverse. While the method in Paper 4 is related to midpoint assessment and is more attached to recent developments in environmental economics, such as the social costs of carbon by Ricke et al. (2018) and the ESVD (de Groot et al., 2020a, 2020b, 2012). The approach in Paper 3 is connected to endpoint results in LCA. In GLAM it was decided that only consistent weighting at endpoint level is valid. Thus, based on this criterion, the method developed in Paper 3 should be favored compared to the method developed in Paper 4. Based on the integration of research from environmental economics, Paper 4 would be preferable to Paper 3. It enables a synthesis of research in environmental economics and LCA, by using the values that are collected in the ESVD (de Groot et al., 2020a, 2020b) for an LCA study for the first time.

In the following, the development of the environmental criticality assessment of resources is discussed in relation to the derived methods in this thesis. An overview is provided in Table 5.

Table 5: Overview of existing criticality methods that include environmental assessments

Method	**LCA Impact categories covered**	**Absolute or relative**	**Material specific**	**Aggregation method**	**Country specific**	**Regionalized inventory**
Graedel et al (2012)	All impact categories from ReCiPe relevant for human health and ecosystems	Relative (per kg)	Yes	Equal weighting	No	No
ESSENZ (Bach et al., 2016)	Climate change, eutrophication, acidification, ozone depletion and formation of photochemical oxidants (smog)	Absolute	Yes	None	No	No
SCARCE (Paper 2 and Bach et al. (2017))	Climate change, water use, sensitivity of biodiversity, ADP	Relative (per kg)	Yes	Equal weighting	Partially (water and sensitivity of biodiversity)	No
Manhart et al. (2019)	Water stress Index (water use), cumulative raw material demand, cumulative energy demand	Relative	Yes	None	Yes	No
Eheliyagoda et al. (2020)	None- only Environmental performance index and regulatory quality from world governance indicators	Not applicable	No	Equal weighting	Yes	Not applicable

Paper 3	All ReCiPe impact categories apart from endpoint to freshwater and marine ecosystems	Absolute (produced mass)	Yes	Monetary	For DALY one value was chosen, species impacts were adjusted depending on the vulnerability of ecosystems.	Inventory as regionalized as databases allowed (Ecoinvent, 2021; Sphera Solutions Inc., 2021), documented in supplementary material of Paper 3
Paper 4	Climate change, water use, land use, resource depletion	Absolute (imported and produced mass)	Yes	Monetary	Yes, for water use, and land use, no for climate change and resource use	No

The methods are compared based on the following characteristics: whether they are material-specific, whether they determine absolute or relative damages, which aggregation method they use, whether they determine country specific damages and whether they use a regionalized inventory.

The environmental criticality assessment of Graedel et al. (2012) has been the first criticality assessment that includes environmental implications of mining. Paper 2, 3 and 4 assess the likelihood of supply restrictions induced by environmental impacts, including the depletion impacts caused by material extraction and processing. Graedel et al. (2012) take a different path- they assess the environmental impact- considering all impact categories and how they affect the ReCiPe endpoints human health and ecosystems. They leave out the impact on resources, as they claim that this is already covered by other assessments. Additionally, they rely on equal weighting and relative impacts per kilogram.

The first criticality assessment that focused only on environmental impacts was published in 2019 (Manhart et al., 2019). It proposes 11 indicators. Some of these are LCA impact category indicators. However, many indicators are semi-quantitative and also do not have an approach to prioritize certain materials, as the assessment does not aggregate the indicators and provides no approach for hotspot analysis. However important aspects such as acid mine drainage and radioactivity of ores are covered. Many of the proposed categories are currently not or only indirectly assessed with LCA and thus the two existing methods could be used to complement one another and taken an inspiration for further development.

Eheliyagoda et al. (2020) assess criticality with a focus on the environmental dimension- but without LCA indicators. They relied on the Environmental Performance Index and the sub indicator regulatory quality from the world governance indicators. Santillán-Saldivar et al. (2021b) base their assessment also on LCA and the GeoPolRisk method. Their endpoint assessment is also monetized, but via the possible economic damage of a supply risk to the production of the product system. Their approach differs compared to the research developed here as they see the LCA indicators and the criticality assessment as a part of a holistic Life Cycle Sustainability Assessment instead of a criticality assessment with a sustainability perspective. These proposals and methodological developments are not in conflict. Criticality assessment should be a part of Life Cycle Sustainability Assessment, while criticality assessment should also consider environmental and social impacts.

Generally, it would be desirable to have a method that covers many impact categories, that assesses absolute damages, material and country specific (from inventory and method perspective) with a reasonable aggregation method. Methodologically the Papers 3 and 4 have contributed to these goals, while the most significant remaining gap is the modification of available inventory regionalization and specificity. Furthermore, it would be of value to complement the monetary environmental damage assessment (that represents an inside-out perspective as defined by Cimprich et al. (2019)) with the economic damages of supply disruptions to products as proposed by Santillán-Saldivar et al. (2021b) (which represents an outside-in perspective). Such an integration would allow to assess the impacts of material demand on the environment, but also on the economic activity of producing the product itself in monetary terms.

4.3 Strengths and Weaknesses of Developed Approaches

The strengths and weaknesses of the conducted studies and presented papers are outlined in the following. They are more connected to methodological strengths and weaknesses and not to the underlying uncertainties, which has been discussed in section 4.1. Partially they repeat aspects that have been already described in the discussion of the papers, but they are summarized here again to provide an overview.

One strength of the thesis is the wide coverage of monetization methods in the review Paper. It covered all currently used monetization methods, where the documentation allowed. Further, the first time a link between the qualitative and the quantitative comparison of the monetization factors could be established. However, the conversion of monetary values per emission flow to one homogenous impact category indicator was necessary for the quantitative comparison. This is a severe challenge as the conversion factors depended on the used characterization factors. The applied conversion factors can be found in the supplementary material of Paper 1. These vary between the different impact assessment methods. They had a significant impact on the results, especially for example for photochemical ozone creation potential. A quantitative determination how severe this effect is, was not conducted for all impact categories. This is a major shortcoming of the review and a severe research gap in LCA. A quantitative comparison of characterization factors for most impact categories would be very useful, but would require an in-depth review of each impact category.

An additional weakness was that not all monetization methods for the impact category resources were covered. This was due to the fact, that only methods that covered several impact categories were included in the review and did not explicitly assess all impact assessment methods for resource depletion that monetized. Therefore e.g., the future effort method and the Hotelling's rule method that have been used in Paper 3 and 4 respectively have not been assessed in the review. However, one finding of Paper 1 was also verified with Paper 3 and 4: that the ranges of the derived values are large, even if the same methods with a high and low estimate are applied. For example, the range of Paper 3 from the high to the low estimate resulted in a 12-fold increase, while for Paper 4 this resulted in a nearly 7-fold increase.

The strength of the SCARCE method was that all sustainability dimensions are covered in this case study, which is the first criticality assessment on European level that does so. Additionally, the method is covering an economic free trade zone. It can be assumed that such a trade zone shares the associated supply risk. Shortcomings of the approach are that it relies on equal weighting and thus aggregates each dimension to a single score so that every dimension can be plotted in the criticality matrix. An additional weakness is that the applied method cannot predict supply restrictions with a certain probability quantification. This is the case for all criticality assessments. A connected shortcoming is that is not clearly identifiable that a certain material will become a problem. However, Paper 2 identified a very high supply risk for natural gas and petroleum oil, which could be verified by recent geopolitical events and energy price increases in Europe (Liadze et al., 2022). Furthermore, the identification of equal weighting as a weakness led to the development of a monetized environmental criticality assessment in Paper 3 and 4.

Additional shortcomings for the environmental dimension of Paper 2 are that for the water scarcity and sensitivity of local biodiversity only the characteristics of the country, where the materials are produced are considered. That means that if two materials are only produced in Brazil, they will have the same score for water scarcity and sensitivity of local biodiversity. Further, the CO_2 emissions are not country specific, thus if a material is produced in a country with a more CO_2 intensive energy mix, this is not accounted for. This is because the CO_2-emissions per kilogram relied mostly on global data. This could not be improved, due to the lack of more regionalized inventory data. Additionally, the environmental impacts were quantified per kilogram and then compared- precious metals and other materials of the kind have the highest impacts. For an absolute approach, the impact would need to be multiplied by the absolute used mass, which has been used for Paper 3 and 4, to account for the occurring impacts.

Furthermore, LCA is not able to assess the impact on local cultural values that might be affected, or important cultural sites that are destroyed for mining purposes. This kind of assessment is beyond the scope of a normal environmental LCA study and should be assessed in Social LCA (Benoît Norris et al., 2020) and ideally also integrated in criticality assessment. For example, upon the discovery of gold in Indonesia, scientist observed higher prevalence of conflicts and a change of behavior (Meisanti et al., 2012). Such socio-cultural effects should be included in criticality assessment.

In this paragraph the strength and weaknesses of the developed methods in Paper 3 and 4 are discussed and presented. One advantage is that the derived impacts are expressible in a monetary metric that can be compared to other goods and economic activity. Additionally, it was possible to underline the geographical distribution of the damages by mapping the damages to the different countries. Furthermore, the aggregation approach through monetization did not rely on equal weighting nor questionnaires, but on monetizing environmental impacts. However, some of the underlying monetization factors were obtained through questionnaires. Another strength is that the assessment is always material-specific opposed to Paper 2 and that the damages are absolute not relative damages. However, the data quality of the inventory data remains a challenge. For a more reliable result, regionalized inventory data or ideally primary data is necessary to account for local differences of mining activities.

A weakness of the derived methods is that they reduce environmental damages to a monetary score, which suggests interchangeability of natural and physical capital and thus only follows a weak sustainability perspective (Steen, 2006). Sandel (2012) shows how such an approach might crowd out non-market norms, which can lead to negative environmental and social consequences. Additionally, the quantified monetary impacts do not say anything about the value of species or human health as such, but rather show how much value humans assign to them, as money is a unit that humans have made to order their social relations.

Moreover, additional uncertainties, next to the already prevalent uncertainties of the used data, are introduced through the economic assessment. For the method in Paper 3 the monetization of biodiversity relies on a meta-study of biodiversity valuation that depended on European values. Global data would be preferable for Paper 3. The important impacts on freshwater and marine biodiversity had to be neglected in both studies. For the method in Paper 4 only the impact of land use on ecosystem services is considered via the integration of values from the ESVD, while all other impact categories are neglected. Therefore, both methods lead to an underestimation. Further, the approach in Paper 3 omits the feedback of environmental impacts on economic activity. For example, the impact of climate change only assesses impacts on ecosystems and human health but not on productivity and the destruction of physical capital through floods and storms. This is also a problem for most other impact categories. The DALY values rely on an outdated and disputed approach depending on income, which relies on similar assessment as the human capital approach. A strength of the derived method in Paper 4 is its integration of environmental economic research and LCA. The ESVD (de Groot et al., 2020b, 2020a, 2012) was used for the first time to value damages that were quantified with LCA

The main advantage of the method application in Paper 3 is that it becomes possible to judge mining activities from a national cost-benefit perspective to explain, why it will be difficult to phase out the extraction of certain activities. It was identified, that extraction is beneficial from a cost-benefit perspective for certain countries, if only domestic damages are considered as they can outsource a large share of the induced climate change costs to other countries.

The main strengths of Paper 4 were that it considered how environmental impacts affect economic activity. Amongst these considered impacts are reduced yields induced by water scarcity, productivity decrease from temperature increase and reduced production value due to reduced availability of materials, as well as reduced ecosystem services that also impact economic activities indirectly. Furthermore, it was explored and explained how this assessment could be integrated into criticality assessment.

In Paper 3 the monetization was done via the monetization of endpoint values, while in Paper 4 mid-points were monetized. This might reduce the consistency of the derived values. Additionally fewer impact categories were covered. For example, in Paper 3 ecotoxicity was covered, which was neglected in Paper 4. For the quantitative comparison, toxicity showed a hotspot for the case study of low-carbon development (see section 4.1 Figure 11), however toxicity also has the risk of uncertain data and a high number of uncharacterized substances. Thus, the approach in Paper 4 leads to an underestimation and the coverage of additional impact categories would be desirable. However, the results can be questioned if uncertain and less-developed impact categories dominate the results.

The assessment of land use with the ESVD represents an additional weakness. While it was a desirable step to use these values for the first time in LCA, they were still used in a very reductive way. Only land occupation was assessed and an average value for every country based on the prevalent biomes in that country was used. Then it was assumed that the ecosystem service cannot be delivered as long as the land is occupied. Land use change was neglected as well as reduced ecosystem services after land use relaxation. It would be desirable to have respective time series on how high the delivery of ecosystem services is, after relaxation of the land occupation. Additionally, the assessment on biome level is not satisfactory. For site-specific assessments, discrete geographical data that account for the respective mine sites and its biological characteristics would be needed. Possibly the areas affected by land use change through mining could be identified by satellite images and remote sensing.

Finally, it is important to reflect on what monetization can deliver for LCA in particular and sustainability assessment in general. Monetization has many risks but also significant benefits, which is why it is continually used and a drive to monetization by several international companies can be observed (Value Balancing Alliance, 2022). However, one of the main benefits of LCA -that the impact categories show conflicts of interest- might be blurred through monetization that suggests, for example, a certain amount of CO_2 emission reduction equals a certain increase in water use and that the caused and avoided damages in a trade-off situation are equivalent. These shortcomings and radical simplifications have to be kept in mind and should be evaluated with critical distance.

An additional challenge is that monetization (even if outlined as monetization of environmental impacts in this thesis) always touches economic activity and social impacts on humans- as money always refers to objects that are part of our socio-economic system. Thus, monetization has great potential to be used in sustainability assessment but there is also a risk of double counting, when monetized LCA results are integrated in a comprehensive Life Cycle Sustainability Assessment. Furthermore, monetization cannot assess intrinsic value, as money is a human concept that other species do not understand nor use. An additional shortcoming is that weighting is never purely natural science-based (Finnveden, 1997). Monetary weighting, also the methods developed in this thesis, rely on environmental economics, which is a social science.

4.4 Possible Future Research Agenda

In this subchapter, possibilities for additional research are outlined. As shown in the storyline and research questions, the monetized LCA results could be used to complement criticality assessment.

All dimension of the used criticality assessment SCARCE could be aggregated by monetization. Such an approach would allow for a homogenous aggregation method. This could be based on Paper 4 and the method by Vogtländer et al. (2019) or by Santillán-Saldivar et al. (2021b). Furthermore it remains an open question how to best integrate environmental concerns into criticality assessment, as environmental and social impacts can lead to supply risk (Le Billon, 2001). Glöser et al. (2015) propose to integrate environmental and social concerns into supply risk, while Bach et al. (2017, 2016) keep the assessment of environmental and social impacts separate, which was also the approach followed in this thesis and by Graedel et al. (2012).

Another branch of research to explore would be to derive possibilities to reduce environmental and social pressures and criticality through a certification and responsible sourcing scheme (Pelzeter et al., 2022; Young, 2018). On a similar account, it should be further investigated how recycling can reduce criticality, building on established research (Santillán-Saldivar et al., 2021a). These approaches can be brought into dialogue and it could be quantified how recycling and certification will affect environmental costs and supply risk.

Another possibility for further research would be to integrate the monetized results into integrated assessment models. These could then optimize not just for CO_2 abatement, but also for other impacts including supply risks and derive an optimal mitigation pathway that also considers water use and land use as well as toxicity impacts. More impact categories could be integrated, if additional monetization factors become available. These need to be communicated to policy makers so that they do not only concentrate on carbon emission reductions but also on other significant environmental impacts. This pathway was started already (Baumstark et al., 2021; Rauner et al., 2020), but the impact of resource use and how it affects different countries was neglected. Further the combination of monetized LCA results with energy system models could be developed based on existing research (Algunaibet et al., 2019). It would be of particular interest to identify the different optimizations with varying monetization methods, i.e., if a certain technology mix is preferred over another depending on the monetization method.

Additionally, more case studies are needed, in which the developed approach is coupled with MFA and also predictive MFA, which can yield information and insights regarding the geographical distribution of environmental impacts induced by digitalization or the introduction of new material intensive products. The developed assessment method could also be applied to time series of material production, to identify the historical development of environmental impacts.

The monetized environmental impact assessment coupled with LCA and MFA seems most promising, as then not only the impacts of extraction but also of e.g., burning fossil fuels could be geographically tracked and mapped, which can further explain why certain patterns of use persist. For such an approach more granular process data from LCA would be necessary, as well as a global scale and up-to-date MFA. As both of these are difficult to obtain, such an approach could be initially started with just one material, where the data availability is good. Ideally such an assessment is carried out in cooperation with an industry association, to obtain the necessary primary data.

The developed assessment method can be easily modified through different monetization factors or newer impact assessment methods. Monetization in LCA can further profit from developments in environmental economics and the most valuable and necessary link here is to improve the knowledge and collaborate related to ecosystem services. It would be imaginable to further integrate ecosystem service valuation for more impact categories than just land use – and to also integrate the country-specific or even Ecoregion-specific values (as the biome perspective in Paper 4 is too rough).

The uncertainty and the variance of the monetization methods is not assessed in as much detail as possible. With the currently non-standard distribution no values can be recommended with a scientific argument of significance. For the carbon price the price range diverged based on more and more research in relation to it (with the maximum proposal around 10.000/t CO_2 (Archer et al., 2020)). Thus, taking the average as Schneider-Marin and Lang (2020) has done, risks that the fact of divergence is blurred by averaging. To understand such a distribution and accept it (as no convergence is expected) would require a better reflection of the underlying values that drive the divergence. One way to better reflect these values would be to investigate, which monetization method fit which convictions, building on some first reflections in this direction in LCA (Freidberg, 2018; Steen, 2006). Then companies or LCA practitioners have a better guidance, which method to choose if they hold particular values. Some companies might commit to the 1.5°C target and thus if they chose a monetization method for CO_2, they should seek to use a CO_2-price that is more or less consistent with a carbon tax that is believed to enable reaching the 1.5°C target. Or a company that commits to equality and diversity should value a DALY lost in one country equally to another country irrespective of the average income. Therefore, these aspects and qualitative criteria could be connected to certain moral convictions, similar to the cultural perspectives in ReCiPe.

One additional aspect that this thesis did not explore is the impact of dissipation, which would be interesting to study especially in relation to the finite stocks and how the environmental costs of transferring ores to an in-use stock compares to dissipation. For dissipation flows, quantitative estimates exist (Charpentier Poncelet et al., 2021), which could be used.

Further, the total economic value perspective of environmental impacts was taken for Paper 3, but it was not possible to quantify the distributive effects. Thus, it needs to be explored whether the environmental costs of mining rather affect poorer parts of the population or richer parts. This information is relevant for the impact on human welfare, while site-specific studies will be most promising. For Peru, for example, it has been shown that mining has net positive effects on wealth, but negative distributive effect (Loayza and Rigolini, 2016). Such an analysis would be worthwhile to explore in other countries, but also on global level.

A general challenge related to monetization is that it is sometimes unclear whether something is an economic or environmental damage. The productivity decrease due to temperature increase could be perceived as an effect of an environmental emission. However, the impact is also economic and social, as it reduces health and labor productivity, if people have to a work at excess temperatures. This research need has been outlined in Paper 1 as a challenge for the review and should be further explored.

5 Conclusion

This thesis contributed to the monetization in LCA and applied it to environmental criticality assessment. Regarding research question 1, it provided an overview of existing monetization methods, and identified the main methods. Further, it showed how a choice of a monetization method affects the prioritization of impact categories and compared the methods quantitatively by applying them to a case study (average emissions of an average EU citizen). This resulted in significant differences between the methods and a non-normal distribution of monetization factors. The review provides information regarding the questions, if monetization is a valid option for a practitioner and if yes, which method could be chosen.

Additionally, application fields beyond weighting were explored for the monetization of environmental impacts. Two monetization methods for LCA were developed, while the first one monetized the classical LCA-AoPs, the second one drew on environmental economics and monetized mid-point results. The monetization in LCA was used to derive an environmental criticality assessment and to be applied in a cost-benefit-analysis. These methodological developments were applied in two case studies. Unlike former methods that assessed the environmental criticality relative to the materials (environmental impact per kilogram), this thesis delivers a quantitative absolute damage-based assessment of materials to assess environmental criticality. These are expressed with a monetary metric.

The methods were applied to quantify the environmental costs induced by material production at global level and to the material demand induced by the EU's low-carbon development. With these two case studies, research question three could be answered, that identified the main hotspots of current material production to still lie with bulk materials such as steel and fossil energy carriers. Additionally, the geographical distribution of costs was tracked, which mostly take place in China. Furthermore, it was possible to compare economic gains and environmental losses of material mining and production on global scale from a total economic value perspective. The quantified monetary damages for global material production ranged from €0.4 trillion/yr (low) to €5 trillion/yr (high), while the damages for the EU's low-carbon development ranged from €13.1 billion to €74.8 billion, with €38.9 billion as the medium estimate per year.

These developed assessment methods and case studies deliver knowledge of environmental impacts of material use that can be used by policy makers to reduce the environmental pressures of material production along the supply chain and make trade relationships more stable. These assessments will enable the prioritization of environmental pressures that can be a threat to the stability of a trade partnership. Moreover, they can support the reduction of up-stream impacts that might come into conflict with a countries' environmental goals. It was observed that the monetized damages were different and prioritized changing materials as compared to the non-monetized relative approaches. The monetized assessments provide information regarding the environmental hotspots of a global material mining and processing that goes beyond a pure mass-based assessment. Such knowledge can help to reduce environmental impacts of globally growing material use and through this also enable more stable trade partnerships.

It remains a challenge how high the variance of the derived monetization factors is. The choice in this thesis to quantify a high and a low estimate should be maintained in future research and sustainability assessment. The values in the review did not converges- but also the values in the own developed methods did not converge either, which underlines the uncertainty of monetization approaches. Therefore, the informed practitioner should choose the monetization method that fits their value choices and goals for the specific case study and make these choices more explicit.

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Appendix

Supplementary Material Paper 1

This section of the appendix contains the supplementary material of the following publication:

Paper 1: Arendt, Rosalie; Bachmann, Till M.; Motoshita, Masaharu; Bach, Vanessa; Finkbeiner, Matthias: Comparison of Different Monetization Methods in LCA: A Review. *Sustainability* **2020,** *12*, 10493. https://doi.org/10.3390/su122410493.

Supplementary material

Table S1. Impact category indicators that were used for comparison, CFs and UCFs used for unit conversion to convert all impact categories to the same units (all from ReCiPe [1] if not otherwise specified)

Impact categories	Impact category indicator for comparison	CFs	UCF (derived from the CFs)
Climate change	kg CO_2-e	None	None
Depletion of stratospheric ozone	kg CFC-11-e	None	None
Acidification (only for MMG for "land and water"	kg SO_2-e	For mol H+-e: 1.37 SO_2/mol H+-e based on [2]	0.73 mol H+-e/SO_2
Eutrophication	€/kg PO_4 eq	3.03 PO_4-e/P	0.33 P/PO_4
Formation of tropospheric ozone	€/kg C_2H_4-e	0.070 kg C_2H_4/kg C_2H_2 0.496 kg C_2H_4/kg NMVOC 2.76 kg kgC_2H_4/NO_x (only based on ReciPe human health)	14.28 kg C_2H_2/kg C_2H_4 2.016 kg NMVOC/ kg C_2H_4 0.36 kg NO_x/C_2H_4
Abiotic depletion of non-fossil resources	€/kg Sb-e	None- some methods supplied only kg Sb-these were assumed to be the same as Sb-e	None
Abiotic depletion of fossil Resources	€/MJ	MJ/kg dependent on the caloric value from different energy carriers based on [3]	Kg/MJ dependent on the caloric value from different energy carriers based on [3]
Human toxicity cancer	€/CTUh (cancer) then transferred to DALY as explained in the section 2.2. of the main paper	1.81E-07 CTUh/kg 1,4-DCB to air (Usetox) 1.8E-07 CTUh/kg 1,4 DCB to soil	5524861 kg 1,4-DCB to air/CTUh 5555555 kg 1,4 DCB to soil/CTUh/

		9.26E-07 CTUh/ kg Vinylchloride to air	1079913 kg Vinylchloride to air/CTUh
		0.004076541 CTUh/ kg Benzo(a)Pyrene	245 kg Benzo(a)Pyrene/CTUh
Human toxicity non-cancer	€/CTUh (non-cancer) then transferred to DALY as explained in the section 2.2. of the main paper	5.42E-08 CTUh/kg 1,4-DCB to air (UseTox)	18450184 kg 1,4-DCB to air/CTUh
		7.06E-08 CTUh/kg 1,4 DCB to soil (UseTox)	14164305 kg 1,4 DCB to soil/CTUh
		2.67E-06 CTUh/ kg Vinylchloride to air	374531 kg Vinylchloride to air/CTUh
Particulate matter	kg PM2.5 e	0.28 kg $PM_{2.5}$ e/kg PM_{10} .(based on TRACI [4]	3.57 kg PM_{10}/kg$PM_{2.5}$-e
Ionising radiation human health	kBq kg U235 e	0.01 kBq kg U235 e/ Bq C14 Based on [5]	100 Bq C14/kBq kg U235
Ecotoxicity terrestrial	kg 1,4 DB-e to industrial soil	1.023 kg 1,4 DCB to industrial soil/kg DCB to agricultural soil	0.977 kg DCB to agricultural soil/kg 1,4 DCB to industrial soil/
		0.0031 kg 1,4 DCB to industrial soil/kg TEG to soil (from IMPACT2002+ [6])	322.58 TEG to soil/ kg 1,4 DCB to industrial soil/kg
Ecotoxicity freshwater	CTUe	9.83E+02 CTUe/ kg 1,4 DCB to freshwater (UseTox)	0.001017 kg 1,4 DCB to freshwater/ CTUe
		9924279 CTUe/ kg Cu to freshwater (UseTox)	1.00763E-07 Cu to freshwater/ CTUe
		3.87E-01 CTUe/kg TEG to freshwater (UseTox)	2.58 kg TEG to freshwater/ CTUe
Ecotoxicity marine	1,4-dichlorobenzen emitted to seawater	None	None
Water scarcity	m^3 H_2O	None	None

Land use occupation	m^2/a – no differentiation for different kinds of occupation	None	None
Soil organic matter	kg of C deficit	None	None
Land use transformation	m^2- no differentiation for different kinds of occupation	None	None
Marine eutrophication	Kg N-e	0.23 kg N/kg NO_3	4,34 kg NO_3/ kg N

Table S2. Overview of established links between methods (per impact category) and AoPs; links in brackets are partially established

Methods	Impact category	Human health	Agricultural production	Ecosystem	Resources	Working capacity	Buildings and materials	Human wellbeing	Abatement costs	Societies WTP
Bruyn et al. (2018)	**Climate change**	**x**		**x**	**x**					
Huijbregts et al. (2017)	**Climate change**	**x**		**x**						
Ecotax	Climate change									x
Ecovalue	Climate change	x	x	x		Working capacity (if based on FUND 3.3-3.9)				
MMG method	Climate change								x	
Environmental prices	Climate change								x	
Stepwise	Climate change	x	(x)- positive impact	x	x					

Methods	Impact category	Human health	Agricultural production	Ecosystem	Resources	Working capacity	Buildings and materials	Human wellbeing	Abatement costs	Societies WTP
EPS	Climate change	x	x	x		x (working capacity, migration)	x			
EVR	Climate change								x	
Trucost	Climate change	x	x	x		(x) included in the FUND model which is also a basis in estimating SCC				
LIME3	Climate change	x		x						
According to Bruyn et al. (2018)	**Acidification**	**x**		**x**			**x**			
Huijbregts et al. (2017)	**Acidification**			**x**						
EVR	Acidification								x	
Ecotax	Acidification									x
MMG method	Acidification			x			x			
Environmental prices	Acidification	(x)	x	x			x			
Stepwise	Acidification			x						
EPS	Acidification		x	x			(x) (include only CO_2-			

Methods	Impact category	Human health	Agricultural production	Ecosystem	Resources	Working capacity	Buildings and materials	Human wellbeing	Abatement costs	Societies WTP
							emissions of steel that has to be replaced due to corrosion)			
Trucosts	Acidification	x		x						
Ecovalue	Acidification			x			x	x		
Bruyn et al. (2018)	**Ozone Depletion**	**x**		**x**				**x**		
Huijbregts et al. (2017)	**Ozone Depletion**	**x**								
EVR	Ozone Depletion								x	
Ecotax	Ozone Depletion									x
MMG method	Ozone Depletion	x	x (including wood production)							
Environmental prices	Ozone Depletion	x	x	(x)						
Stepwise	Ozone Depletion	x				x				
EPS	Ozone Depletion	x								
Bruyn et al. (2018)	**POCP**	**x**		**x**			**x**			
Huijbregts et al. (2017)	**POCP**	**x**		**x**						

Methods	Impact category	Human health	Agricultural production	Ecosystem	Resources	Working capacity	Buildings and materials	Human wellbeing	Abatement costs	Societies WTP
EVR	POCP								x	
Ecotax	POCP									x
MMG method	POCP	x								
Environmental prices	POCP	x		(x)			(x)			
Stepwise	POCP	x	x	x		x				
EPS	POCP	x	x (crops and wood)							
Trucost	POCP	x		x						
LIME3	POCP	x								
Ecovalue	POCP	x	x							
Bruyn et al. (2018)	**Eutrophication**			**x**				**x**		
Huijbregts et al. (2017)	**Eutrophication**			**x**						
Ecotax	Eutrophication									x
Ecovalue	Eutrophication		(x)	(x)				Partly (recreation value)		
Environmental Prices	Eutrophication			(x)						
EPS	Eutrophication		x	x						
EVR	Eutrophication								x	
MMG	Eutrophication			x				x	x	
Stepwise	Eutrophication			x						
Trucost	Eutrophication	x		x				x		
Ecovalue	Marine Eutrophication			(x)				x		

Methods	Impact category	Human health	Agricultural production	Ecosystem	Resources	Working capacity	Buildings and materials	Human wellbeing	Abatement costs	Societies WTP
Environmental Prices	Marine Eutrophication			(x)						
EPS	Marine Eutrophication		x	x						
Stepwise	Aquatic Eutrophication			x						
Trucost	Marine Eutrophication	x		x				x		
Bruyn et al. [7]	**Particulate matter**	**x**					**x**			
Huijbregts et al. [1]	**Particulate matter**	**x**								
Ecovalue	Particulate matter	x								
Environmental Prices	Particulate matter	x					x			
EPS	Particulate matter	x								
EVR	Particulate matter								x	
MMG	Particulate matter	x								
Stepwise	Particulate matter	x				x (working capacity)				

Methods	Impact category	Human health	Agricultural production	Ecosystem	Resources	Working capacity	Buildings and materials	Human wellbeing	Abatement costs	Societies WTP
Trucosts	Particulate matter	x								
LIME3	Particulate matter	x								
Bruyn et al. [7]	**Ionizing radiation**	**x**						**(x)**		
Huijbregts et al. [1]	**Ionizing radiation**	**x**								
Environmental Prices	Ionizing radiation	x								
EPS	Ionizing radiation	x								
Stepwise	Ionizing radiation	x				x (working capacity)				
MMG method	Ionizing radiation	x								
Huijbregts et al. [1]	**Water use**	**x**		**x**						
EPS	Water use		X (based on costs for drinking and irrigation water)							
EVR	Water use								x	
MMG	Water use								x	
Trucost	Water use	x		x						

Methods	Impact category	Human health	Agricultural production	Ecosystem	Resources	Working capacity	Buildings and materials	Human wellbeing	Abatement costs	Societies WTP
Ligthart 2019 damage costs	Water use		x							
Ligthart 2019 abatement costs	Water use								x	
LIME3	Water use	X (induced by agricultural production loss and domestic water shortage)								
Bruyn et al. [7]	**Land use/ transformation**			**x**	**x**			**x**		
Huijbregts et al. [1]	**Land use/ transformation**			**x**						
Environmental Prices	Land use/ transformation			(x)				(x)		
EPS	Land use/ transformation	x (working capacity)	x (drinking water renewal)	x						
EVR	Land use/ transformation			x (take into account bio factor and scenic beauty)					x	
MMG	Land use/ transformation			x					x	

Methods	Impact category	Human health	Agricultural production	Ecosystem	Resources	Working capacity	Buildings and materials	Human wellbeing	Abatement costs	Societies WTP
Stepwise	Land use/ transformation			x						
LIME3	Land use			x	x (NPP)					

Table S3. Monetary values per impact category for all assessed methods

Impact categories	**Global warming**	**Acidification**	**Strat Ozone depletion**	**Freshwater Eutrophication**	**Marine Eutrophication**	**Photochemical oxidation**	**Mineral resources**	**Fossil resources**	**Human Toxicity**	**Ionizing radiation**	**Freshwater Ecotoxicity**	**Terrestrial Ecotoxicity**	**Marine Ecotoxicity**	**Particulate matter**	**Land use**	**Land transformation**	**Soil organic matter**	**Water use**
Unit	**2019 €/kg-CO_2-e**	**2019 €/Kg SO_2-e**	**2019 €/Kg CFC-11-e**	**2019 €/PO_4-e**	**2019 €/ kg NO_3-e**	**2019 €/ kg C_2H_4-e**	**2019 €/ kg Sb-e**	**2019 €/MJ**	**2019 €/DALY**	**2019 €/ kg kBq U235-e**	**2019 €/CTUe-e**	**2019 €/ kg1,4-DCB emitted to soil**	**2019 €/ kg 1,4-DCB emitted to marine water**	**2019 €/ kg PM 2.5 -e**	**2019 €/m²/a**	**2019 €/m²**	**2019 €/Kg C-e**	**2019 €/m³**
Stepwise	0.107281697	0.19388	129.25506	1.5510607	0.572	0.7238283		0.005170	19542.42	0.0025	2.371346e-05	0.45889487		87.89344	0.1551			
LIME3	0.00745					0.000004201402	821.3	0.0011						7.71	0.00322			0.00153

Impact categories	Global warming	Acidification	Strat Ozone depletion	Freshwater Eutrophication	Marine Eutrophication	Photochemical oxidation	Mineral resources	Fossil resources	Human Toxicity	Ionizing radiation	Freshwater Ecotoxicity	Terrestrial Ecotoxicity	Marine Ecotoxicity	Particulate matter	Land use	Land transformation	Soil organic matter	Water use
Environmental Prices	0.059316800	7.8390400	128.904	0.6432624	3.2592800	2.3185484			42054.7257	0.0483128	3.848708e-05	9.10712000	7.744720e-03	146.72000	0.132			
EPS	0.137288000	0.0056	11.74808	0.0128904	0.0138336	1.330960e+00	19063	Various 0.0105 (max) 0.00525 (min)		0.000587				160.34	3.7 (max-industrial use) 0.00019 (min meadow)			0.0021 (drinking water) 0.001 (irrigation water)
MMG (max, central, andmin value)	0.106873343 0.053436671 0.026718336	0.9404854 0.459555 0.2351214	106.8734 52.47481 26.71834	64.1240057 21.3746686 7.0536406		7.0536406 0.5129920 0	6.658209 1.667224 0	0.0069 0 0	Non-cancer: 285156 57031 11406 cancer:	0.00309 0.001037 0.000342	1.977157e-04 3.954314e-05 7.897940e-06			90.84234 36.33694 13.57291	2.5 (max) 0.0000587 (min)	117.56 28.85 7.37 (all tropical rainforest)	6.412401e-06 1.496227e-06 3.633694e-07	0.2137 0.0716 0.0235

Impact categories	Global warming	Acidification	Strat Ozone depletion	Freshwater Eutrophication	Marine Eutrophication	Photochemical oxidation	Mineral resources	Fossil resources	Human Toxicity	Ionizing radiation	Freshwater Ecotoxicity	Terrestrial Ecotoxicity	Marine Ecotoxicity	Particulate matter	Land use	Land transformation	Soil organic matter	Water use
									2472 43 6181 0 1545 2									
Ecotax 2002 (max, min)	0.062 31259 3	1.7803 598	118. 6906 5	2.82582 66		763.283 9382 76.3283 938		0.01 48 0	6562 9.48		1.251 404e- 02 6.123 697e- 03	17.04 95071 8	1.318 455e- 06 5.993 878e- 02					
Ecovalue14 (max, central, min)	0.480 90742 6 0.244 74752 9 0.008 58763 3	2.5762 898		57.5371 384	7.729	55.2259 331 37.2775 049 19.3290 766		0.02 06 0.01 03 0.00 034	1885 78.36 1083 65.07 771.2 8				1.030 516	83.72				
Ligthart 2019		-															1.068 733e- 01	5.428 (damage)

Impact categories	Global warming	Acidification	Strat Ozone depletion	Fresh water Eutrophication	Marine Eutrophication	Photochemical oxidation	Mineral resources	Fossil resources	Human Toxicity	Ionizing radiation	Freshwater Ecotoxicity	Terrestrial Ecotoxicity	Marine Ecotoxicity	Particulate matter	Land use	Land transformation	Soil organic matter	Water use
																	(abatement) 3.056578e-02 (damage)	33.91 16.58 (both abatement)
EVR	0.119441934	9.0096286	0	4.2937316		9.3082334	8.169192	Various 0.0179 (max) 0.0062 (min)	82452.42		3.527596e-05			36.038	6.17 (max-heterogeneous agriculture) 0 (min, e.g. natural forest)	7.72 (max in countries with high biodiversity) 0.1544 (min in countries with low biodiversity)		Various 1.029 (max) -1.029 (min)

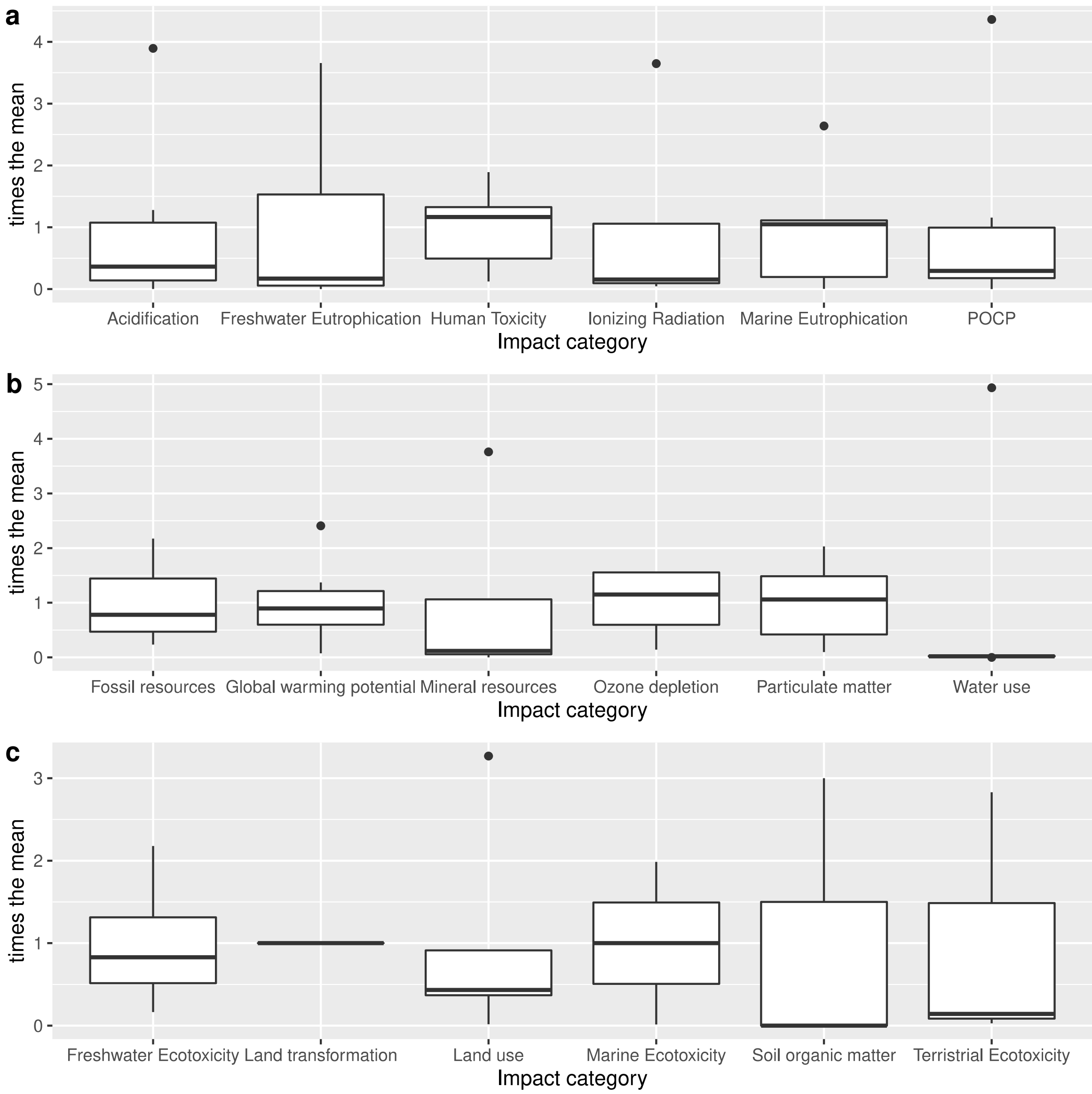

Figure S1. Distribution of monetary values per impact category without Ecotax and EVR

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Supplementary Material Paper 2

This section of the appendix contains the supplementary material of the following publication:

Paper 2: Arendt, Rosalie; Muhl, Marco; Bach, Vanessa; Finkbeiner, Matthias: Criticality Assessment of Abiotic Resource Use for Europe– Application of the SCARCE Method. *Resources Policy* **2020** 67 (August): 101650. https://doi.org/10.1016/j.resourpol.2020.101650.

Supplementary material

The supplementary material contains an outline of the SCARCE method (section one), a small fraction of the results namely the per category (and not per material) comparison of global and European supply risks and the comparison with the critical raw material assessment of the European Union (section three for a methodological comparison and section four for the results comparison). The main part (section one) is dedicated to a detailed outline of the SCARCE method. It contains details on the calculation of the sub dimension results for supply risk, vulnerability, compliance with social standards and compliance with environmental standards. In the third section of the supplementary material, the categories used in the SCARCE method are compared to the categories that the EU criticality method considered. Further, a short comparison to the SCARCE case study of Germany with the results obtained in this study (displayed in the main paper) is performed in chapter 5.

Content

1. Further Details on the SCARCE Method and its Application

In this chapter, the SCARCE method is introduced. The information in section one of this supplementary material is mostly contained in the publication by Bach et al. (2016, 2017), but is provided to ease the understanding of how the sub dimension result are obtained in its application to Europe. After an overview of the method is provided, the first subchapter focuses on how the import mix in the SCARCE method is determined followed by the different sub dimensions that are visualized in Figure S 1. For all sub dimensions, the categories are described as well as the calculation formula for the scaling of the indicators and the calculation of the overall sub dimension result. The SCARCE method consists of the following (sub-) dimensions (Details can be found in the associated publication of Bach et al. (2017)):

- Criticality:
    - Supply risk
    - Vulnerability

- Societal acceptance
    - Compliance with social standards
    - Compliance with environmental standards

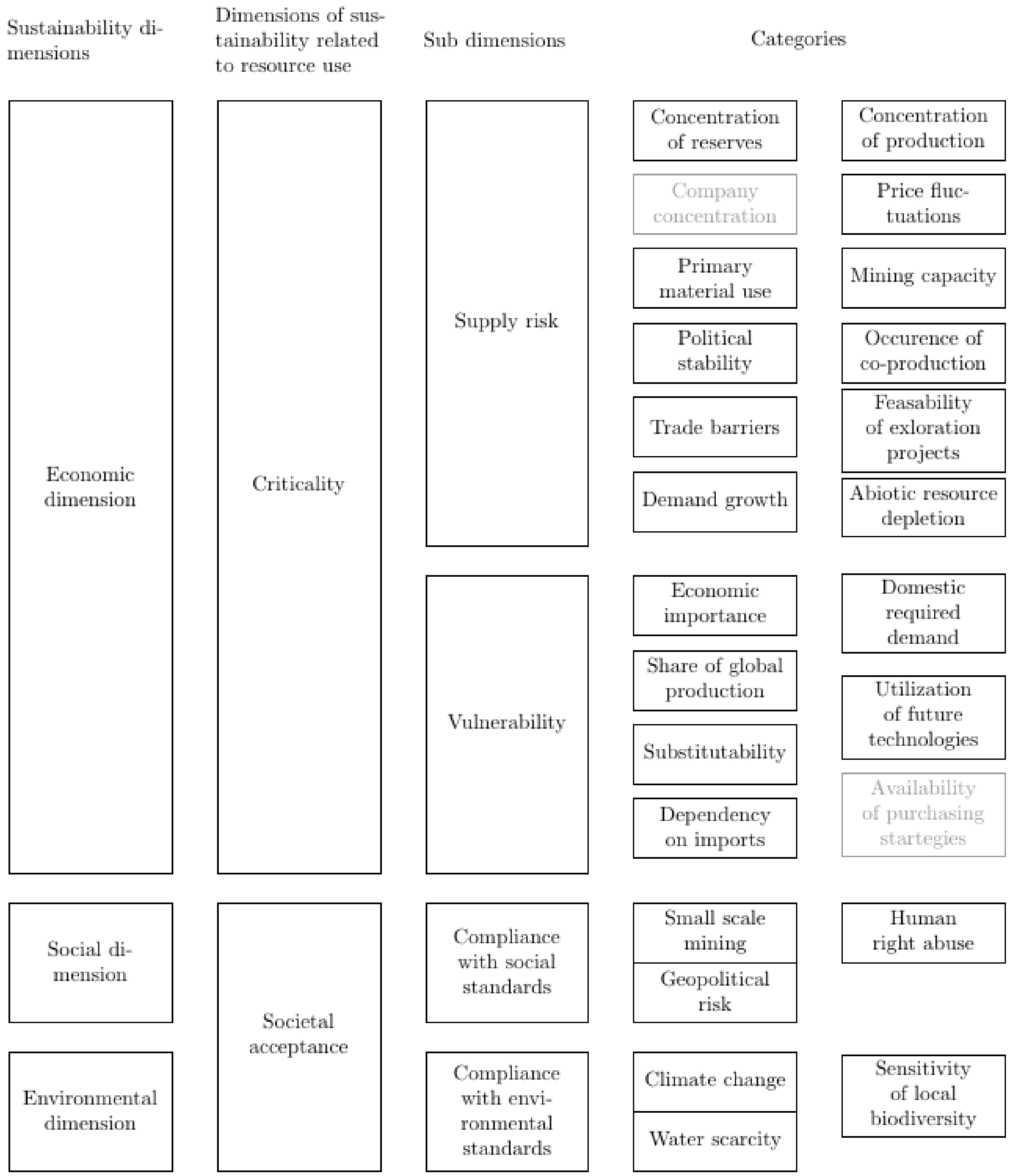

Figure S 1 Dimensions and categories of SCARCE adapted from Bach et al. (2017) *;grey categories were not considered in SCARCE application to Europe which is explained in step one of the method section in the main paper*

Each sub dimension consists of several categories (and associated indicators), which address relevant aspects of the sub dimension. These indicators are calculated for all assessed resources individually. Further information on the categories and on the calculation of the corresponding indicators is provided for in the following.

1.1 Determining the Import mix in SCARCE

The import mix is ascertained by the import data into Europe for the respective materials that we extracted from comtrade/comext (UN Statistics Division, 2016) and the world production mix (U.S. Geological Survey, 2017) that we obtained during the data collection. We excluded all non-producing countries that export to Europe from the import mix, since they do not produce the resources themselves, but have imported them previously (if this step is not taken e.g. 57 % of cobalt supplied to Europe was from Japan, even though cobalt is not mined in Japan). We assumed that all non-producing countries export the average world production mix, since we were not able to obtain data with higher granularity by tracing back the trade flows from the exporting countries within a reasonable timeframe. This aspect is explained in more detail in Bach et al. (2017). The European import mix is thus based on UN Statistics Division (2016) and the U.S. Geological Survey (2017), if the data was not available by the U.S. Geological survey it is based on data from the British Geological Survey (2019).

1.2. Supply Risk

The sub dimension supply risk contains eleven categories that are shortly described in the following:

- Concentration of production, reserves and company concentration: High concentrations refer to few countries and/or companies that mine and trade resources, which can lead to potential supply disruptions (Rhoades, 1993).
- Feasibility of exploration projects: Political and societal factors (taxation, environmental regulations, implementation of regulations, or infrastructure) can influence the opening of mines up to a point where mine development is delayed or cancelled (Wilson et al., 2013).
- Political stability: Governance instability in the raw material producing countries (e.g., potential revolutions but also corruption or financial crises) may interrupt production and lead to potential supply restrictions (Kaufmann et al., 2011).
- Occurrence of co-production: Due to metals being mined as companion metals within host metal ore bodies, their availability can be restricted when the host metal is no longer produced in the same quantity (Angerer et al., 2009).
- Mining capacity: When the remaining time to extract a certain resource in already developed mines worldwide considering current conditions (e.g., technological and economic feasibility) is too short, potential supply restrictions may occur.
- Primary material use: When secondary materials are only used to a certain extent, more primary materials need to be extracted, which can lead to potential supply restrictions (Graedel et al., 2011).
- Demand growth: If the demand increases significantly, current production might not be able to keep up and potential supply restrictions might occur.
- Price fluctuations: Significant unexpected price fluctuations can lead to higher prices of a resource that a company can no longer afford and therefore potential supply restrictions might occur (Federal Institute for Geoscience and Natural Resources, 2014).
- Trade barriers: When barriers to material trade exist (e.g., due to export duties), certain countries/companies might not have access to the needed materials (Lawrence et al., 2008).

For the sub dimension supply risk also the indicator abiotic resource depletion (van Oers and Guinée, 2016) is applied, which considers the whole available resource not only the reserves like the category *mining capacity*. The formulas to determine the indicator results in the application of SCARCE to Europe (and thus not for availability of purchasing strategies and company concentration as displayed in *Figure S 1*) are displayed in Table S 1 and in Table S 2 and Table S 3.

The indicators *trade barriers, mining capacity, political stability, feasibility of exploration projects* and *concentration of production* are import-based, which means that their calculation is based on data from actual European sourcing. The import mix and country shares are determined as explained in 1.1 of the supplementary material.

For the category *feasibility of exploration projects* we made an adjustment, since the indicator (the Policy Potential Index (PPI)) is only applicable to countries that have reserves (and not to producing countries that do not have reserves, but only produce the material). In order to consider the PPI only for countries that have reserves, we compared the import mix (the list of countries that export to Europe) with the global reserves and excluded all exporting countries that did not have reserves. For the share of exporting countries without reserves, we assumed the PPI of the global reserves (by determining an average). For the exporting countries, which have reserves, the PPI values of the exporting countries were multiplied with the import share of all European imports of the respective country as shown in Table S 1.

The indicators are calculated with the formulas displayed in Table S 1 for the resource *i* from exporting countries *x*. Imports are always referring to imports into Europe from a country *x*:

Table S 1 Indicator calculation for the categories in the sub dimension supply risk

Category	Formula
Country concentration of reserves	$$\sum_{x=1}^{N} s_{x,i}^2$$ While *s* stands for the share of reserves that country *x* has of resource *i*. *N* is the amount of countries that have reserves of resource *i*. The formula corresponds to the Herfindahl Hirschman Index.
Country concentration of production	$$\sum_{x=1}^{N} s_{x,i}^2$$ While s stands for the share of imports into Europe that country *x* has of resource *i*. *N* is the amount of countries that export resource *i* to Europe. The formula corresponds to the Herfindahl Hirschman Index.
Feasibility of exploration projects:	$$\sum_{x} (import[\%]_{x,i} \cdot PPI_{x,i})$$ PPI stands for the policy potential index
Political stability	$$\sum_{x} (import[\%]_{x,i} \cdot WGI_{x,i})$$ Only for government effectiveness, regulatory quality, rule of law and control of corruption
Mining capacity	$$\sum_{x} (import[\%]_{x,i} \cdot \frac{reserves[t]_{x,i}}{imports[t]_{x,i}})$$

Trade barriers	$$\sum_x (import[\%]_{x,i} \cdot ETI_x)$$ ETI stands for Enable Trade Index
Demand growth	$$\frac{\sum_1^5 \left(\frac{global\ production\ of\ year_{i,n+1}}{global\ production\ of\ year_{i,n}} - 1\right)}{4}$$
Price fluctuations	$$\sqrt{\frac{1}{m-1} \times \sum_{t=1}^{12} \left(ln\left(\frac{P_{i,t}}{P_{i,t-1}}\right) - R^2\right)^2} \times \sqrt{12}$$ *m* is amount of price data, *P* are the month values for the respective resource, *t* is the time, *p* is the monthly return of the respective metal, while *R* is the mean return over the year
Primary material use	$$100\% - recycled\ content_i$$

Occurrence of co-production is a semi quantitative indicator that is determined as displayed in Table S 2.

Table S 2 Indicator values for the category occurrence of co-production

Qualitative criteria as reported by Angerer et al. (2009)	Quantitative indicator used in SCARCE
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

To determine the final indicator results of the categories, three steps are carries out: i) determination of the indicator value for each category and individual resource (formulas for the indicators are displayed in Table S 1)

ii) indicator values are set in relation to the category specific targets (for details on how these targets were set up, see Bach et al. (2017, 2016)) to determine the Distance-to-Target (DtT) value based on the ecological scarcity approach (Müller-Wenk and Ahbe, 1990) with the following formula for the resource *i* and the category *c*:

$$DtT - value_{i,c} = \left(\frac{indicator\ value_{i,c}}{target}\right)^2 \tag{S1}$$

DtT values smaller than 0.8 are set to zero. If the indicator value is below the target value, it is set to zero meaning no supply risk. This means, that the determined indicator value is set in relation to a target value (which is predefined in the SCARCE method and was obtained by stakeholder interviews).

iii) scaling of the DtT values within each category to between zero and one.

$$indicator\ result_{i,c} = \left(\frac{DtT - value_{i,c} - DtT - value_{min,c}}{DtT - value_{max,c} - DtT - value_{min,c}}\right) \tag{S2}$$

The advantage of scaling between zero and one is that materials are well comparable; while the disadvantage is that, the order of magnitude of the difference is not so clearly visible anymore.

The supply risk of resource *i* can be determined by summing up all indicators of the supply risk categories for resource *i* in which all categories are weighted equally (see equation S3). This summed result is used for the sub dimension specific discussion of the results (section 3.1.2).

$$Supply\ Risk_{i,unscaled} = \sum_{c}(DtT - value_{i,c}) \quad \text{(S3)}$$

As a final step, the supply risk results are also scaled between zero and one to plot them in the criticality matrix:

$$Supply\ Risk_i = \frac{Supply\ Risk_{i,unscaled} - Supply\ Risk_{min,unscaled}}{Supply\ Risk_{\ max,unscaled} - Supply\ Risk_{min,unscaled}} \quad \text{(S4)}$$

This scaling ensures that only the highest supply risk value has the value one and the minimum value hast the value zero and thus can be plotted in the criticality matrix.

1.3. Vulnerability

The sub dimension vulnerability reflects how sensitive a country reacts to potential supply risks of raw materials. The vulnerability is determined by considering the following seven categories based on existing vulnerability methods (e.g.: Erdmann et al. (2011), Graedel et al. (2012), Knašytė et al.(2012)).

- Economic importance: economic profits of the imported raw materials
- Domestically required demand: imported amount of raw material
- Availability of purchasing strategies: between countries mining the raw materials and countries utilizing the raw materials to reduce potential supply risks
- Utilization of future technologies: demand of specific raw materials by future technologies and consumption patterns
- Share of global production: the share of imported raw materials is set in relation to the global production to identify material competition
- Dependency on imports: domestic production is determined and set in relation to the imported amount, which reflects the dependency on imports
- Substitutability: how well raw materials can be substituted

The calculation or determination of the indicators for the categories of the sub dimension vulnerability, that were used in the application of SCARCE to Europe (thus excluding *availability of purchasing strategies*) are displayed in Table S 3, Table S 4 and Table S 5.

Table S 3 Formulas for indicator calculation considered in the sub dimension vulnerability

Category	Calculation formula
Economic importance	$\left(\sum_{y} share[\%]_{y,i} \cdot value\ added_{y,i}\right) \cdot Primary\ material\ use_i$ *y* stands for all sectors that the material *i* is used in. The primary material use is not sector specific.
Domestically required demand	$imported\ material\ [t]_i$

Share of global production	$\frac{European\ imports\ [t]_i}{Global\ production\ [t]_i}$
Dependency on imports	$1 - \frac{European\ production[t]_i}{Imported\ amount[t]_i}$

We collected updated data for the indicator values of the categories *utilization in future technologies* and *substitutability* (both categories are not Europe specific). For determining the indicator value of the category *utilization in future technologies* we classified the abiotic resources into five tears (see below). The classification is based on peer reviewed literature (Grandell et al. (2016)) and government reports mainly by the U.S. Geological Survey Minerals Yearbook from 2015 or 2016 (U.S. Geological Survey, 2018a) and from the reports by Marscheider-Weidemann et al. (2016) and Erdmann and Behrendt (2011). To translate qualitative information into a quantitative indicator the classification according to Table S 4 was performed.

Table S 4 Indicator determination for the category utilization in future technologies

Indicator value	Characteristics
0	No indication that the material will be used in specific future technologies or so large overall consumption that the use will be comparably irrelevant
0.3	Potential of material is mentioned (because of interesting properties) but not in significant known quantities or technologies
0.5	The material will be needed in future technologies that are still under development and it is unlikely that its use in these technologies will dominate the demand of the material
0.7	The material will be needed in key future technologies like electro mobility, low carbon technologies, artificial intelligence, robotics and might dominate current supply
1	The material will be needed in key future technologies like electro mobility, low carbon technologies, artificial intelligence, robotics and might exceed current supply

In order to determine the indicator value of the category *substitutability,* we classified the abiotic resources into four tears. The characteristics of the semi-quantitative indicators are possibility of technological substitution, quality decrease and price increase. The values were updated based on peer reviewed literature by Graedel et al. (2015) and government reports by U.S. Geological Survey (2018b). To translate qualitative information into a quantitative indicator the classification according to Table S 5 was used.

Table S 5 Indicator determination for the category substitutability

Indicator value	Characteristics
0	Substitution exist and no quality decrease and no higher costs
0.3	Substitution exist and no quality decrease, but higher costs
0.5	Substitution exist, but comes with some quality decrease
0.7	Substitution exist, but comes with quality decrease
1	Barely or no substitute exists

After the indicator values of the vulnerability categories are determined, the indicator result is calculated based on formula S5.

$$indicator\ result_{i,c} = \left(\frac{indicator\ value_{i,c} - indicator\ value_{min,c}}{indicator\ value_{max,c} - indicator\ value_{min,c}}\right) \tag{S5}$$

Further, the vulnerability of resource *i* can be determined by summing up all indicator results of the vulnerability categories for resource *i* (see equation S6). This result is used for the sub dimension specific discussion of the results (see section 3.1.3 in the main paper).

$$Vulnerability_{i,unscaled} = \sum_{c} indicator\ results_{i,c} \tag{S6}$$

The vulnerability results are scaled between zero and one, so they can be plotted in the criticality matrix (see equation S7):

$$Vulnerability_{i} = \frac{Vulnerability_{i,unscaled} - Vulnerability_{min,unscaled}}{Vulnerability\ _{max,unscaled} - Vulnerability_{min,unscaled}} \tag{S7}$$

This scaling ensures that only the highest vulnerability value has the value one and the minimum value has the value zero.

1.4. Societal Acceptance

To assess resource use in the context of sustainable development social and environmental aspects have to be considered next to criticality. To assess social aspects a new approach was developed by Bach et al. (2016), which considers social aspects relevant for consumers. To determine the sub dimension compliance with social standards, overall three categories were identified as being significant:

- Small scale mining: quantifies the percentage of resources being sourced in small scale mining operations, which face social challenges, e.g. unsafe working conditions based on Brower et al. (1979)
- Geopolitical risk: determines the state of peace within the sourcing country, including the probability of armed conflicts based on Kaufmann and Kraay (2015) and The Institute for Economics and Peace (2015)
- Human rights abuse: considers the amount of forced labor, child labor and overall torture in the sourcing country based on Benoît-Norris et al. (2012) and Cingranelli and Richards (2010)

Reputational risks can also be associated with environmental aspects, which have been on the agenda related to product life cycle as well, especially with regard to mining (Garcia et al., 2017; Hanna et al., 2016)). Thus, three categories are considered to assess the compliance with environmental standards:

- Sensitivity of local biodiversity: assesses how sensitive the biodiversity in the sourcing country is based on Winter et al. (2018)
- Water scarcity: considers how high the local water scarcity is and its impact on ecosystems and human use based on Berger et al. (2014)
- Climate change: considers the amount of greenhouse gases emitted in the mining process of the resource based on Huijbregts et al. (2017)

Details on how these categories are quantified can be found in in the publication by Bach et al. (2017). The formulas for calculating the indicators for resource *i* exported from or produced by country *x* for the categories of the sub dimension compliance with social standards are provided in Table S 6, while those for the calculation of the categories for the sub dimension compliance with environmental standards are provided in Table S 7.

Table S 6 Indicator calculation for the categories of the sub dimension compliance with social standards

Categories	Calculation formula
Small scale mining	$$\sum_x import[\%]_{x,i} \cdot \; share\ of\ ssm_{x,i} \; \cdot (share\ of\ artisanal\ ssm_{x,i} + 1)$$ Ssm = small scale mining
Geopolitical risk	$$\sum_x import[\%]_{x,i} \cdot (GI_{\ x} + GPI_x)^2$$ GI = World Governance Indicators for voice and accountability and political stability and no violence; GPI = Global Peace Index
Human right abuse	$$\sum_x import[\%]_{x,i} \cdot (CL_x + FL_x + CIRI_x)^2$$ CL = Child labor based on Social Hotspot database; FL = Force labor also based on social hotspot database, torture is based on Cingranelli- Richards Human Rights Physical Integrity Rights Index (CIRI)

Table S 7 Indicator calculation for the categories of the sub dimension compliance with environmental standards

Categories	Calculation formula
Water scarcity	$$\sum_x import[\%]_{x,i} \cdot WDI_x$$ Import share of European import mix of resource *i* from country *x* multiplied by WDI (Water depletion index) from country *x*
Climate change	$$(CCHH_i + CCE_i)^2$$ CCHH=climate change impact to human health of resource *i* by applying ReCiPe end point CCE = Climate change impacts to ecosystems by resource *i* applying ReCiPe endpoint
Sensitivity of local biodiversity	$$\sum_x import[\%]_{x,i} \cdot (SE_x + CS_x + ES_x)^2$$ With SE = scarcity of ecoregions in country *x*; CS = conservation status in country *x*, ES = number of endemic species in country *x*

The final indicator results of the sub dimensions compliance with social standards and compliance with environmental standards are determined as follows:

i) Determination of intermediate indicator results for the considered categories and resource by inserting the data into the formulas displayed in Table S 6 and Table S 7.

ii) Scaling the category specific results between 0-1,

$$indicator\ result_{i,c} = \frac{indicator\ value_{\ i,c} - indicator\ value_{\ min,c}}{indicator\ value_{\ max,c} - indicator\ value_{\ min,c}} \tag{S8}$$

iii) Summing up the scaled indicator results of the considered resource *i* for the categories of the sub dimension compliance with social standards and compliance with environmental standards considering the respective categories (as shown in equation S9 and S10);

$$Compliance\ with\ social\ standards_i = \sum_c indicator\ results_{i,c} \tag{S9}$$

$$Compliance\ with\ environmental\ standards_i = \sum_c indicator\ results_{i,c} \tag{S10}$$

As environmental and social indicators are not plotted in the criticality matrix, scaling of the sub dimension is not necessary. The obtained results are then sorted in descending order. The top five highest results of the sub dimensions are environmental and social hotspots.

2. Comparing the Results of European Supply Risk Categories with the Global Supply Risk Categories

In this section, the import-based supply risk category-results are compared for Europe and the world. Figure S 2 shows the aggregated difference of the European import-based supply risk categories and the ones obtained by the assessment of the average global production. The emphasis is not on the result per resource but to see whether the categories have a lower or higher performance for Europe than for the global supply risks or vice versa.

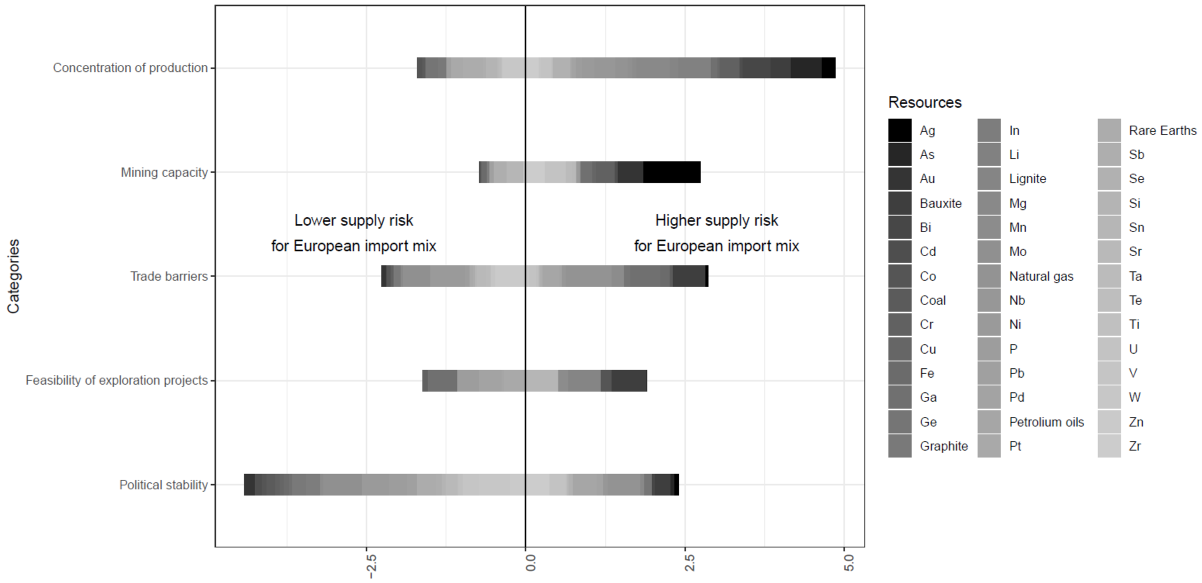

Figure S 2 Aggregated difference of import-based categories for Europe and global supply risk; with the categories on the y-axis

European imports have slightly higher trade barriers compared to the global production mix. This is mainly influenced by bauxite, gallium and natural gas imports into Europe. The countries from which Europe is receiving these resources show high trade barriers (Guinea, China and Russia). The concentration of production and mining capacity are higher for Europe. One explanation is that some of Europe's trade relationships are dominated by one country only. Comparably, the global production portfolio is more diverse. Examples are silver and bauxite. In addition, the mining capacity of Europe's importing countries is lower compared to the global average, especially for silver and gold. However, countries that import into Europe tend to have higher political stability (and thus a lower indicator score), since many materials that are mainly produced in China are only imported in very limited amounts into Europe from China, like molybdenum, rare earths, tungsten and zinc. If the reserves of the political more stable countries are exhausted the European economy might become more dependent on material imports from political unstable countries.

3. Comparison of SCARCE with the EU Method

Initiated by the European Raw Materials Initiative (European Commission, 2008), a list of critical raw materials for the EU is derived on a regular basis. The latest and third assessment was published in 2017 (Deloitte Sustainability et al., 2017). The study identified 26 raw materials and groups of raw materials out of 61 as critical. The underlying assessment is based on a methodology developed by the European Commission (Pennington et al., 2017). In this chapter the main differences between the methods developed by the European Commission (Pennington et al., 2017) and SCARCE are discussed.

For simplicity reasons, considered aspects in the EU method are allocated to categories in the SCARCE method. This was done to avoid terminology confusion. Based on a detailed analysis of the EU critical raw material study we derived an overview, which compares the categories considered in the SCARCE method with the categories that are considered in the EU method. Further, we determined differences in the data sources (see Table S 8). Categories that show an overlap are marked in green, while categories that have no correspondence are highlighted in red in the EU method column.

Table S 8 Comparison of the categories considered in the SCARCE method and the critical raw material study for the European Union

Sub dimensions	Considered categories /indicators in		Comments
	SCARCE	EU method	
Vulnerability/ Economic importance	Economic Importance	Economic Importance	The calculation method and data sources of this category are similar, but in the EU method the substitution of the material is also considered, for which SCARCE has a separate indicator that is part of the vulnerability dimension
	Substitutability	Substitution index of a raw material related to economic importance	EU method mainly considered the cost performance of possible substitutes while SCARCE considers if substitutes exist and whether there is performance decrease or cost increase
	Dependency on imports	Import reliance (considered under supply risk)	Similar approach, but other use of it in the calculation: import reliance of EU method strongly influences the supply risk dimension while it is a separate indicator in SCARCE
	Share of global production		Missing in EU method
	Domestic required demand		Missing in EU method
	Utilization in future technologies		Missing in EU method
Supply Risk	Country concentration of production/ HHI	HHI = Herfindahl-Hirschman Index (used as a proxy for country concentration)	Similar approach, differences in the determination of the import mix and thus different results

	Political stability/WGI	WGI = scaled World Governance Index (used as a proxy for country governance)	WGI is also used for the political stability indicator in SCARCE
	Occurrence of co-production	SISR = substitution index related to supply risk	Different approach: SCARCE considers whether the resource itself is a co-product, EU method considers if substitute is a co-product
	Trade barriers/ETI	t = trade parameter adjusting WGI	Different data sources
	Primary material use	EOLRIR = end-of-life recycling input rate	The same approach, but differing data sources
	Country concentration of reserves		Missing in EU method
	Feasibility of exploration projects		Missing in EU method
	Demand growth		Missing in EU method
	Mining capacity		Missing in EU method
	Price fluctuations		Missing in EU method

Based on this confrontation some methodological differences were derived that are further specified in the following:

- Different geographical scope: SCARCE additionally includes Switzerland, Norway and Iceland as part of the European single market with similar availability constraints.
- Different categories: SCARCE considers additional vulnerability categories like *share of global production*, *domestic required demand* and *utilization in future technologies* as well as supply risk categories (*concentration of reserves, feasibility of exploration projects, demand growth*, *mining capacity* and *price fluctuations*).
- Different allocation of categories to dimensions: The EU method considers the category import reliance under the sub dimension supply risk, whereas SCARCE considers it (*dependency on imports*) under the sub dimension vulnerability.
- Differences in calculation of import mix: SCARCE calculates the import mix based on world production mix and Europe's import mix (see section 1.1 in the supplementary material). The exclusion of all non-producing countries that export to Europe from the European import mix is essential as it ensures that only countries directly importing into Europe are assessed. The EU method does not exclude non-producing countries that export to the EU from the import mix. Instead, it weights them according to their import reliance. For example in the case of 100 % import reliance, the supply risk would be determined by 50 % based on the global supply mix and 50 % based on the EU import mix.
  - Different definitions of categories:
    - Dependency on imports (SCARCE)/ Import reliance (EU method): In comparison to the EU method, SCARCE does not subtract the exports from the imports in its determination of the dependency on imports, which the EU method does.

- Consideration of different resource types: Both methods take into account metals, metalloids and minerals. The EU method additionally includes biotic resources, whereas SCARCE includes fossil fuels.
- Consideration of social and environmental impacts: In comparison to SCARCE, the current EU method did not include social nor environmental impacts of resource extraction.
- Presentation of the results: The EU method determined the criticality lines (thresholds) empirically from the first report in 2011 (European Commission, 2011) and defined one criticality zone. Criticality lines in SCARCE are adapted from Glöser et al. (2015) and represent low, medium and high criticality. The different definitions of criticality also have an effect on the classification of the criticality zones and the identified materials.

4. Comparison of Results of the Study Based on the EU Method with SCARCE Results

In the following a detailed comparison of materials that obtained different results in the study based on the EU method and the study performed in this paper are provided for tungsten, indium, silicon and gallium:

Tungsten: Based on the underlying data in SCARCE the imports are mainly from Canada, whereas for the study based on the EU method Russia is the country that exports the largest amount of the resource to Europe (Deloitte Sustainability et al., 2017). The difference can be explained by the use of different reference years: whereas SCARCE is using data with reference year 2015, the study based on the EU method considers average data from 2010 to 2014.

Indium: The differences can be explained by use of different reference years as well. In SCARCE, 41 % of the supply comes from Canada (additional 21 % from China and 17 % from Japan), whereas in the case of the study based on the EU method only 5 % are imported from Canada, 5 % from Japan but 41 % from China. A different classification also occurs from the different definitions of criticality zones in both methods.

Silicon: In the study based on the EU method silicon is classified as a critical raw material, but very close to being classified as non-critical. The differences can be explained by the different definitions of criticality lines like in the case of indium.

Gallium: Differences in the outcomes occur due to different underlying data and considered categories. In the study based on the EU method gallium China imported 50 % into the EU, whereas the underlying data in the SCARCE method shows 85 % of imports from China. Furthermore, gallium shows high values for the category utilization in future technologies, which is not included in the EU method.

5. Comparison with the Case Study of Germany

The SCARCE method was previouly applied to Germany (Bach et al., 2017) and yielded different results compared to the European case study. Generally the method is not suited for inter country comparison but rather for a country or region to track its own criticality. Some categories like share of global production will naturally rise if the assessed country or region is bigger or more populated. However, the dependency on imports will decrease if a broader geographical scope is analyzed since it will contain more resource deposits. There are some materials that have different results that are outlined shortly:

1) Petroleum oils: According to Bach et al. (2017) Germany imports 17 % from Kazakhstan and 24 % from European countries like Norway, the United Kingdom and the Netherlands. While Europe imports 34 % from Russia, Germany only imported 18 % from Russia, which resulted in an overall lower supply risk (since Russia has low political stability and high trade barriers). Nigera that has low feasability of exploration projects is close to abscent from the German import mix (2%), but has 10 % of the European import mix. This results in a higher supply risk of petroleum oils for Europe than for Germany.

2) Natural gas: According to Bach et al. (2017) 17 % of German imports came from Kazakhstan. Further Germany imported 24 % of natural gas from European countries like Norway, the Netherlands and the United Kingdom, which are not accounted as imports into Europe (as they are part of Europe), but limit the import dependency. Europe imports its natural gas to 60 % from Russia and 28 % from Algeria. These countries have low political stability and high trade barriers. This results in a higher supply risk of natural gas for Europe than for Germany.

3) Tungsten: Most tungsten is imported from Bolivia to Germany while European imports are dominated by Canada (but also not as concentrated as the German imports). German tungsten imports have thus lower political stability, higher production concentration and higher trade barriers. Tungsten has a social hotsport in the German assessment, because of Bolivia's high prevalence of small scale mining. Small scale mining is not relevant for the Canadian tungsten imports into Europe.

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Supplementary Material Paper 3

This section of the appendix contains the supplementary material of the following publication:

Paper 3: Arendt, Rosalie; Bach, Vanessa; Finkbeiner, Matthias: The global environmental costs of mining and processing abiotic raw materials and their geographic distribution. *Journal of Cleaner Production* **2022** https://doi.org/10.1016/j.jclepro.2022.132232

Supplementary Material: The Global Environmental Costs of Mining and Processing Abiotic Raw Materials Supplementary Material and Their Geographical Distribution

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This supplementary material contains:

- A brief description of the Scope and limitation of LCA and ReCiPe (section 1)
- The documentation of the system boundaries (Figure S 1) and used processes from LCA databases (Table S 9) (section 2)
- The sources of the employment statistics for the calculation of the damages per job (section 3)
- All impact categories, that were not discussed in the main paper for the low estimate and the contribution of each material to the total domestic costs of the countries with the highest costs (section 4.1)
- All results for the high estimate (section 4.2)

1. Brief Description of Scope and Limits of LCA and ReCiPe

LCA is an assessment method to assess the environmental impacts of products along their life cycle. It does not aim to replace a site-specific environmental impact assessment (e.g. the assessment whether local laws and protected areas are respected). The impact assessment method ReCiPe assesses impacts at midpoint and at endpoint level. The covered impacts in this study are acidification, climate change, eco- and human toxicity, particulate matter, photochemical oxidation potentials, land use, water use, freshwater eutrophication and ionizing radiation. We have used a damaged based assessment, thus the impact at the areas of protection are measured. The effect on cultural value etc. is neglected in environmental LCA, which leads to an underestimation of the impacts to society.

2. Processes

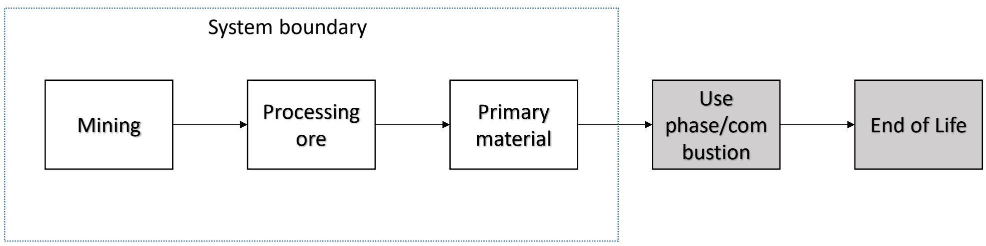


Figure S 3: Simplified illustration of the system boundaries for the LCA of the materials

Table S 9: Underlying processes to quantify impacts from LCA databases; the EU and RER processes were used for all EU countries, and RME for all countries of the middle east and RNA for the U.S.A. and Canada

Material	Process	Database

Aluminium	CA: Aluminium ingot mix IAI 2015 IAI/Sphera CN: Aluminium ingot mix IAI 2015 IAI/Sphera EU28+EFTA: Primary aluminium production ingot mix Europe (2015 only produced in Europe) European Aluminium GLO: Aluminium ingot mix IAI 2015 IAI/Sphera RME: Aluminium ingot mix IAI 2015 IAI/Sphera RU: Aluminium ingot mix IAI 2015 IAI/Sphera	GaBi
Silver	GLO: Silver mix agg ts RNA: Silver ts SE: Silver ts	GaBi
Arsenic	GLO: market for sodium arsenide ecoinvent 3.7.1	Ecoinvent
Hard coal/Coal	EU-28 Hard coal mix agg ts AT: Hard coal mix ts AU: Hard coal mix ts BE: Hard coal mix ts BR: Hard coal mix ts CA: Hard coal mix ts CN: Hard coal mix ts DE: Hard coal mix ts ES: Hard coal mix ts EU-28: Hard coal mix ts FI: Hard coal mix ts FR: Hard coal mix ts GB: Hard coal mix ts IE: Hard coal mix ts IN: Hard coal mix ts IT: Hard coal mix ts JP: Hard coal mix ts LU: Hard coal mix ts NL: Hard coal mix ts NO: Hard coal mix ts NZ: Hard coal mix ts PL: Hard coal mix ts PT: Hard coal mix ts RO: Hard coal mix ts RU: Hard coal mix ts	GaBi

	SE: Hard coal mix ts SI: Hard coal mix ts SK: Hard coal mix ts TH: Hard coal mix ts TR: Hard coal mix ts TW: Hard coal mix ts US: Hard coal mix ts ZA: Hard coal mix ts	
Graphite	CN: graphite production, battery grade ecoinvent 3.7.1 GLO: market for graphite, battery grade ecoinvent 3.7.1	Ecoinvent
Gold	GLO: Gold (primary) agg ts SE: Gold ts	GaBi
Cadmium	GLO: Cadmium ts GLO Metal production, technology mix, production mix, at plant, 8.65 g/cm3, 112.4 g/mol	GaBi
Cobalt	GLO Cobalt, refined (metal) agg	GaBi
Chromium(Ferrochromium)	ZA Ferro chrome agg ts	GaBi
Copper	DE: Copper mix (99,999% from electrolysis) ts GLO: Copper mix (99,999% from electrolysis) ts SE: Copper ts	GaBi
Iron/steel	DE: Steel cold rolled coil 2,5 mm ts <t-agg> EU: Steel cold rolled coil worldsteel EU: Steel finished cold rolled coil worldsteel GLO: Steel cold rolled coil worldsteel GLO: Steel finished cold rolled coil worldsteel RNA: Steel finished cold rolled coil	GaBi
Gallium	GLO market for gallium, semiconductor-grade agg ecoinvent 3.7.1	Ecoinvent
Indium	GLO market for indium agg ecoinvent 3.7.1	Ecoinvent

	RER: indium production ecoinvent 3.7.1	
Niobium	BR Ferro Niobium agg ts	GaBi
Lithium	GLO: market for lithium ecoinvent 3.7.1	Ecoinvent
Magnesium	CN Magnesium agg ts	GaBi
Manganese	ZA: Manganese ts [Metal production] production mix, at plant 99,9% concentrated	GaBi
Molybdenum	GLO: Ferromolybdenum (FeMo)	IMOA
Nickel	GLO Nickel mix agg ts	GaBi
Phosphorous	EU-28: Raw phosphate (32% P2O5) Fertilizers Europe GLO: Rock phosphate mix (32,4 % P2O5) ts	GaBi
Lead (Pb)	DE: Lead (99,995%) ts RNA: Lead (99,995%) ts SE: Lead (99,995%) ts	GaBi
Palladium (Pd)	GLO Palladium mix agg ts	GaBi
Platinum	GLO Platinum mix agg ts	GaBi
Antimony	CN Antimony agg ts	GaBi
Selenium	GLO: market for selenium ecoinvent 3.7.1 RER: selenium production ecoinvent 3.7.1 RoW: selenium production ecoinvent 3.7.1	Ecoinvent
Rare earths	CN: rare earth oxides production, from rare earth oxide concentrate, 70% REO ecoinvent 3.7.1 GLO: market for rare earth oxide concentrate, 70% REO ecoinvent 3.7.1 RoW: rare earth concentrate production, 70% REO, from bastnäsite ecoinvent 3.5	Ecoinvent
Silicon	GLO: Ferro silicon mix (90% Si) ts	GaBi
Tin (Sn)	GLO Tin agg ts	GaBi

Sr	GLO market for strontium carbonate agg ecoinvent 3.7.1	Ecoinvent
Ta	GLO market for tantalum, powder, capacitor-grade agg ecoinvent 3.7.1	Ecoinvent
Te	GLO: market for tellurium, semiconductor-grade ecoinvent 3.7.1 CA-QC: tellurium production, semiconductor-grade ecoinvent 3.7.1	Ecoinvent
Ti	GLO: Titanium ts	GaBi
Zn (Zinc)	DEU: zinc RNA: zinc	GaBi
Zr (Zirconium)	AU: zirconium oxide production ecoinvent 3.7.1 GLO: market for zirconium oxide ecoinvent 3.7.1	Ecoinvent
Petroleum oil	GLO:average crude oil (own average calculation) AE: Crude oil mix Sphera AL: Crude oil mix Sphera AR: Crude oil mix Sphera AT: Crude oil mix Sphera AU: Crude oil mix Sphera AZ: Crude oil mix Sphera BA: Crude oil mix Sphera BE: Crude oil mix Sphera BG: Crude oil mix Sphera BR: Crude oil mix Sphera CA: Crude oil mix Sphera CH: Crude oil mix Sphera CL: Crude oil mix Sphera CN: Crude oil mix Sphera CO: Crude oil mix Sphera CZ: Crude oil mix Sphera DE: Crude oil mix Sphera DK: Crude oil mix Sphera DZ: Crude oil mix Sphera EC: Crude oil mix Sphera EG: Crude oil mix Sphera EG: Crude oil mix Sphera ES: Crude oil mix Sphera	GaBi

	FI: Crude oil mix Sphera FR: Crude oil mix Sphera GB: Crude oil mix Sphera GR: Crude oil mix Sphera HR: Crude oil mix Sphera HU: Crude oil mix Sphera ID: Crude oil mix Sphera IE: Crude oil mix Sphera IN: Crude oil mix Sphera IQ: Crude oil mix Sphera IR: Crude oil mix Sphera IT: Crude oil mix Sphera JP: Crude oil mix Sphera KR: Crude oil mix Sphera KZ: Crude oil mix Sphera LK: Crude oil mix Sphera LT: Crude oil mix Sphera MA: Crude oil mix Sphera MX: Crude oil mix Sphera MY: Crude oil mix Sphera NG: Crude oil mix Sphera NL: Crude oil mix Sphera NO: Crude oil mix Sphera NZ: Crude oil mix Sphera OM: Crude oil mix Sphera PE: Crude oil mix Sphera PH: Crude oil mix Sphera PL: Crude oil mix Sphera PT: Crude oil mix Sphera RO: Crude oil mix Sphera RS: Crude oil mix Sphera RU: Crude oil mix Sphera SA: Crude oil mix Sphera SE: Crude oil mix Sphera SG: Crude oil mix Sphera SK: Crude oil mix Sphera TH: Crude oil mix Sphera TM: Crude oil mix Sphera TR: Crude oil mix Sphera TT: Crude oil mix Sphera TW: Crude oil mix Sphera UA: Crude oil mix Sphera US: Crude oil mix Sphera UZ: Crude oil mix Sphera VE: Crude oil mix Sphera	

	ZA: Crude oil mix Sphera	
Lignite	CA: Lignite mix ts CIS: Lignite mix ts DE: Lignite mix ts ES: Lignite mix ts EU-28: Lignite mix ts FR: Lignite mix ts GR: Lignite mix ts HU: Lignite mix ts IN: Lignite mix ts PL: Lignite mix ts RO: Lignite mix ts SI: Lignite mix ts SK: Lignite mix ts TH: Lignite mix ts TR: Lignite mix ts US: Lignite mix ts	GaBi
Uranium	GLO market for uranium ore, as U agg ecoinvent 3.7.1.	Ecoinvent
Natural gas	AT: Natural gas mix ts AU: Natural gas mix ts BE: Natural gas mix ts BR: Natural gas mix ts CA: Natural gas mix ts CH: Natural gas mix ts CN: Natural gas mix ts DE: Natural gas mix ts ES: Natural gas mix ts EU-28: Natural gas mix ts FI: Natural gas mix ts FR: Natural gas mix ts GB: Natural gas mix ts GR: Natural gas mix ts HU: Natural gas mix ts IE: Natural gas mix ts IN: Natural gas mix ts IT: Natural gas mix ts JP: Natural gas mix ts LT: Natural gas mix ts LU: Natural gas mix ts LV: Natural gas mix ts NL: Natural gas mix ts NO: Natural gas mix ts	GaBi

	NZ: Natural gas mix ts PL: Natural gas mix ts PT: Natural gas mix ts RER: Natural gas PlasticsEurope RO: Natural gas mix ts SE: Natural gas mix ts SI: Natural gas mix ts SK: Natural gas mix ts TH: Natural gas mix ts TR: Natural gas mix ts TW: Natural gas mix ts UA: Natural gas mix ts US: Natural gas mix ts VE: Natural gas mix ts ZA: Natural gas mix ts	

3. Sources for employment statistics

Table S 10:Statistics for employments to calculate damages per job

Country Name	ISO3-digit Alpha	Employment	Source
Angola	AGO	150000	(Hilson, 2016)
Albania	ALB	99000	(CEIC Data, 2021)
United Arab Emirates	ARE	2200000	(TDRA, 2021)
Argentina	ARG	103000	(Argentina; Instituto Nacional de Estadística y Censos (Argentina), 2021)
Armenia	ARM	9000	(Avinyan, 2021)
Australia	AUS	252100	(Commonwealth of Australia, 2021)
Austria	AUT	6451	(Statistik Austria, 2021)
Azerbaijan	AZE	40592	(Hampel-Milagrosa and Siba, 2019)
Burundi	BDI	10000	(Matthysen, 2015)
Burkina Faso	BFA	200000	(Hilson, 2016)
Bolivia	BOL	150000	Estimate based on (Bocangel, 2001)
Brazil	BRA	1741100	(Instituto Brasileiro de Mineração, 2021)
Botswana	BWA	12773	(Statistics Botswana, 2016)
Central African Rep.	CAF	400000	(Hilson, 2016)
Canada	CAN	409000	(Garside, 2020)
Chile	CHL	239000	(Consejo minero, 2021)
China	CHN	6000000	(Gunson and Jian, 2001)

Côte d'Ivoire	CIV	100000	(Hilson, 2016)
Dem. Rep. of the Congo	COD	200000	(Hilson, 2016)
Germany	DEU	38000	(Eurostat, 2021)
Eritrea	ERI	400000	(Hilson, 2016)
Ethiopia	ETH	500000	(Hilson, 2016)
Finland	FIN	13000	(Finnish Minerals Group, 2021)
France	FRA	13200	(Eurostat, 2021)
Gabon	GAB	2000	(Ministère de l'Economie et des Finances-Direction générale du Trésor Francais, 2020)
United Kingdom	GBR	54000	(Eurostat, 2021)
Ghana	GHA	1100000	(Hilson, 2016)
Guinea	GIN	300000	(Hilson, 2016)
India	IND	2300000	(The Economic Times, 2019)
Liberia	LBR	100000	(Hilson, 2016)
Madagascar	MDG	500000	(Hilson, 2016)
Mozambique	MOZ	100000	(Hilson, 2016)
Niger	NER	450000	(Hilson, 2016)
Nigeria	NGA	500000	(Hilson, 2016)
Peru	PER	177000	(Gálvez Delgado et al., 2020)
Philippines	PHL	28992	(Philippine Statistic Authority, 2018)
Russian Federation	RUS	1117000	(EMIS, 2019)
Rwanda	RWA	35000	(Matthysen, 2015)
Saudi Arabia	SAU	250000	(Puri-Mirza, 2021)
Sierra Leone	SLE	300000	(Hilson, 2016)
South Sudan	SSD	200000	(Hilson, 2016)
Turkey	TUR	92000	
United Rep. of Tanzania	TZA	1500000	(Hilson, 2016)
Uganda	UGA	150000	(Hilson, 2016)
USA	USA	663000	(Burea of Economic Analysis (U.S.A.), 2020)
South Africa	ZAF	451427	(Minerals Council South Africa, 2020)
Zambia	ZMB	65000	(IndustriALL, 2019)
Zimbabwe	ZWE	500000	(Hilson, 2016)

4. Additional results

4.1. Not discussed impact categories low estimate

In the following, the impact categories that were not discussed in the main paper are outlined. The impacts are described in descending order (the highest costs are outlined first). In Figure S 4, the costs of the impact category land use are shown. Steel production in China and manganese mining in South Africa mostly cause them.

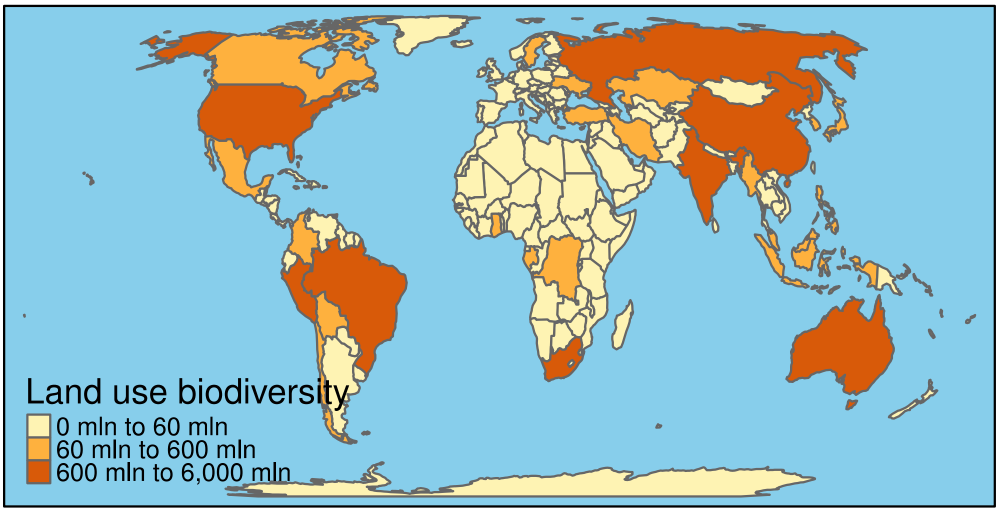


Figure S 4: cost distribution in Euro [€] for the impact category land use and its impact on biodiversity, low estimate

Other countries with high land use impacts are the U.S.A. (coal), Russia (coal), Peru (zinc), Brazil (manganese), India (coal) and Australia (manganese and coal). In Figure S 5 the impacts on terrestrial ecotoxicity is shown. Most of the displayed impacts are caused by rare earth extraction and zirconium.

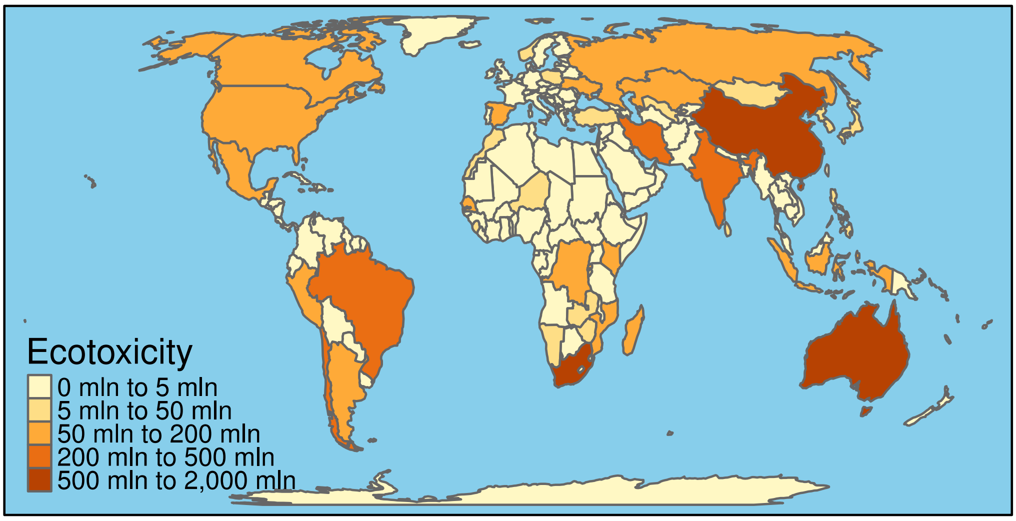


Figure S 5: Cost distribution of the impact category terrestrial ecotoxicity in euro [€], low estimate

In Figure S 6 we see the distribution of non-cancerous human toxicity. All countries with high impacts have impacts from extraction of crude oil, but also magnesium and aluminum play a role for China.

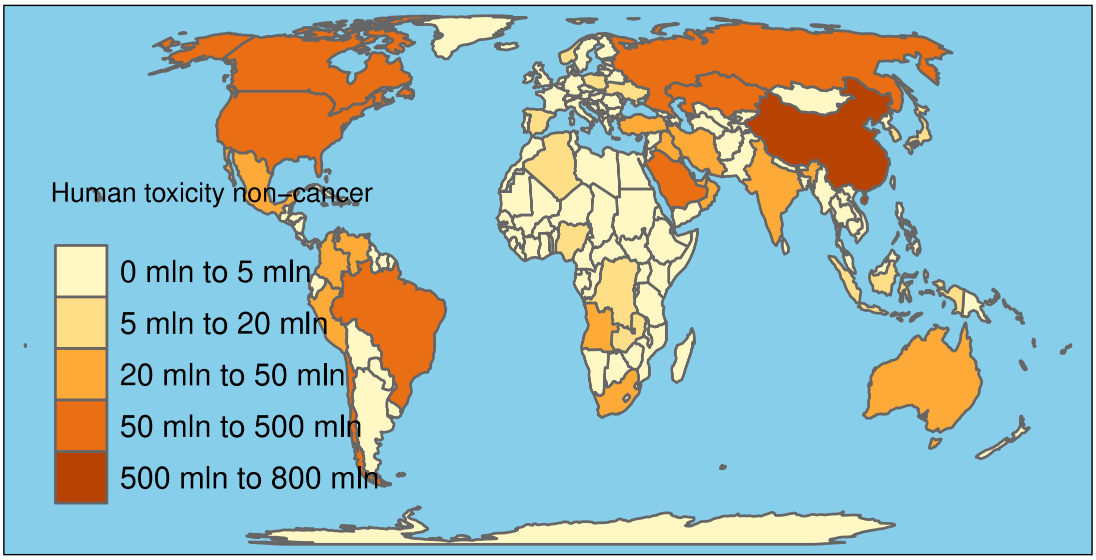


Figure S 6: Cost distribution of the impact category human toxicity [non-cancer] in euro [€], low estimate

In Figure S 7, the impacts from water use on biodiversity are visualized. The countries with the highest costs are China (associated with magnesium, aluminum and manganese production), Russia (aluminum and manganese), South Africa (manganese and chromium) and Australia (manganese and coal).

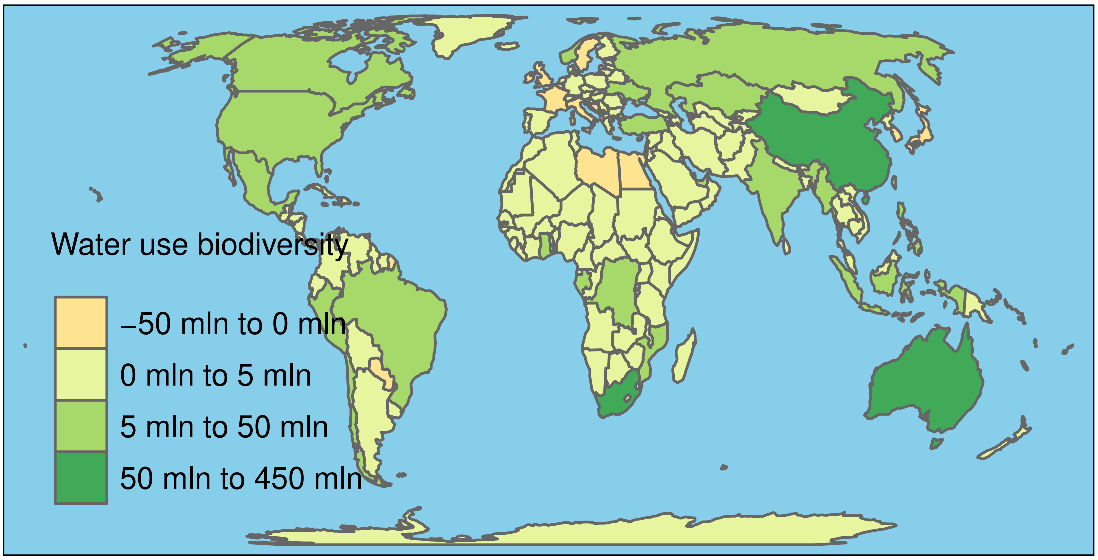


Figure S 7: Cost distribution of the impact category water use [impact on biodiversity] in euro [€], low estimate

In Figure S 8, the damages of photochemical oxidation on human health are displayed. The country with the highest damages is China, mostly due to coal, iron, aluminum and magnesium production there.

Figure S 8: Cost distribution of the impact category POCP [impact on biodiversity] in euro [€], low estimate

In Figure S 9, we see the impact distribution of the impact category "human toxicity cancer". Here impacts are also dominated by petroleum oil extraction that contributes to impacts in Russia, the U.S.A., Canada and Brazil. Impacts in China stem from aluminum and magnesium production.

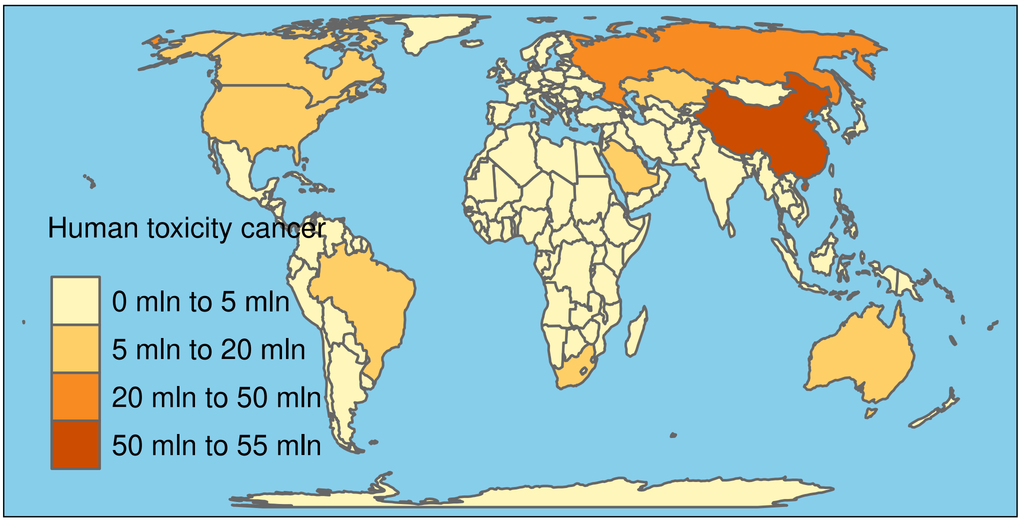


Figure S 9: Cost distribution of the impact category human toxicity [cancer] in euro [€], low estimate

In Figure S 10, the impacts on human health from water use is shown. China has the highest impacts that are caused by aluminum, coal and magnesium production.

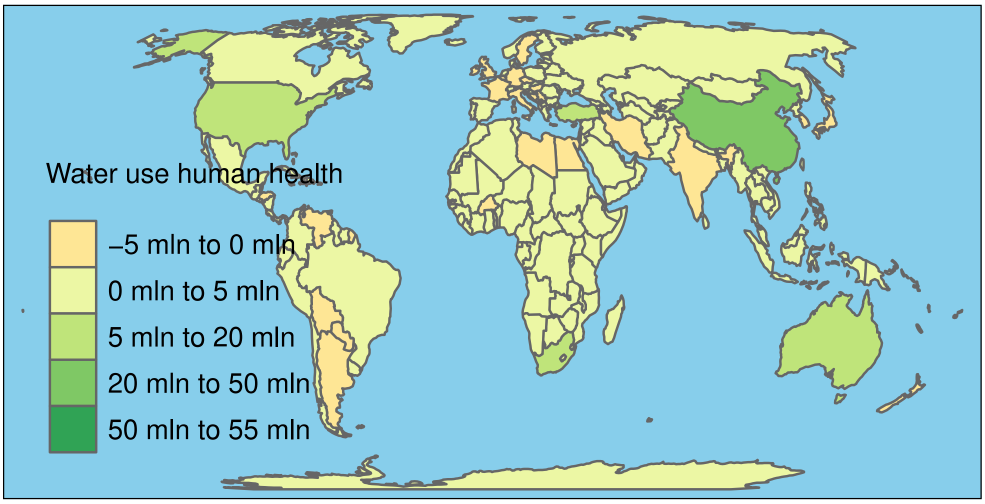


Figure S 10: Cost distribution of the impact category water use [human health] in euro [€], low estimate

In Figure S 11, the impacts of freshwater eutrophication are shown. For China, graphite, iron and coal cause the costs. For Australia and South Africa, the decisive material is zirconium, while for India it is also graphite.

Figure S 11: Cost distribution of the impact category freshwater eutrophication in euro [€], low estimate

In Figure S 12, we map the impacts caused by ionizing radiation. These damages are all dominated by uranium extraction and processing.

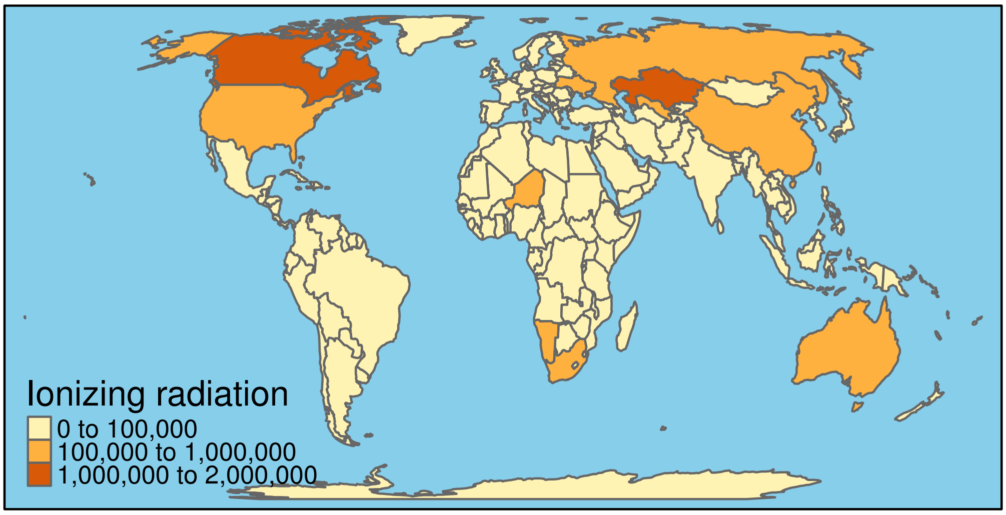


Figure S 12: Cost distribution of the impact category ionizing radiation in euro [€], low estimate

Cost distribution for domestic costs for the countries with the highest domestic costs (does not include contribution of climate change impacts) are presented for China (Figure S 13), India (Figure S 14), Russia (Figure S 15), Brazil (Figure S 16), South Africa (Figure S 17) and Australia (Figure S 18).

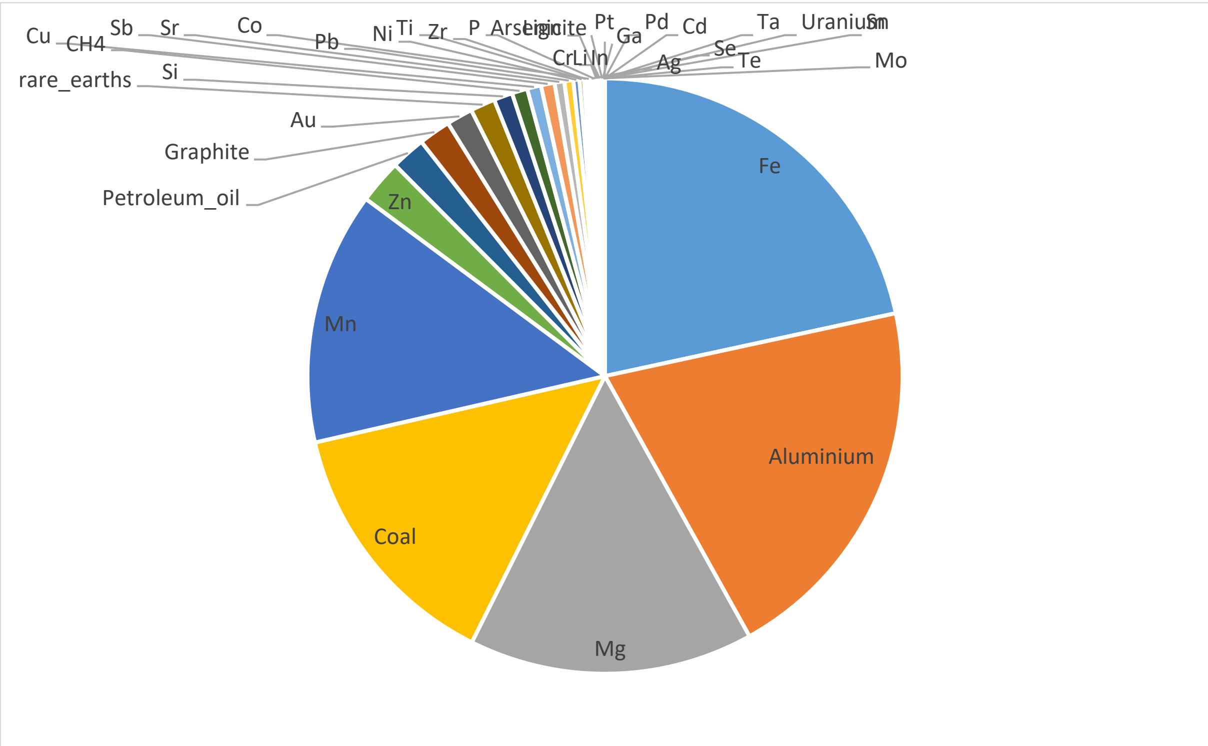


Figure S 13: Contribution of the different materials to China's domestic costs

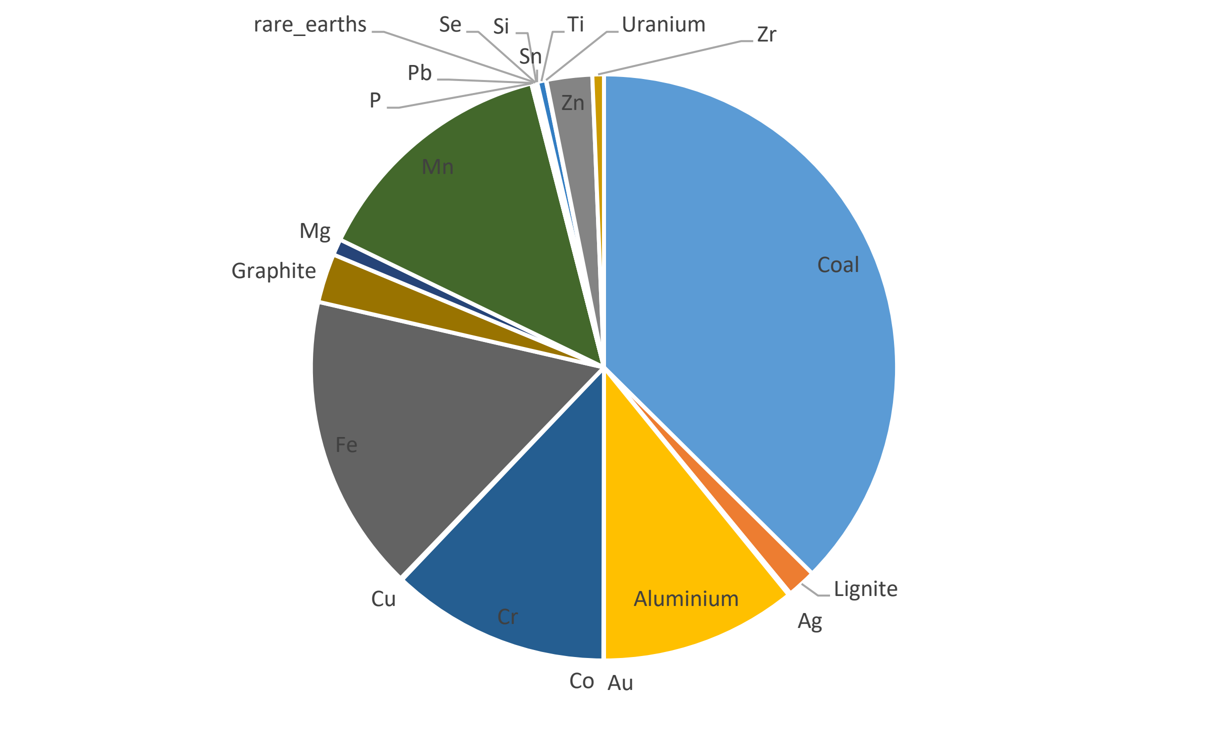


Figure S 14 Contribution of the different materials to India's domestic costs

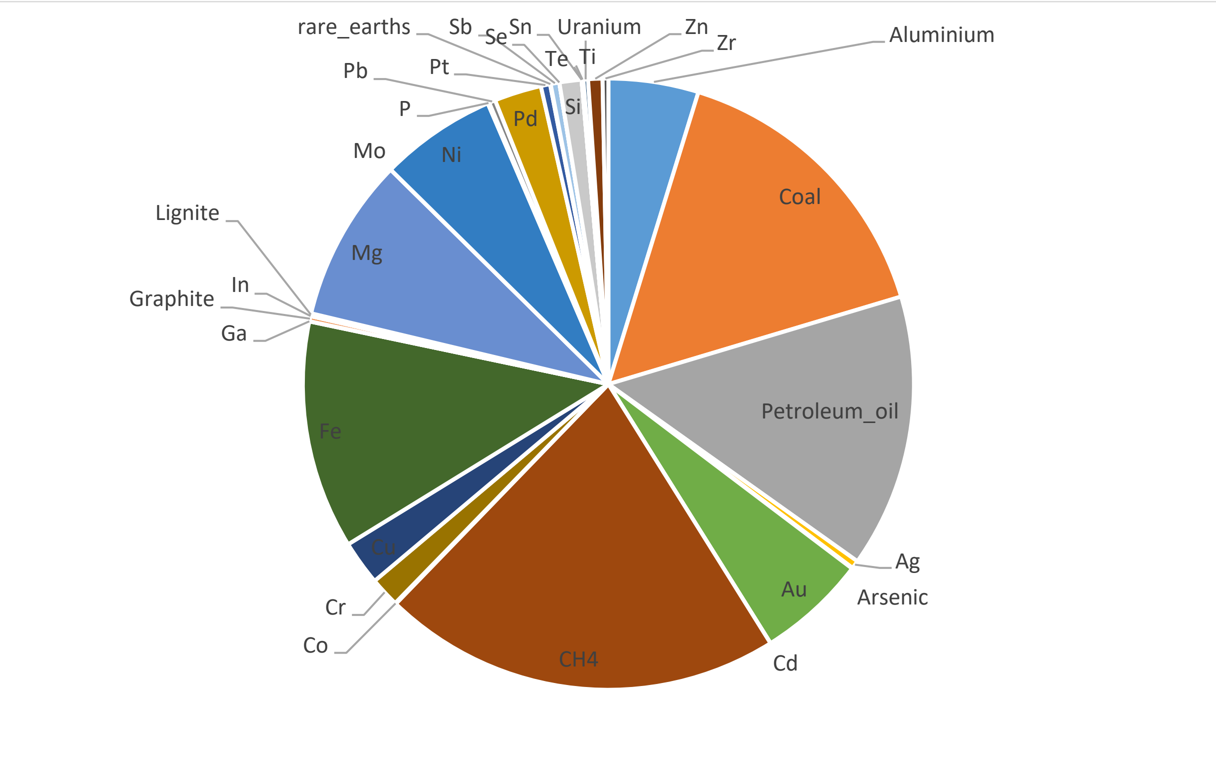


Figure S 15 Contribution of the different materials to Russia's domestic costs

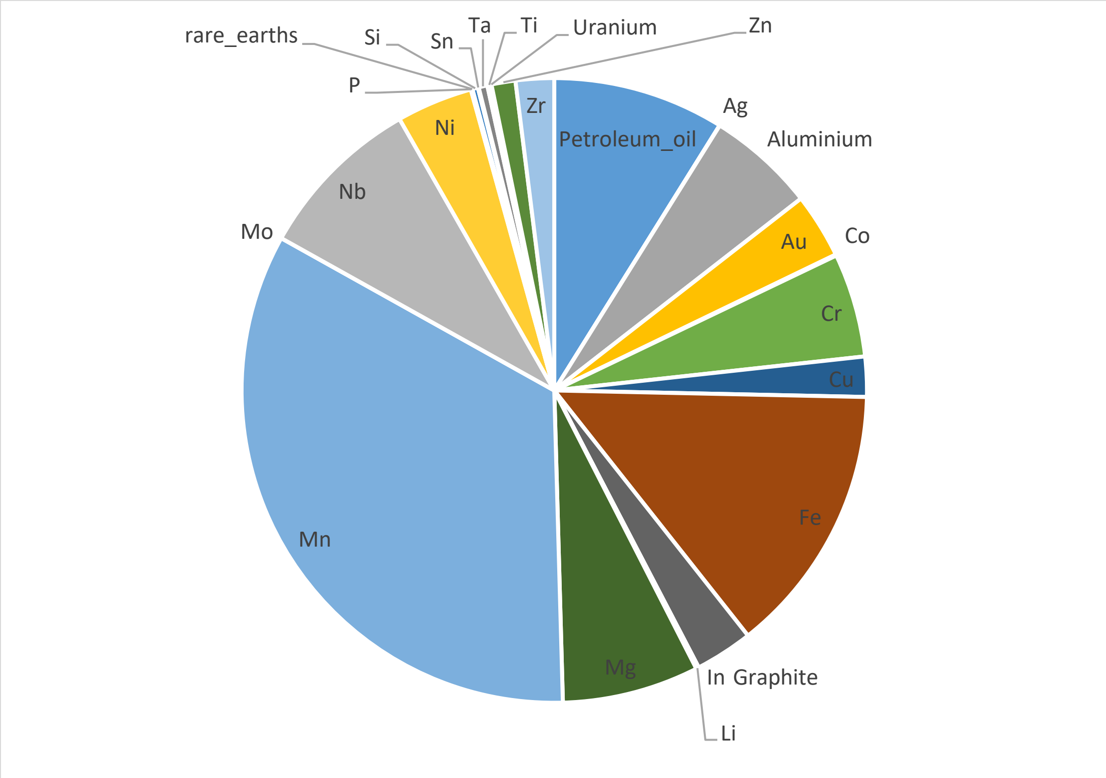


Figure S 16 Contribution of the different materials to Brazil's domestic costs

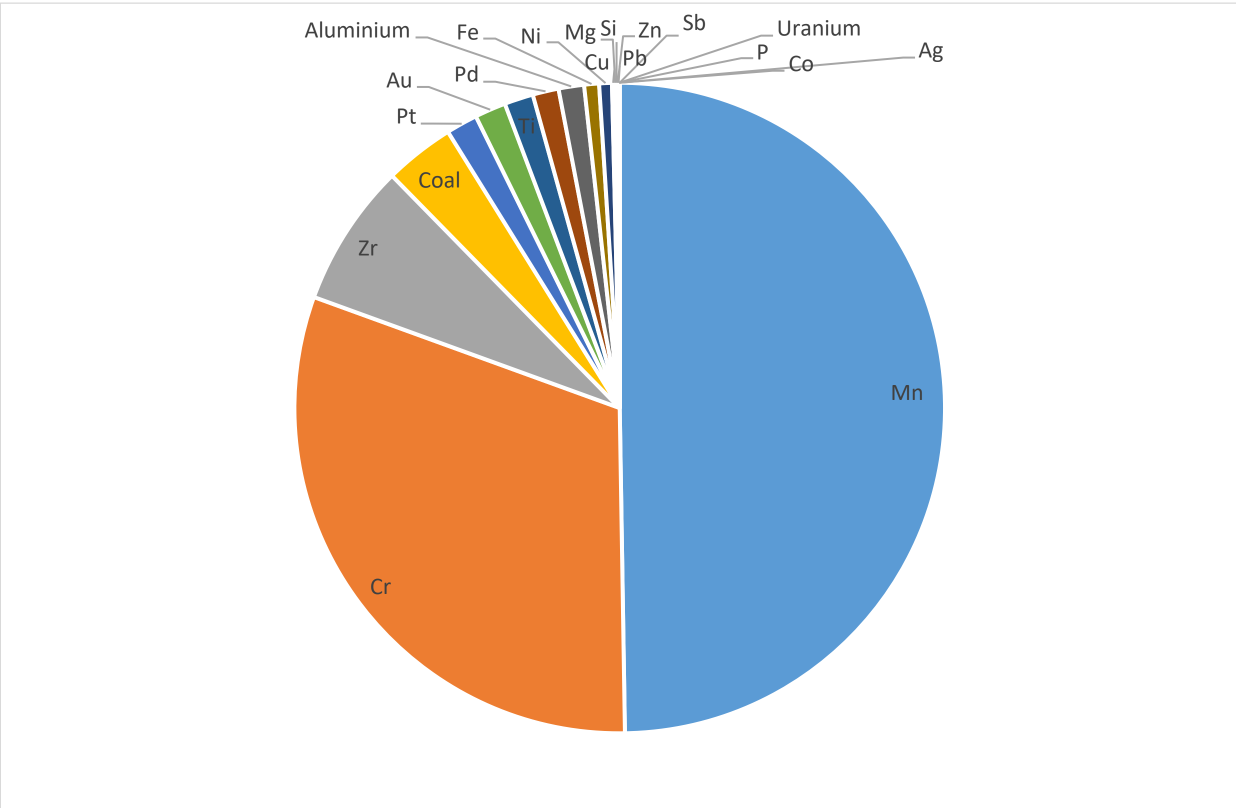


Figure S 17 Contribution of the different materials to South Africa's domestic costs

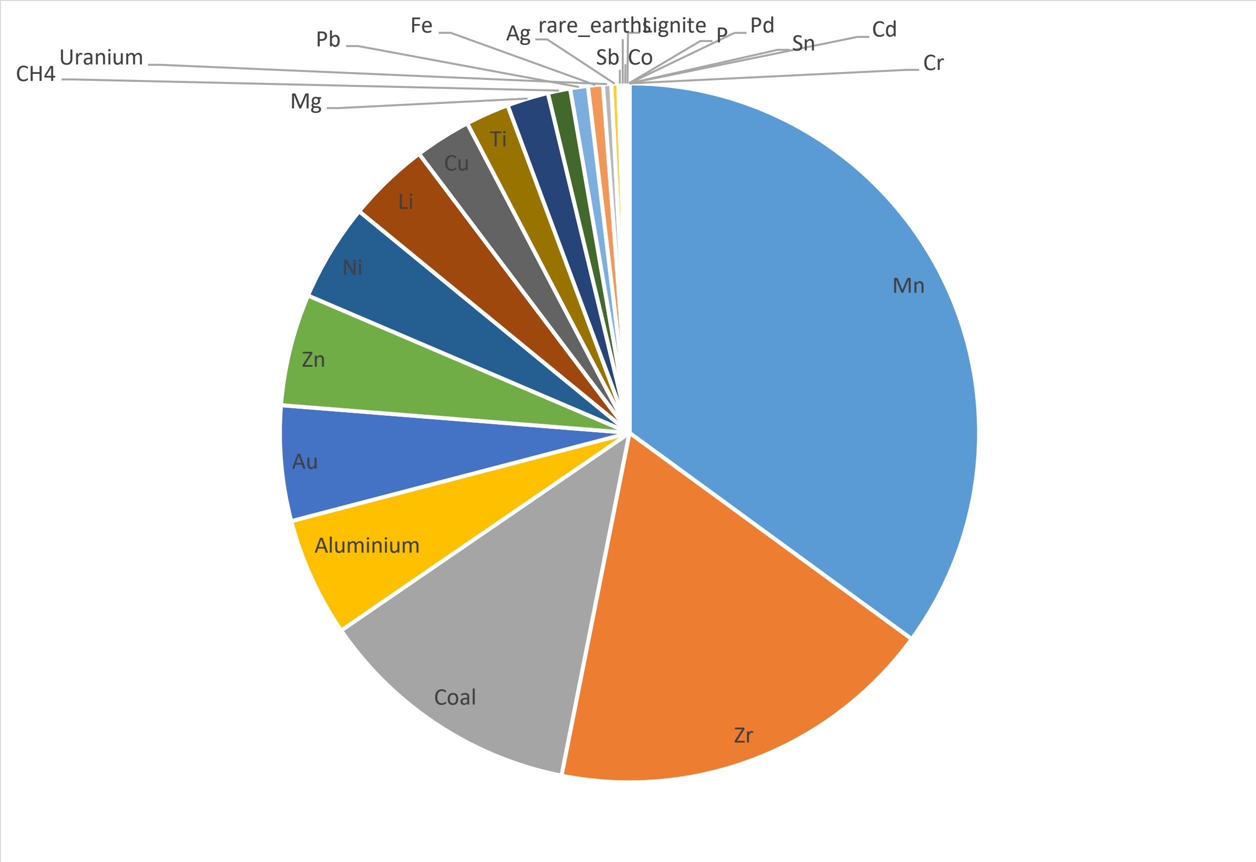


Figure S 18 Contribution of the different materials to Australia's domestic costs

4.2. Results of the high estimate

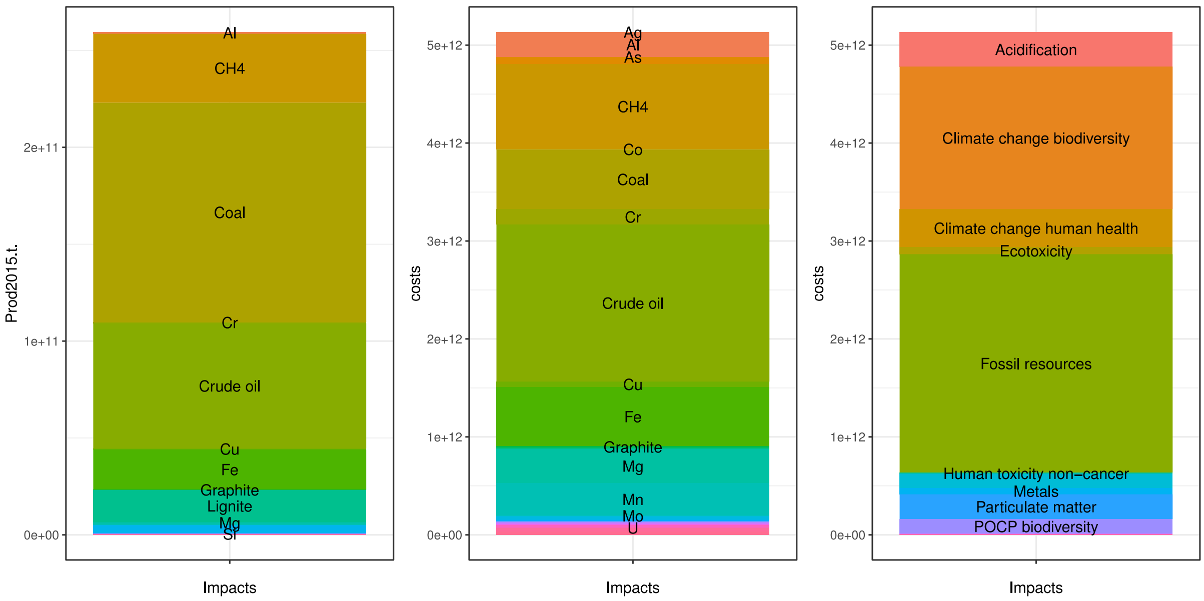


Figure S 19: mass share of materials (left) cost share of materials (middle), share of impacts (right)- high estimate

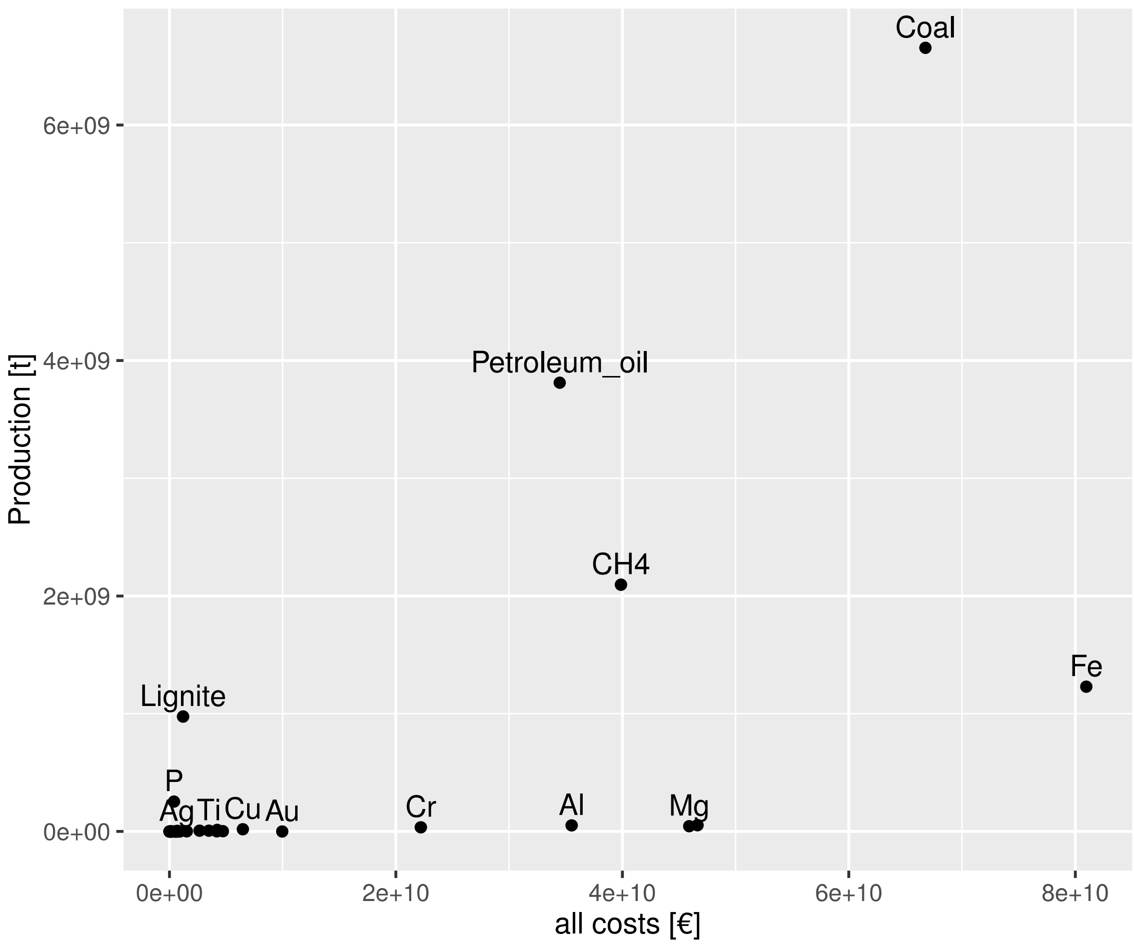


Figure S 20: Relation of environmental costs to mass for the analyzed materials, high estimate

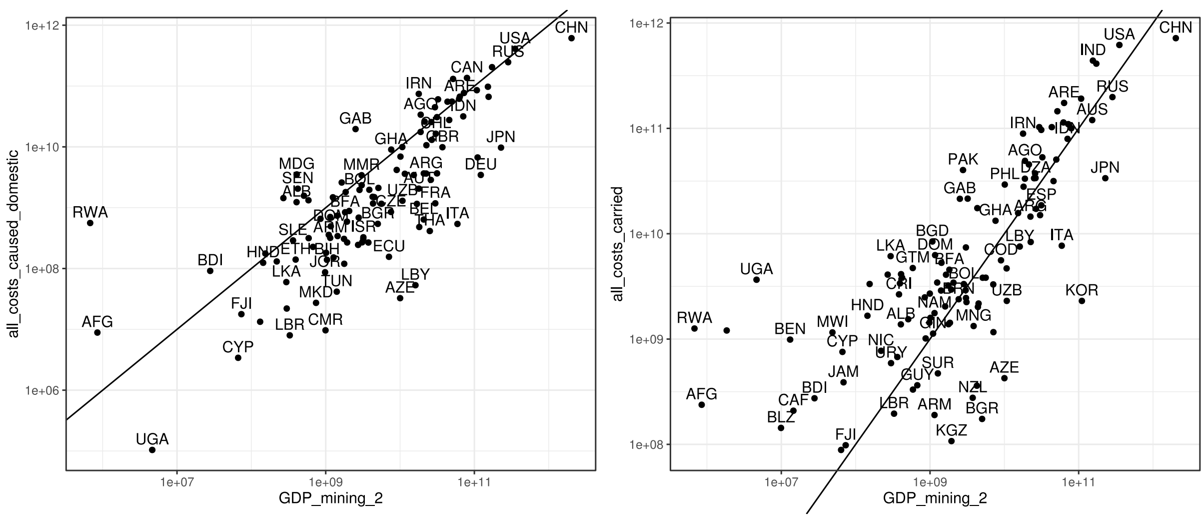


Figure S 21: GDP gain vs domestic costs (left), GDP gain vs damage costs (right), high estimate double logarithmic scale

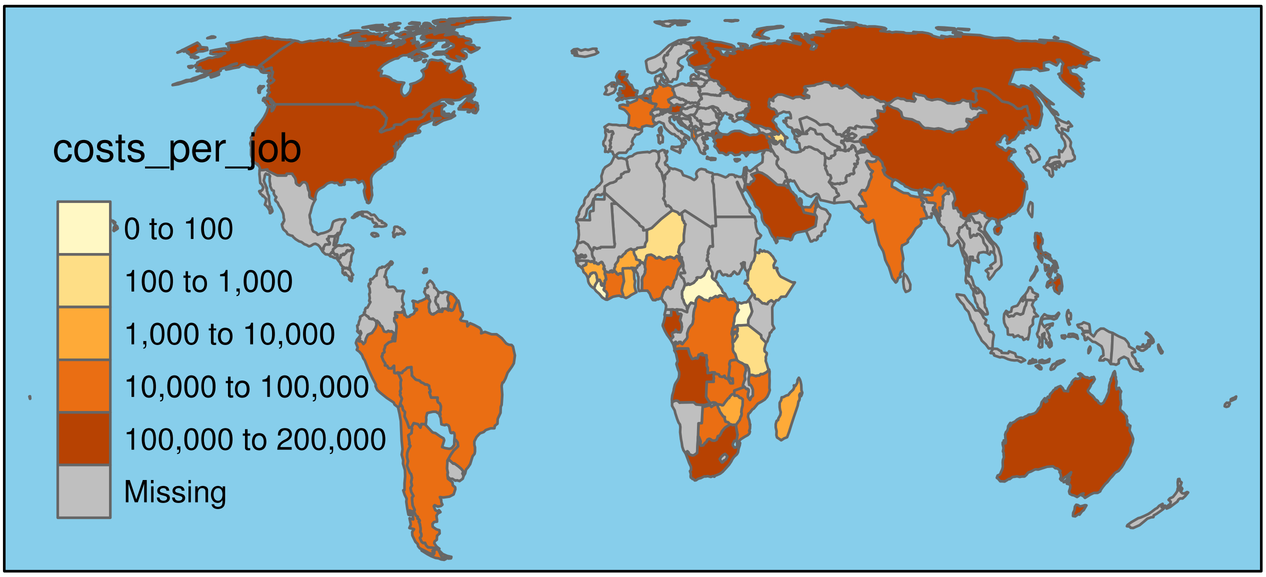
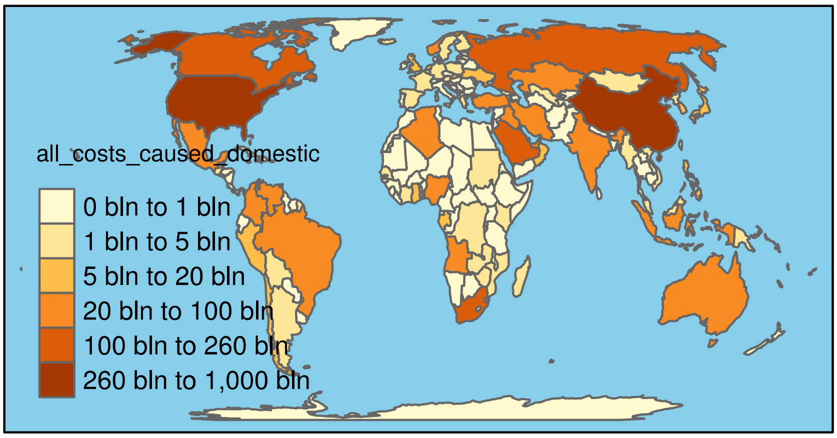
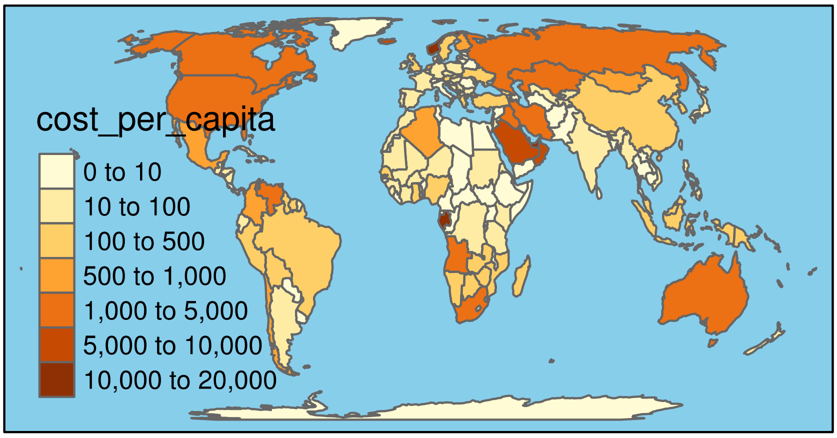
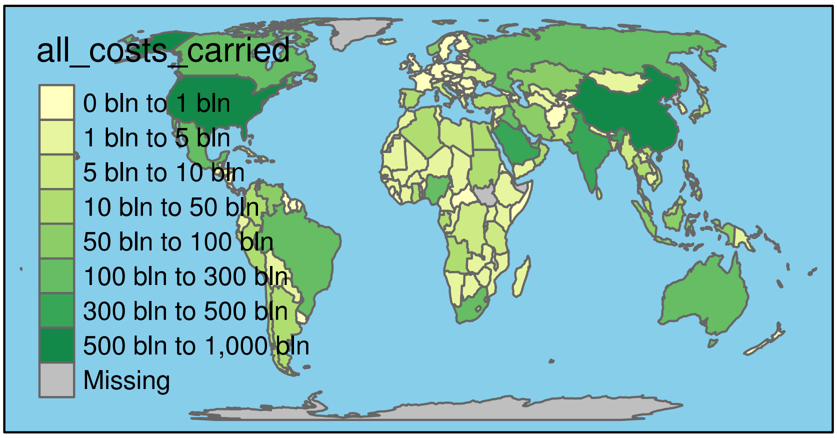


Figure S 22: Environmental costs per job in [€], high estimate


all_costs_caused_domestic
0 bln to 1 bln
1 bln to 5 bln
5 bln to 20 bln
20 bln to 100 bln
100 bln to 260 bln
260 bln to 1,000 bln


cost_per_capita
0 to 10
10 to 100
100 to 500
500 to 1,000
1,000 to 5,000
5,000 to 10,000
10,000 to 20,000


all_costs_carried
0 bln to 1 bln
1 bln to 5 bln
5 bln to 10 bln
10 bln to 50 bln
50 bln to 100 bln
100 bln to 300 bln
300 bln to 500 bln
500 bln to 1,000 bln
Missing

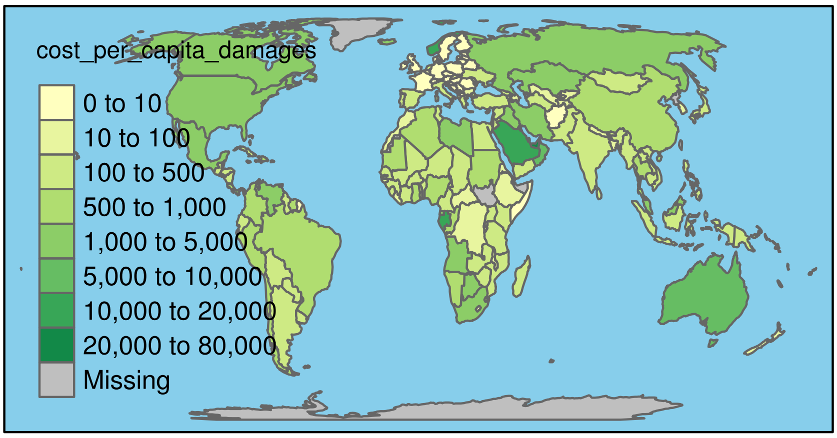

cost_per_capita_damages
0 to 10
10 to 100
100 to 500
500 to 1,000
1,000 to 5,000
5,000 to 10,000
10,000 to 20,000
20,000 to 80,000
Missing

Figure S 23: All domestic costs; all domestic costs per capita; all damage costs; all damage costs per capita (from top to bottom) for the high estimate

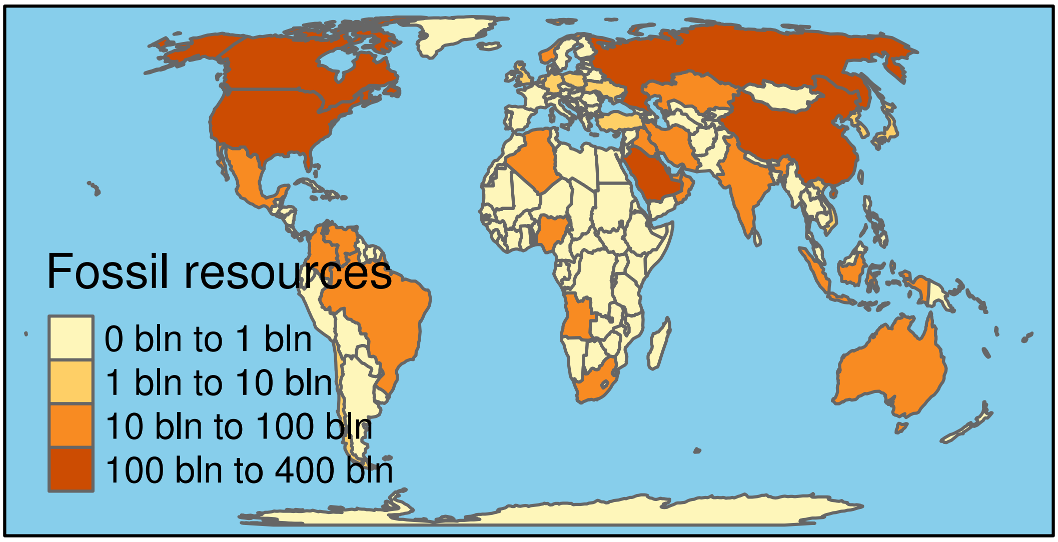


Figure S 24: costs caused by exhaustible fossil resource extraction, high estimate

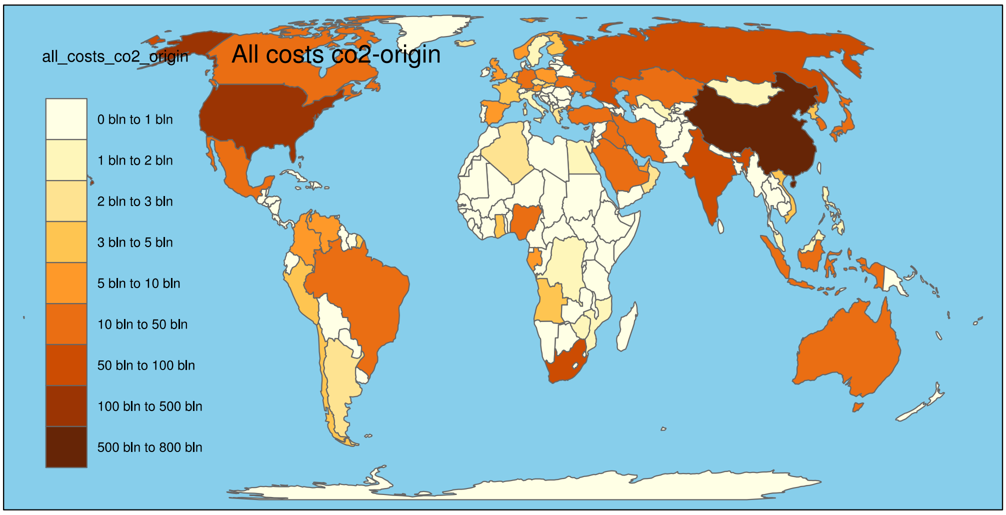


Figure S 25: All origin costs of the CO_2-emissions, high estimate

Figure S 26: damage costs caused by metal extraction, high estimate

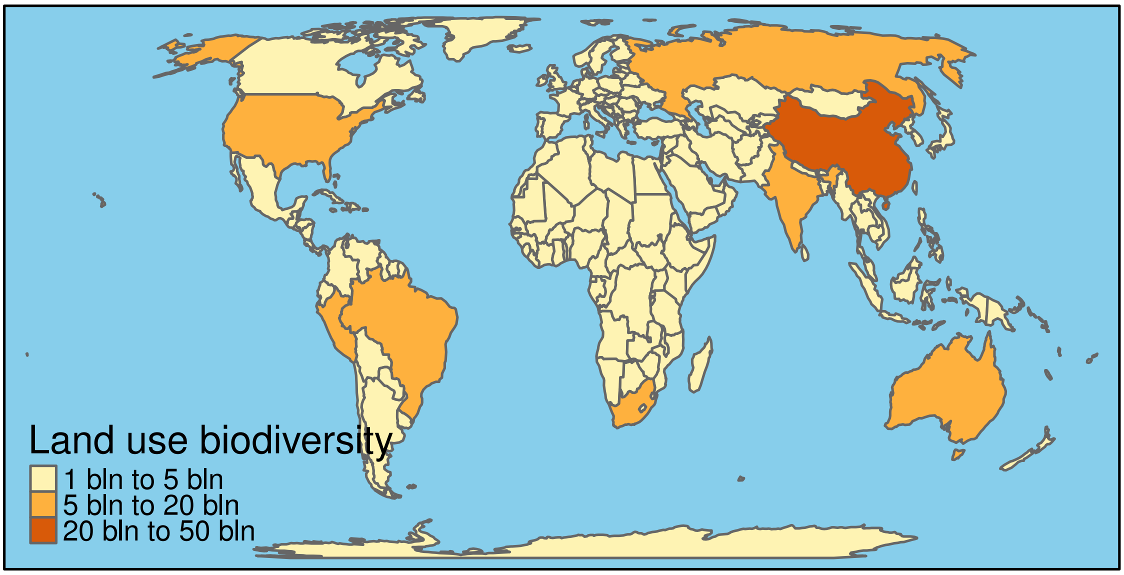


Figure S 27: damage costs caused by land use and its impact on biodiversity, high estimate

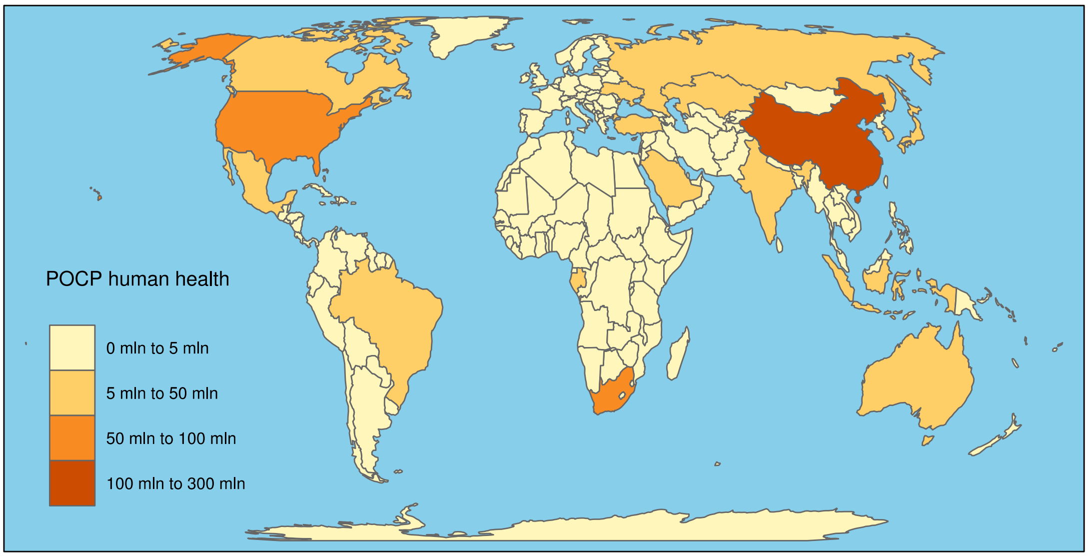


Figure S 28: damage costs caused by POCP and its impact on human health, high estimate

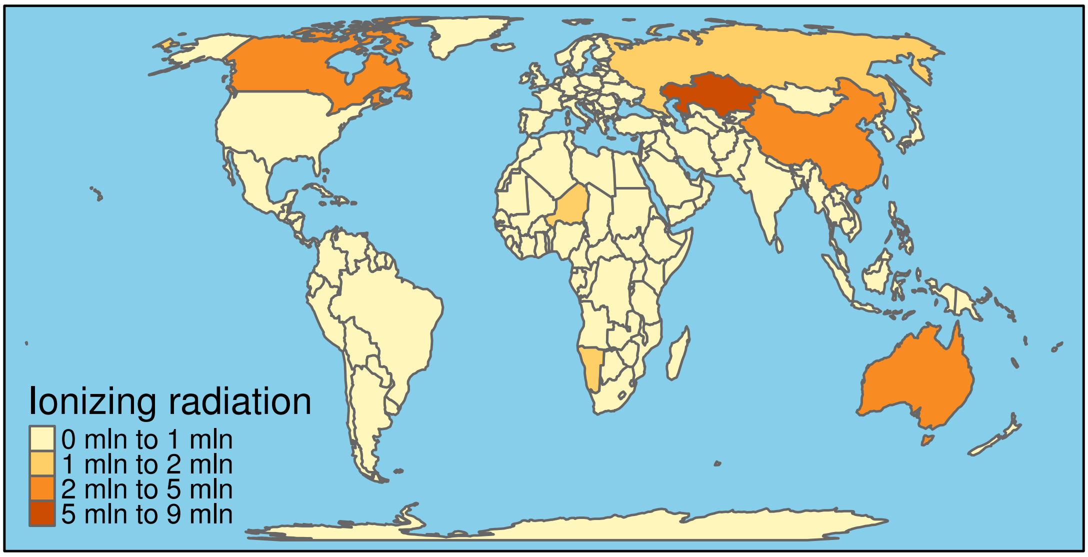


Figure S 29: Impacts from ionizing radiation in [€], high estimate

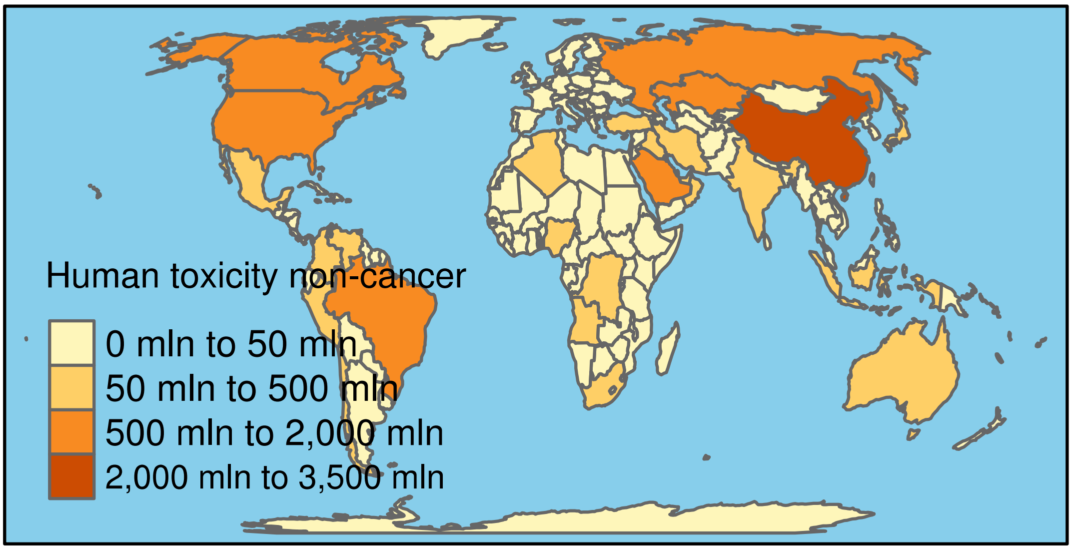


Figure S 30: costs caused by human toxicity non-cancer [€], high estimate

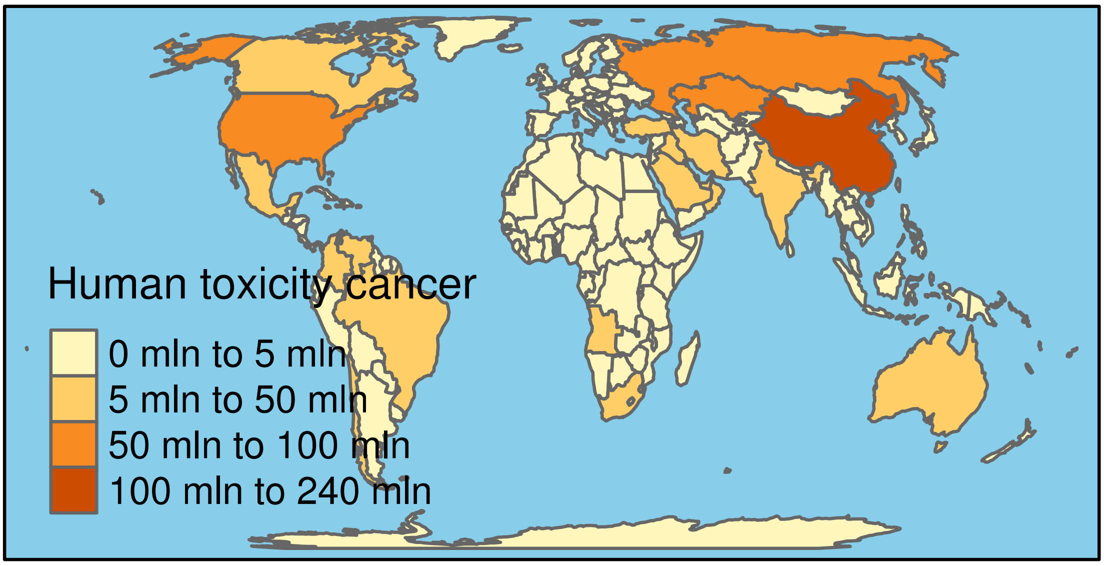


Figure S 31: Costs caused by human toxicity, cancer in [€], high estimate

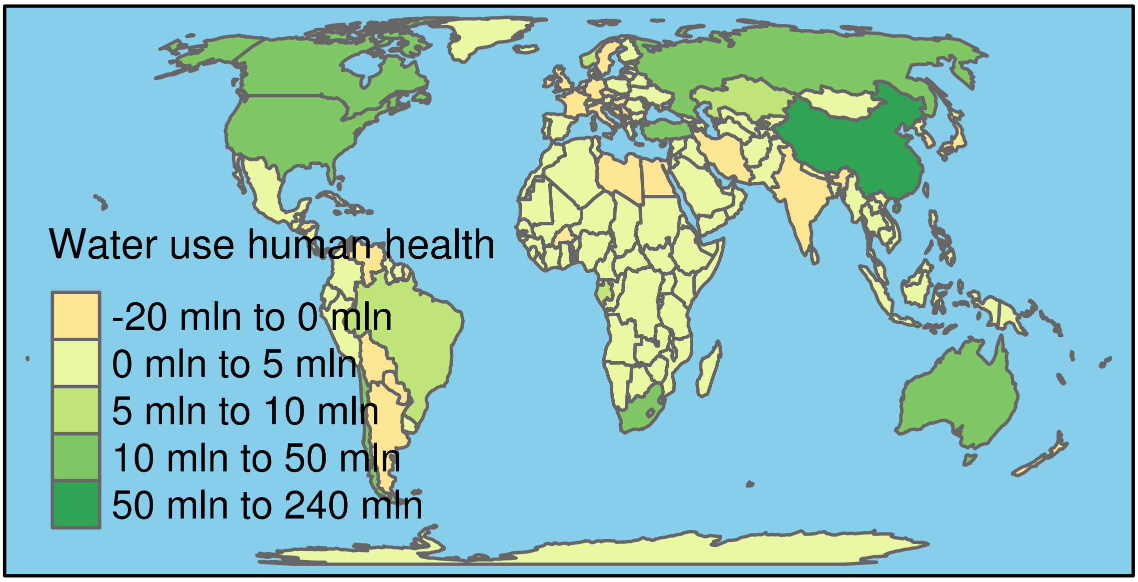


Figure S 32: costs caused by water use to human health in [€], high estimate

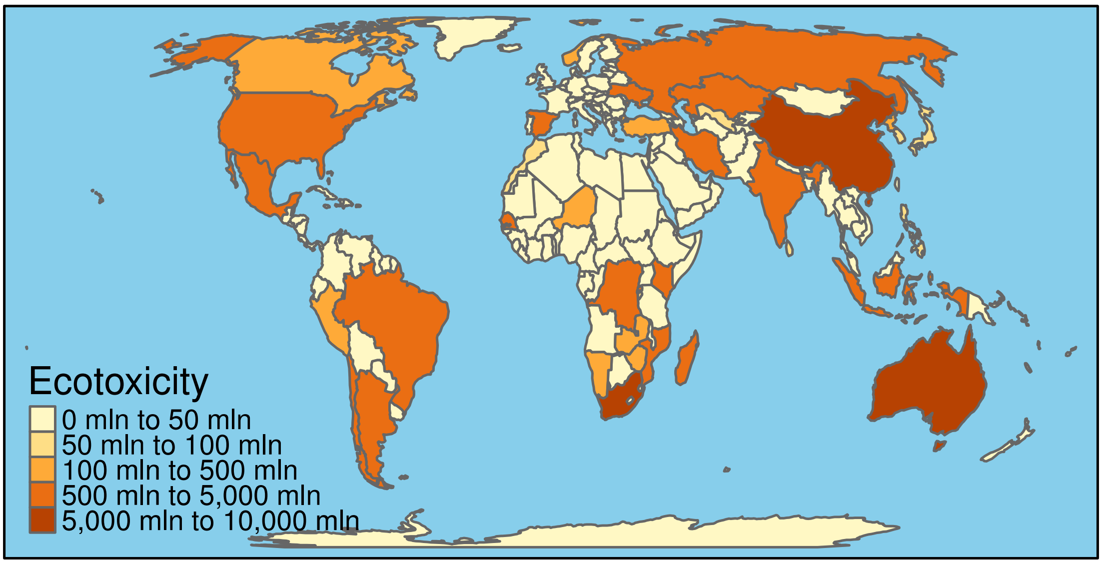


Figure S 33: Costs caused by terrisitrial ecotoxicity in [€], high estimate

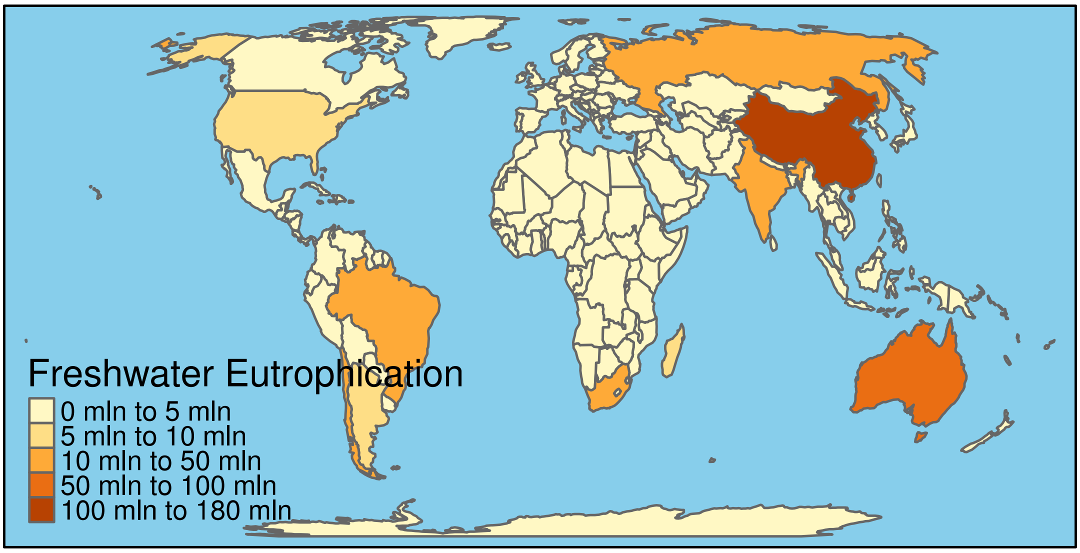


Figure S 34: costs caused by freshwater eutrophication in [€], high estimate

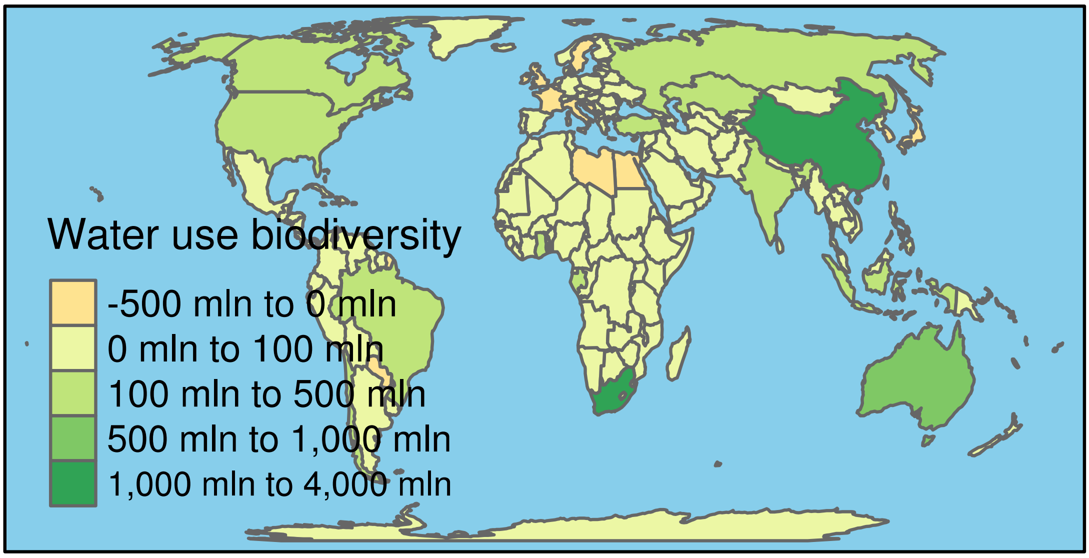


Figure S 35: costs caused by water use to biodiversity, high estimate

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Supplementary Material Paper 4

This section of the appendix contains the supplementary material of the following publication (Supplementary Material 1 and 2):

Paper 4: Arendt, Rosalie; Bach, Vanessa; Finkbeiner, Matthias: Environmental Costs of Abiotic Resource Demand for the EU's Low Carbon Development. *Resources Conservation and Recycling* **2022** 180 (May): 106057 https://doi.org/10.1016/j.resconrec.2021.106057

Supplementary Material 1 Environmental costs of abiotic resource demand for the EU's low-carbon development

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This supplementary material contains:

- Documentation of underlying processes for the LCA (section 1)
- Calculation details on the scaling for the biodiversity valuation for water use (section 2)
- The assessed low-carbon technologies, their cost shares and their associated material demands (section 3)
- A plot that shows the relation of costs to mass per material (section 4)
- A figure that shows the origins of the carbon emissions associated with mining and a map that shows the origin of the abiotic depletion potential costs (section 5)
- The visualization of all 17 scenarios that have been assessed, next to the medium, non-recycling scenario and 0 % discount rate that has been presented in the main paper (section 6)

1. Documentation of LCA Processes

The underlying process and origination databases are shown in the following table:

Table S 11: underlying processes to quantify impacts from LCA databases

Material	Process	Database
Silver	GLO: Silver mix agg ts Metal production from electrolysis consumption mix, to consumer solid, density: 10.49 g·cm−3,	GaBi
Graphite	GLO market for graphite, battery grade agg ecoinvent 3.5 2011:Manufacture of basic chemicals 05.03.2019	Ecoinvent
Cobalt	GLO Cobalt, refined (metal) agg CDI CDI 01.01.2020 hydro- and pyrometallurgical processes production mix, at plant 99% Co	GaBi

Chromium(Ferrochromium)	ZA Ferro chrome agg ts Metal production 05.03.2019 primary production production mix, at plant 60 % chrome, high carbon ~ 6%	GaBi
Copper	GLO Copper mix (99,999% from electrolysis) agg ts Metal production 21.08.2020 from electrolysis consumption mix, to consumer 99,999% Cu	GaBi
Iron/steel	DE Steel cold rolled coil 2,5 mm t-agg ts Metal production 21.08.2020 BF route production mix, at plant2,5 mm	GaBi
Gallium	GLO market for gallium, semiconductor-grade agg ecoinvent 3.5 2011:Manufacture of basic chemicals 01.11.2018	Ecoinvent
Indium	GLO market for indium agg ecoinvent 3.5 2011:Manufacture of basic chemicals 05.03.2019	Ecoinvent
Lithium	GLO market for lithium agg ecoinvent 3.5 2011:Manufacture of basic chemicals 05.03.2019	Ecoinvent
Manganese	ZA: Manganese ts [Metal production] production mix, at plant 99,9% concentrated	GaBi
Molybdenum	GLO: Ferromolybdenum (FeMo)	IMOA
Nickel	GLO Nickel mix agg ts Metal production 21.08.2020 ore mining and processing, roasting, reduction, magnetic separation	GaBi

	production mix, at plant 99.9% Nickel	
Platinum	GLO Platinum mix agg ts Metal production 21.08.2020 primary production production mix, at plant solid, density. 21,45 g/cm3 (20 °C)	GaBi
Selenium	GLO market for selenium agg ecoinvent 3.5 2011:Manufacture of basic chemicals 05.03.2019	Ecoinvent
Rare earths	RoW rare earth oxides production from bastnäsite concentrate agg ecoinvent 3.5 2011:Manufacture of basic chemicals 05.03.2019	Ecoinvent
Silicon	GLO: Ferro silicon mix (90% Si) ts	GaBi/sphera
Te	GLO market for tellurium, semiconductor-grade agg ecoinvent 3.5 2011:Manufacture of basic chemicals 05.03.2019	Ecoinvent
Zn (Zinc)	RNA Redistilled zinc agg ts Metal production 21.08.2020 ore mining and beneficiation, electrolysis, remelting production mix, at plant 7.14 g/cm3, 65.41 g/mol	GaBi

2. Calculation Detail for methods: Impact Assessment and calculation of Water Biodiversity Impacts

Table S 12: Sources of impact assessment method and derived cost factors

Impact category	Impact assessment method	Monetization	Monetization factor units
Climate change	IPCC (Stocker et al., 2013)	(Ricke et al., 2018)	€/t carbon (vector of countries)
Abiotic resources	Based on (Huppertz et al., 2019), and (UNIDO, 2020)		€/kg material
Water use, human health	(Motoshita et al., 2018)	(European Commission, 2009)	€/m³

Water use, economic	(Ligthart and van Harmelen, 2019) scaled with the water depletion index (WDI) (Berger et al., 2018)		€/m³
Water use, biodiversity	Pfister et al. (2009)	(Kuik et al., 2008) as used in (Bruyn et al., 2018) adjusted with country factors of (Winter et al., 2019, 2018) and (Lindner et al., 2019)	€/species*yr
Land use	Area*time	(de Groot et al., 2012) and (de Groot et al., 2020a, 2020b) , biomes based on (WWF, 2012)	€/m²*yr

We explain here the calculation details of the method section, that refers to step 4 and the details of scaling the species*year value to monetize impact of water use on biodiversity. The species year value from Kuik et al (2008) was used, as for the upper estimate in de Bruyn et al (2018). The value from de Bruyn et al (2018) was transformed to the global species year value through dividing by the global species density for terrestrial species (Goedkoop et al., 2008) (equation S1).

$$a_{global}\left[\frac{€}{species \cdot yr_{global}}\right] = b\left[\frac{€}{yr \cdot m^2}\right] \cdot c^{-1}\left[\frac{species}{m^2}\right] \tag{S1}$$

Where a_{global} stands for the global monetary value of species*yr, that is obtained by dividing *b* (the monetary value of affecting a species on a respective square meter for one year) by *c* that stands for the global species density. The country weighted Ecoregionfactor (EF) were used to scale the values. The EF by Lindner et al (2019) was scaled on a country level according to equation S2, where $EF_{i,c}$ stands for the ecoregion factor of ecoregion *i* in country *c* and $q_{i,c}$ stands for the percent share of ecoregion *i* in country *c*.

$$EF_c = \sum_{i}^{n} EF_{i,c} \cdot q_{i,\,c} \tag{S2}$$

This EF_c is used to scale the species year value for different countries. We assume that the ecoregion factor would correlate with higher species density, and an according effect on the associated potential disappeared fraction due to water use. The species year value for each country a_c is derived by multiplying the global species year value a_{global} obtained from equation S1 with the ration of the ecoregion factor for country *c* (obtained from equation S2) with the average ecoregion factor shown in equation S3.

$$a_c\left[\frac{€}{species \cdot yr_c}\right] = \frac{EF_c}{EF_{av}} \cdot a_{global}\left[\frac{€}{species \cdot yr_{global}}\right] \tag{S3}$$

The obtained species year values per country are supplied in the supplementary material 2 and used for the monetization of water use impacts on ecosystems.

3. Technologies and shares of costs

Table S 13: Renewable energy technologies and shares of total costs

Technologies	Share of costs [%]	Materials needed [t] (based on Bobba et al. (2020) medium demand)
Electro-mobility batteries	45.8	110000 cobalt 1800000 graphite 130000 lithium 83000 manganese 1100000 nickel
Electro mobility –traction motors	22.6	420000 aluminium 260000 copper 2900000 iron 5500 rare earths 140000 silicon
PV modules	11.7	110 silver 410000 aluminum 30 cadmium 250000 copper 5 gallium 10 germanium 20 indium 30 selenium 109000 silicon 40 tellurium
Wind turbines	9.4	63000 aluminum 32000 chromium 150000 copper 1200000 iron 48000 manganese 21000 nickel 3830 rare earths 330000 zinc
Fuel cells (platinum only) for mobility	1.8	30 platinum
Renewable energy batteries	8.4	44000 cobalt 290000 graphite 28000 lithium 41000 manganese 97000 nickel

Fuel cells (platinum only) for renewable energy	0.1	2 platinum

4. Relation of mass to costs

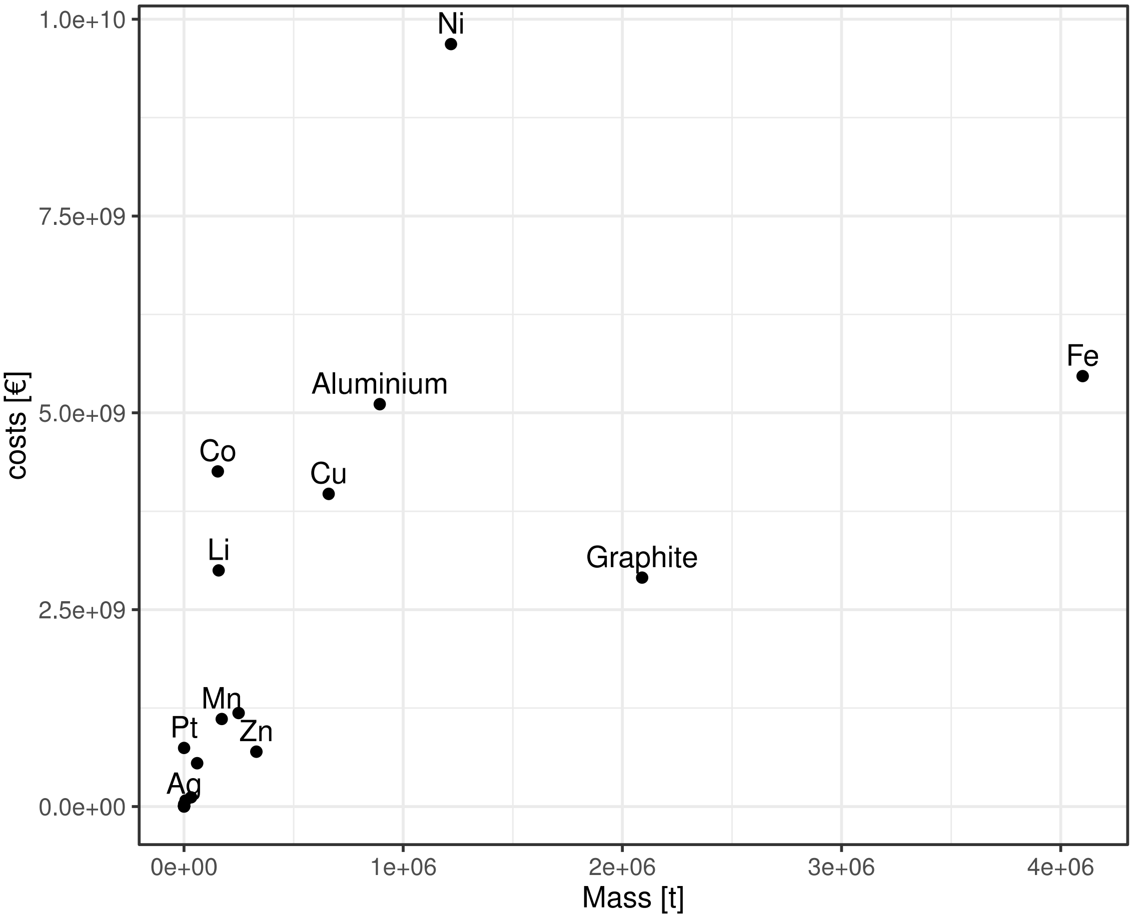


Figure S 36: relation of imported mass and caused costs for the assessed materials

5. Origins of GHG emissions and material depletion costs

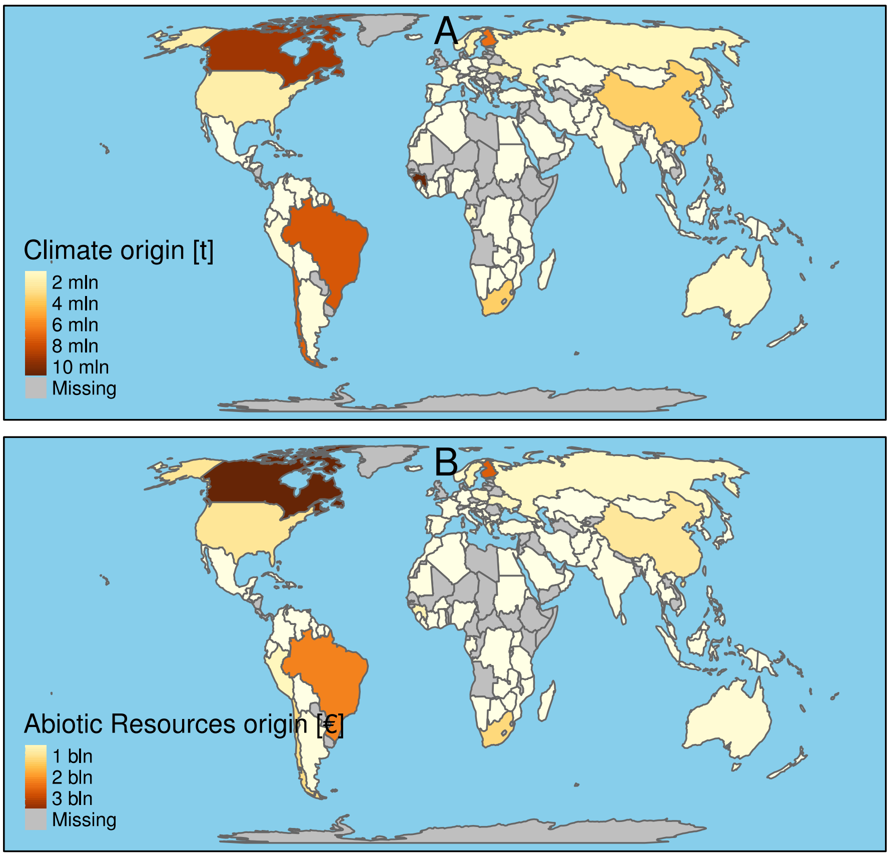


Figure S 37: Origins of GHG-emissions (panel A) and origins of environmental costs due to material depletion (B)

6. Scenarios

The varying parameters are the projected material demands (medium, high and low demand), whether recycling was considered to fulfill the demand and whether the damages, that will only accrue in 2050 were discounted with the NPV method (with 0 %, 1 % and 3 % respectively). The highest scenario maximum demand, no recycling and no discounting has the highest associated costs with 86 billion € annually in 2050, while the lowest scenario is minimum demand recycling and 3 % discount rate with around 6.7 billion € annually. The import mix is not varied.

6.1 Medium Demand

6.1.1. No Recycling

6.1.1.1. 1 % discount rate

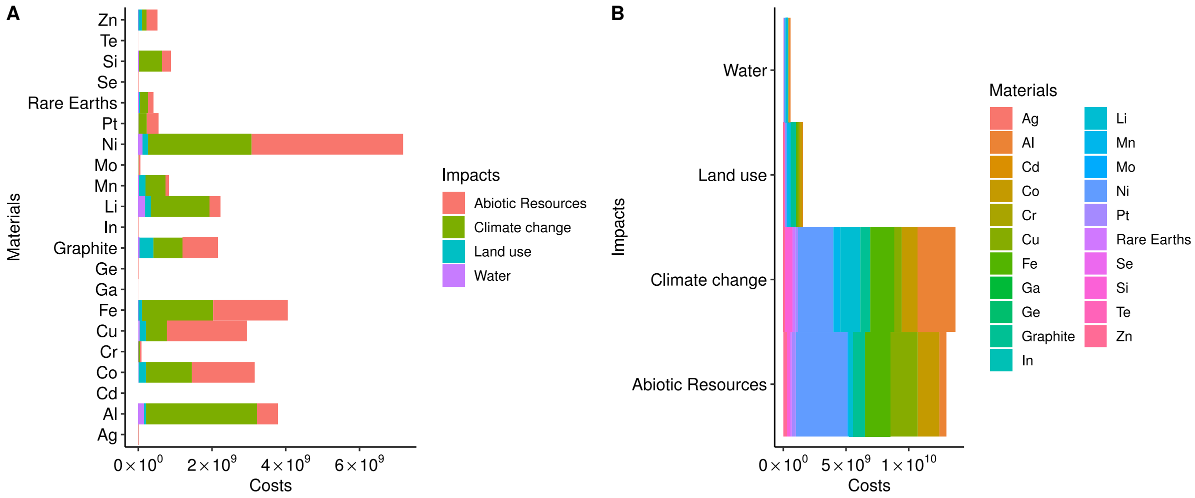
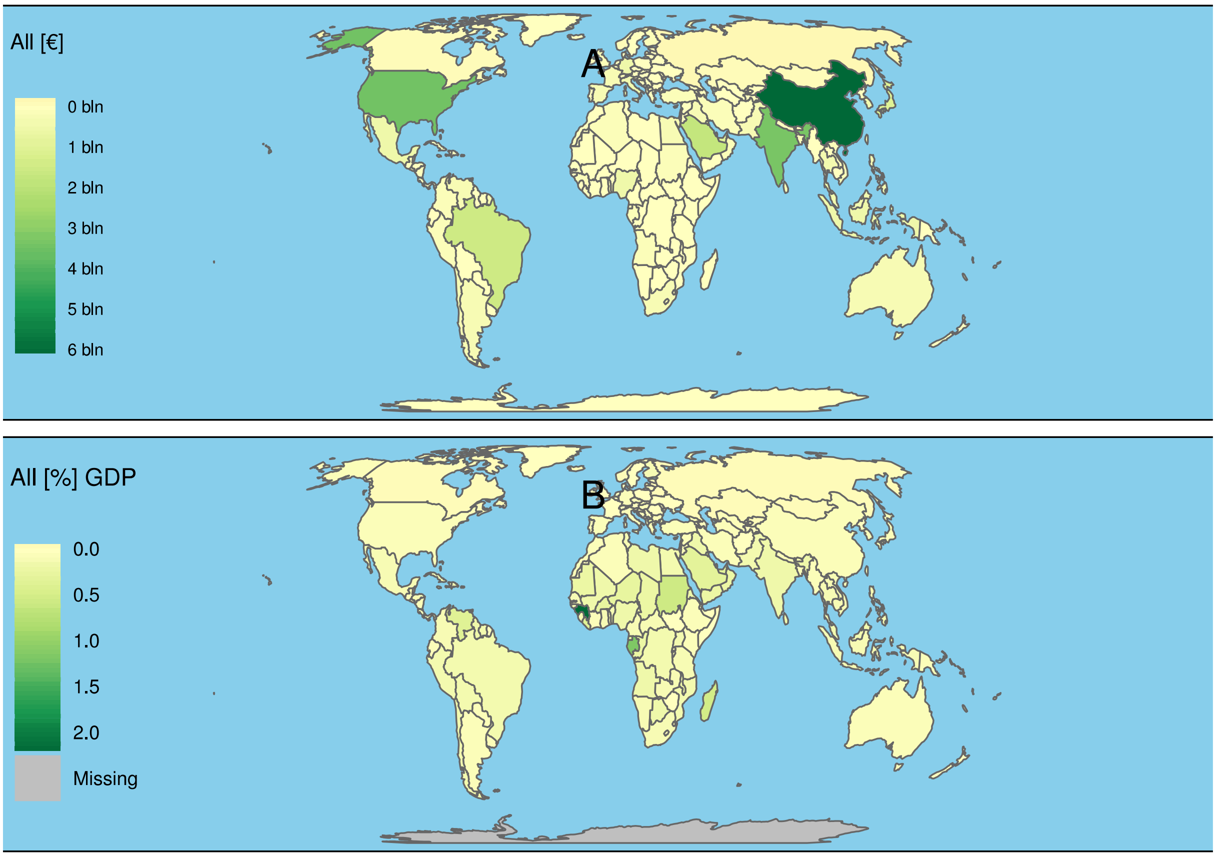


Figure S 38 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for medium demand scenario and no recycling and 1% discount rate

Figure S 39 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for medium demand scenario and no recycling and 1% discount rate



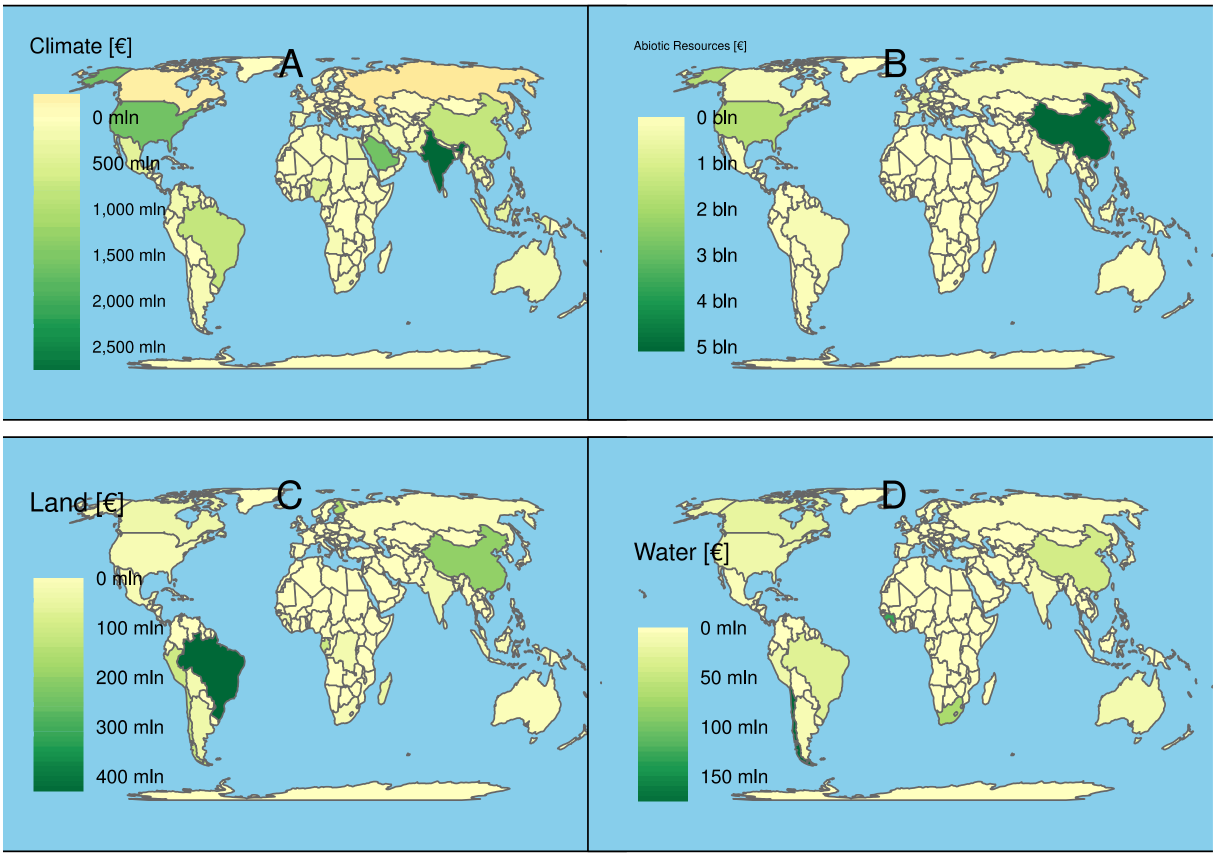


Figure S 40 Damages due to climate change, resource depletion, land use and water use for medium demand scenario and no recycling and 1% discount rate

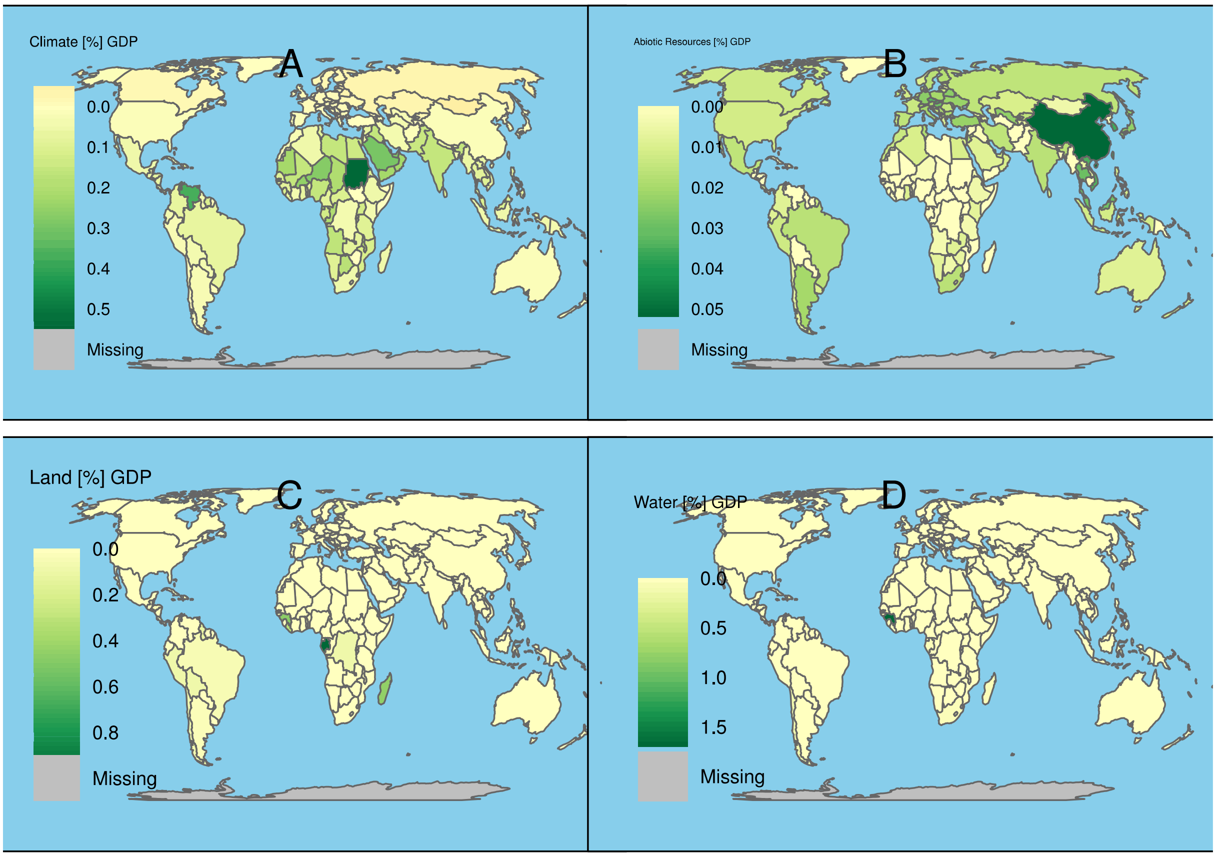


Figure S 41: Damages related to GDP for climate change, abiotic resource depletion. land use and water use for medium demand scenario and no recycling and 1% discount rate

6.1.1.2. 3 % discount rate

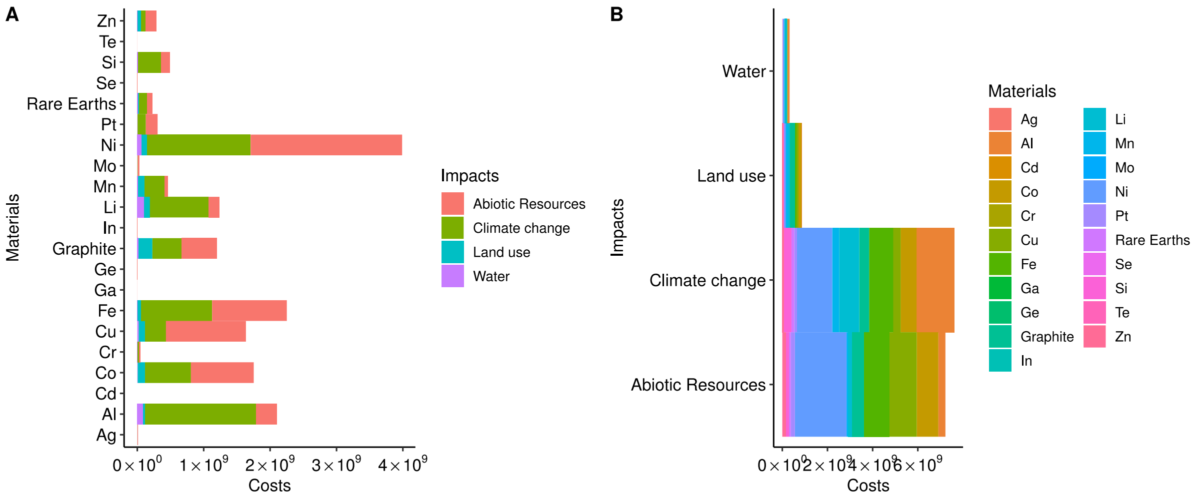


Figure S 42 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for medium demand scenario and no recycling and 3% discount rate

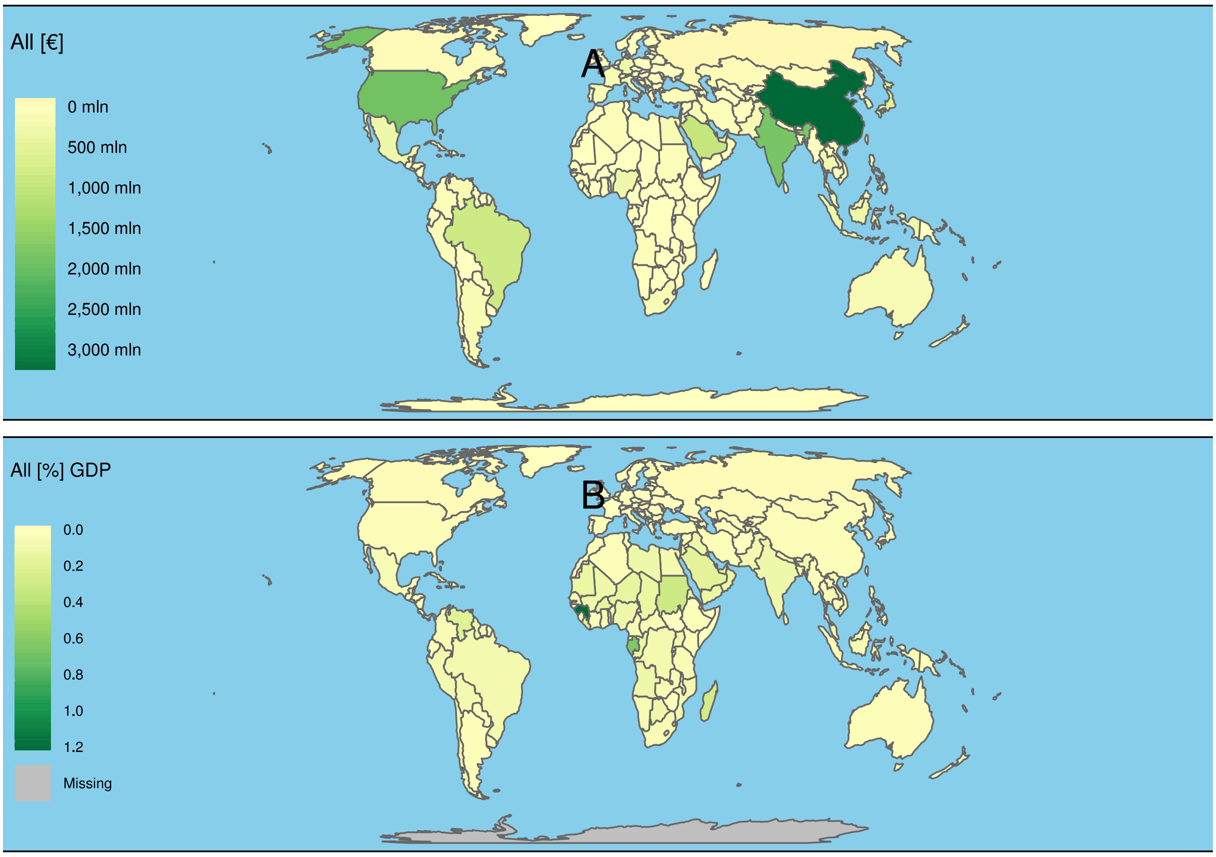


Figure S 43 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for medium demand scenario and no recycling and 3% discount rate

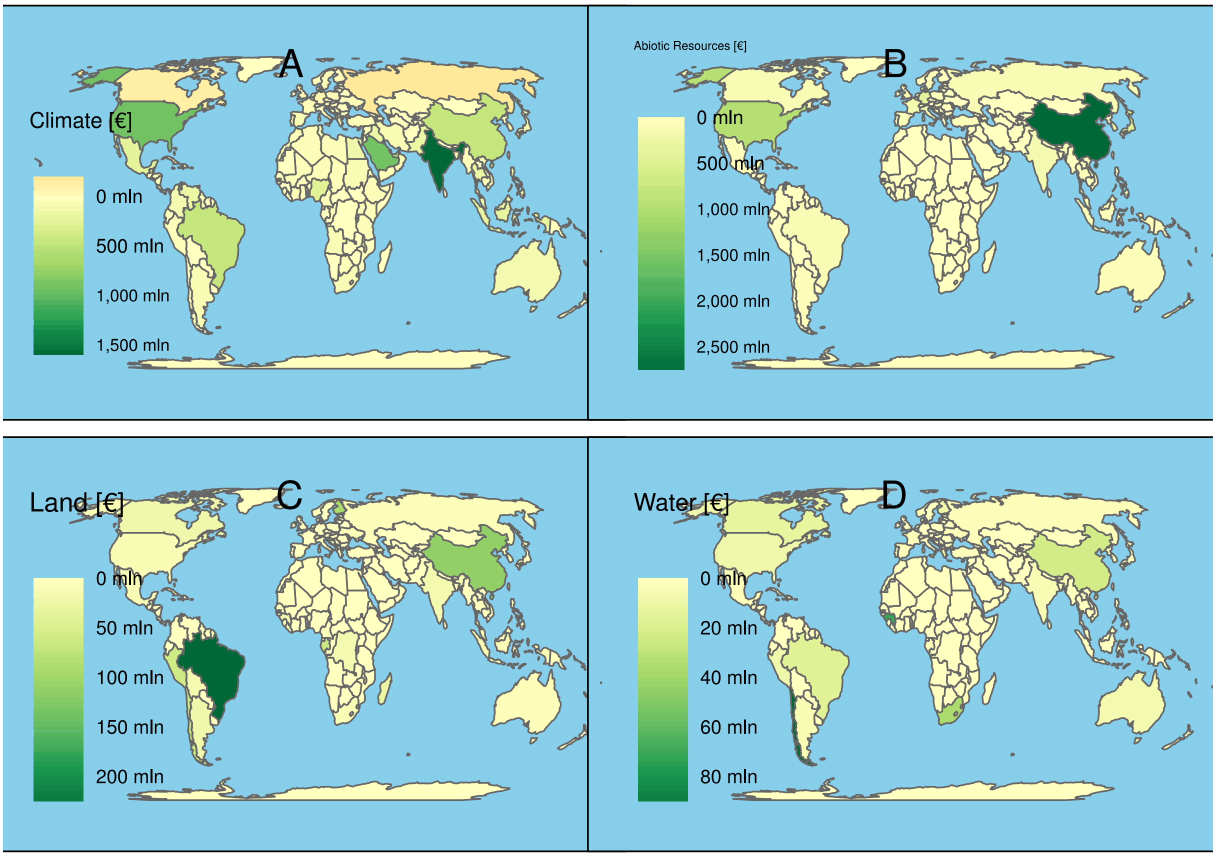


Figure S 44 Damages due to climate change, resource depletion, land use and water use for medium demand scenario and no recycling and 3% discount rate

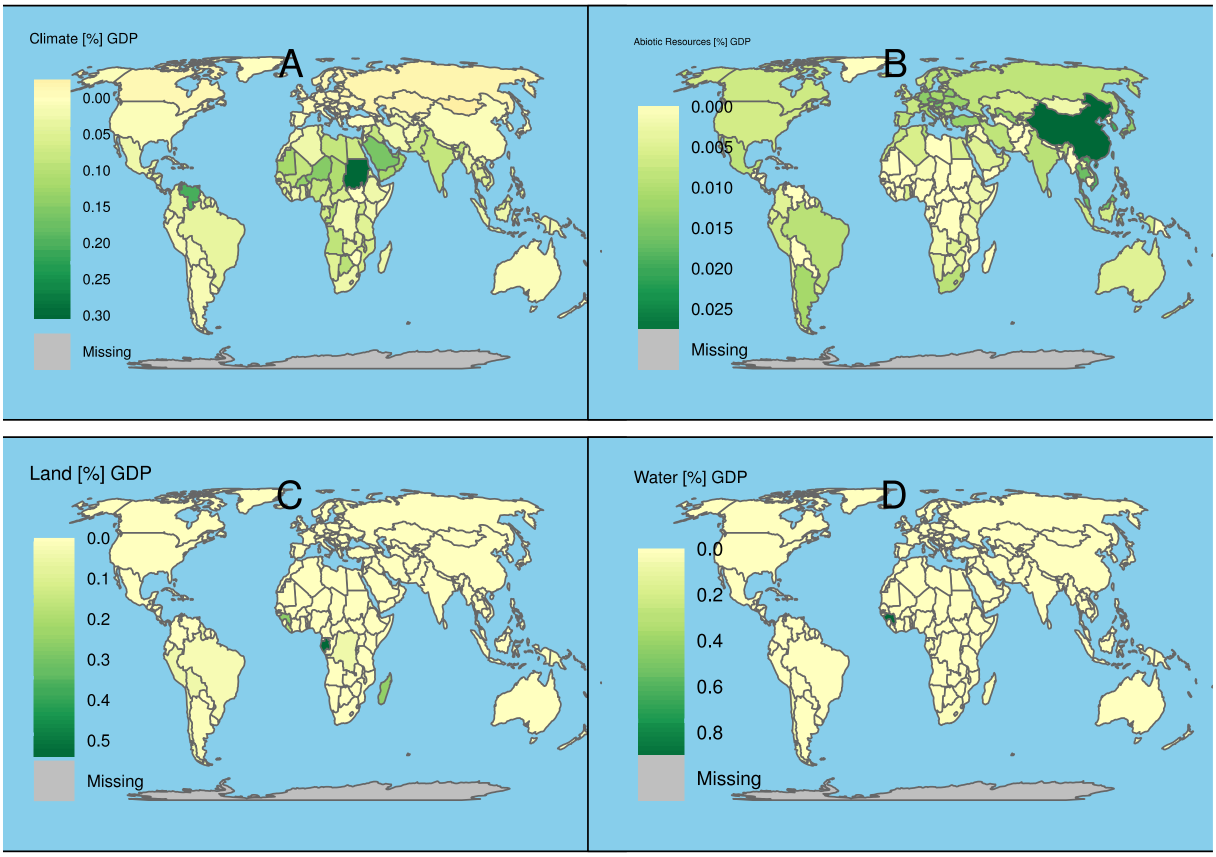


Figure S 45 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for medium demand scenario and no recycling and 3% discount rate

6.1.2. Recycling

6.1.2.1. No discounting

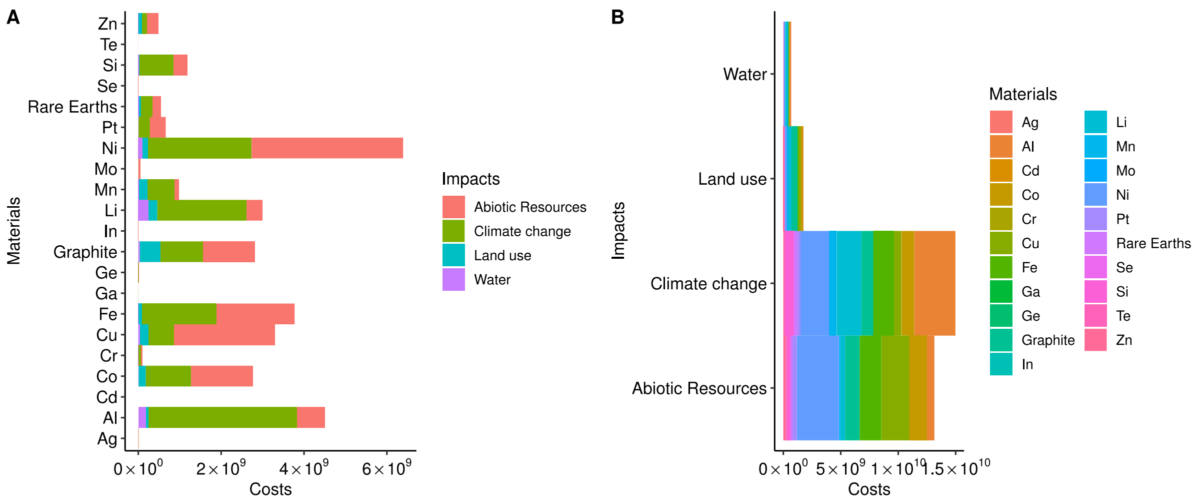


Figure S 46 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for medium demand scenario and recycling and no discounting

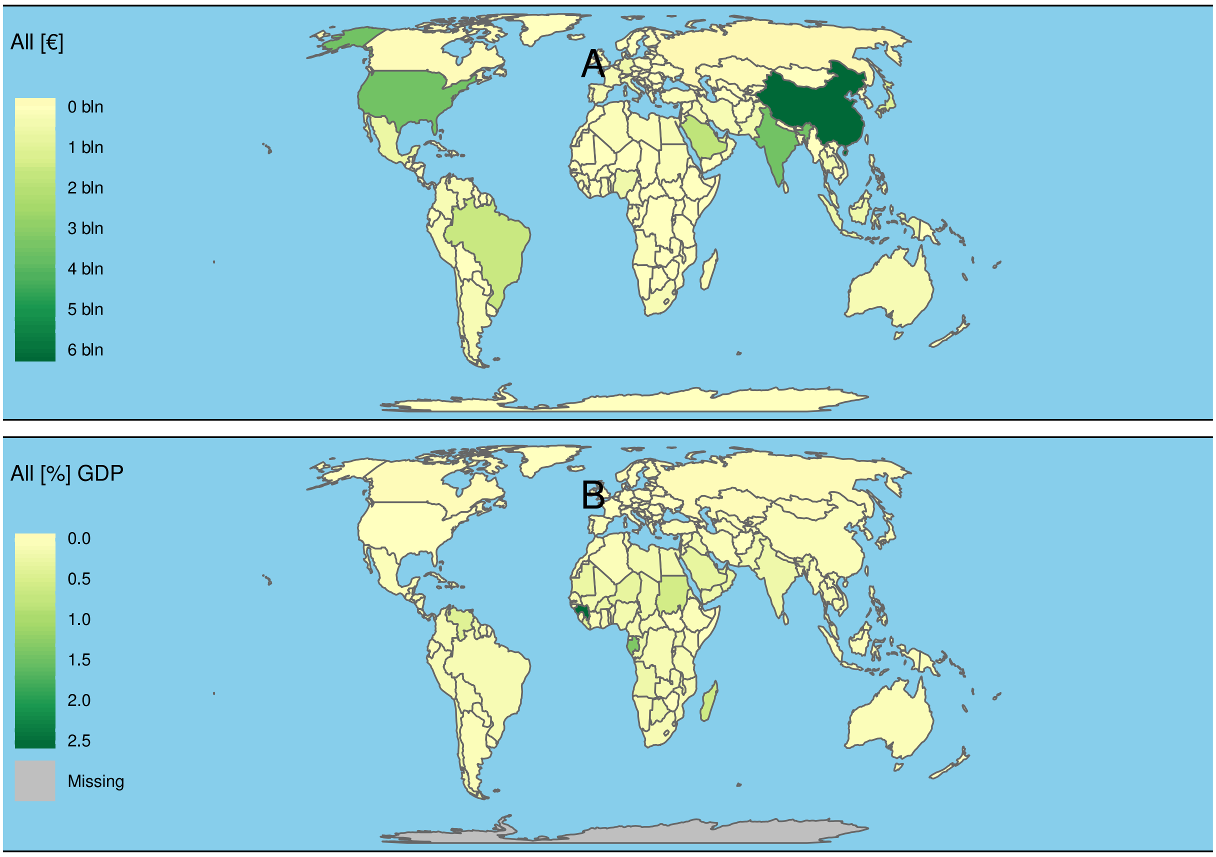


Figure S 47 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for medium demand scenario and recycling and no discounting

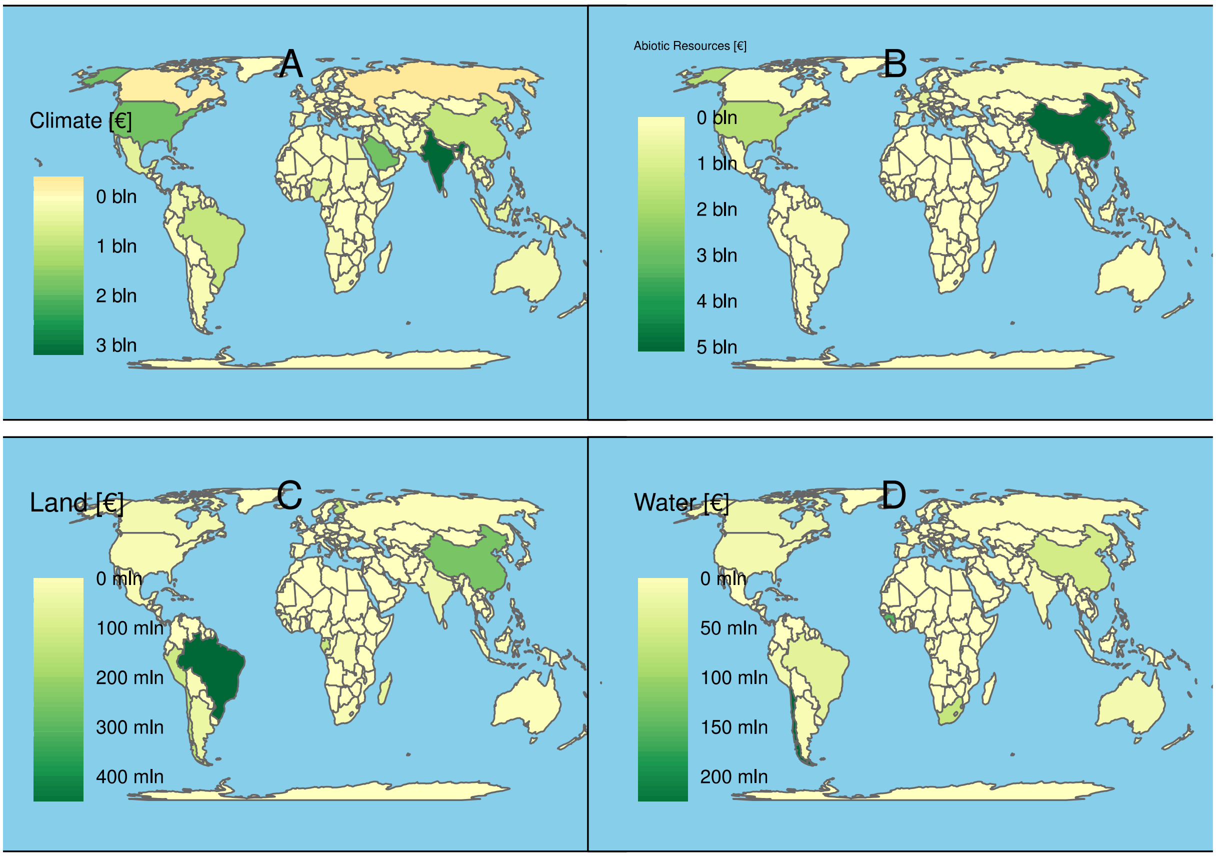


Figure S 48 Damages due to climate change, resource depletion, land use and water use for medium demand scenario and recycling and no discounting

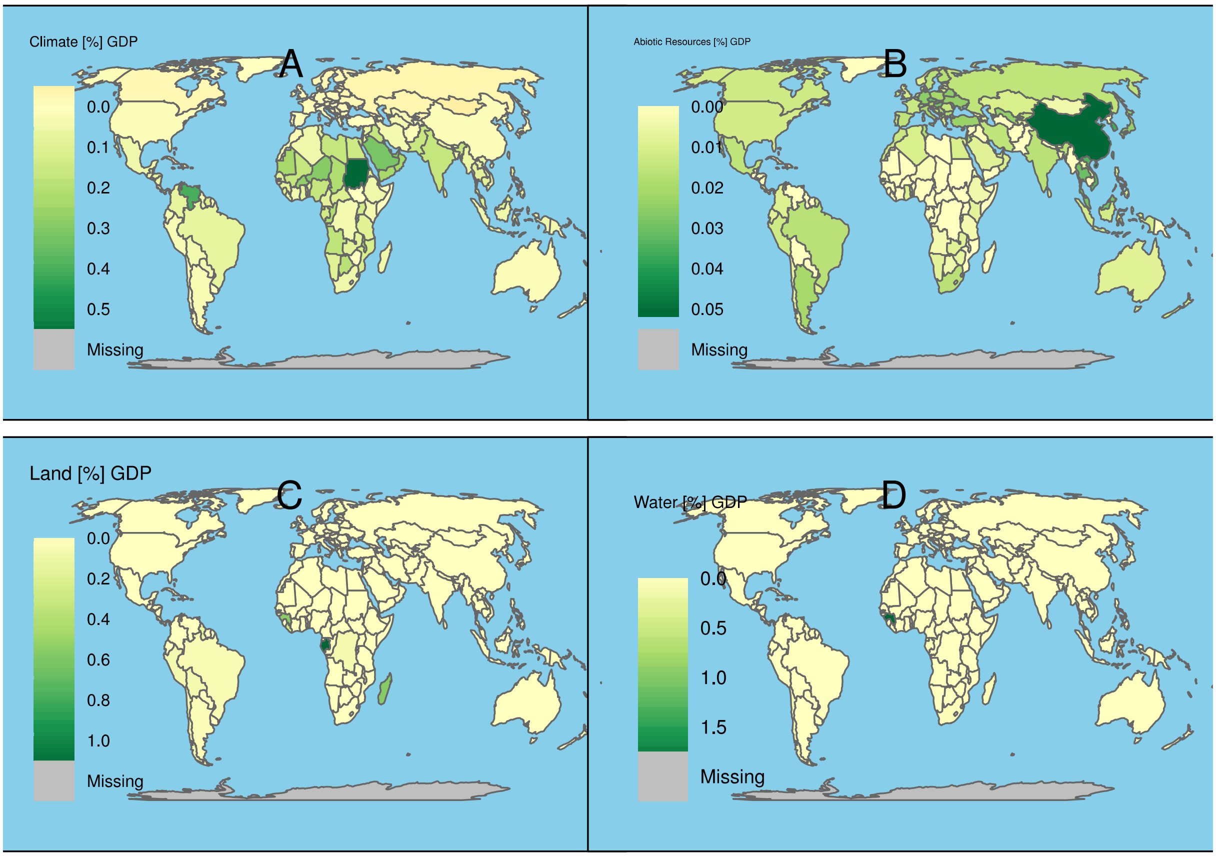


Figure S 49 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for medium demand scenario and recycling and no discounting

6.1.2.2. *1% discount rate*

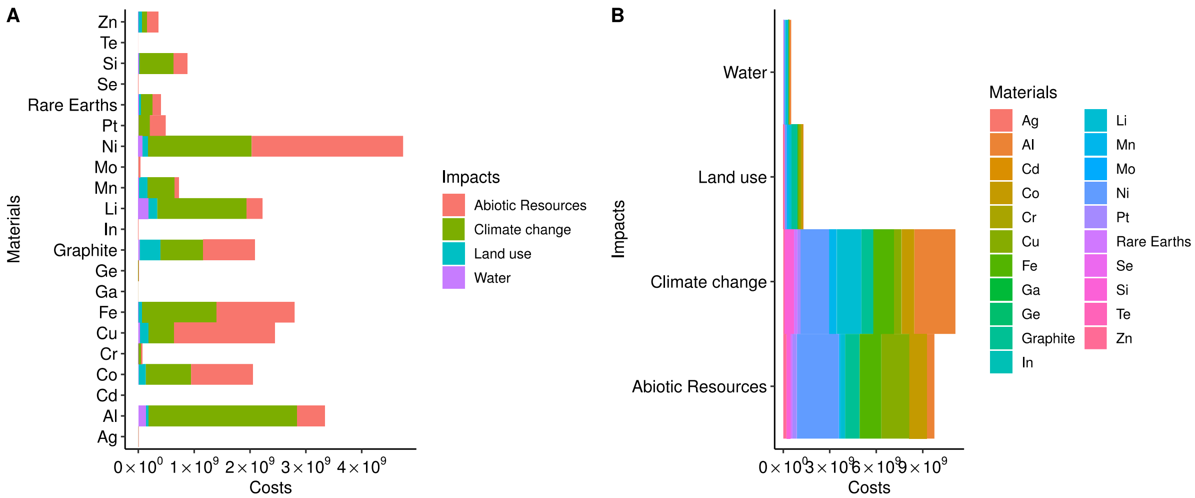


Figure S 50 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for medium demand scenario and recycling and 1% discount rate

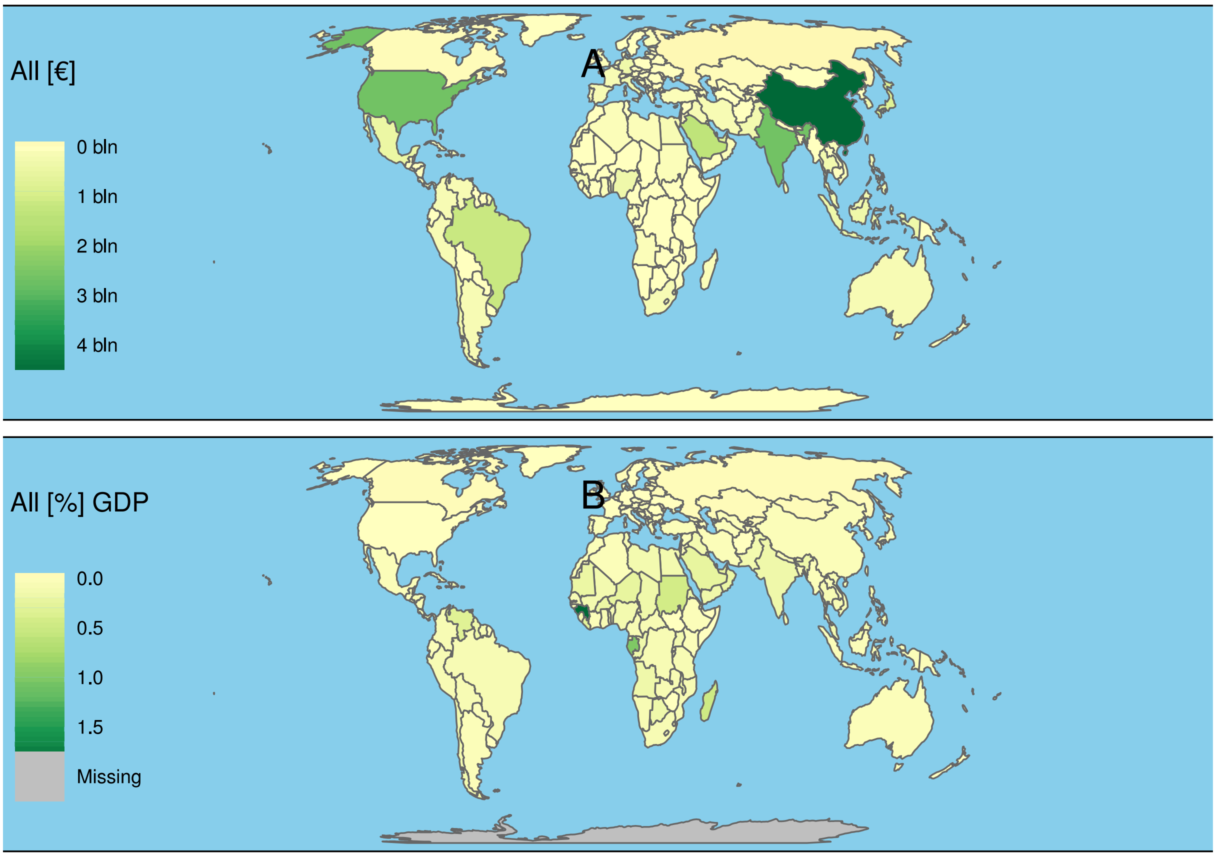


Figure S 51 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for medium demand scenario and recycling and 1% discount rate

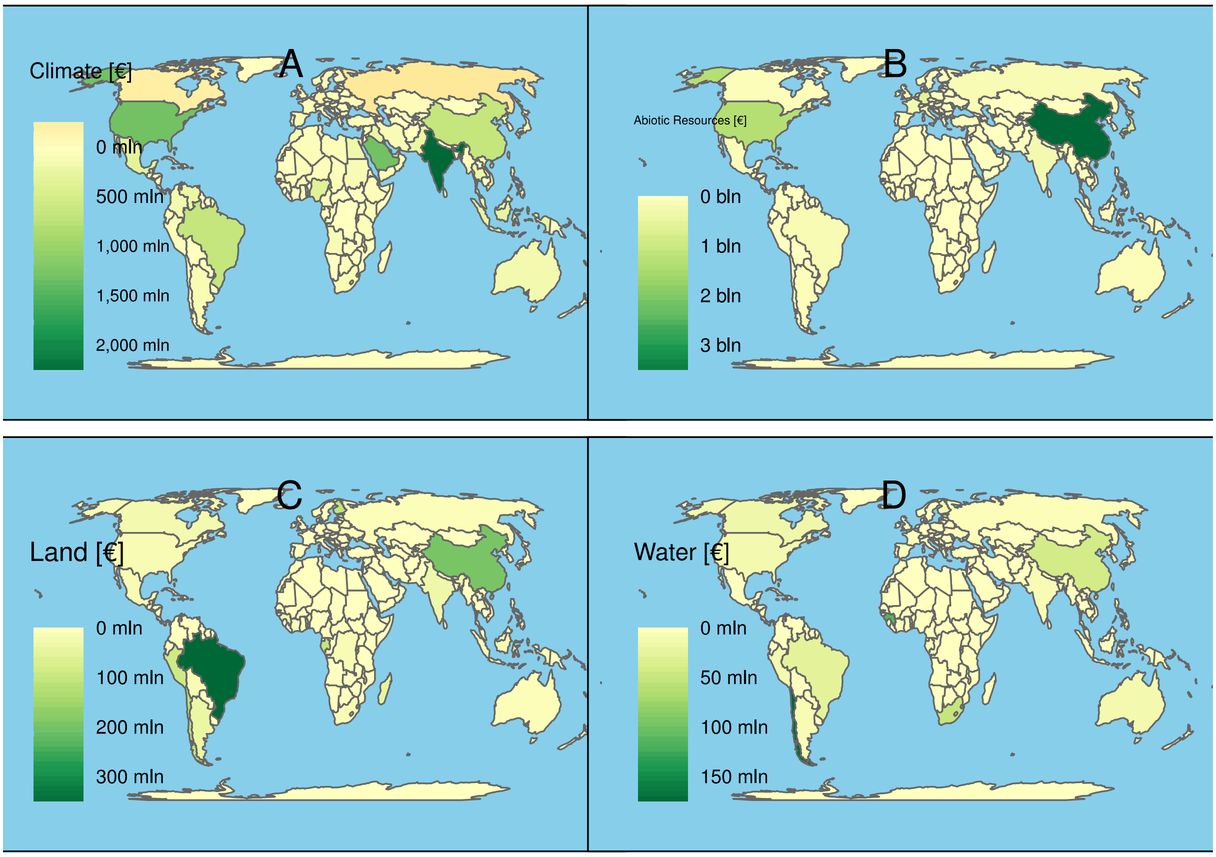


Figure S 52 Damages due to climate change, resource depletion, land use and water use for medium demand scenario and recycling and 1% discount rate

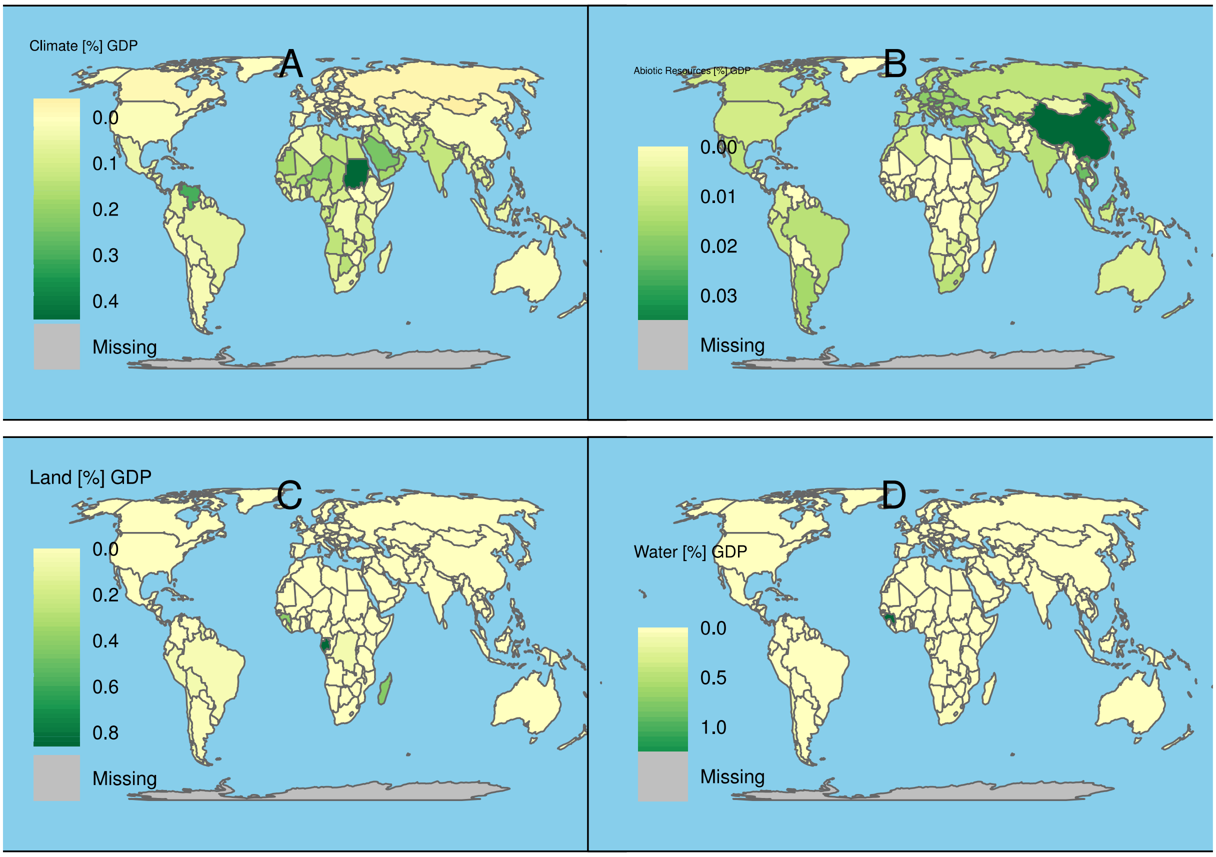


Figure S 53 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for medium demand scenario and recycling and 1% discount rate

6.1.2.3. 3 % discount rate

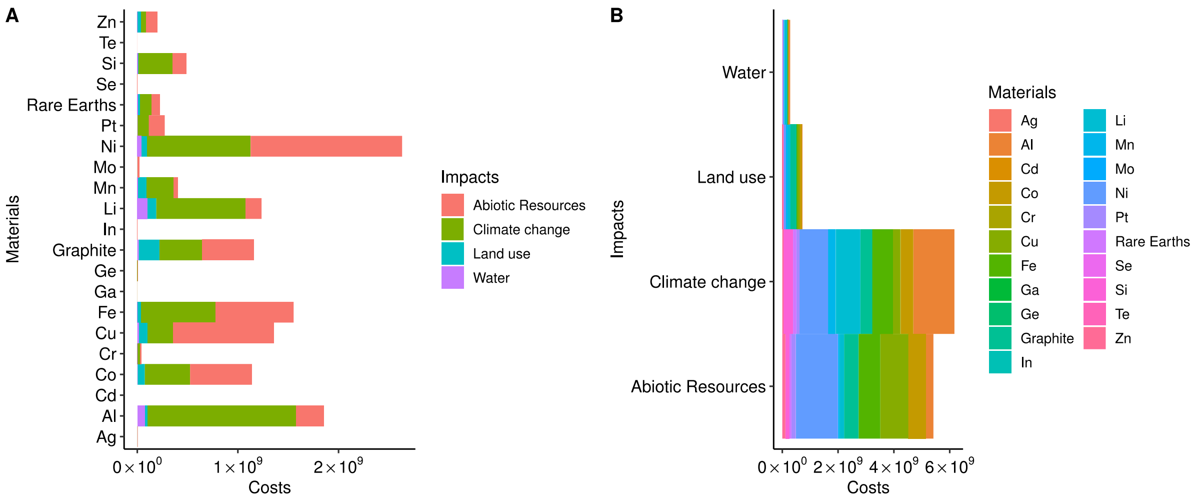


Figure S 54 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for medium demand scenario and recycling and 3% discount rate

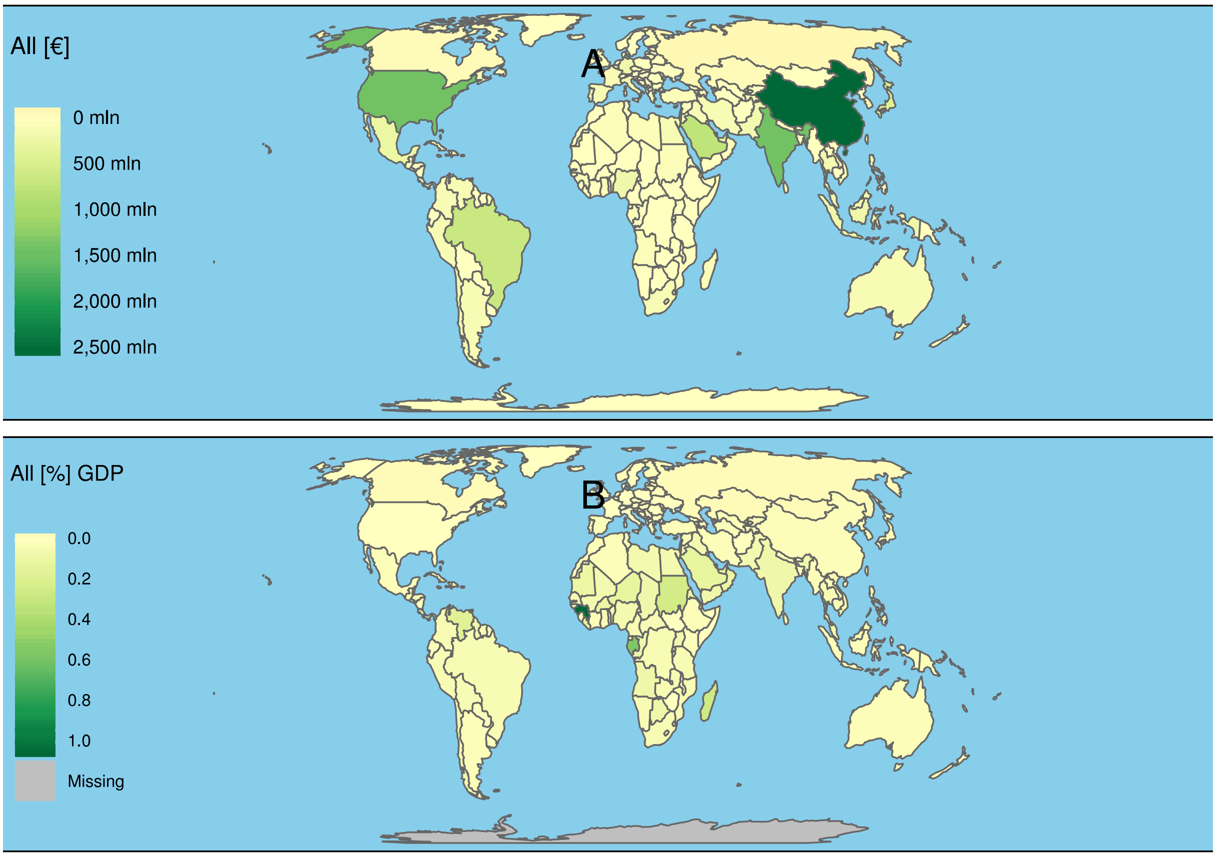


Figure S 55 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for medium demand scenario and recycling and 3% discount rate

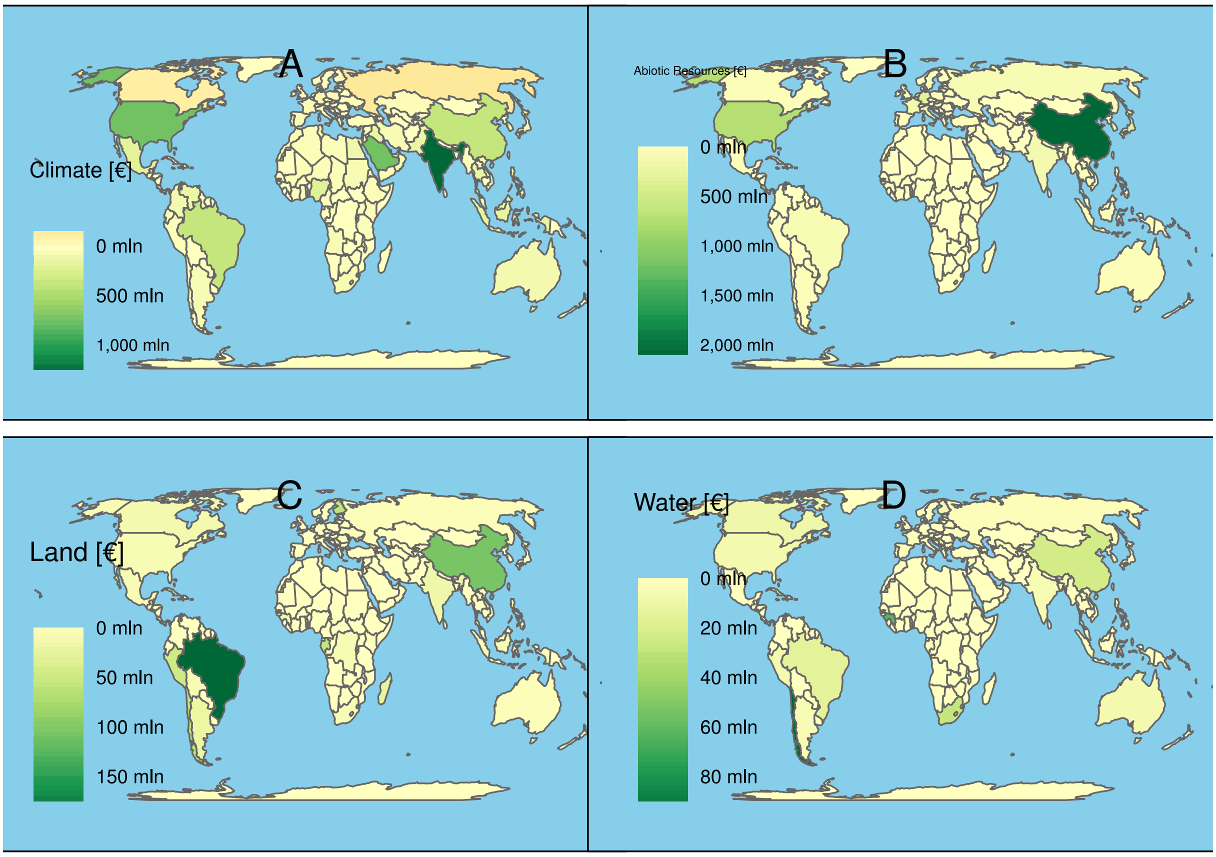


Figure S 56 Damages due to climate change, resource depletion, land use and water use for medium demand scenario and recycling and 3% discount rate

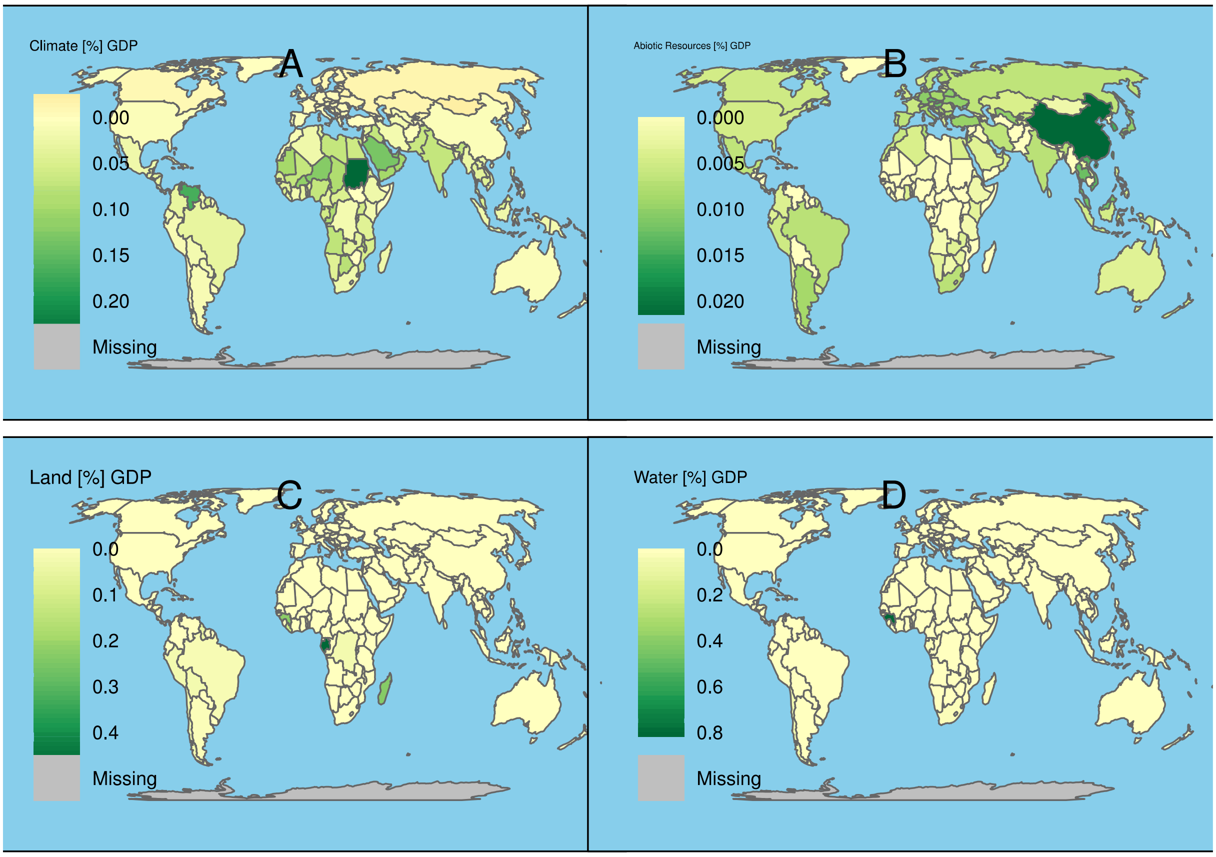


Figure S 57 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for medium demand scenario and recycling and 3% discount rate

6.2. Minimum Demand

6.2.1. No Recycling

6.2.1.1. No discounting

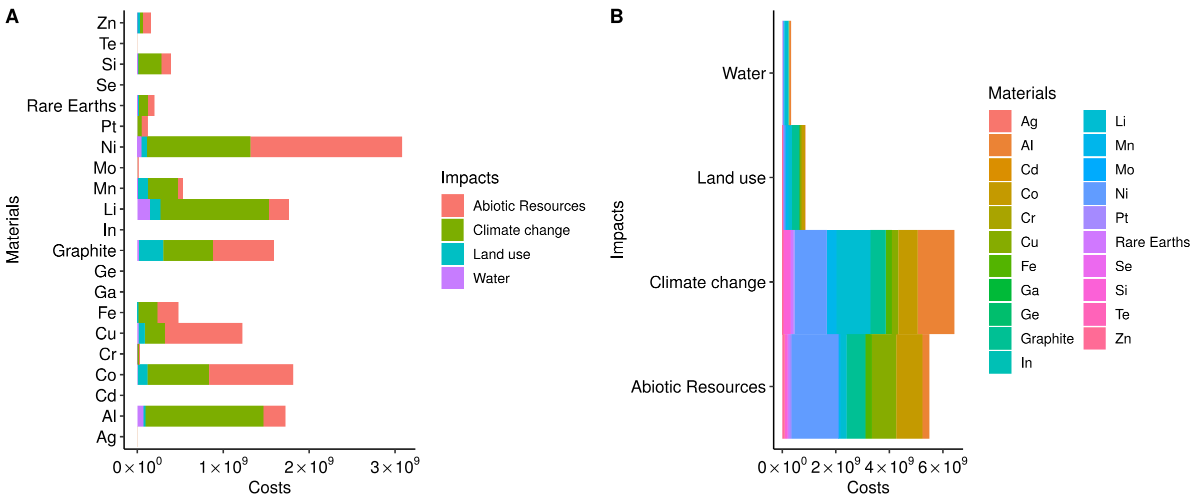


Figure S 58 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for minimum demand scenario and no recycling and no discount rate

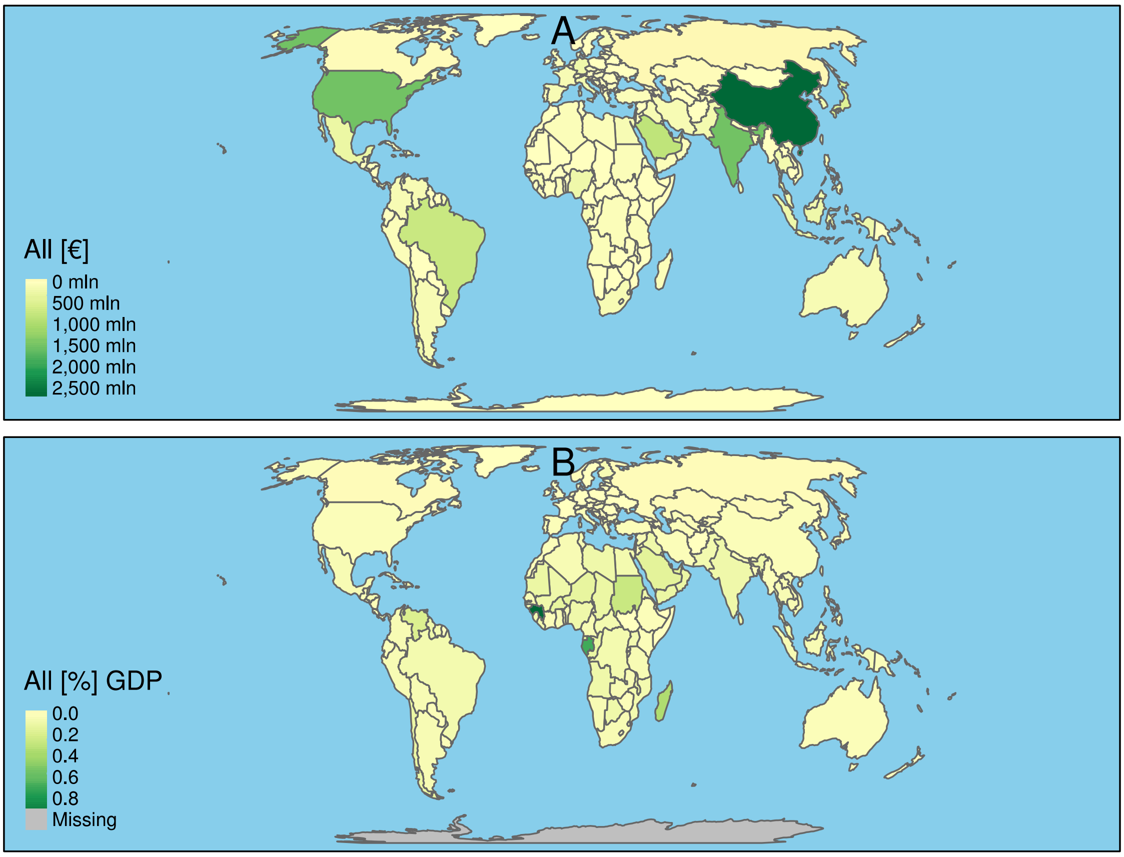


Figure S 59 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for minimum demand scenario and no recycling and no discount rate

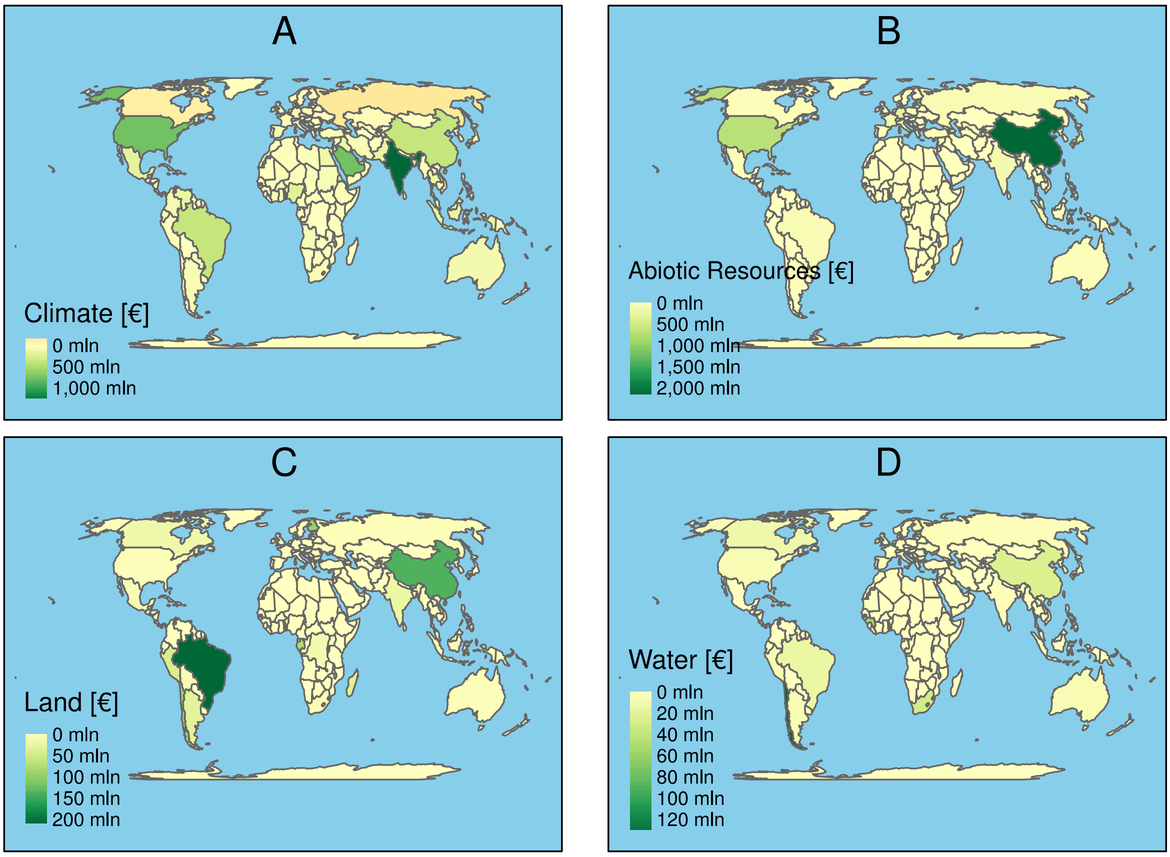


Figure S 60 Damages due to climate change, resource depletion, land use and water use for minimum demand scenario and no recycling and no discount rate

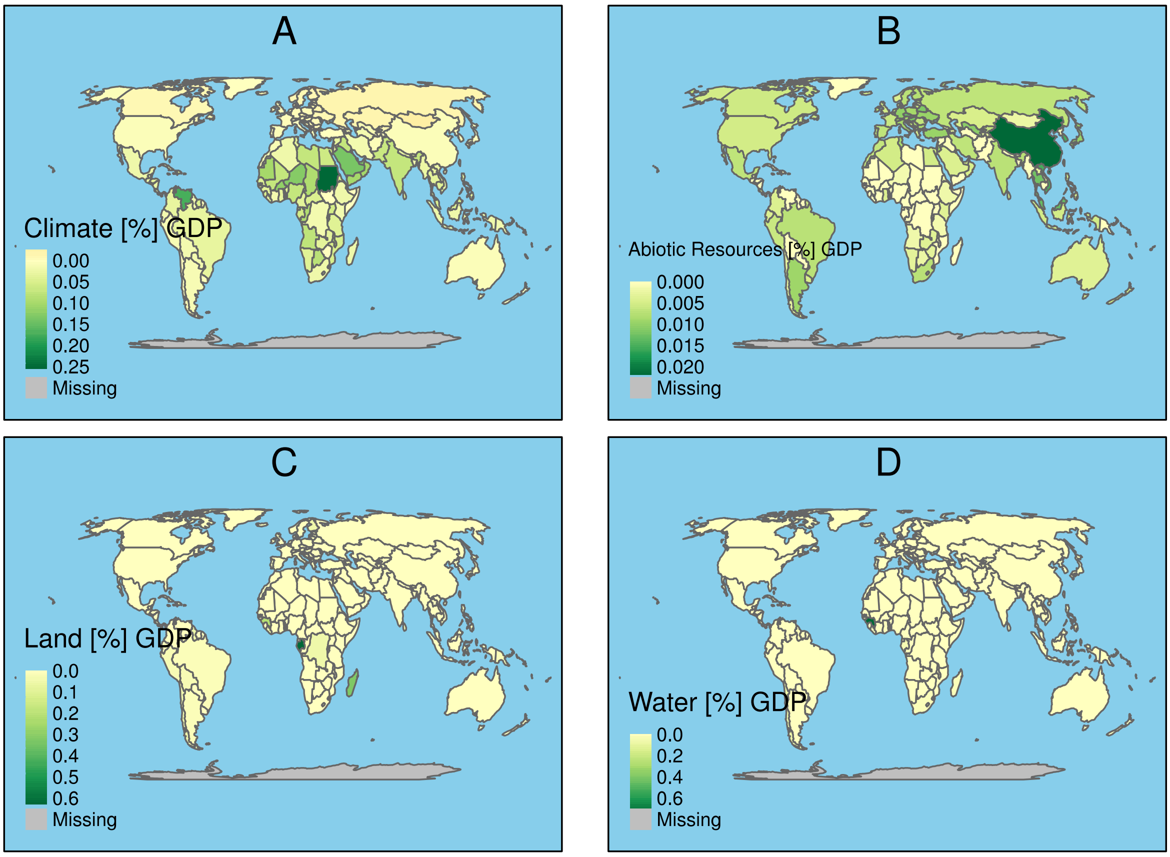


Figure S 61 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for minimum demand scenario and no recycling and no discount rate

6.2.1.2. 1% discount rate

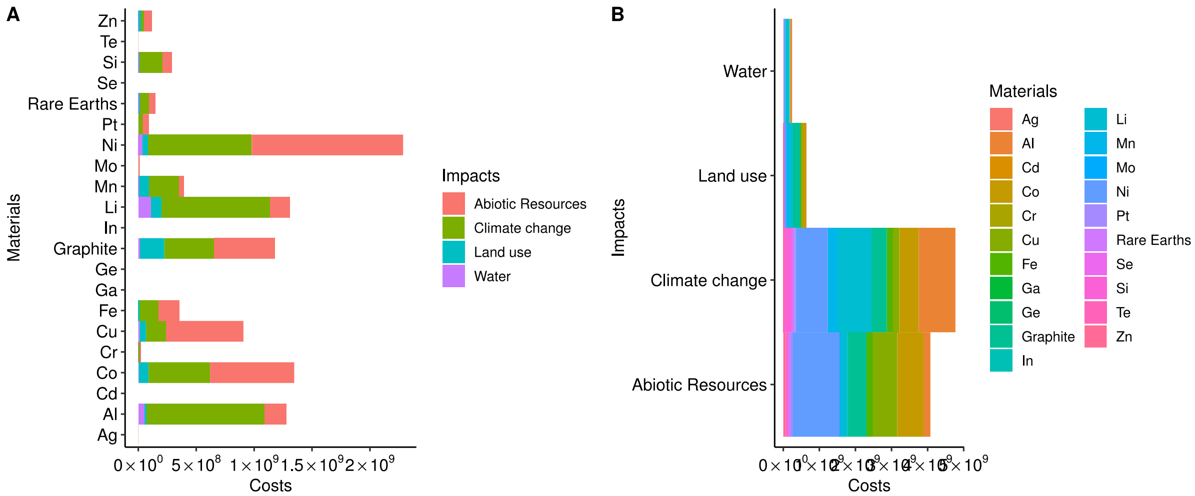


Figure S 62 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for minimum demand scenario and no recycling and 1% discount rate

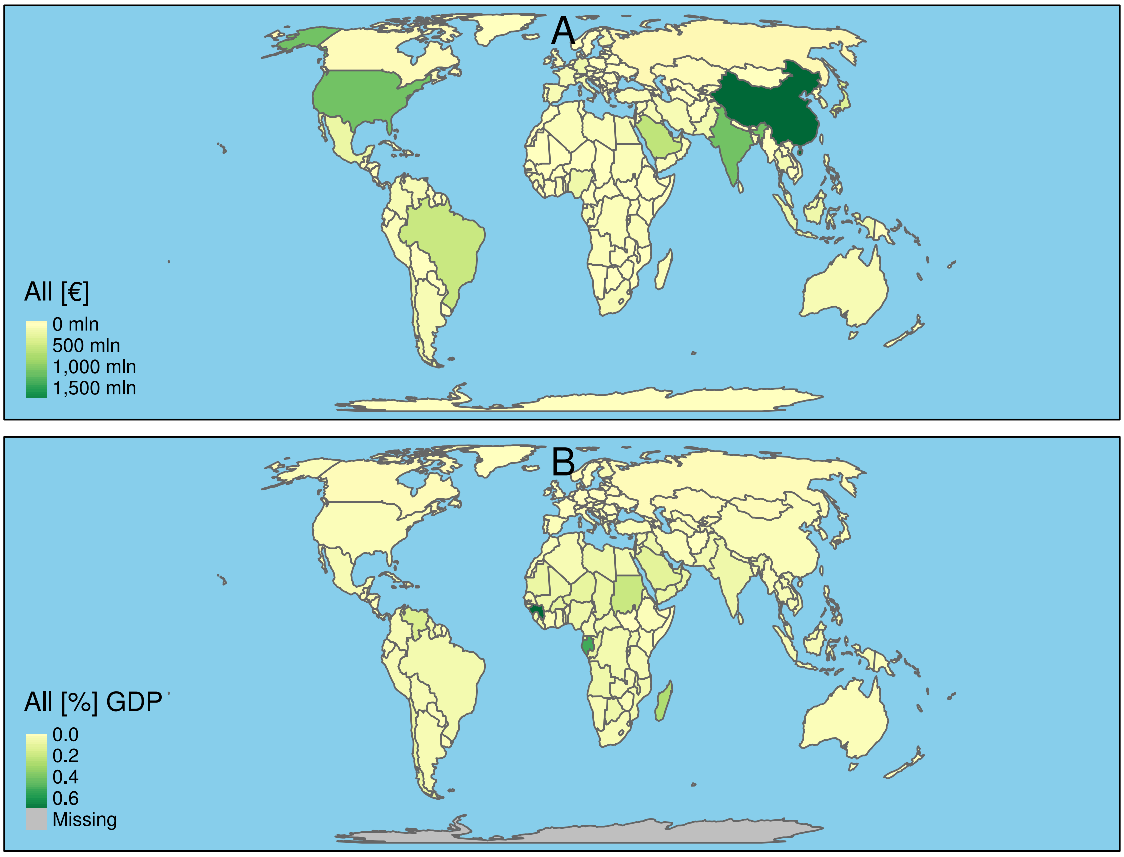


Figure S 63 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for minimum demand scenario and no recycling and 1% discount rate

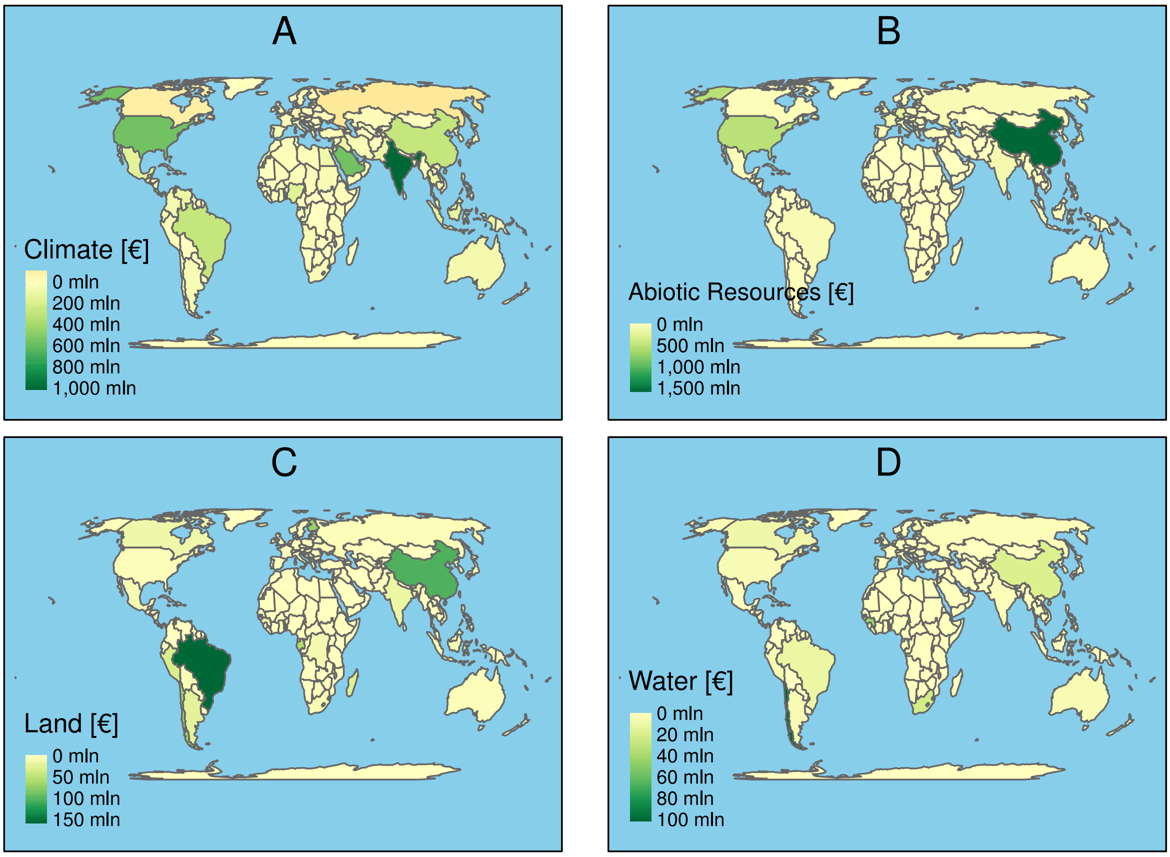


Figure S 64 Damages due to climate change, resource depletion, land use and water use for minimum demand scenario and no recycling and 1% discount rate

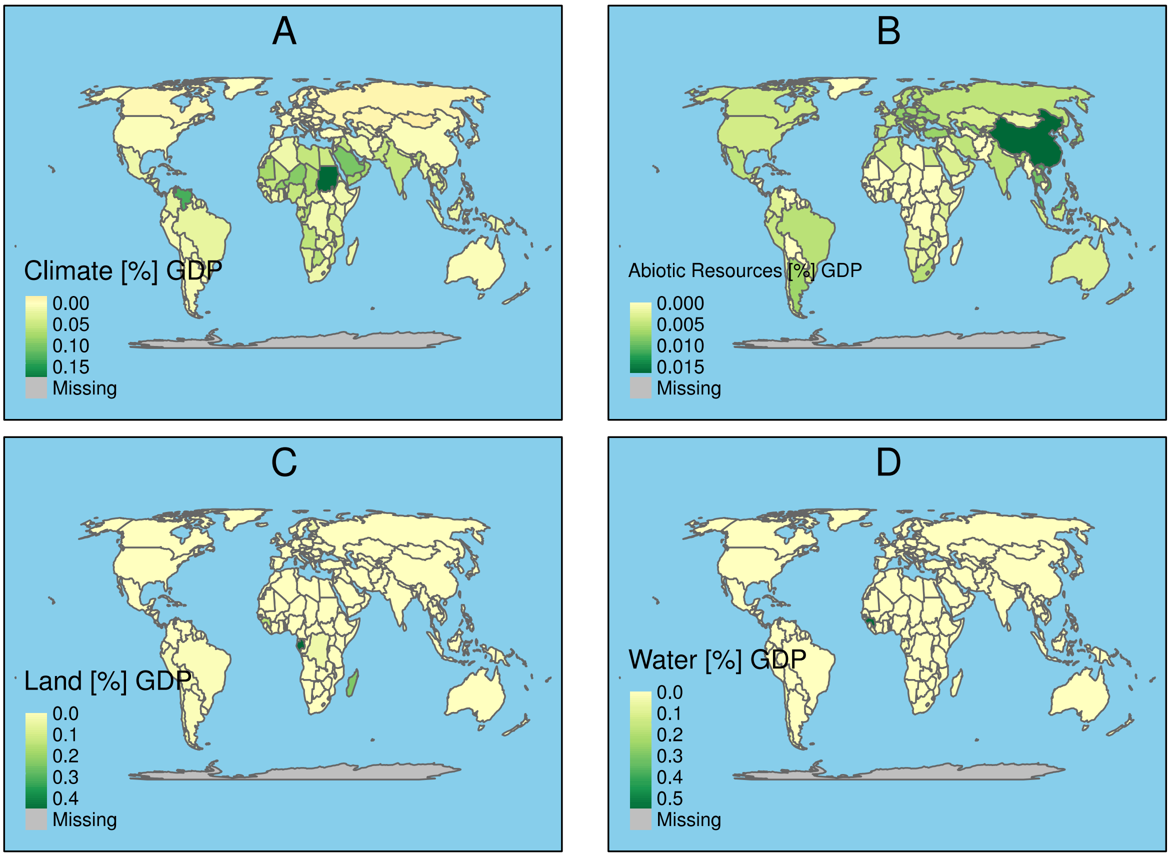


Figure S 65 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for minimum demand scenario and no recycling and 1% discount rate

6.2.1.3. 3 % discount rate

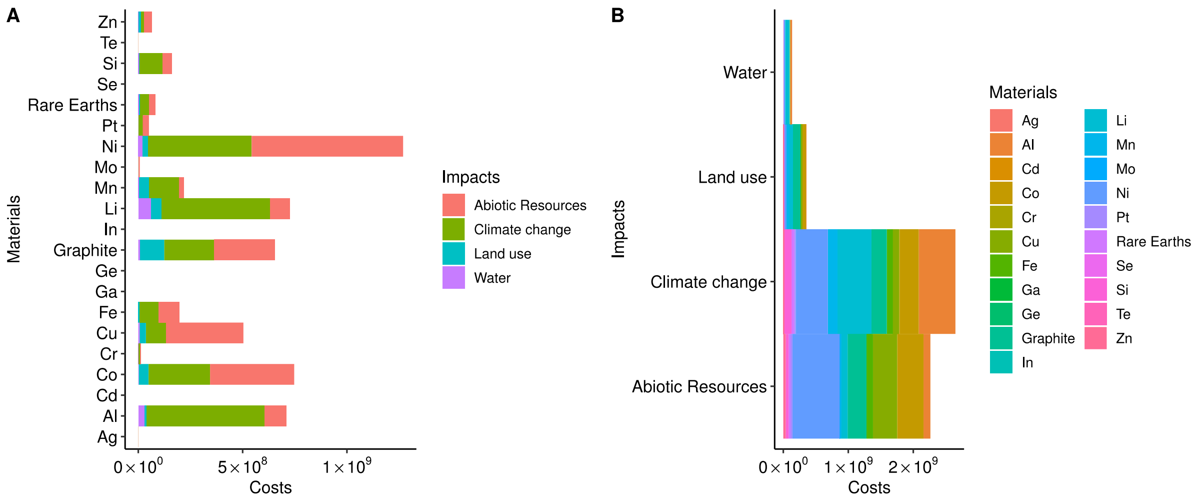


Figure S 66 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for minimum demand scenario and no recycling and 3% discount rate

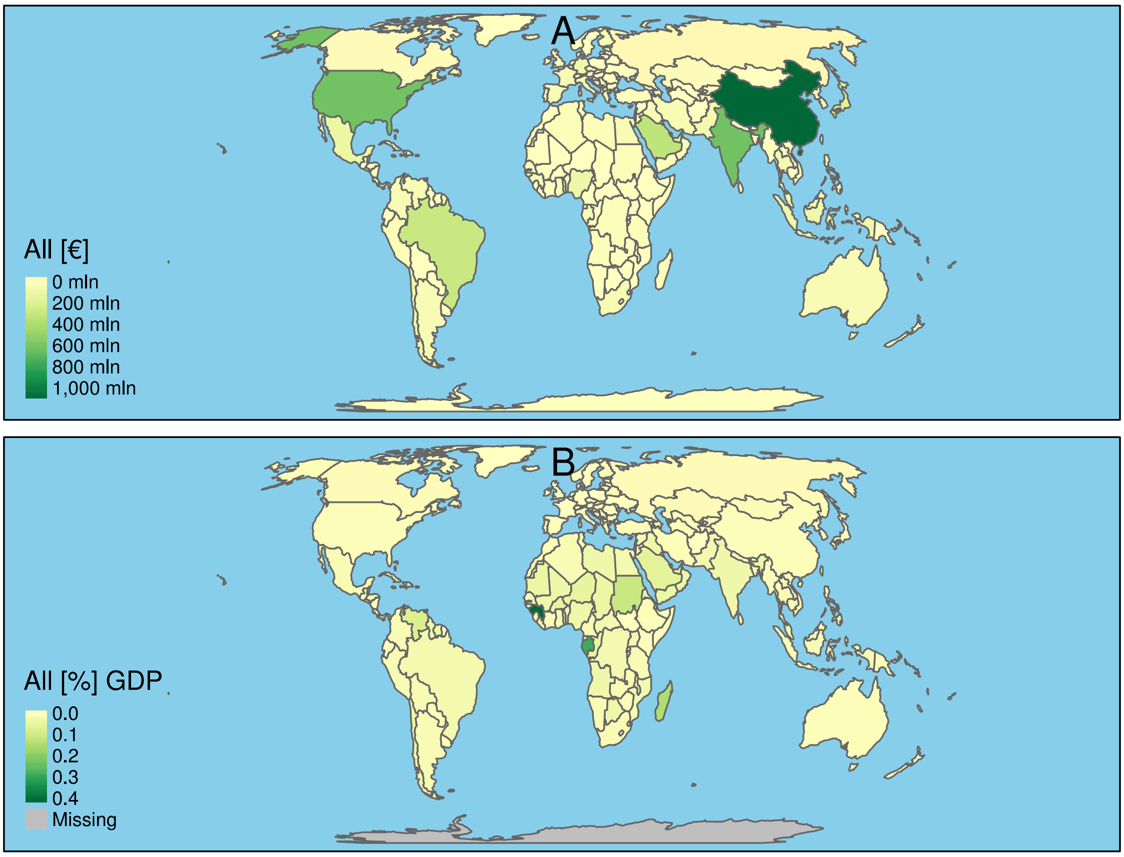


Figure S 67 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for minimum demand scenario and no recycling and 3% discount rate

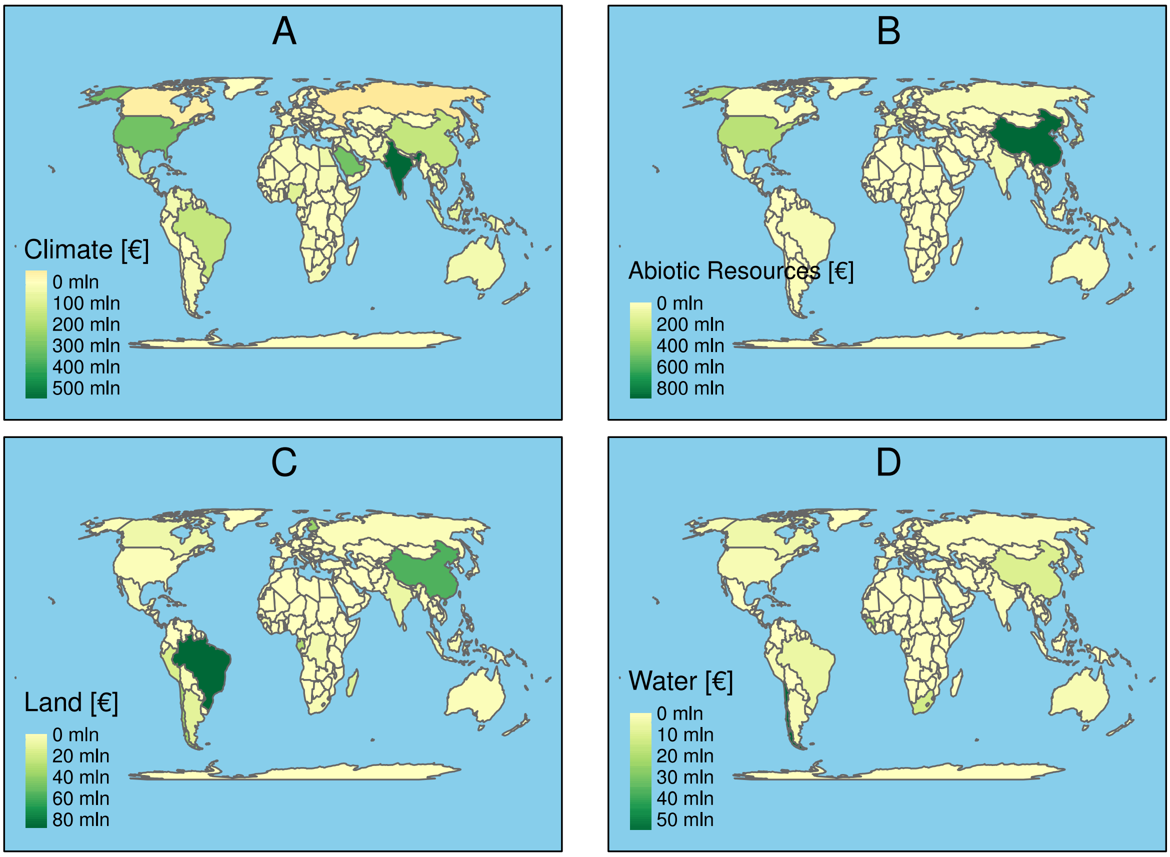


Figure S 68 Damages due to climate change, resource depletion, land use and water use for minimum demand scenario and no recycling and 3% discount rate

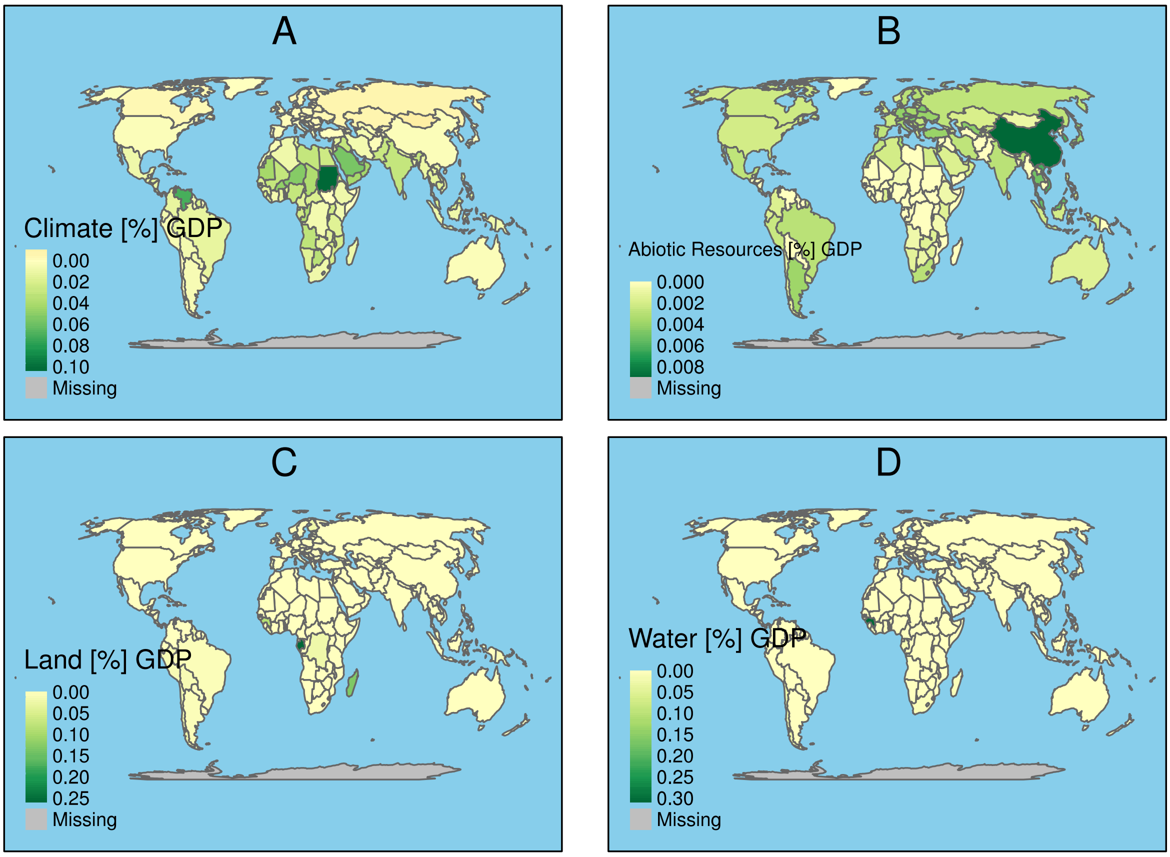


Figure S 69 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for minimum demand scenario and no recycling and 3% discount rate

6.2.2. Recycling

6.2.2.1. No discounting

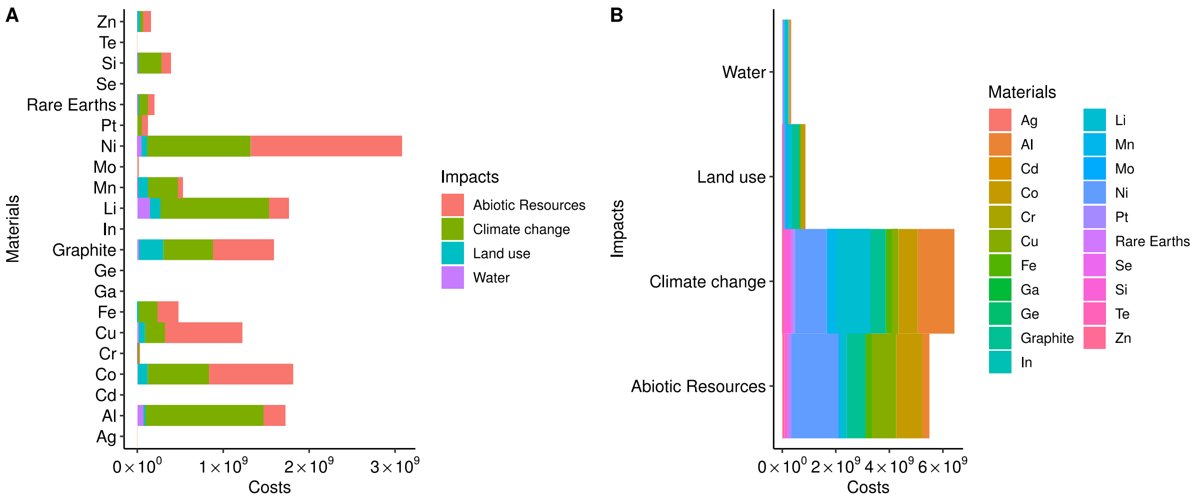


Figure S 70 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for minimum demand scenario with recycling and no discount rate

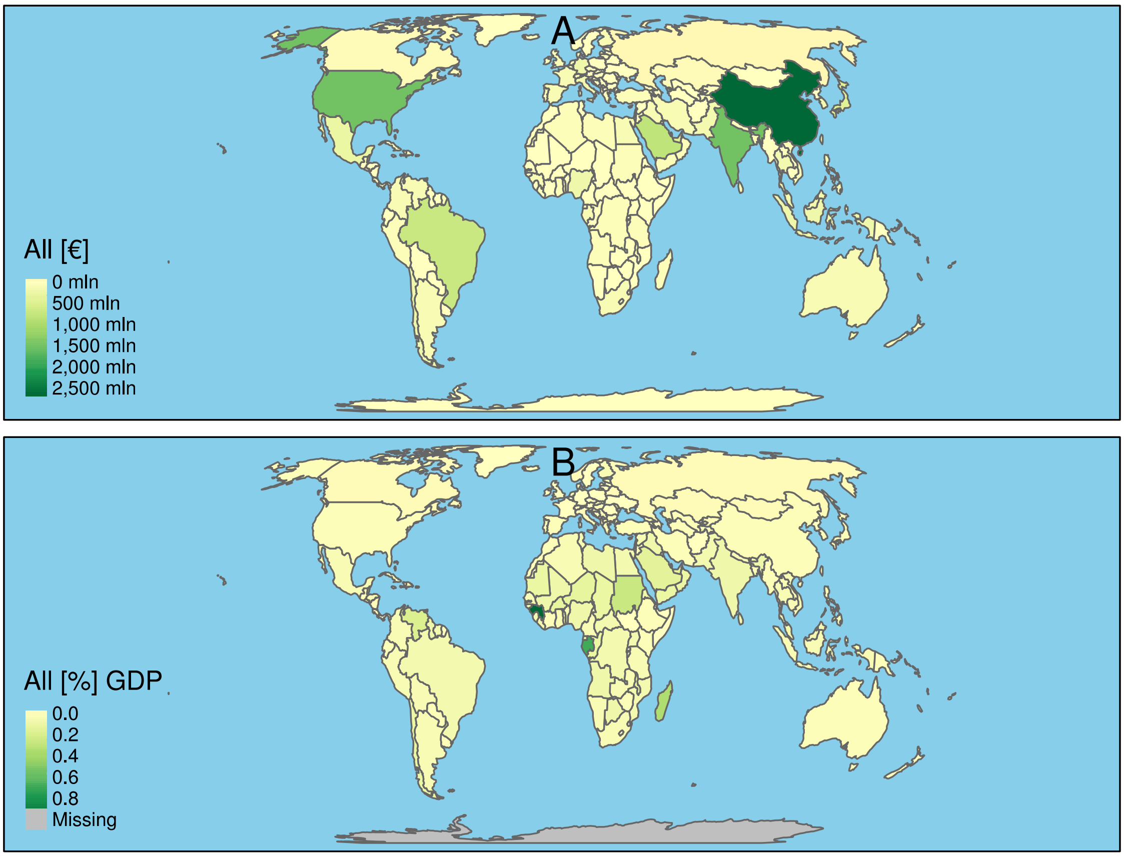


Figure S 71 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for minimum demand scenario with recycling and no discount rate

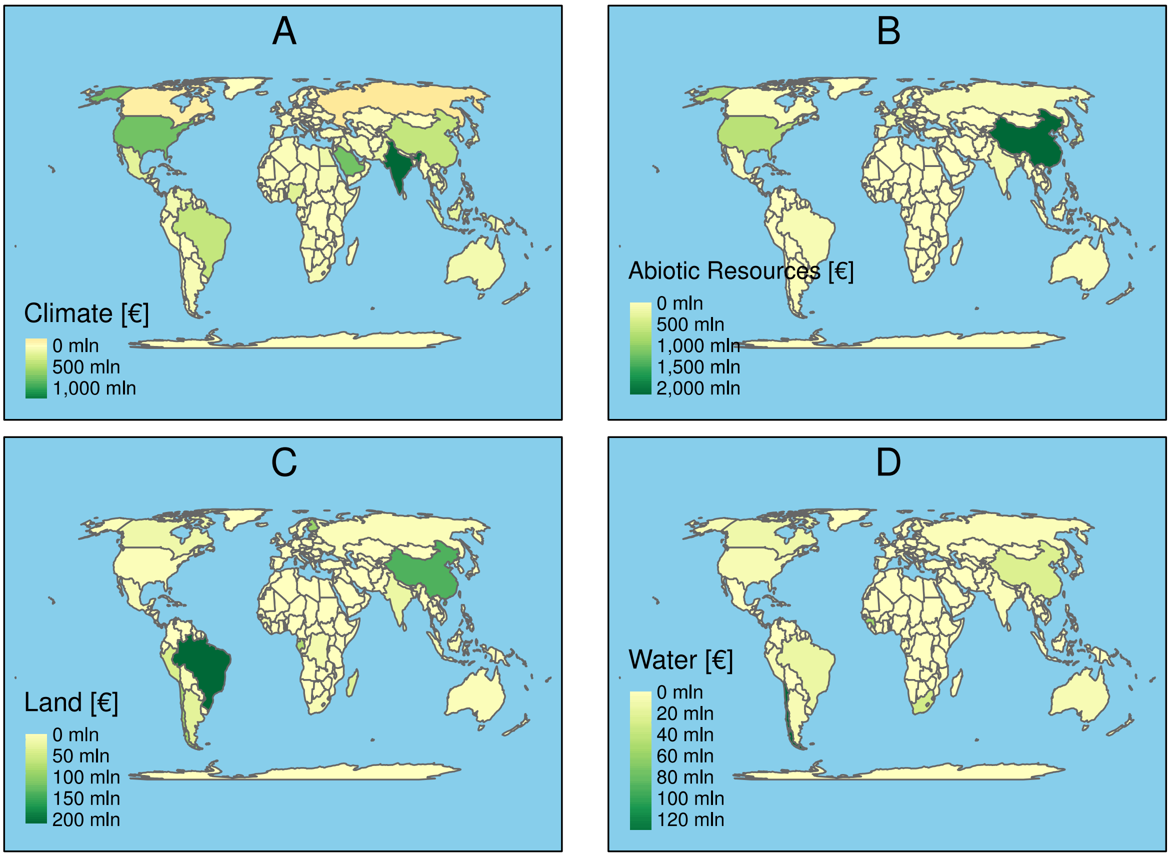


Figure S 72 Damages due to climate change, resource depletion, land use and water use for minimum demand scenario with recycling and no discount rate

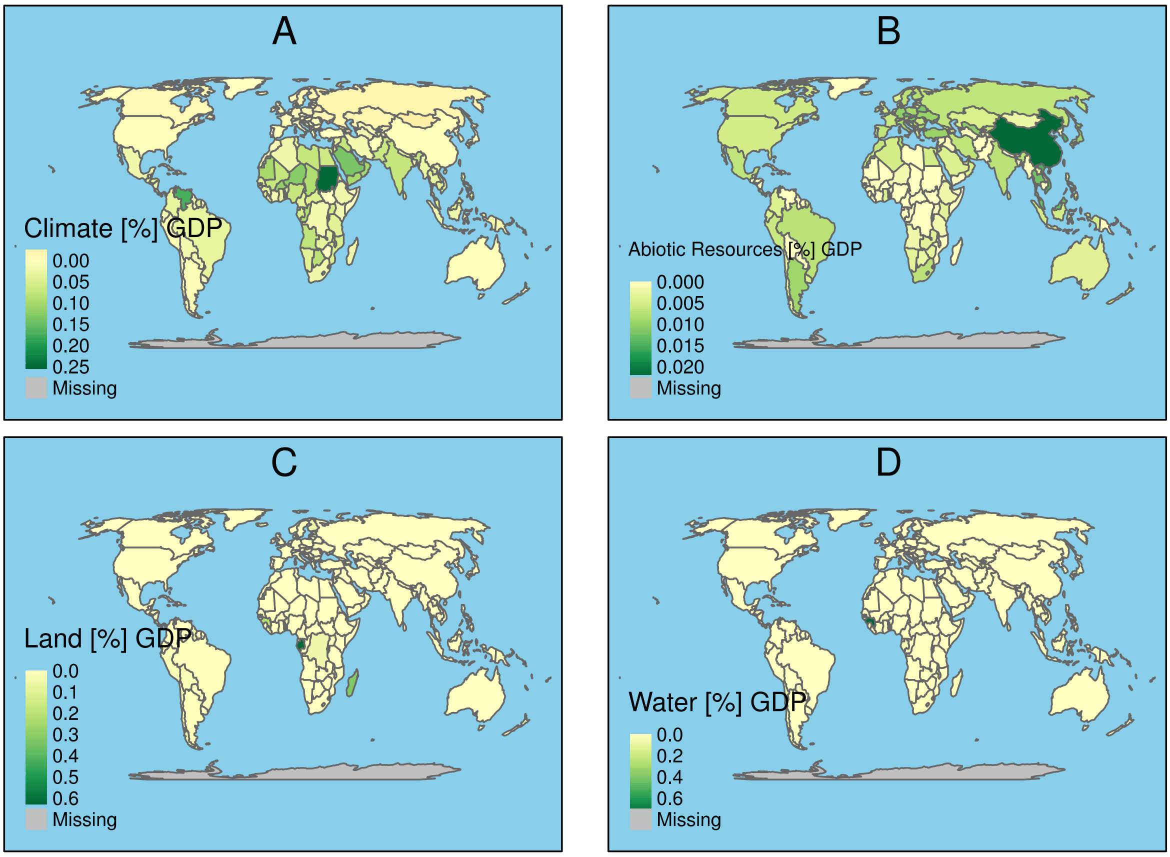


Figure S 73 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for minimum demand scenario with recycling and no discount rate

6.2.2.2. 1% discount rate

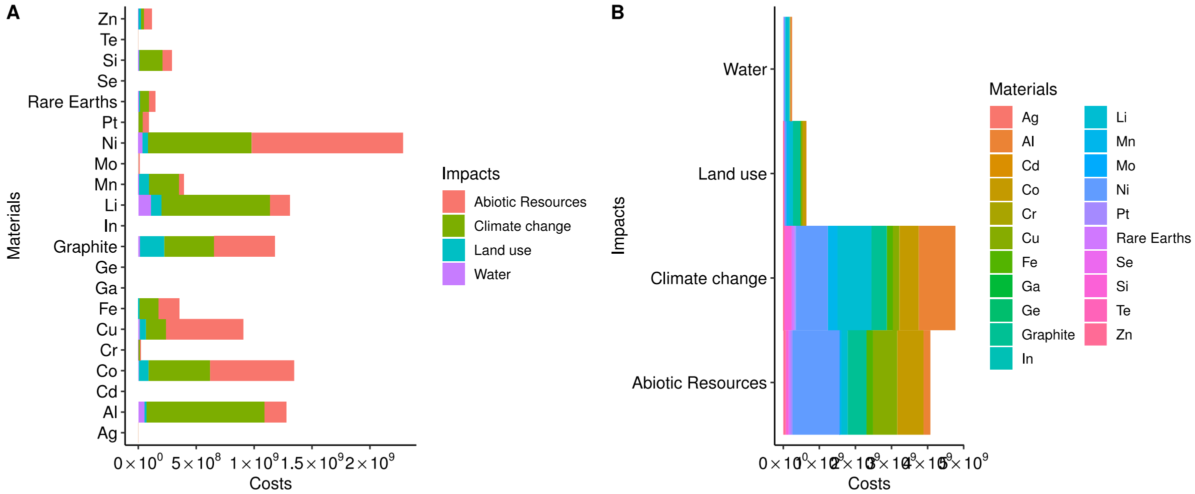


Figure S 74 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for minimum demand scenario with recycling and 1% discount rate

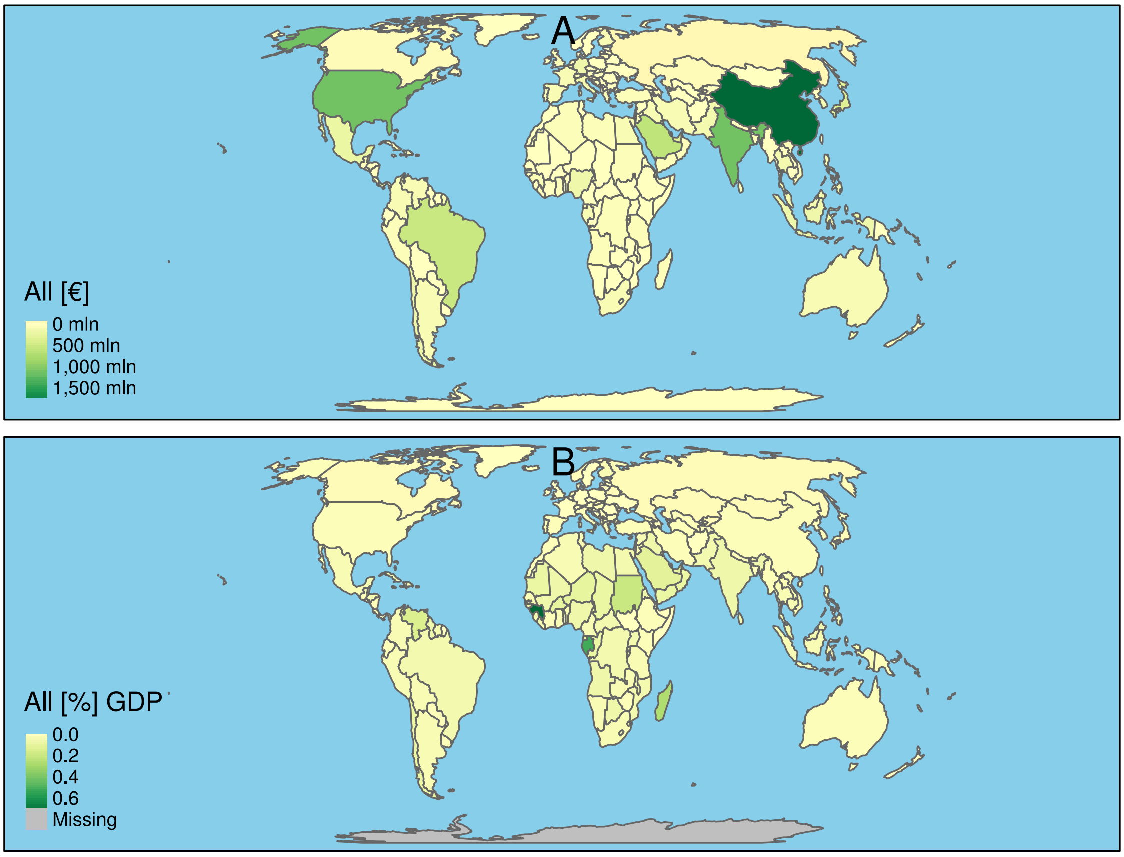


Figure S 75 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for minimum demand scenario with recycling and 1% discount rate

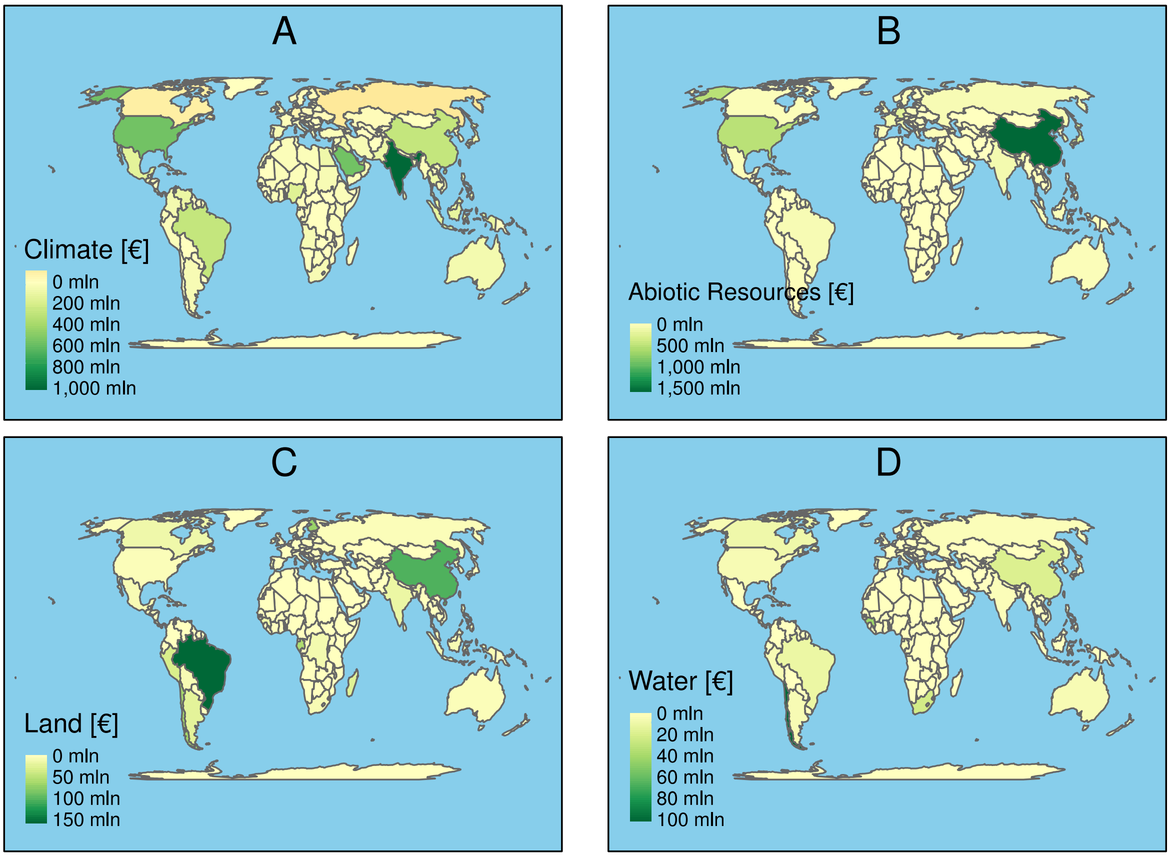


Figure S 76 Damages due to climate change, resource depletion, land use and water use for minimum demand scenario with recycling and 1% discount rate

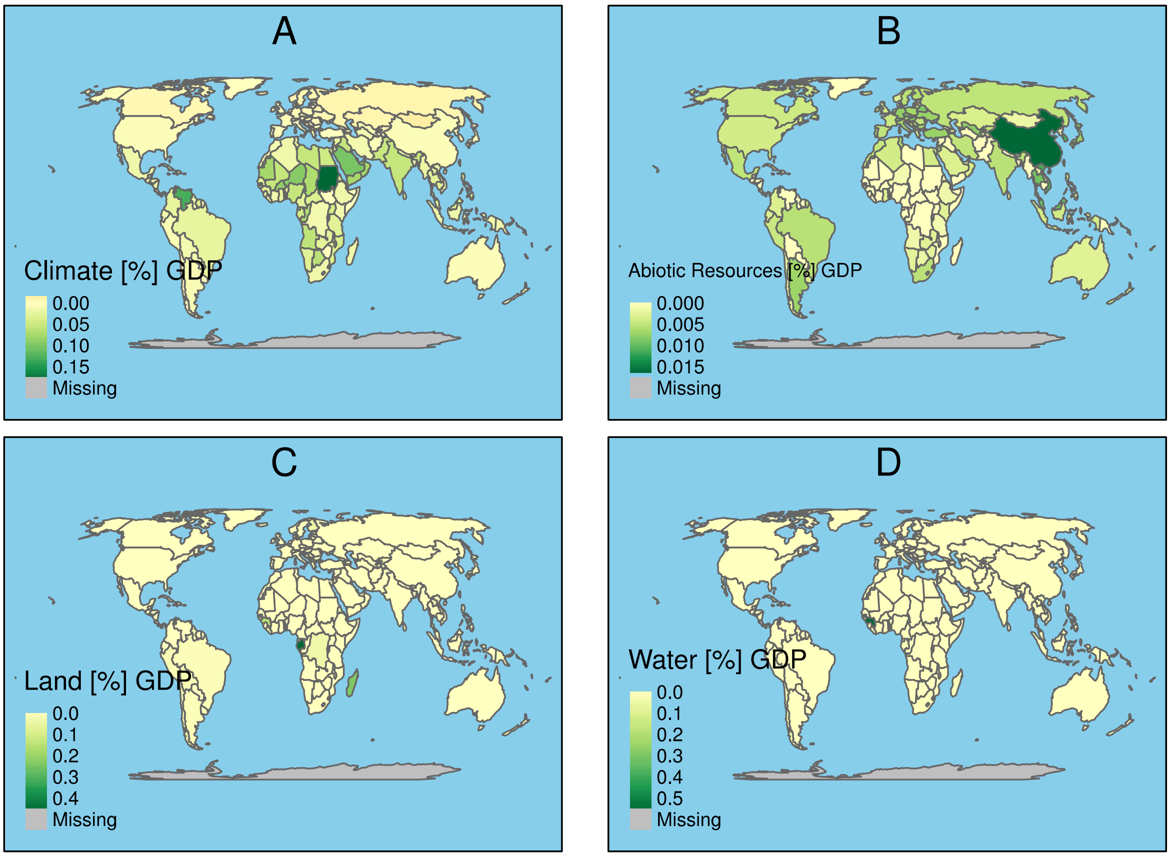


Figure S 77 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for minimum demand scenario with recycling and 1% discount rate

6.2.2.3. 3 % discount rate

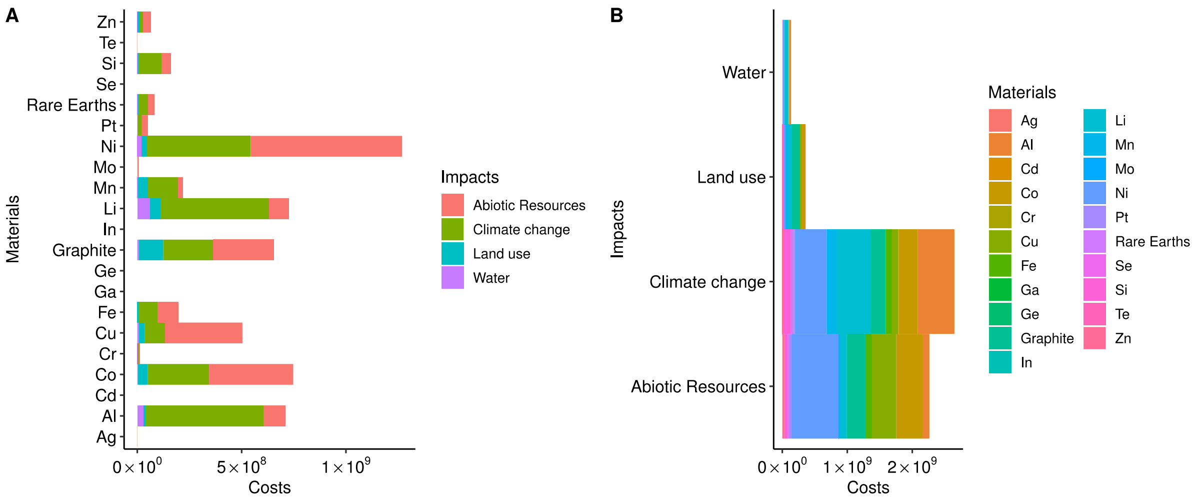


Figure S 78 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for minimum demand scenario with recycling and 3% discount rate

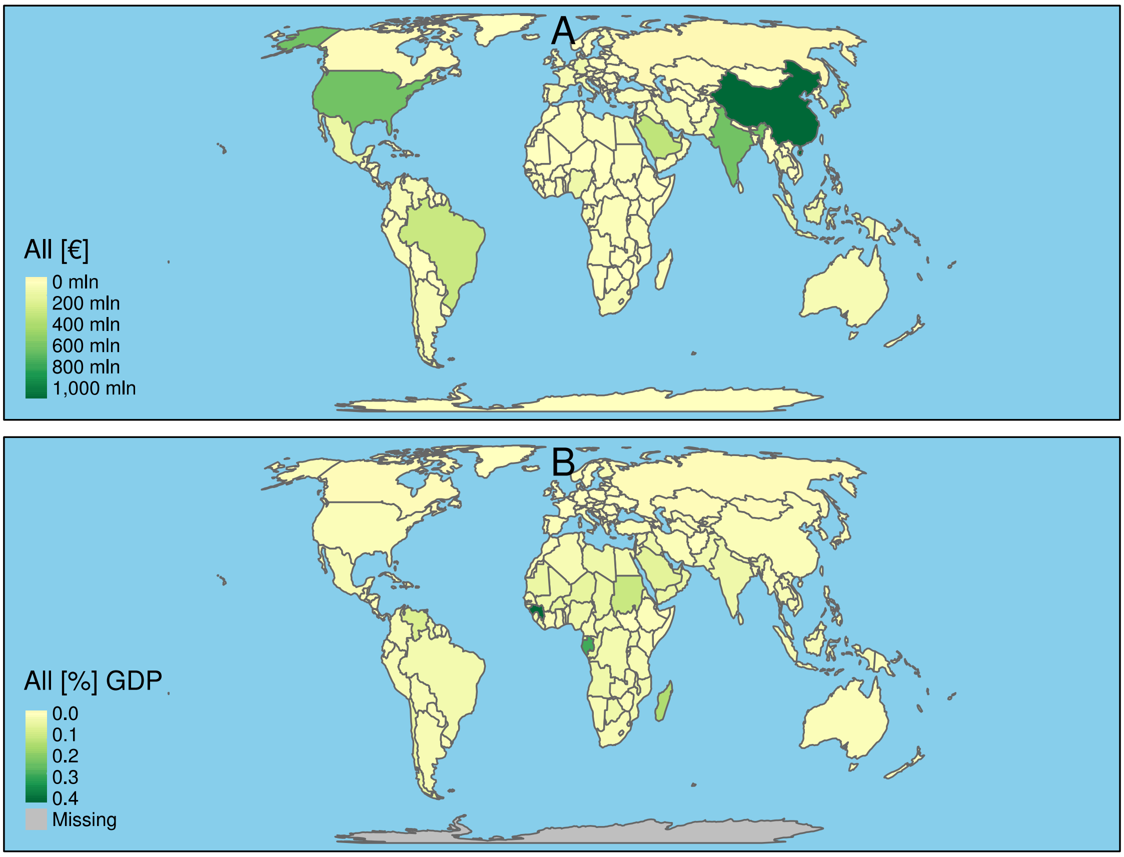


Figure S 79 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for minimum demand scenario with recycling and 3% discount rate

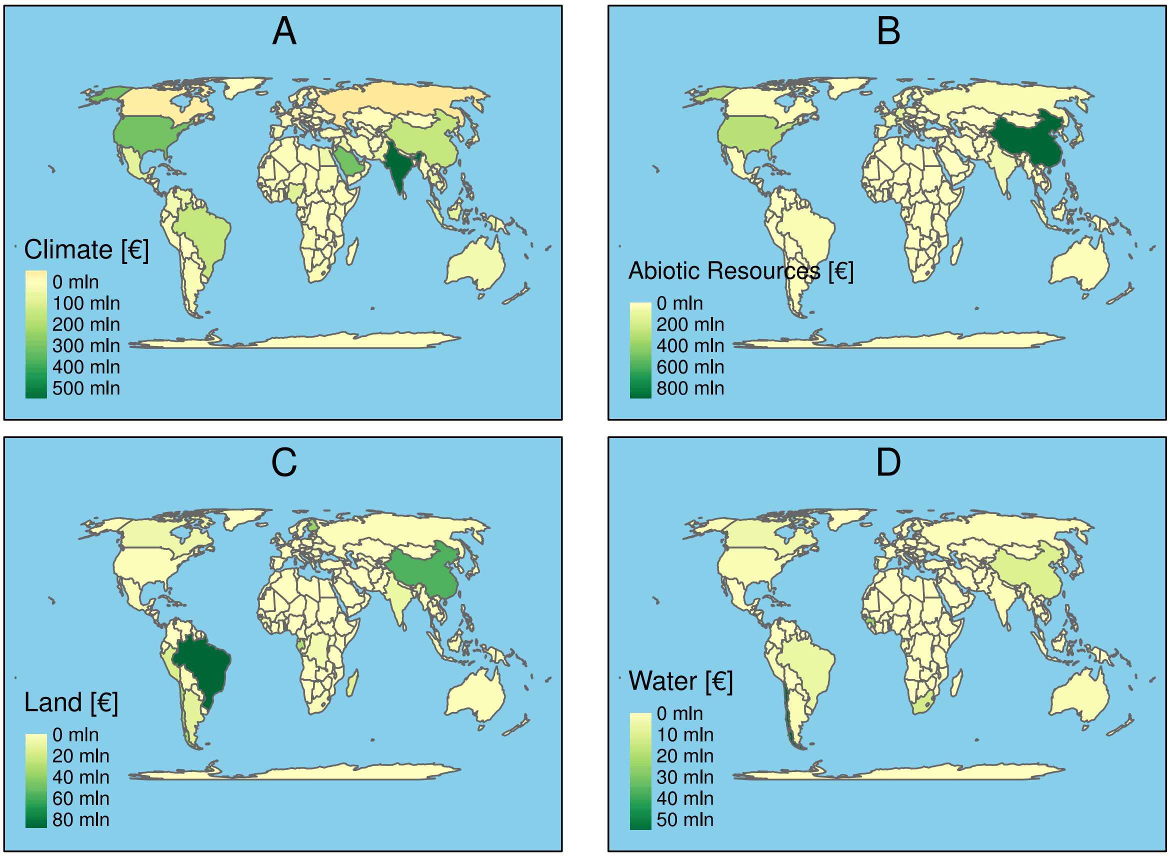


Figure S 80 Damages due to climate change, resource depletion, land use and water use for minimum demand scenario with recycling and 3% discount rate

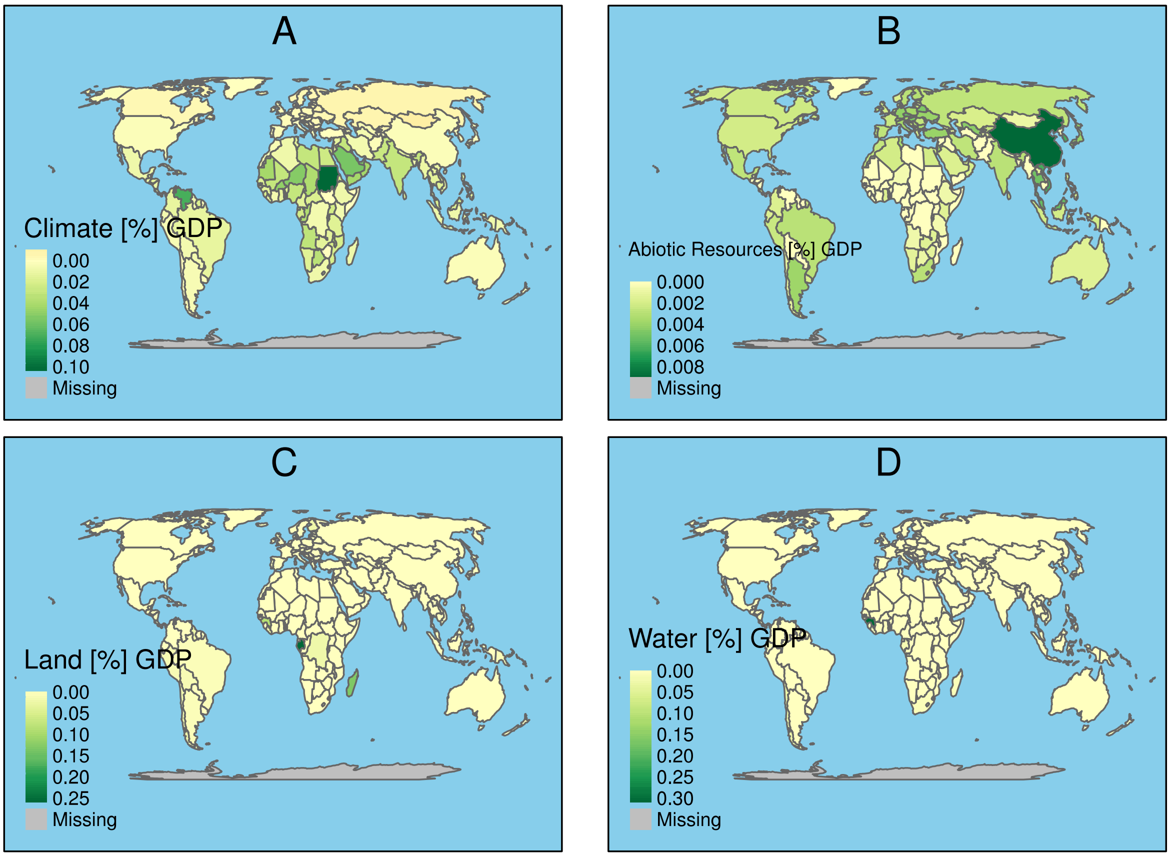


Figure S 81 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for minimum demand scenario with recycling and 3% discount rate

6.3. Maximum Demand

6.3.1. No Recycling

6.3.1.1. No discounting

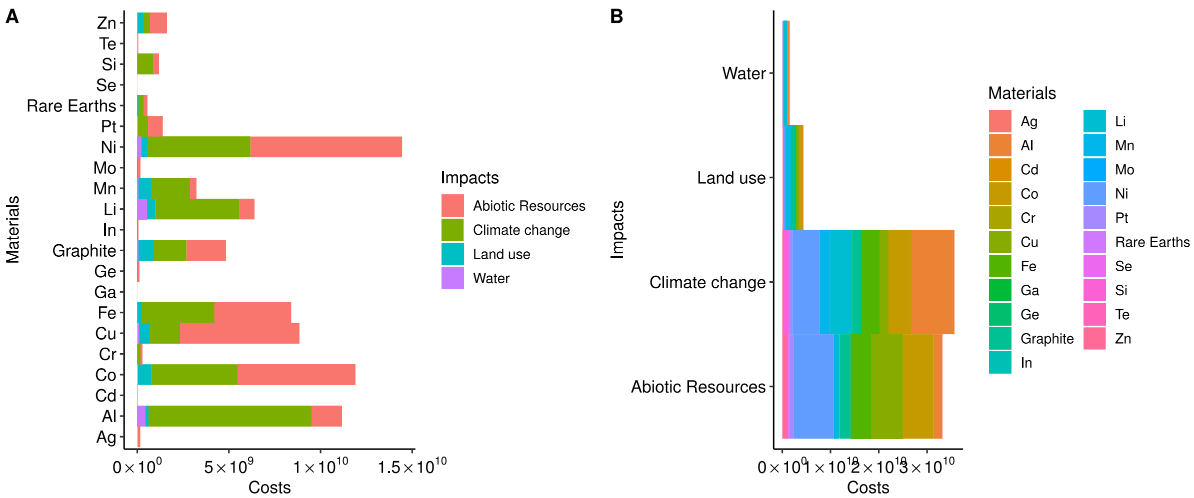


Figure S 82 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for maximum demand scenario without recycling and no discount rate

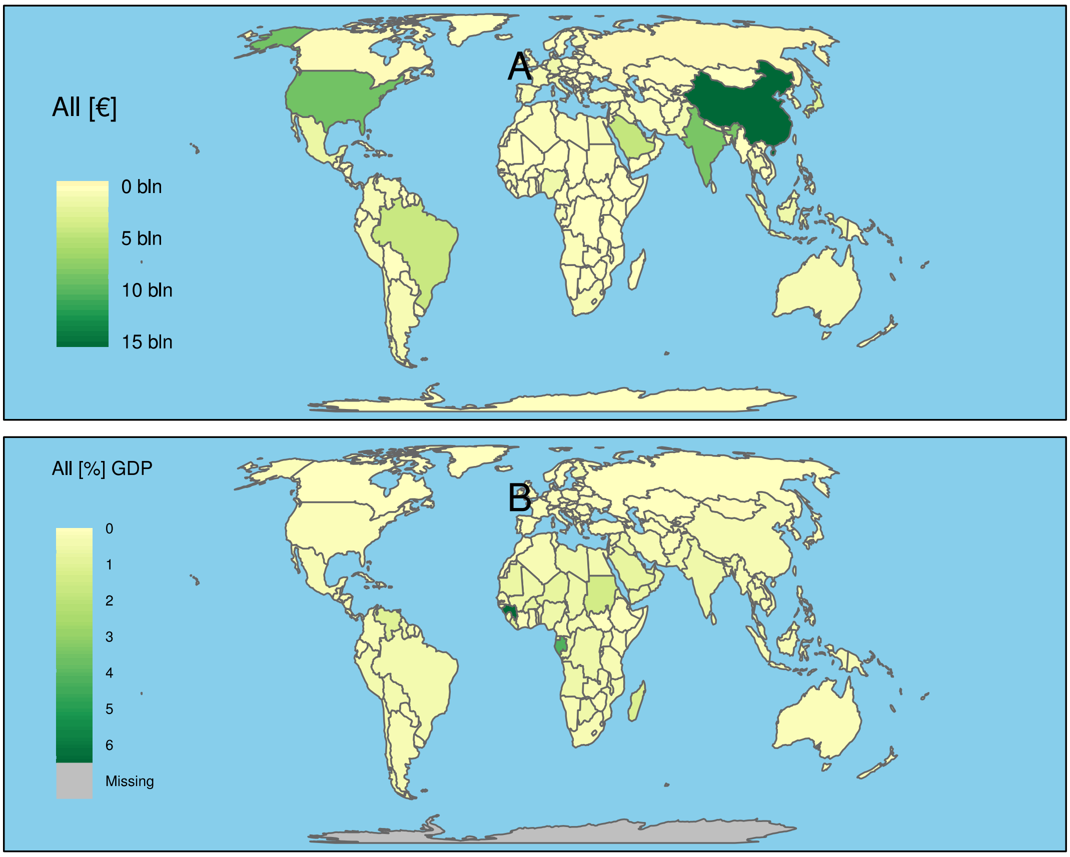


Figure S 83 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for maximum demand scenario without recycling and no discount rate

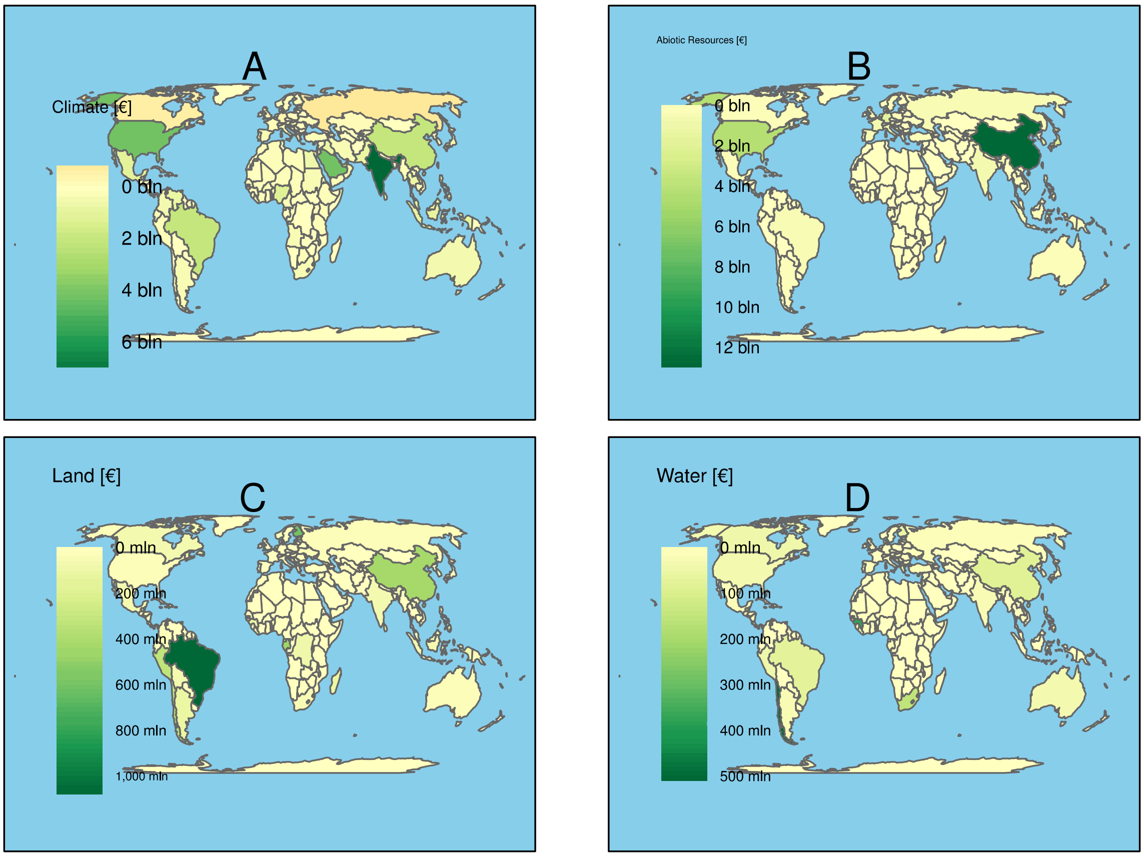


Figure S 84 Damages due to climate change, resource depletion, land use and water use for maximum demand scenario without recycling and no discount rate

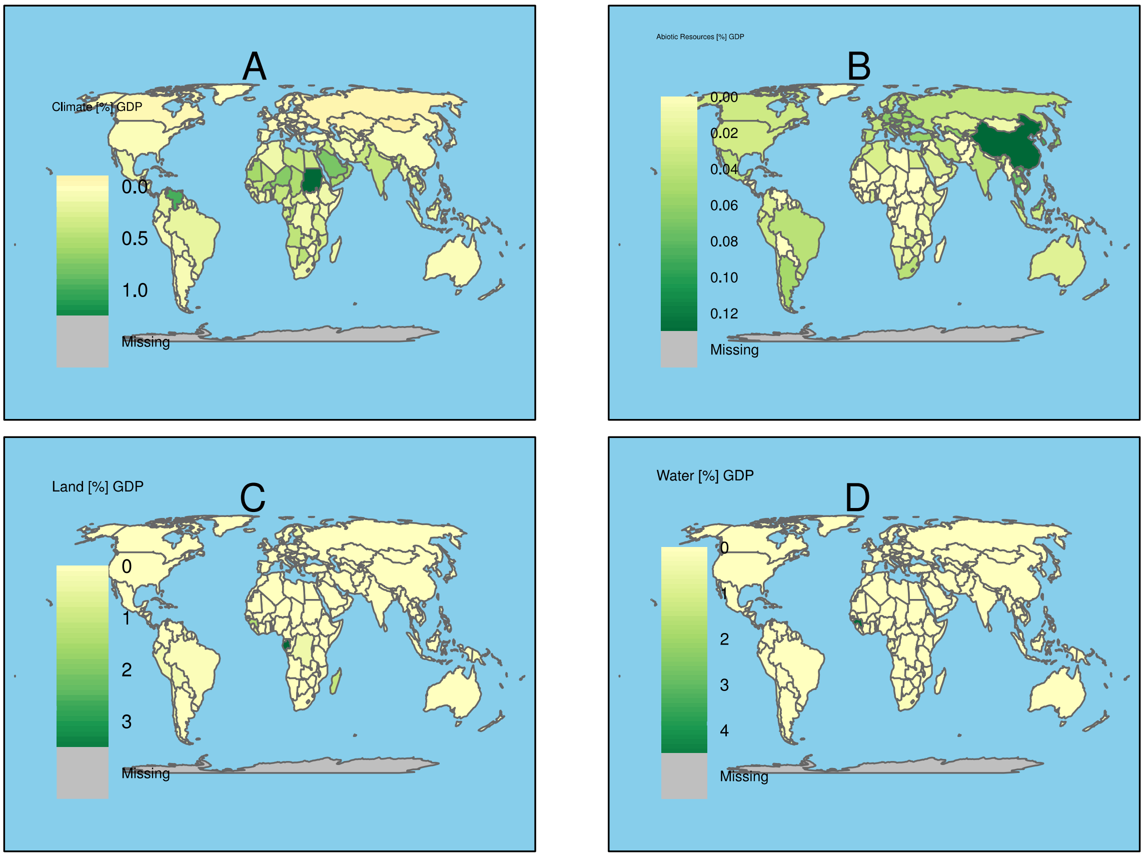


Figure S 85 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for maximum demand scenario without recycling and no discount rate

6.3.1.2. 1% discount rate

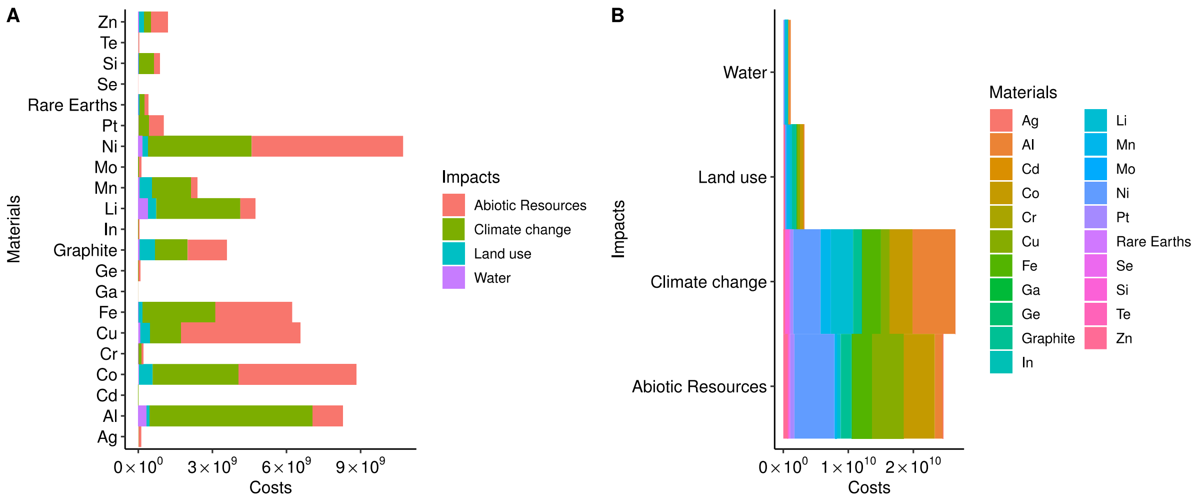


Figure S 86 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for maximum demand scenario without recycling and with 1% discount rate

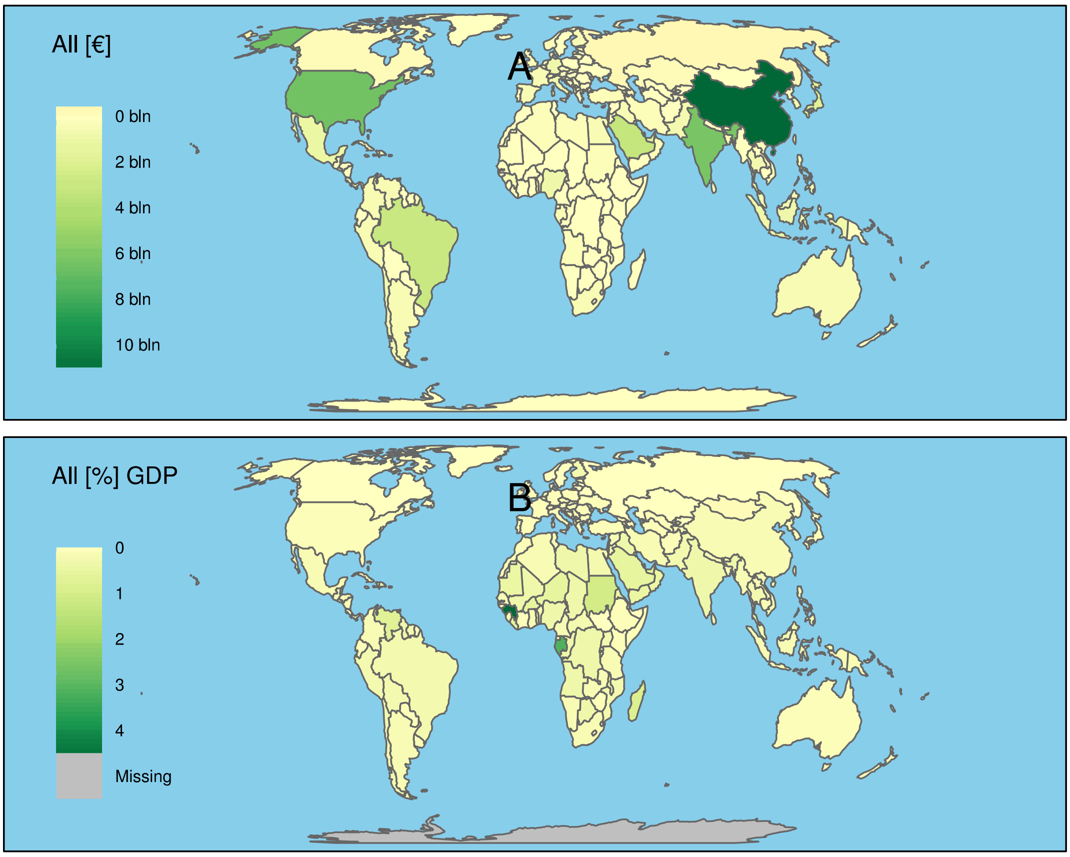


Figure S 87 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for maximum demand scenario without recycling and with 1% discount rate

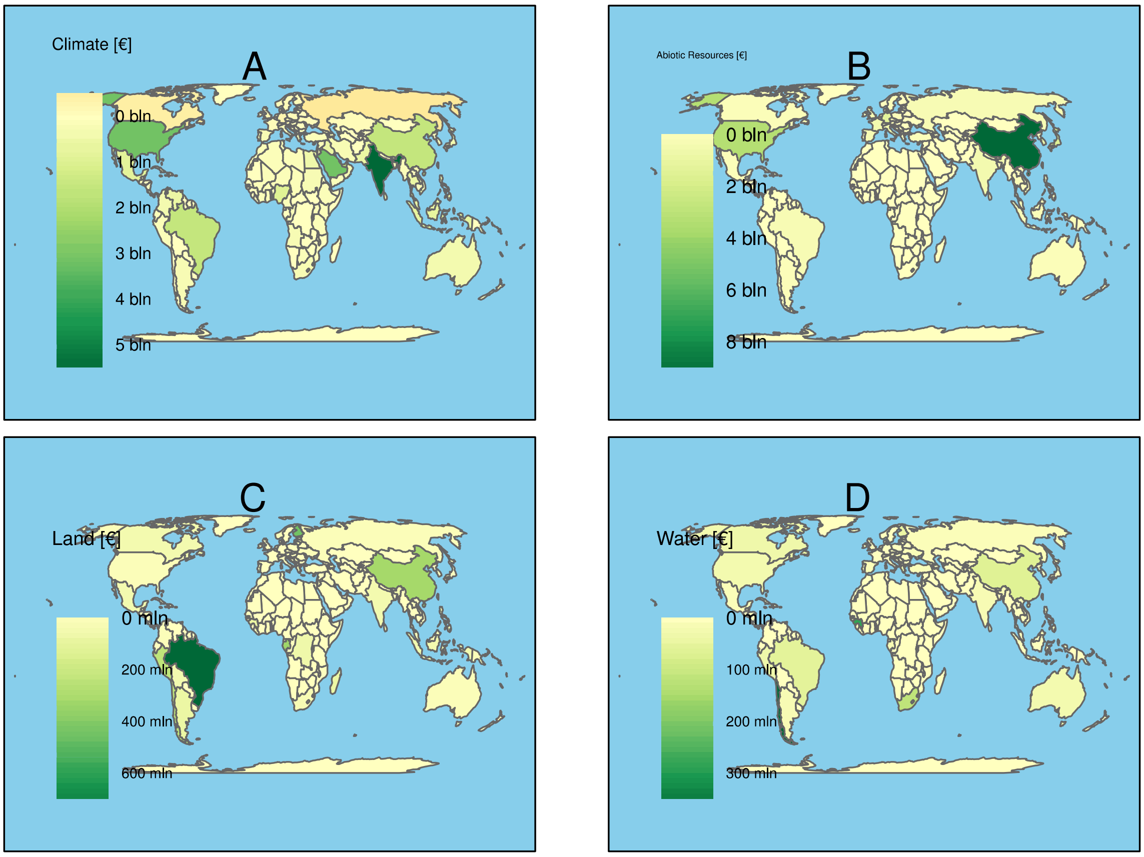


Figure S 88 Damages due to climate change, resource depletion, land use and water use for maximum demand scenario without recycling and with 1% discount rate

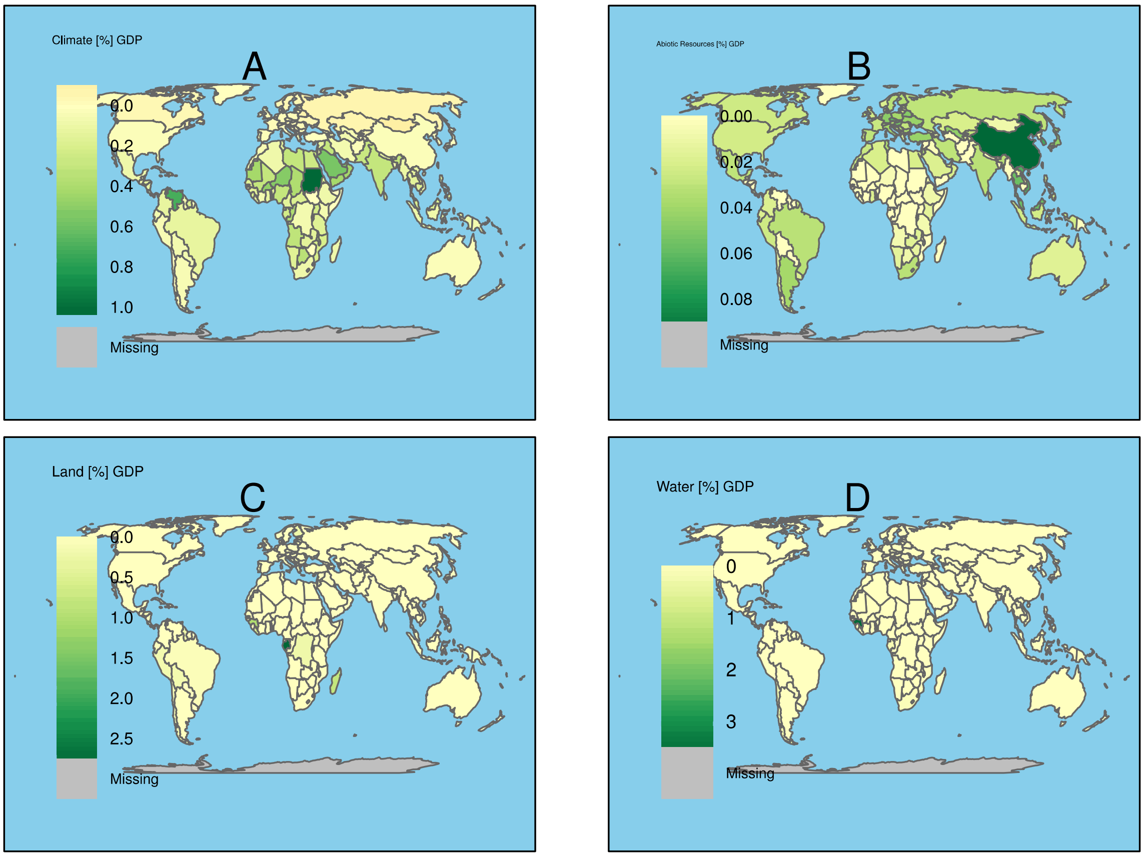


Figure S 89 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for maximum demand scenario without recycling and with 1% discount rate

6.3.1.3. 3 % discount rate

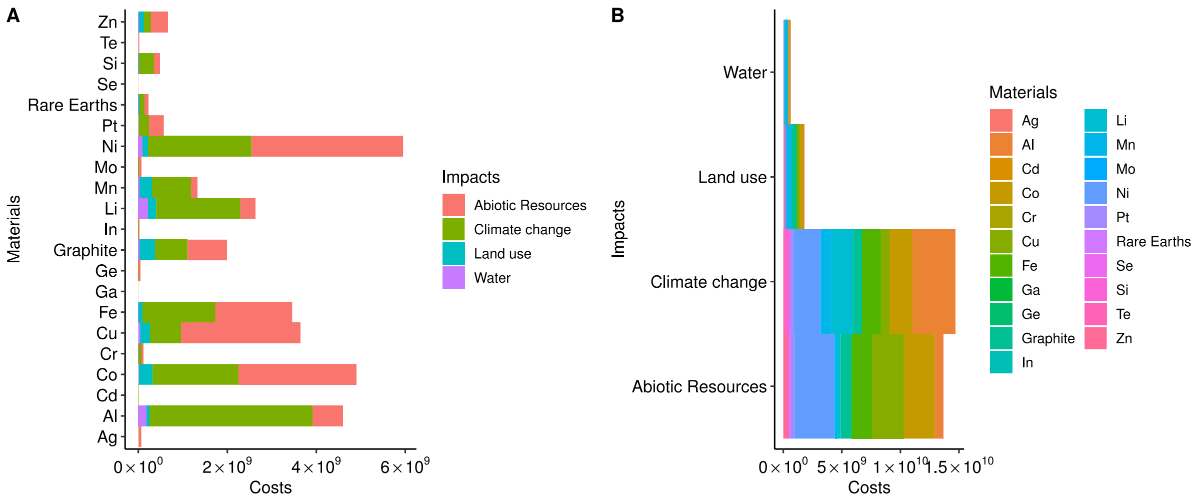
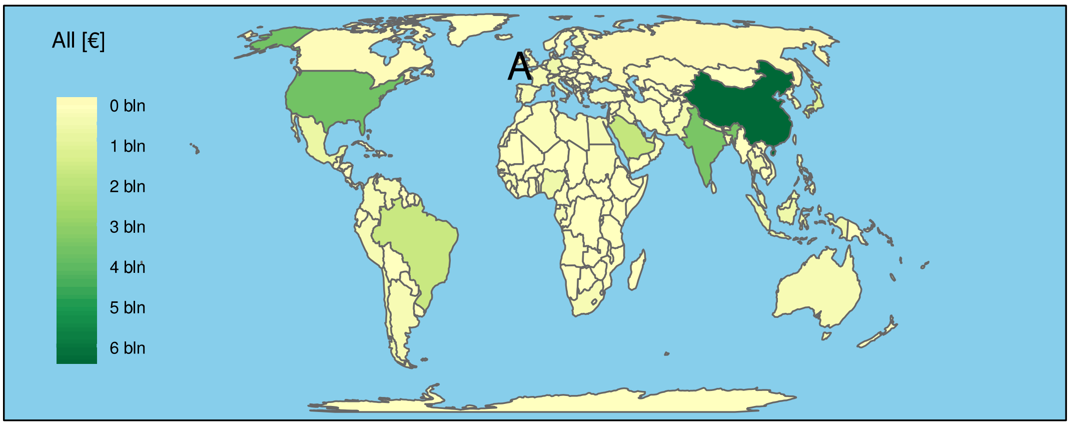


Figure S 90 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for maximum demand scenario without recycling and with 3% discount rate



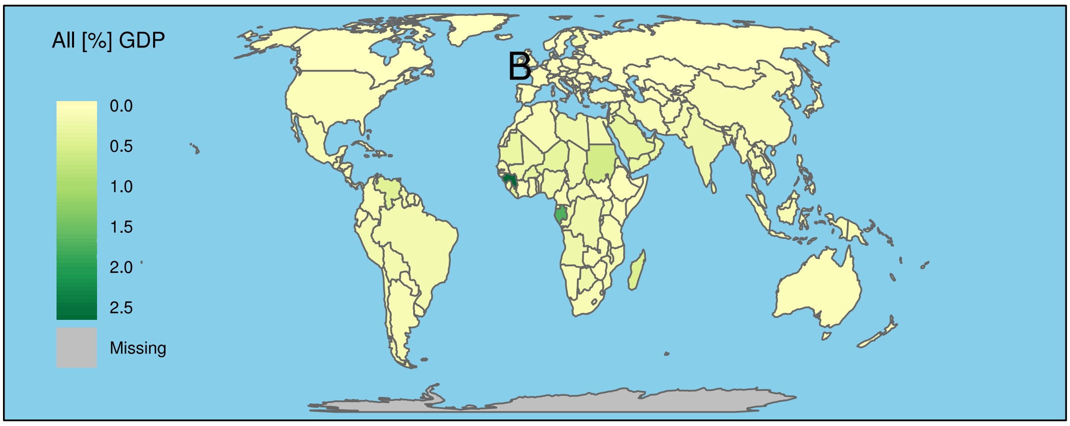


Figure S 91 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for maximum demand scenario without recycling and with 3% discount rate

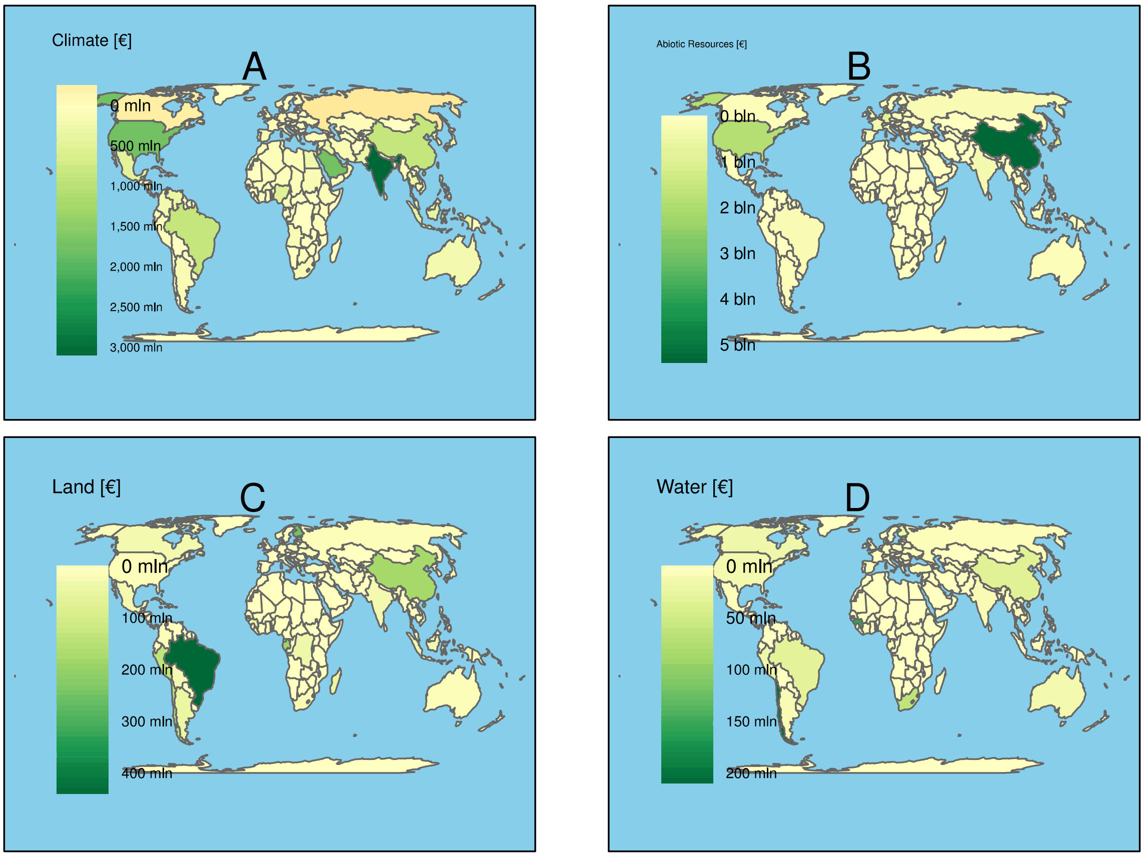


Figure S 92 Damages due to climate change, resource depletion, land use and water use for maximum demand scenario without recycling and with 3% discount rate

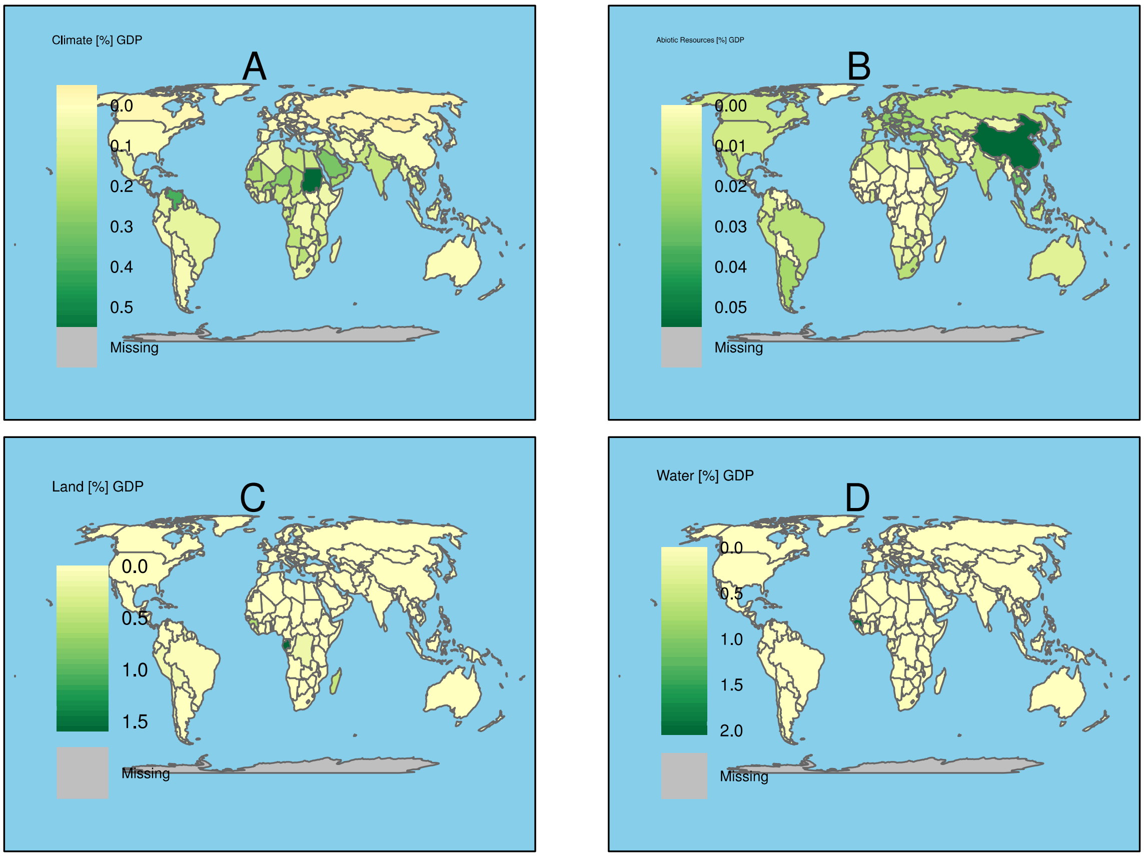


Figure S 93 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for maximum demand scenario without recycling and with 3% discount rate

6.3.2. Recycling

6.3.2.1. No discounting

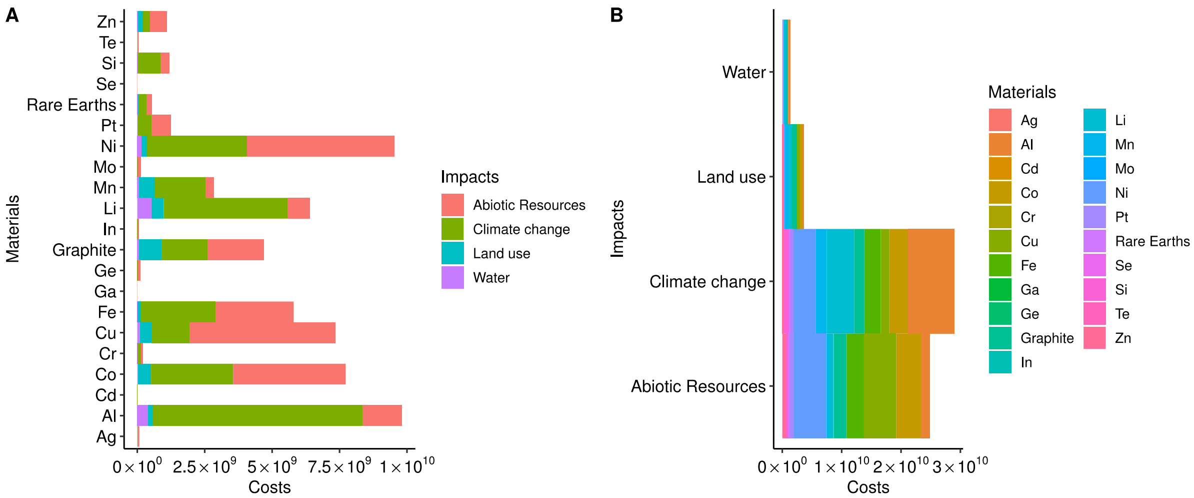


Figure S 94 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for maximum demand scenario with recycling and without discount rate

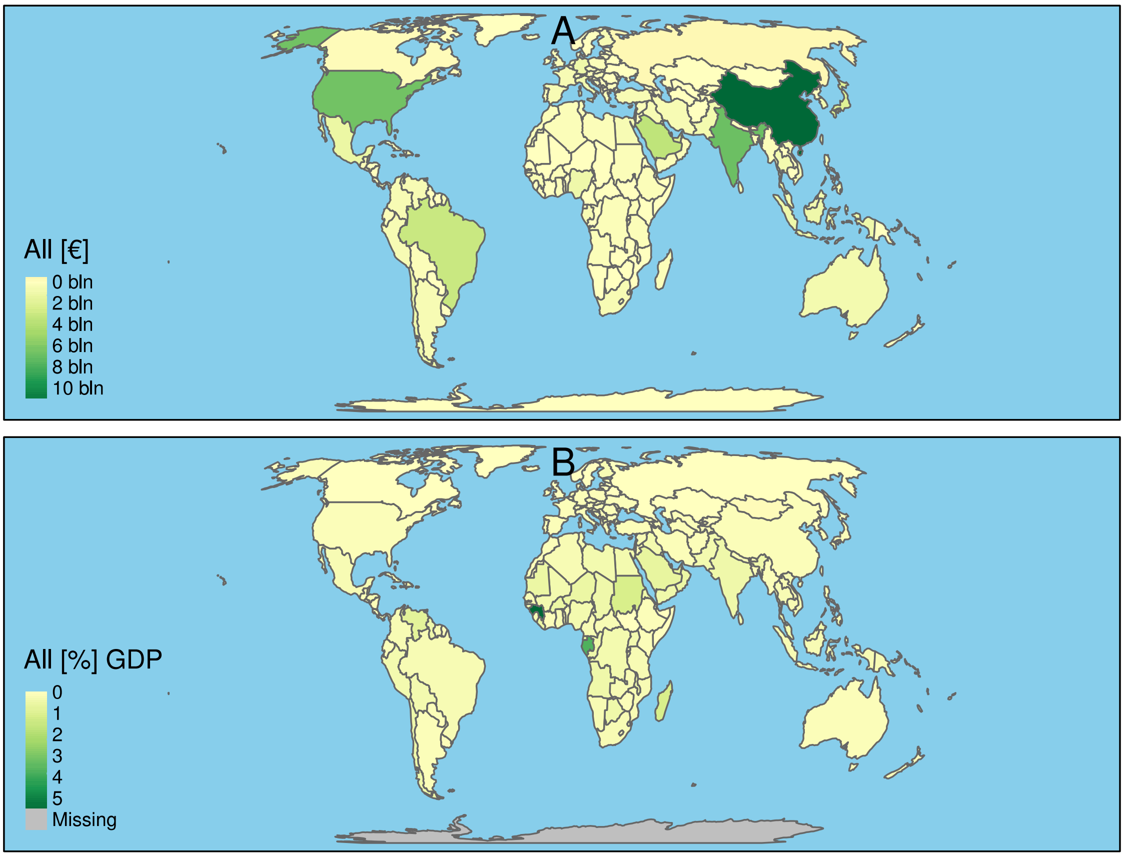


Figure S 95 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for maximum demand scenario with recycling and without discount rate

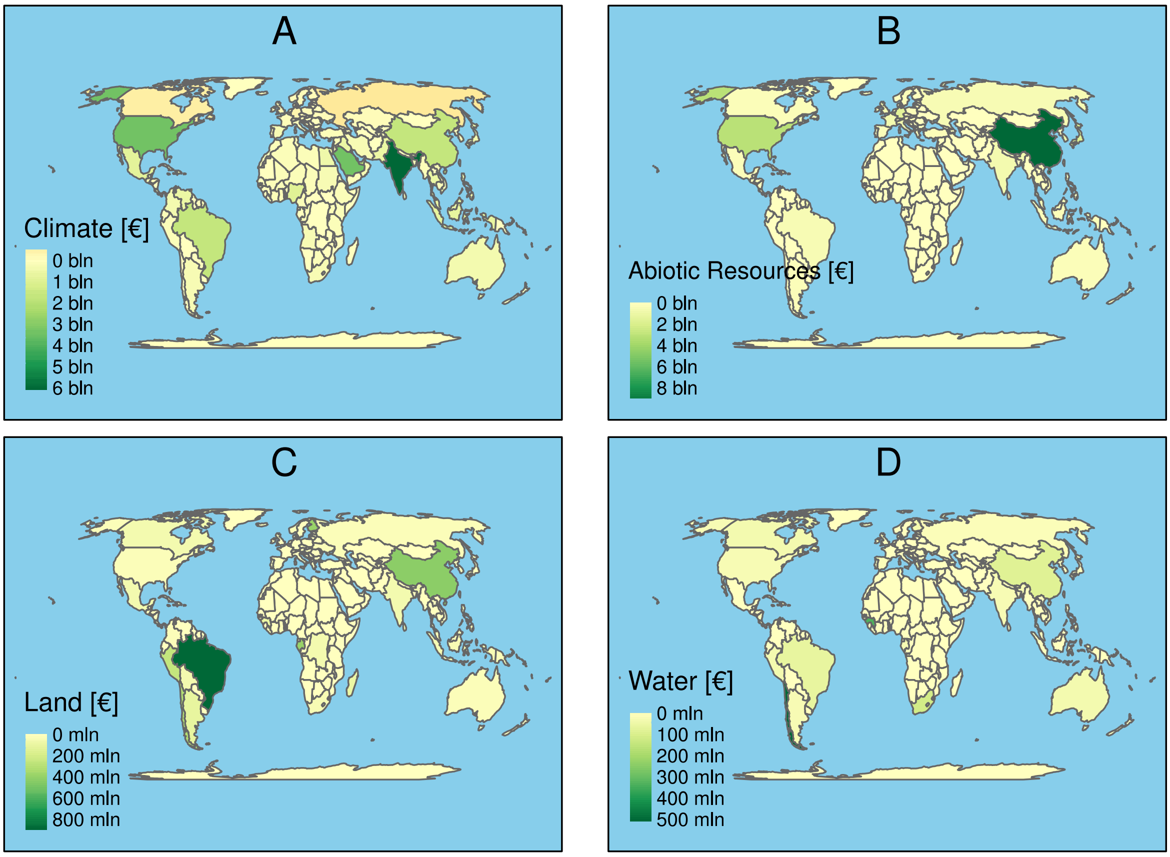


Figure S 96 Damages due to climate change, resource depletion, land use and water use for maximum demand scenario with recycling and without discount rate

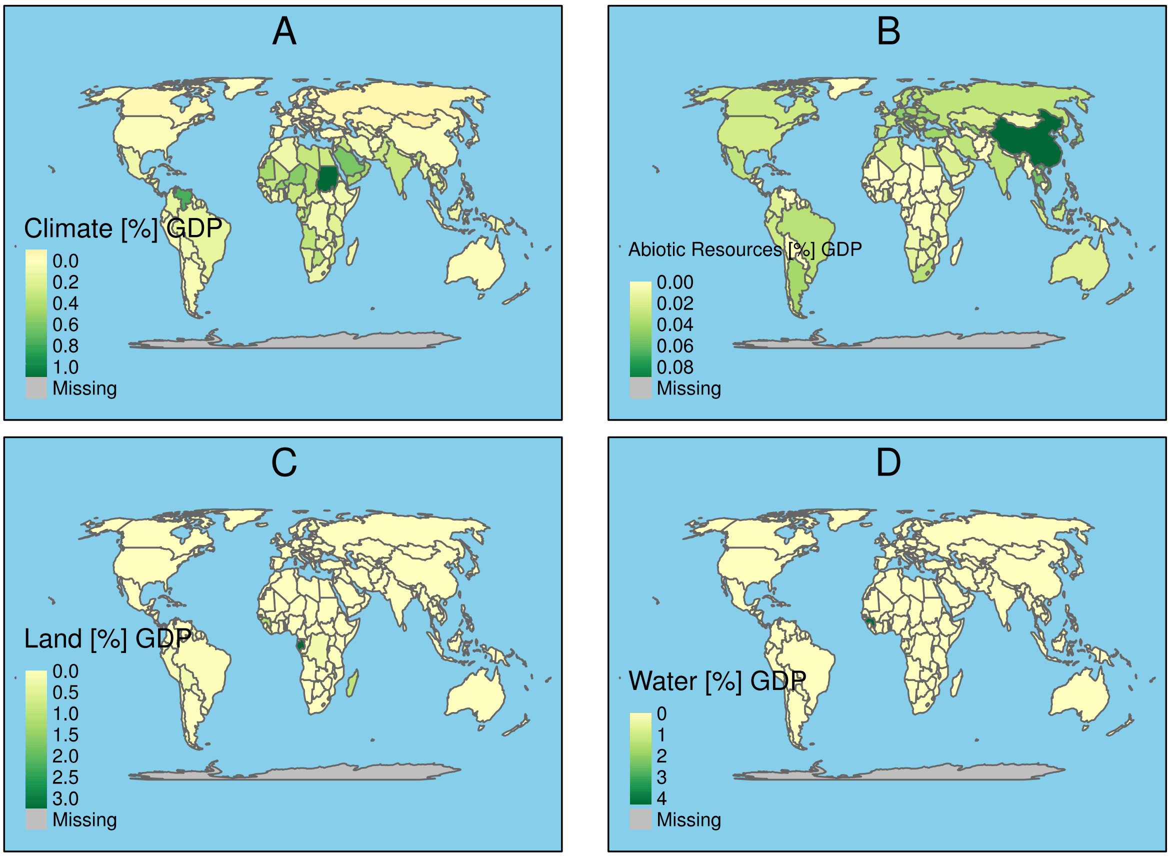


Figure S 97 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for maximum demand scenario with recycling and without discount rate

6.3.2.2. 1% discount rate

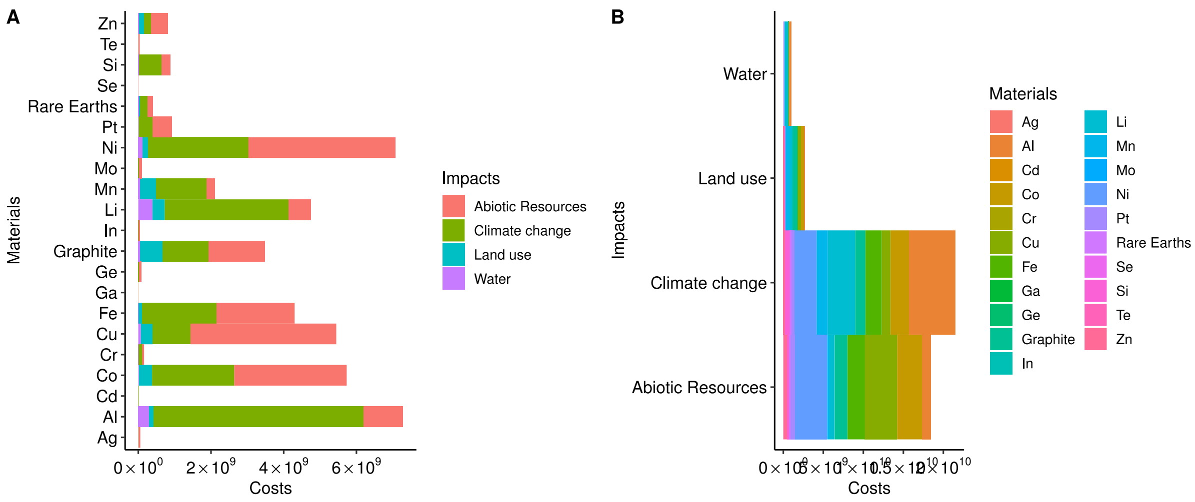


Figure S 98 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for maximum demand scenario with recycling and with 1% discount rate

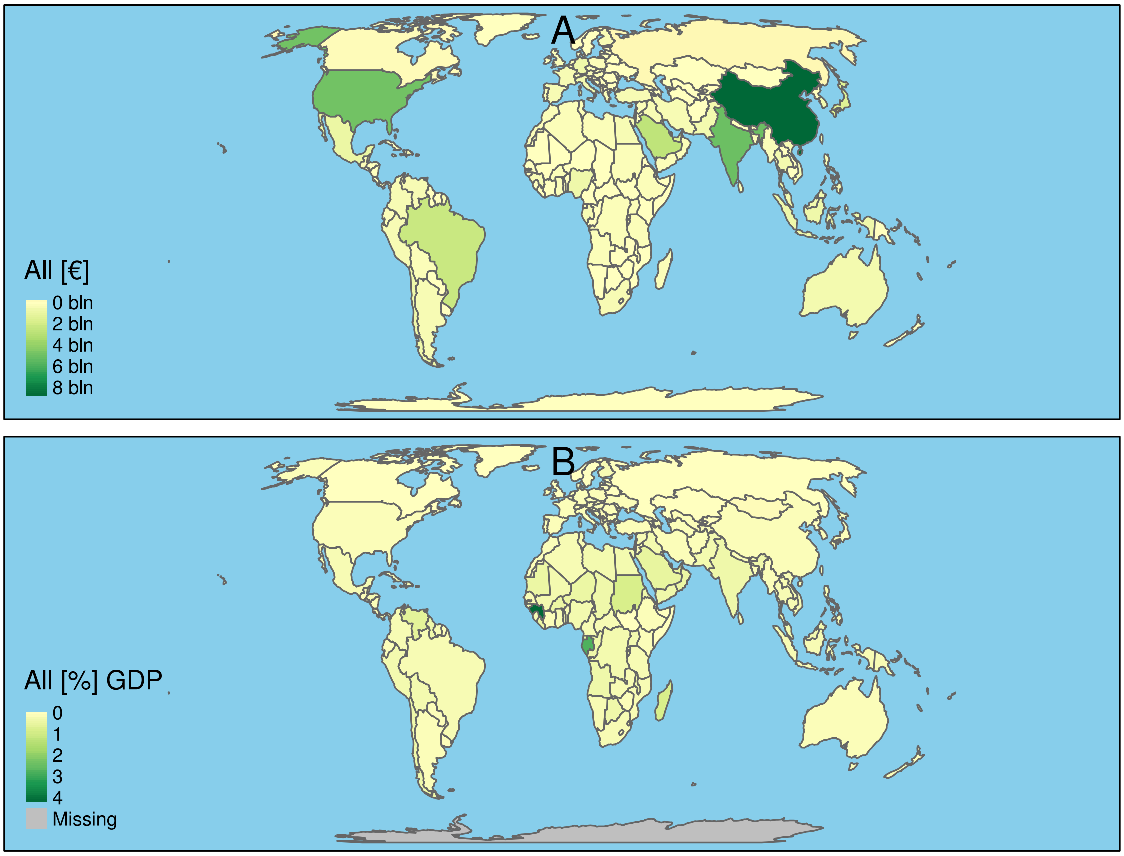


Figure S 99 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for maximum demand scenario with recycling and with 1% discount rate

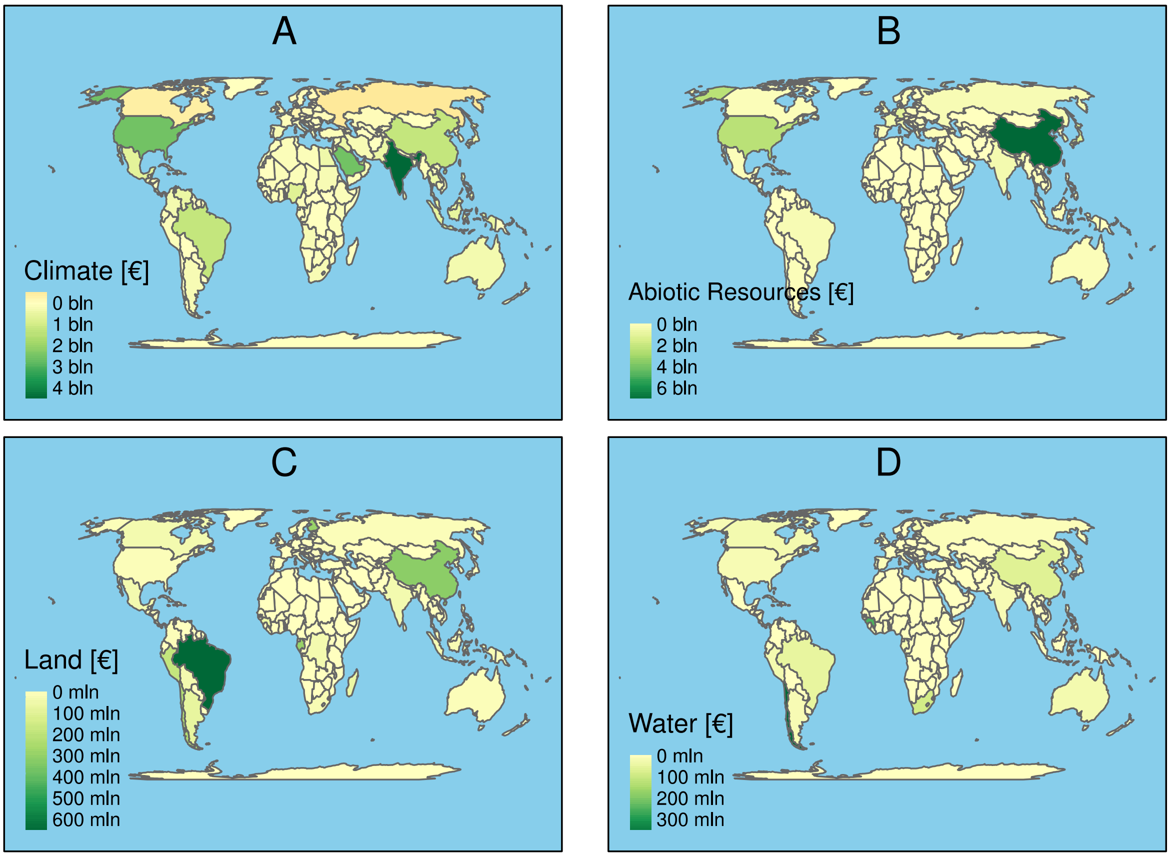


Figure S 100 Damages due to climate change, resource depletion, land use and water use for maximum demand scenario with recycling and with 1% discount rate

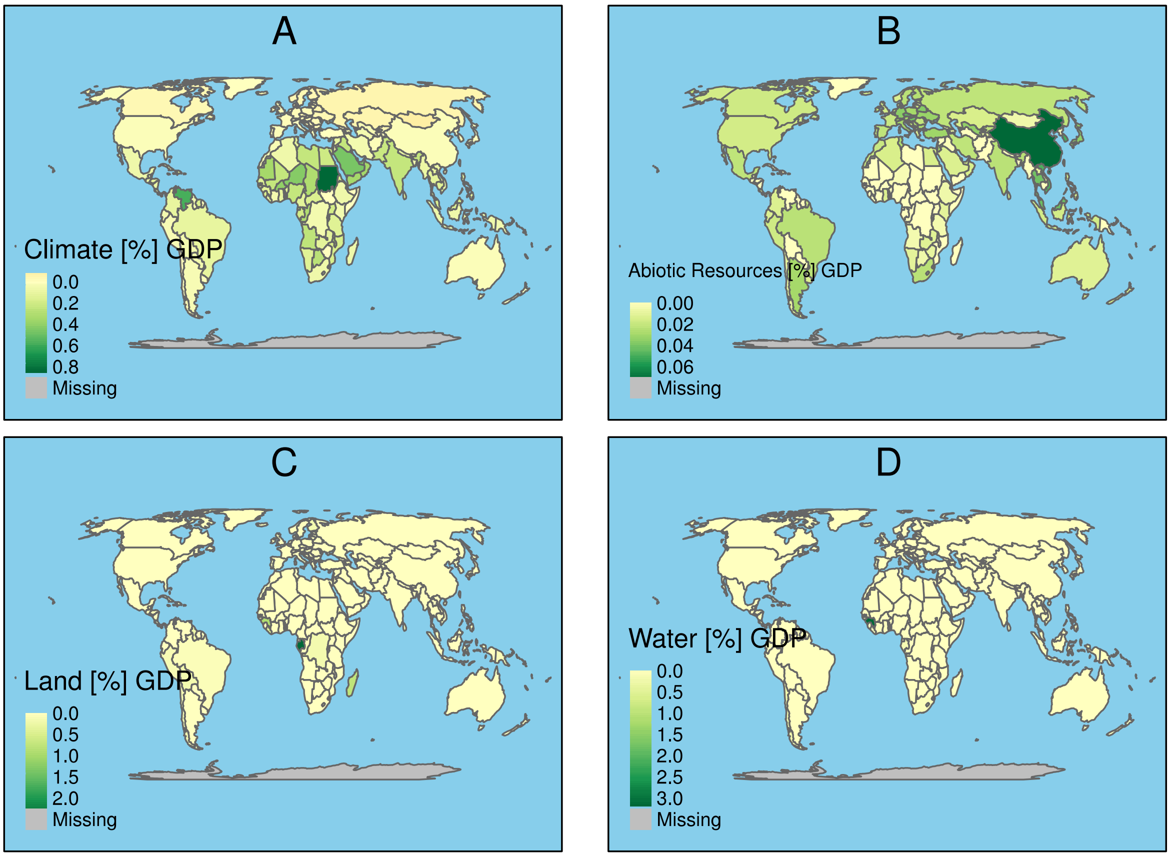


Figure S 101 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for maximum demand scenario with recycling and with 1% discount rate

6.3.2.3. 3 % discount rate

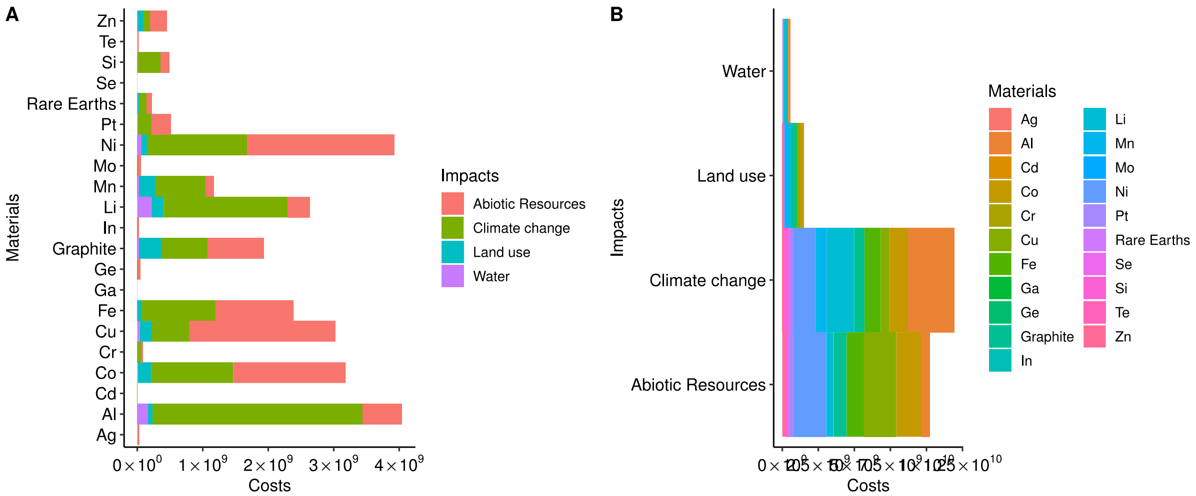


Figure S 102 Costs per material, distribution of costs per impact, Cost per impact and distribution of materials based in the impacts for maximum demand scenario with recycling and with 3% discount rate

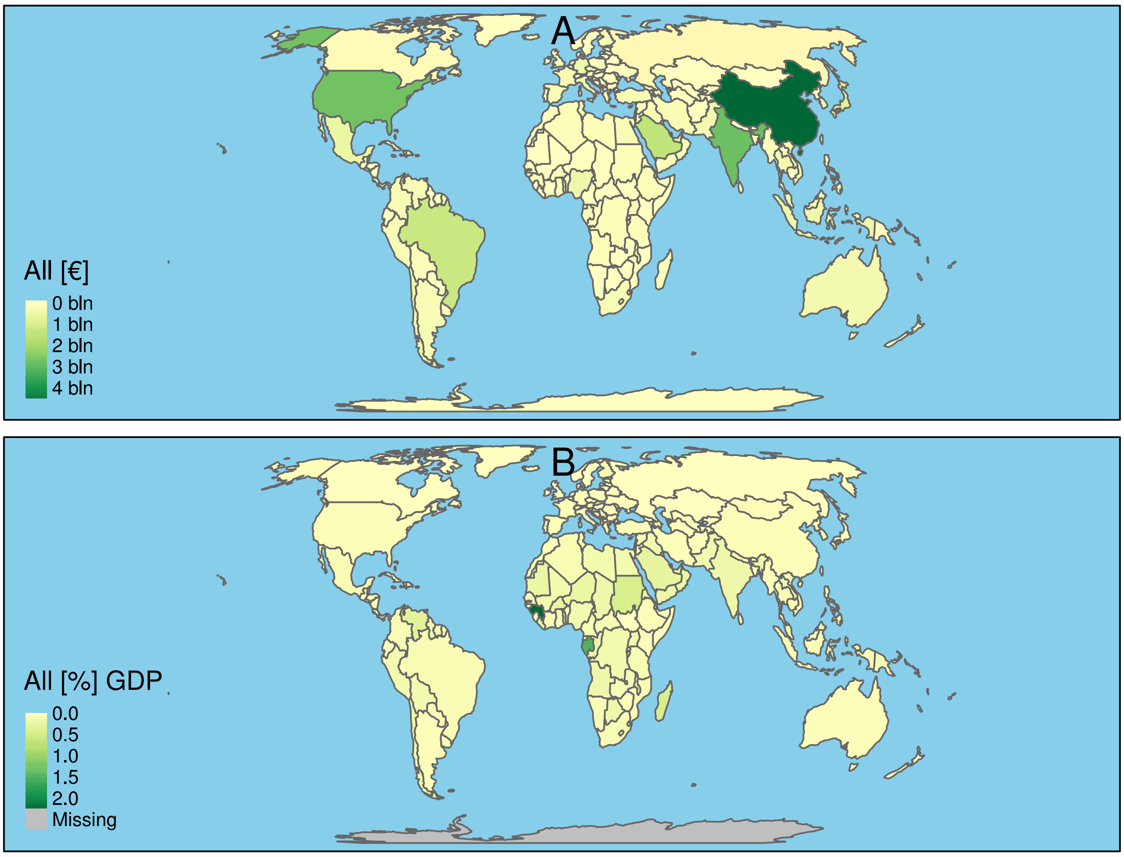


Figure S 103 Total environmental costs including damages due to climate change, water use, land use and material depletion in €$_{2018}$ (A) as well as the ratio of the environmental costs in [%] (B) for maximum demand scenario with recycling and with 3% discount rate

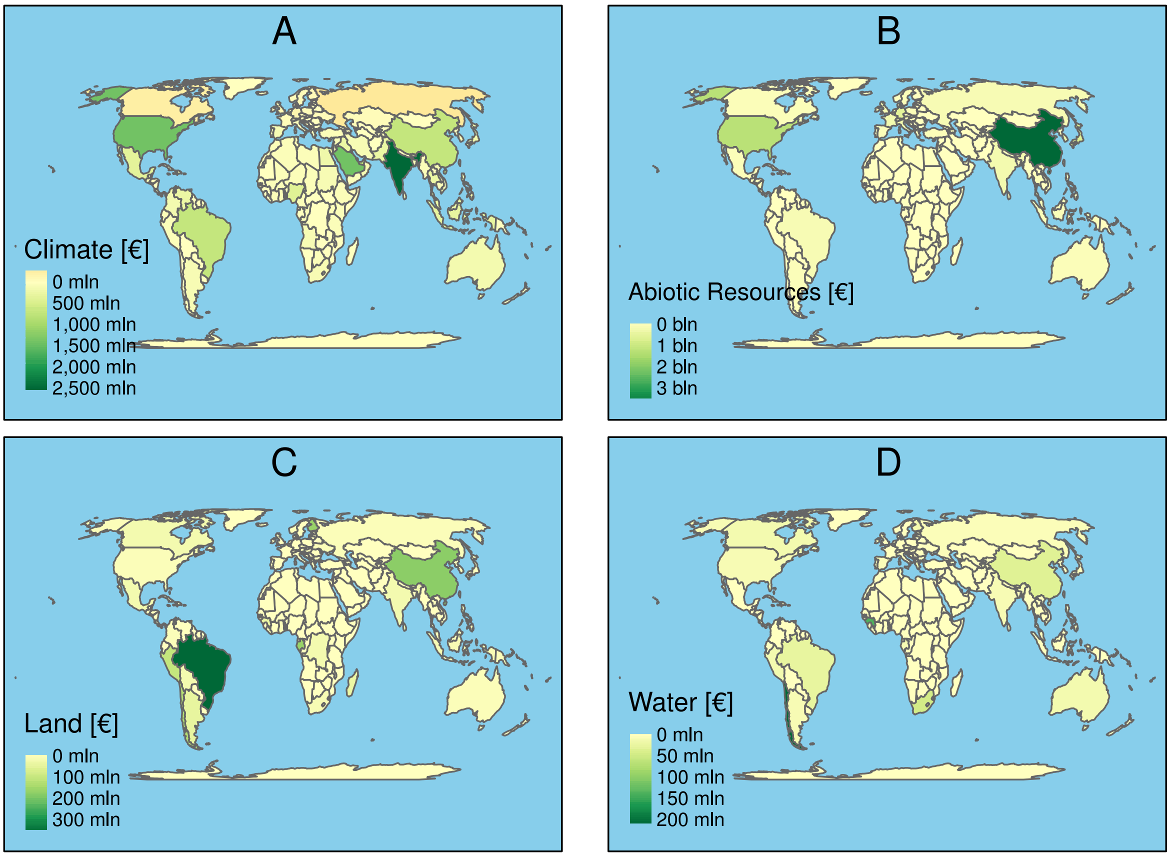


Figure S 104 Damages due to climate change, resource depletion, land use and water use for maximum demand scenario with recycling and with 3% discount rate

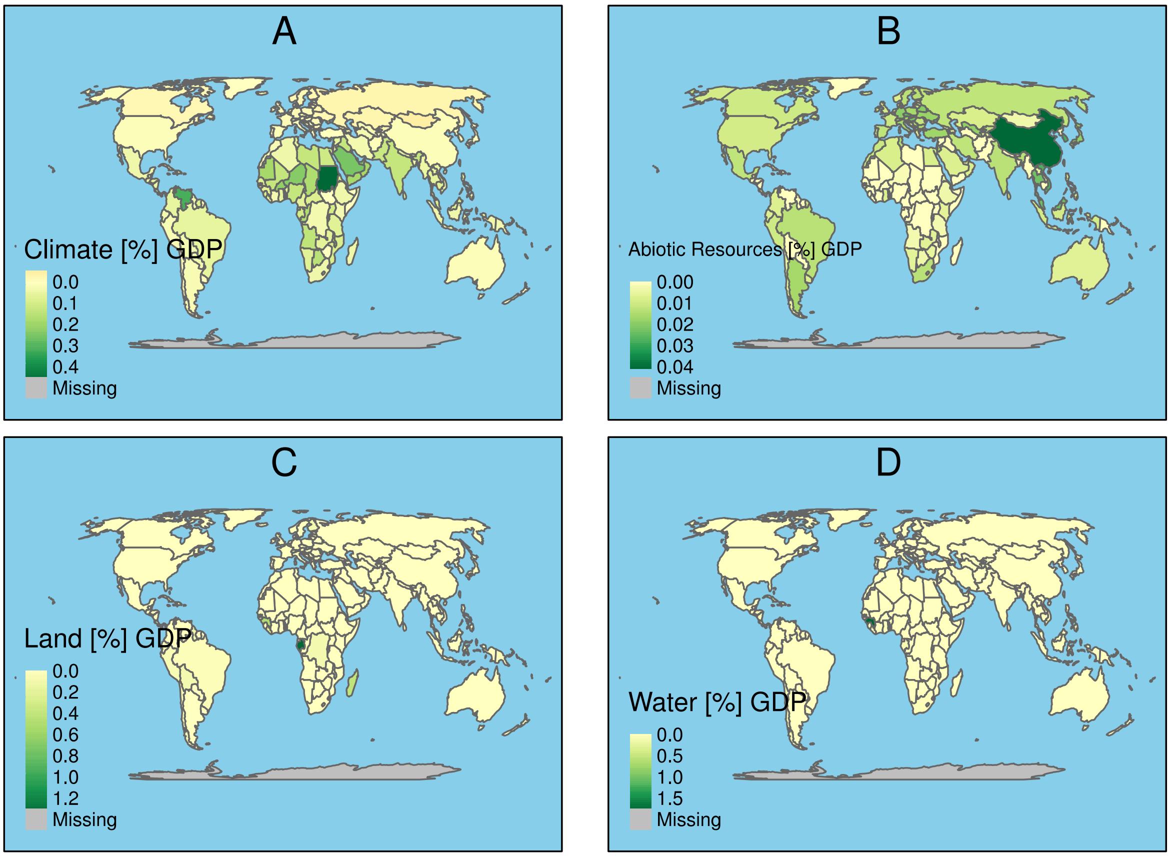


Figure S 105 Damages related to GDP for climate change, abiotic resource depletion. land use and water use for maximum demand scenario with recycling and with 3% discount rate

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Supplementary Material 2 Environmental Costs of Abiotic Resource Demand for the EU's Low-Carbon Development

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This supplementary material contains:

- Material demands and their link to certain technologies (Table S 14)
- Derived cost factors that were described in the method part of the main paper (Table S 15)
- Intermediate results for environmental quantities (Table S 16)
- Cost distribution per material (Table S 17)
- Cost distribution per country and related to GDP (Table S 18)

Table S 14: Material demands and associated technologies

Assessed material	Low demand [tonnes]	Medium demand[tonnes]	High demand[tonnes]	Technology
Co	38000	110000	290000	emobility
Li	48000	130000	260000	emobility
Ni	420000	1100000	1500000	emobility
Mn	19000	83000	260000	emobility
Graphite	700000	1800000	2700000	emobility
Co	63000	44000	140000	renewable energy batteries
Li	45000	28000	77000	renewable energy batteries
Ni	160000	97000	270000	renewable energy batteries
Mn	59000	41000	130000	renewable energy batteries
Graphite	480000	290000	780000	renewable energy batteries
Pt	1	2	10	fuelcells renewables
Aluminium	22000	63000	140000	wind turbines

Borates	10	80	360	wind turbines
Cr	10000	32000	74000	wind turbines
Cu	47000	150000	350000	wind turbines
Dysprosium	30	310	1400	wind turbines
Mn	15000	48000	110000	wind turbines
Mo	2100	6600	15000	wind turbines
Neodymium	260	2900	13000	wind turbines
Ni	7400	21000	48000	wind turbines
Praseodymium	40	510	2300	wind turbines
Fe	380000	1200000	2800000	wind turbines
Terbium	10	110	450	wind turbines
Zn	110000	330000	760000	wind turbines
Aluminium	210000	420000	510000	traction motors emobility
Borates	160	360	440	traction motors emobility
Cu	130000	260000	320000	traction motors emobility
Dysprosium	500	1100	1400	traction motors emobility
Neodymium	2100	3300	4100	traction motors emobility
Praseodymium	690	1100	1400	traction motors emobility
Si	64000	140000	170000	traction motors emobility
Fe	140000	2900000	3500000	traction motors emobility
Aluminium	110000	410000	1300000	PV
Cd	1	30	600	PV
Cu	68000	250000	800000	PV
Ga	0	5	40	PV
Ge	0	10	100	PV
In	0	20	170	PV
Se	1	30	350	PV

Si	18000	109000	399000	PV
Ag	20	110	660	PV
Te	1	40	690	PV
Pt	5	30	50	fuel cells emobility

Table S 15: Cost factors derived for the method part

ISO3-digit Alpha	cost of one m3 of water (human health)[€/m³]	cost of species yr (for water environment)[€/species*yr]	cost of one m3 blue water (economic costs)[€/m³]	land use costs [€/m²*yr]
ABW		64728104,99	1,929406975	6,856319022
AFG	0,139278383	39621050,29	1,478264815	0,10087346
AGO	0,139278383	62855627,19	1,197943754	0,447160596
AIA		16982591,8	0,86	1,341534671
ALB	0,139278383	34392171,12	1,112445066	0,5383
AND	0,139278383	34088155,26	0,86	0,5383
ANT		43851351,35	1,937247774	
ARE	0,139278383	26221356,81	2,84636194	0,077010918
ARG	0,139278383	60981834,52	0,621867614	2,694358638
ARM	0,030508397	58740167,13	1,324418686	0,35546952
ASM		0	0,00325672	11,9076
ATA		0	0,86	
ATF		0	0,86	0,082176611
AUS	0	55941667,98	0,70192144	0,24328273
AUT	0	26718628,91	0,015150661	0,5383
AZE	0,032652328	52469525,39	1,453363025	0,239905128
BDI	0,038385262	67682305,91	0,503337978	3,348583463
BEL	0	17424838,95	0,721855371	0,5383
BEN	0,015732691	72254408,01	1,20311588	0,306219064
BES		41486694,7	0,86	4,596501208
BFA	0,021200063	70696528,27	1,577031408	0,1597
BGD	0,352748286	30429350,7	0,815805811	11,44255857
BGR	0,006492681	40582329,13	0,846582914	0,536477534
BHR	0,139278383	21738035,93	2,618060581	0,0769
BHS	0,139278383	49712077,07	0,082483811	9,894593856
BIH	0,000824552	36861329,22	0,078297789	0,5383
BLM	0	59486214,85	0,8375	7,8052
BLR	0,001443889	32116204,04	0,274927084	0,5383
BLZ	0,139278383	75627977,64	0,295820879	11,40551257
BMU		0	0,060608985	11,9076
BOL		64545963,7	0,372419385	7,902014216
BRA	0,002348686	71475378,05	0,494020609	7,749847752
BRB	0,139278383	17632547,73	0,127171731	0,5383

ISO3-digit Alpha	cost of one m3 of water (human health)[€/m³]	cost of species yr (for water environment)[€/species*yr]	cost of one m3 blue water (economic costs)[€/m³]	land use costs [€/m²*yr]
BRN	0,139278383	43851351,35	0,003295745	
BTN	0,139278383	57259972,66	0,256832186	1,885987689
BWA	0,062721434	59350645,91	2,045260406	0,507706803
CAF	0,005898192	65640291,55	0,762762018	1,390661205
CAN	0	26029488,91	0,019348723	0,367174264
CCK	0	0	0,8375	11,9076
CHE	0	24971265,6	0,06353578	0,5383
CHL	0	51242741,47	1,540379484	0,393655781
CHN	0,119136982	48123019,61	1,302803762	2,217180389
CIV	0,139278383	69500010,63	0,635662968	5,617336973
CMR	0,006210508	69096123,48	0,992253003	6,335263112
COD	0,139278383	70053837,4	0,489049493	5,966293558
COG	0,006187792	78541955,19	0,454223756	8,09575645
COK	0,139278383	0	0,070710262	11,9076
COL	0,013751655	68672595,88	0,162315536	9,849131035
COM	0	37060358,72	0,320238216	11,9076
CPV	0,139278383	43851351,35	2,374190918	
CRI	0,000460318	62465549,26	0,296267517	11,80687049
CSK	0	23238506,78	0,085345707	0,5383
CUB	0,139278383	44074489,09	1,251096029	10,91942088
CUW		35793472,87	0,8375	2,974645957
CXR		0	0,058024071	11,9076
CYM		49883285,63	0,492541967	9,699682165
CYP	0	26030864,26	1,613973454	0,5383
CZE	0	23238506,78	0,085345707	0,5383
DEU	0	20067459,44	0,061650367	0,5383
DJI	0,139278383	32740668,53	3,051345025	0,0769
DMA	0,139278383	25401716,68	0,186081104	9,080987152
DNK	0	16086526,03	0,410581014	0,5383
DOM	0,047507934	61821752,43	0,805308337	11,74697191
DZA	0,169557378	27054752,48	1,763086042	0,166616263
ECU	0,054609222	77091075,47	0,65407493	10,65142732
EGY	0,676955804	24227347,85	2,619642979	0,427817209
ERI	0,269756461	50004222,08	2,669256037	1,649936217
ESH	0	26876656,3	1,771787266	0,086727127
ESP	0	33202099,34	1,21287297	0,5383
EST	0	39219875,18	0,020021137	0,5383
ETH	0,278232141	61786415,17	1,115949654	2,525221849
FIN	0	54998878,83	0,20672322	0,533918734
FJI	0,001151343	54008700,41	0,003489719	11,9076
FLK	0	74307226,07	0,113053966	0,1597
FRA	0	19517609,04	0,550754947	0,5383

ISO3-digit Alpha	cost of one m3 of water (human health)[€/m³]	cost of species yr (for water environment)[€/species*yr]	cost of one m3 blue water (economic costs)[€/m³]	land use costs [€/m²*yr]
FRO		32745272,31	0,003253342	0,1597
FSM	0,139278383	0	0,86	11,9076
GAB	0,007489445	74981439,83	0,392085241	9,761332989
GBR	0	24198450,97	0,154972742	0,5383
GEO	0,008468588	58042188,4	0,759654357	0,462771784
GHA	0,032707066	69374106,78	0,850938003	4,111757404
GIB		31869426,34	0,86	0,5383
GIN	0,027873198	72939298,22	1,583871599	2,534523182
GLP		29416049,6	0,201985637	6,367734712
GMB	0,006702446	71677895,92	2,015317926	0,868876994
GNB	0,00485661	70966550,24	1,805789239	2,193086102
GNQ	0,139278383	77377576,64	0,274025915	11,76599882
GRC	0	36300321,82	1,324725052	0,5383
GRD	0,139278383	23908780,73	0,383619269	10,82033642
GRL		1682565,02	0,8375	0,077952235
GTM	0,004210834	75982092,63	0,706813543	11,59716358
GUF		70359793,05	0,239654527	11,7929268
GUM		0	0,00325651	11,9076
GUY	0,013117609	69678449,06	0,153766871	11,13675484
HKG	0,119136982	48123019,61	0,224162964	2,217180389
HMD		0	0,8375	0,0818
HND	0,009726616	71059073,52	0,710749141	11,79479238
HRV	0	30909032,38	0,8375	0,5383
HTI	0,022766748	62039357,43	0,67788716	11,7621516
HUN	2,66873E-05	27912333,48	0,047077869	0,5383
IDN	0,184057157	67468006,95	0,274969015	11,69156377
IND	1,612202419	39875194,62	2,065500621	8,123057712
IOT		11804688,33	0,8375	11,9076
IRL	0	26789562,85	0,049904933	0,5383
IRN	0,139278383	36939008,68	1,743273764	0,241506988
IRQ	0,139278383	32777271,7	1,711129306	0,420036829
ISL		46694375,83	0,022822383	0,490530884
ISR	0,139278383	32865542,62	1,900769535	0,255611765
ITA	0	19768535,63	0,941555431	0,5383
JAM	0,000180644	54183881,9	0,941555431	11,74708002
JOR	0,056686054	30914360,73	2,008569492	0,126217695
JPN	0	25665126,22	0,171692474	0,652903507
KAZ	0,005085367	39734515,77	0,897845103	0,113571604
KEN	0,019146465	63180656,43	0,842914102	1,715463035
KGZ	0,611912119	44519974,7	1,320441196	0,171791612
KHM	0,38270203	53822332,31	0,807382957	11,88381355
KIR	0,139278383	0	0,003253298	11,9076

ISO3-digit Alpha	cost of one m3 of water (human health)[€/m³]	cost of species yr (for water environment)[€/species*yr]	cost of one m3 blue water (economic costs)[€/m³]	land use costs [€/m²*yr]
KNA	0,139278383	50851035,77	0,062587974	8,341608863
KOR	0	38096206,28	0,088147546	0,5383
KWT	0,139278383	22077734,67	2,413685633	0,0769
LAO	0,139278383	66662326,54	0,790201954	11,9076
LBN	0,037511039	42634604,4	1,61793608	0,5383
LBR	0,011061043	72057865,35	0,469432219	11,83319881
LBY	0,139278383	26490122,83	2,172089534	0,101334445
LCA	0,139278383	19043636,12	0,188610703	11,55929614
LKA	0,204930675	51863776,97	0,724382456	11,43197713
LSO	0,417766388	59196008,13	1,990534718	0,1597
LTU	0	35046750,11	0,00780616	0,5383
LUX	0	20289055,82	0,054919441	0,5383
LVA	0	39219875,18	0,021007798	0,5383
MAC	0,119136982	0	1,302803762	11,9076
MAR	0,168789837	42985179,79	1,771787266	0,438264709
MDA	0	32961763,96	0,704155573	0,452949093
MDG	0,081137942	97170474,96	0,784698613	9,39175604
MDV	0,139278383	11804688,33	0,543524987	11,9076
MEX	0,048153164	60273336,17	1,799343632	6,857354738
MHL	0,139278383	0	0,03788664	11,9076
MKD		0	1,13916166	11,9076
MLI	2,256312184	47128539,44	2,542773991	0,297530065
MLT	0,139278383	23990090,96	1,587522289	0,5383
MMR	0,139278383	43851351,35	0,997838912	11,45504943
MNE	0	38901547,42	0,86	0,5383
MNG	0,006392443	42960560,47	0,360282134	0,16241065
MNP	0	0	0,8375	11,9076
MOZ	0,028593614	67693398,22	0,972579751	2,56624122
MRT	0,121084805	36895460,43	2,793911281	0,132289935
MSR	0	17932604,87	0,00328183	9,252089732
MTQ	0	24805230,43	0,058763707	8,916657523
MUS	0	0	0,145453871	11,9076
MWI	0,003647571	55765706,57	1,49171922	0,43307739
MYS	0,003647571	75161490,81	0,086998326	11,65022764
MYT	0	37060358,72	0,8375	11,9076
NAM	0,139278383	51421664,28	2,176084055	0,199685143
NCL	0	68737009,27	0,199771816	11,9076
NER	0,286070191	41079948,4	2,31344424	0,128056275
NFK	0	0	0,8375	11,9076
NGA	0,139278383	69111078,97	1,339890278	1,967123866
NIC	0,010771708	62039241,78	1,004060676	11,02540522
NIU	0,139278383	0	0,003253363	11,9076

ISO3-digit Alpha	cost of one m3 of water (human health)[€/m³]	cost of species yr (for water environment)[€/species*yr]	cost of one m3 blue water (economic costs)[€/m³]	land use costs [€/m²*yr]
NLD	0	16444116,7	0,369123336	0,5383
NOR	0	52542189,06	0,003946332	0,308750953
NPL	2,563566227	50868416,33	1,924319848	4,332269885
NRU	0,139278383		0,003255309	
NZL	0,139278383	47296782,7	0,048579433	0,404816651
OMN	0,139278383	24777805,32	3,010066414	0,083736202
PAK	1,210736902	37243639,47	2,085816885	0,328216815
PAN	0,000333436	65574033,21	0,480662095	11,70577855
PCI	0	0	0,8375	11,9076
PCN	0	0	0,8375	11,9076
PCZ	0	0	0,8375	11,9076
PER	0,803547698	71984238,54	1,388892936	8,565974696
PHL	0,421305829	58718830,61	0,246705904	11,9076
PLW	0,139278383	0	0,8375	11,9076
PNG	0,139278383	68587529,11	0,01087133	11,2499266
POL	0	25924149,17	0,02508166	0,5383
PRK	0,139278383	53437214,42	0,184322937	0,5383
PRT	0	31441727,1	1,188911389	0,5383
PRY	0,002222984	68116724,57	0,166112974	7,843233762
PSE	0	57735587,56	2,017577549	11,9076
PYF	0	0	0,8375	11,9076
QAT	0,139278383	25032606,3	2,733199011	0,0769
REU	0	0	0,119304641	11,9076
ROU	0,001224357	33585529,22	0,351605276	0,498895589
RUS	0	54600964,35	0,176929152	0,415476531
RWA	0,091741978	58135859,52	0,284544581	5,262235059
SAU	0,139278383	26862975,54	2,85368452	0,0769
SCG	0,001316635	36058677,79	0,301820205	0,5383
SDN	0,297505485	49089856,99	1,559235431	0,505444158
SEN	0,070668359	68133854,42	2,44772474	0,220851756
SGP	0,139278383	63539291,96	0,004895521	11,9076
SGS	0	0	0,8375	0,0818
SHN	0	0	0,686501537	0,1597
SLB	0	78773481,27	0,003253288	11,9076
SLE	0,018824194	71636924,02	1,020882081	8,484357496
SLV	0,002146401	66435197,76	1,25860577	11,73953877
SMR	0,139278383	17974579,41	0,8375	0,5383
SOM	0,139278383	61090643,83	2,037344772	0,709151421
SPM	0	0	0,003253332	1
SRB	0,001316635	36058677,79	0,301820205	0,5383
SSD	0,297505485	49089856,99	0,8375	0,505444158
STP	0,139278383	43539714,35	1,247353453	11,9076

ISO3-digit Alpha	cost of one m3 of water (human health)[€/m³]	cost of species yr (for water environment)[€/species*yr]	cost of one m3 blue water (economic costs)[€/m³]	land use costs [€/m²*yr]
SUN	0	0	0,8375	0
SUR	0,002311484	69525700,09	0,256052419	11,74550168
SVK	0	31417037	0,035296587	0,5383
SVN	0	31812912,41	0,044004227	0,5383
SWE	0	49211409,39	0,124532176	0,48848555
SWZ	0,00083587	62983779,08	0,308835661	2,257455714
SXM	0	17632547,73	0,8375	0,5383
SYC	0,139278383	0	0,228919288	6,840668969
SYR	0,139278383	36858084,37	0,8375	0,202515438
TCA	0	48942228,65	3,253275569	10,0591625
TCD	0,038208869	43474960,92	1,872155168	0,162042451
TGO	0,006836955	71648382,25	1,060859909	1,454243878
THA	0,058410033	53391402,2	1,293517343	11,81394145
TJK	0,177389777	37504842,5	1,390246483	0,152979536
TKL	0		0,8375	
TKM	0,077306513	30523258,89	2,002392547	0,081776095
TLS	0,139278383	53866923,14	0,8375	11,9076
TON	0,139278383	0	0,251024635	11,9076
TTO	0	34756330,27	0,276754431	11,76156495
TUN	0,025723771	41371371,45	1,562051695	0,652595719
TUR	0,013366047	45906668,61	1,257023983	0,437774298
TUV	0,139278383	0	0,8375	11,9076
TZA	0,027389546	66600154,87	0,943888854	1,83782748
UGA	0,073642856	50240809,09	0,121489957	1,30788529
UKR	0,020626739	35071853,77	0,62805713	0,38543359
UMI	0		0,8375	
URY	0	61170366,1	0,013340325	0,160208001
USA	0	24595970,28	0,529706002	0,356670831
UZB	0,091034773	33100964,7	1,448976714	0,098165908
VAT	0	17974579,41	0,8375	0,5383
VCT	0,139278383	35227119,81	0,383619269	9,080897051
VDR	0	52927137	0,8375	11,69205194
VEN	0,002622608	53842113,22	0,562721654	7,519812049
VGB	0	23780349,75	0,07126147	4,677741113
VIR	0	23780349,75	0,8375	4,677741113
VNM	0,41718157	52927137	1,023115518	11,69205194
VUT	0,139278383	0	0,008348682	
WLF	0	56980848,71	0,8375	11,9076
WSM	0,139278383	43851351,35	0,003254464	
YEM	0	26663303,12	2,884239704	0,0769
YEM	0,971917804	26663303,12	2,884239704	0,0769
YMD	0	26663303,12	2,884239704	0,0769

ISO3-digit Alpha	cost of one m3 of water (human health)[€/m³]	cost of species yr (for water environment)[€/species*yr]	cost of one m3 blue water (economic costs)[€/m³]	land use costs [€/m²*yr]
YUG	0	0	0,8375	0
ZAF	0,707138552	54607105,22	1,479574989	0,446333724
ZMB	0,020288386	71100897,8	1,524371219	1,419094238
ZWE	0,612004913	62919670,11	1,506493411	0,1597

Table S 16: Intermediate environmental quantities per country; values per material are not disclosed due to licensing issues (Ecoinvent and GaBi database)

ISO country code	Country name	occupied land for mining and processing [m²*yr]	used blue water for mining and processing [m³ H2O]	Co2 emissions (origin) [t-Co2-e]
AFG	Afghanistan	0,19	0,02	0,01
ALB	Albania	24150,57	2288,01	857,49
ARG	Argentina	27648555,25	5590291,53	407699,11
ARM	Armenia	777601,41	219042,85	18697,14
AUS	Australia	42983281,94	11180693,50	1004223,24
AUT	Austria	908718,27	85446,64	149870,70
AZE	Azerbaijan	67,78	14,10	5,13
BEL	Belgium	46725,01	8827,17	890,42
BFA	Burkina Faso	483011,86	81705,74	3882,15
BGR	Bulgaria	3999160,05	914218,11	107429,58
BIH	Bosnia Herzegovina	439335,07	69968,26	12613,25
BOL	Plurinational State of Bolivia	5906785,56	999229,34	47481,71
BRA	Brazil	108290605,22	35705042,19	7549998,38
BTN	Bhutan	320490,00	348586,26	57195,03
BWA	Botswana	27997,88	877,36	311,91
CAN	Canada	189523340,69	108515576,26	9172661,82
CHL	Chile	535098966,86	103169806,69	7238794,91
CHN	China	172631601,82	25232286,70	3643360,57
CIV	Côte d'Ivoire	233151,34	33846,51	9070,38
COD	Democratic Republic of the Congo	7173777,25	155416,05	72559,72
COL	Colombia	750571,00	205928,87	17895,30
CUB	Cuba	318617,86	6380,74	3194,56
CYP	Cyprus	93,79	25,74	2,23
DEU	Germany	183415,77	19808,99	27114,00
DOM	Dominican Republic	323,88	88,87	7,71
DZA	Algeria	111,84	10,52	18,45
ECU	Ecuador	3615,11	992,00	86,07
EGY	Egypt	450,51	48,69	58,71

ERI	Eritrea	887313,68	243483,10	21126,52
ESP	Spain	5374020,83	2986459,51	418853,84
FIN	Finland	589731754,55	23364726,16	6705772,23
FRA	France, Monaco	3007915,74	3212854,42	526381,82
GAB	Gabon	23006264,53	3339812,81	895021,78
GEO	Georgia	3471675,64	952052,82	82728,03
GHA	Ghana	1342321,02	194864,36	52220,84
GIN	Guinea	27040830,01	76874180,09	10721925,12
GRC	Greece	19206747,15	11878053,54	863167,17
GTM	Guatemala	195743,36	33111,73	1573,26
GUY	Guyana	599675,59	1704813,41	237776,61
HND	Honduras	512556,38	86703,46	4119,61
HUN	Hungary	1411189,32	204984,85	54894,72
IDN	Indonesia	3328617,09	1200926,74	97907,27
IND	India	6799913,01	2348115,39	371033,70
IRL	Ireland	4983433,93	843021,72	40058,31
IRN	Iran	41418,72	15263,37	2693,46
ISL	Iceland	314919,95	342536,00	56201,10
ITA	Italy	7609,52	1435,84	146,91
JAM	Jamaica	144384,01	410468,26	57249,52
JPN	Japan	298154,37	177843,32	12998,31
KAZ	Kazakhstan	255368,75	238758,04	32363,59
KOR	Republic of Korea	27036,87	12524,15	976,82
LBR	Liberia	1164633,98	109510,36	192077,69
LKA	Sri Lanka	2890492,29	225506,54	42274,04
MAR	Morocco	2297509,44	514956,55	42903,89
MDG	Madagascar	9076042,99	688685,52	131200,87
MEX	Mexico	5683523,49	971685,71	98135,57
MKD	The Former Yugoslav Republic of Macedonia	2406306,20	519551,30	36164,72
MMR	Myanmar	2021,79	414,31	56,93
MNE	Montenegro	276175,63	46717,56	2219,73
MNG	Mongolia	17741,89	4496,68	553,55
MOZ	Mozambique	14,34	8,96	0,65
MRT	Mauritania	1401473,33	132139,27	230857,69
MYS	Malaysia	1698327,74	2442899,75	328612,96
NAM	Namibia	1219679,04	206374,92	9824,79
NCL	New Caledonia	1235694,42	193640,27	22287,32
NGA	Nigeria	2463,20	415,95	21,31
NLD	Netherlands	8681,24	1695,00	172,52
NOR	Norway, Svalbard and Jan Mayen	34627934,77	6698503,14	1154988,50
NZL	New Zealand	613,97	57,73	101,26
OMN	Oman	2783,09	329,30	95,16
PAK	Pakistan	22079,22	2207,53	784,59

PER	Peru	24380754,75	5873635,49	451420,37
PHL	Philippines	1432607,12	29639,62	14419,95
PNG	Papua New Guinea	702195,74	138312,79	13772,80
POL	Poland	11567617,58	3344142,46	276134,11
PRK	Democratic People's Republic of Korea	867343,35	67967,98	12731,25
PRT	Portugal	11569326,08	2313494,36	164054,50
RUS	Russian Federation	31790698,57	6185460,16	1320401,25
SAU	Saudi Arabia	79414,96	83183,60	10797,99
SDN	Sudan	52,64	7,16	3,01
SLE	Sierra Leone	8198,15	770,87	1352,08
SRB	Serbia	800184,05	135518,40	6455,37
SUR	Suriname	23989,04	68198,26	9511,86
SVK	Slovakia	17,90	7,21	0,78
SWE	Sweden	16672269,94	2269801,68	1690808,67
THA	Thailand	614630,99	372215,37	27078,76
TJK	Tajikistan	3268,77	552,94	26,27
TUN	Tunisia	4555,14	428,33	751,24
TUR	Turkey	7618947,93	1406920,64	121229,72
TZA	United Republic of Tanzania	300,48	82,45	7,15
UGA	Uganda	1,73	0,16	0,29
UKR	Ukraine	29560506,12	3408349,38	1574112,91
USA	USA, Puerto Rico and US Virgin Islands	73052070,21	25110165,48	2021954,82
UZB	Uzbekistan	3537,69	970,76	84,23
VEN	Venezuela	726599,86	68322,08	119834,75
VNM	Viet Nam	11666,62	2591,86	255,18
ZAF	South Africa	68251658,61	30861290,63	3599132,75
ZMB	Zambia	287814,43	13925,32	3327,94
ZWE	Zimbabwe	1675292,60	311704,53	29225,62

Table S 17: Material costs and shares for low-carbon development

Materials	Mass [t]	Mass share [%]	Climate change [€]	Climate change [%]	Abiotic Resources [€]	Abiotic Resources [%]	Land [€]	Land [%]	Water [€]	Water [%]	Full costs [€]	Full costs [%]
Ag	110	0%	2565129	0%	24209753	0%	578045	0%	161945	share water	27514872	0%
Al	893000	9%	4062631428	22%	758859592	4%	84592473	4%	204259686	0%	5110343180	13%
Cd	30	0%	33852	0%	23660	0%	5699	0%	919	30%	64130	0%
Co	154000	2%	1678821560	9%	2297236548	13%	268992766	13%	11712294	0%	4256763167	11%
Cr	32000	0%	74713482	0%	38148702	0%	2796843	0%	1105915	1%	116764942	0%
Cu	660000	7%	755375282	4%	2926152133	17%	240625144	11%	48590920	0%	3970743478	10%
Fe	4100000	41%	2592596793	14%	2732156247	16%	139133827	7%	2548657	6%	5466435524	14%
Ga	5	0%	204581	0%	791708	0%	30240	0%	15834	0%	1042362	0%
Ge	10	0%	2437802	0%	8497868	0%	0	0%	0	0%	10935670	0%
Graphite	2090000	21%	1064449665	6%	1290514325	7%	516252479	25%	35453856	0%	2906670325	7%
In	20	0%	1207456	0%	6052438	0%	100181	0%	35535	4%	7395610	0%
Li	158000	2%	2151903941	12%	388309763	2%	213332823	10%	245380464	0%	2998926992	8%
Mn	172000	2%	733286273	4%	118823423	1%	237168175	11%	21738001	34%	1111015872	3%
Mo	6600	0%	15201928	0%	58506903	0%	987394	0%	1573296	2%	76269520	0%

Ni	1218000	12%	3778589869	20%	5540071805	32%	205894098	10%	158976712	0%	9683532484	25%
Pt	32	0%	302150113	2%	432351951	2%	811399	0%	8148368	15%	743461831	2%
rare_earths	59838	1%	291063682	2%	198643910	1%	41666932	2%	18541031	1%	549915555	1%
Se	30	0%	22232	0%	629087	0%	5269	0%	1422	2%	658010	0%
Si	249000	2%	826449226	4%	325767901	2%	17551092	1%	17467894	0%	1187236113	3%
Te	40	0%	84694	0%	983063	0%	14548	0%	4197	3%	1086501	0%
Zn	330000	3%	162405685	1%	395425963	2%	127545617	6%	10880665	0%	696257931	2%
Sum	10122716		18496194672	100%	17542156742		2098085043		786597610	100%	38923034067	100%
[%]			47.5%		45.1%		5.4%		2%			

Table S 18: Costs per country and impact

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
AFG	2443639,97	2391474,91	53749,5233	0,01299018	0	0,01885839	0,01845581	0,00040257	1,00E-10	3,24E-10
AGO	163285900	160132752	3248925,48	0	0	0,22819566	0,22378906	0,00440659	-1,00E-06	0
ALB	2064249,5	1497989,96	569709,708	8971,39884	4373	0,01930324	0,01400803	0,00517043	8,39E-05	4,09E-05
ARE	1167867295	1122897823	46335427,3	0	0	0,39179161	0,37670543	0,01508618	0	0
ARG	275413340	113765743	104347301	51408658,4	8967767	0,0753639	0,03113077	0,02771177	0,01406743	0,00245393
ARM	-116099,916	-1348130,34	624367,51	190751,444	435318	-0,00132002	-0,01532784	0,0068896	0,00216879	0,00494943

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
ATA	0	0	0	0	0	NA	NA	NA	NA	NA
ATF	0	0	0	0	0	NA	NA	NA	NA	NA
AUS	367730600	234645191	116396892	7216378,06	12903488	0,03635091	0,02319515	0,01116687	0,00071335	0,00127554
AUT	44809796,1	-40146206,8	87248450	337568,618	-57955	0,01394653	-0,01249504	0,02635455	0,00010506	-1,80E-05
AZE	6597714,47	4102162,97	2571317,3	11,2209859	25	0,01983575	0,01233298	0,00750266	3,37E-08	7,43E-08
BDI	2140730,76	1909060,76	238707,023	0	0	0,09984408	0,08903895	0,01080513	0	0
BEL	82647889	-20696323,1	106456378	17357,3016	8783	0,02152983	-0,00539141	0,02691442	4,52E-06	2,29E-06
BEN	10324633,8	10324633,8	0	0	0	0,10261835	0,10261835	-1,00E-06	0	0
BFA	38970134,6	38800344,9	0	53231,7976	116558	0,34370359	0,34220611	0	0,00046949	0,001028
BGD	88162066,6	88162066,6	0	0	0	0,0455685	0,0455685	0	0	0
BGR	11149569	-3849660,11	12694361,4	1480569,33	1198525	0,02384502	-0,00823307	0,02634845	0,00316642	0,00256322
BHS	7990932,12	7990932,12	0	0	0	0,0869184	0,0869184	0	0	0
BIH	1313981,39	-1582018,84	2823485,45	163203,202	-7453	0,00922121	-0,01110223	0,01923042	0,00114532	-5,23E-05

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
BLR	-3711243,16	-17928060,7	14648656,1	0	0	-0,00875663	-0,04230105	0,03354441	0	0
BLZ	1498137,79	1498137,79	0	0	0	0,1134036	0,1134036	0	0	0
BOL	46623331,3	14242767	0	32210497,8	170066	0,16391802	0,05007463	0	0,11324547	0,00059792
BRA	2016101207	1106294975	301400644	579151755	38139043	0,15145573	0,08310828	0,02197466	0,04350766	0,00286512
BRN	24946167,4	24946167,4	0	0	0	0,26043783	0,26043783	0	0	0
BTN	2027967,53	1371959,76	0	417120,718	238887	0,11739404	0,07941937	0	0,02414609	0,01382858
BWA	31245730,5	30478248,4	778011,527	9809,49214	2597	0,23713647	0,23131174	0,00573058	7,44E-05	1,97E-05
CAF	2181901,82	2181901,82	0	0	0	0,13915091	0,13915091	0	0	0
CAN	-92820650,4	-375039931	210879668	48022344,8	29533944	-0,00766049	-0,03095207	0,01689085	0,00396329	0,00243744
CHE	85099486,1	-47015940,1	136128457	0	0	0,0170941	-0,00944418	0,02653829	0	0
CHL	383967656	-15517517,2	16867942,1	145364772	237749722	0,18234667	-0,00736929	0,00777445	0,06903389	0,11290761
CHN	8181995227	1098807353	6972842653	264137312	51765448	0,08340695	0,0112012	0,06898545	0,00269261	0,00052769
CIV	21397908,4	20425441	0	903809,342	68658	0,05224597	0,04987155	0	0,00220678	0,00016764
CMR	28100204,1	28100204,1	0	0	0	0,10286305	0,10286305	0	0	0

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
COD	44555563,3	14766119,2	0	29536629,2	252815	0,16607859	0,05503996	0	0,11009628	0,00094236
COG	21073049,1	21073049,1	0	0	0	0,26650457	0,26650457	0	0	0
COL	235045239	202187397	28416782,7	5101502,73	177276	0,09980702	0,08585463	0,01171087	0,00216625	7,53E-05
CRI	33100193,5	27753180,4	5509429,6	0	0	0,07742544	0,06491812	0,01250732	0	0
CUB	21546268,4	19132079,7	0	2400922,53	13266	0,03051879	0,02709926	0	0,00340074	1,88E-05
XTX	0	0	0	0	0	NA	NA	NA	NA	NA
CYP	8801088,98	7858749,86	970875,638	34,8420306	50	0,04925413	0,04398046	0,0052732	1,95E-07	2,79E-07
CZE	44574503,2	-32558213,4	79475637,2	0	0	0,02536537	-0,01852743	0,0438928	0	0
DEU	715600289	-230330328	974598778	68134,8773	-5385	0,02557157	-0,00823072	0,03380005	2,43E-06	-1,92E-07
DJI	1967952,06	1967952,06	0	0	0	0,0925202	0,0925202	0	0	0
DNK	19447322,6	-25499396,8	46311984,4	0	0	0,00771852	-0,01012055	0,01783907	0	0
DOM	60227887,9	60225129	0	2625,50394	133	0,09971158	0,09970702	0	4,35E-06	2,21E-07
DZA	101784050	85128431	17161508,9	12,8597258	14	0,0821922	0,06874253	0,01344965	1,04E-08	1,13E-08
ECU	41694761,7	32853169,2	9081241,93	26572,8485	1490	0,05490576	0,04326271	0,0116061	3,50E-05	1,96E-06
EGY	310408661	286877350	24245777,9	133,006889	159	0,18630183	0,17217875	0,0141229	7,98E-08	9,52E-08

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
ERI	3693811,19	1757110,55	56886,5603	1010305,59	871185	0,26029986	0,12382215	0,00389057	0,07119541	0,06139173
ESP	367110939	156157537	210874488	1996329,16	4299109	0,03656345	0,01555295	0,02038349	0,00019883	0,00042818
EST	-632764,138	- 4555512,17	4041902,23	0	0	-0,002926	- 0,02106538	0,01813938	0	0
ETH	36738548,3	33795531,3	3032411,62	0	0	0,06175152	0,05680479	0,00494673	0	0
FIN	225823861	- 57803357,2	47248353,9	217289178	20482555	0,11591495	- 0,02967035	0,02353753	0,11153411	0,01051366
FJI	1202506,97	840304,065	373204,888	0	0	0,03051689	0,02132501	0,00919189	0	0
FLK	0	0	0	0	0	NA	NA	NA	NA	NA
FRA	316014354	- 50272343,3	374071104	1117373,77	2125736	0,01605577	- 0,00255419	0,01844519	5,68E-05	0,000108
GAB	179850037	19132092,4	0	154975720	5742225	1,51073911	0,16070945	0	1,30179501	0,04823465
GBR	128810761	-174073778	312084714	0	0	0,00637793	- 0,00861908	0,01499701	0	0
GEO	3094457,89	-340797,32	1051483,87	1108701	1306068	0,02490432	- 0,00274275	0,00821291	0,00892287	0,01051128
GHA	54075428,4	44291161,9	5729662,41	3808836,24	414677	0,11683682	0,09569668	0,01201471	0,00822947	0,00089596
GIN	226788132	8023394,45	0	47296032,5	171468705	2,94511155	0,10419325	0	0,6141948	2,2267235

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
GMB	2365629,24	2365629,24	0	0	0	0,22265452	0,22265452	0	0	0
GNB	926034,798	926034,798	0	0	0	0,0899097	0,0899097	0	0	0
GNQ	30375955,9	30375955,9	0	0	0	0,32402357	0,32402357	0	0	0
GRC	73040884,4	27806793,6	18007045,8	7134879,19	20623009	0,04742747	0,01805572	0,01134778	0,00463288	0,01339109
GRL	0	0	0	0	0	0	0	0	0	0
GTM	37080887,7	35498293,4	0	1566560,77	16033	0,07183238	0,06876661	0	0,00303471	3,11E-05
GUY	7235150,95	1430341,11	0	4608753,26	1196057	0,21405323	0,0423169	0	0,13635079	0,03538555
HND	20297180,7	16080894,8	0	4171962,26	44324	0,11966917	0,09481057	0	0,02459727	0,00026133
HRV	6378077,11	-63662,4371	6637408,58	0	0	0,01471946	-0,00014692	0,01486639	0	0
HTI	3317649,2	3317649,2	0	0	0	0,02943328	0,02943328	0	0	0
HUN	39598550,9	-5538078,83	45854741,2	524225,431	109449	0,03496381	-0,00488988	0,03929419	0,00046287	9,66E-05
IDN	636883988	499082166	112876950	26856218,8	1396235	0,08655415	0,06782653	0,01488804	0,00364983	0,00018975
IND	4362114460	3895578251	431007113	38118106,7	10116963	0,22772743	0,20337156	0,02183772	0,00198998	0,00052816
IRL	33009133,8	-20034532,5	52780616,6	1851234,82	-32227	0,01221801	-0,00741559	0,01896031	0,00068522	-1,19E-05
IRN	214828576	153719338	62924260,4	6902,95084	33065	0,07279938	0,05209117	0,02069466	2,34E-06	1,12E-05

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
IRQ	342301527	340682374	1668334,6	0	0	0,21622918	0,21520637	0,00102281	0	0
ISL	-1162056,15	-3990404,46	2721639,95	106604,356	80337	-0,0063952	-0,0219606	0,01453659	0,00058668	0,00044212
ISR	249390753	221424052	28816194,4	0	0	0,09532006	0,08463086	0,0106892	0	0
ITA	485061328	74692779,9	422828797	2826,76606	1804	0,03284927	0,00505834	0,02779062	1,91E-07	1,22E-07
JAM	5799120,12	4058528	0	1170462,36	570130	0,05227249	0,03658303	0	0,01055039	0,00513907
JOR	20921709,3	14052502,9	7077859,79	0	0	0,0690256	0,04636249	0,02266312	0	0
JPN	1365745234	249891710	1149529659	134337,915	77354	0,03904253	0,00714365	0,03189283	3,84E-06	2,21E-06
KAZ	-33403031,3	-51014310,4	17814479	20014,5547	301952	-0,02638178	-0,0402912	0,01365513	1,58E-05	0,00023848
KEN	33767379,3	27661434,4	6291414,15	0	0	0,0544884	0,04463561	0,0098528	0	0
KGZ	-482216,132	-1680948,88	1235144,48	0	0	-0,00825797	-0,02878632	0,02052835	0	0
KHM	22032188,4	22032188,4	0	0	0	0,12700382	0,12700382	0	0	0
KOR	506400520	-46026833,3	569190709	10043,5963	6191	0,04417028	-0,00401464	0,04818351	8,76E-07	5,40E-07
UNK	0	0	0	0	0	NA	NA	NA	NA	NA
KWT	508266947	495529517	13124331,1	0	0	0,51187376	0,49904594	0,01282782	0	0

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
LAO	11522818,6	10586533,6	964724,858	0	0	0,09090713	0,08352049	0,00738665	0	0
LBN	3965932,78	3965932,78	0	0	0	0,01022078	0,01022078	0	0	0
LBR	11321669,7	1961357,32	0	9510427,59	-150115	0,49191286	0,0852186	0	0,41321658	-0,00652232
LBY	78347679,4	78347679,4	0	0	0	0,21094564	0,21094564	0	0	0
LKA	93637869,8	63330210,8	7333651,97	22803504,1	386697	0,14999175	0,1014441	0,01140094	0,03652729	0,00061942
LSO	-8753,79598	-8753,79598	0	0	0	-0,00048135	-0,00048135	0	0	0
LTU	-1003364,53	-7284792,15	6472227,15	0	0	-0,00264542	-0,01920672	0,0165613	0	0
LUX	-1883899,04	-5699211,59	3931203,37	0	0	-0,00376257	-0,0113826	0,00762003	0	0
LVA	-1804940,97	-4431952,53	2706807,53	0	0	-0,00742844	-0,01824021	0,01081177	0	0
MAR	43472333,7	26601558,7	15425566	694867,869	1205083	0,05214024	0,03190562	0,01795584	0,00083342	0,00144536
MDA	199887,992	-392356,188	610233,707	0	0	0,00247128	-0,00485083	0,0073221	0	0
MDG	66681509,4	6294430,36	0	58823623,4	1563456	0,68177847	0,06435678	0	0,60143629	0,0159854
MEX	753031232	542925013	186641056	26895690,6	2071601	0,0872595	0,06291288	0,02098996	0,00311661	0,00024005

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
MKD	21414069,5	-1245987,56	2339985,95	19773500,2	615553	0,23914978	-0,01391504	0,0253623	0,2208281	0,00687442
MLI	24671551,1	24671551,1	0	0	0	0,20470994	0,20470994	0	0	0
MMR	41636851,2	41620178,8	0	15982,3869	690	0,07742842	0,07739741	0	2,97E-05	1,28E-06
MNE	-294826,593	-673909,042	248134,855	102593,103	35669	-0,00758688	-0,01734195	0,00619711	0,00264007	0,0009179
MNG	-6156671,5	-6717077,77	572087,614	1988,48713	3195	-0,06652415	-0,07257946	0,0059993	2,15E-05	3,45E-05
MOZ	15500492,4	15500451,6	0	25,3940515	15	0,14788875	0,14788836	0	2,42E-07	1,47E-07
MRT	15108522,8	14704179,4	0	127944,042	276399	0,30358432	0,29545961	0	0,00257085	0,00555385
MWI	12079571,9	12079571,9	0	0	0	0,24734897	0,24734897	0	0	0
MYS	474781074	360476573	102096049	13654139,6	1564077	0,18747319	0,14233864	0,03912545	0,00539151	0,00061759
NAM	12590109,8	11053749,1	931251,871	168073,692	464488	0,13254598	0,11637151	0,009515	0,00176944	0,00489003
NCL	12757209	2425292,71	0	10154154,1	177762	0,16427001	0,03122963	0	0,1307514	0,00228898
NER	33251343,3	32963542,5	296542,804	0	0	0,36661005	0,36343692	0,00317312	0	0
NGA	604547876	604543985	0	3343,79453	548	0,21558954	0,21558816	0	1,19E-06	1,95E-07
NIC	6719928,23	6719928,23	0	0	0	0,07285962	0,07285962	0	0	0

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
NLD	71696063,3	-54974112,3	130513447	3224,88788	1005	0,01111025	-0,00851897	0,01962856	5,00E-07	1,56E-07
NOR	-27798116,4	-77730381,1	40951532,3	7378079,36	2809893	-0,00906888	-0,02535883	0,01296622	0,00240703	0,0009167
NPL	15663404,9	15663404,9	0	0	0	0,07604887	0,07604887	0	0	0
NZL	21046261,2	94787,3606	21587762,5	171,518882	-59	0,01433745	6,46E-05	0,0142728	1,17E-07	-4,00E-08
OMN	171523185	163808749	7947354,07	160,823056	1207	0,30449234	0,29079747	0,01369244	2,85E-07	2,14E-06
PAK	412937283	412923066	0	5000,95435	9216	0,18593659	0,18593018	0	2,25E-06	4,15E-06
PAN	47658311,9	46353670,1	1344270,48	0	0	0,10364894	0,10081156	0,00283738	0	0
PER	229980512	43385448,3	26543703,9	144122690	16711171	0,14670514	0,02767569	0,01643314	0,09193622	0,0106601
PHL	278917731	233591521	34525683,4	11772257,9	46077	0,1139042	0,09539392	0,01368392	0,00480755	1,88E-05
PNG	12685615,9	7115786,3	0	5451503,6	118326	0,07452783	0,04180515	0	0,03202751	0,00069516
POL	53090707,2	-68224902,2	119665911	4297112,54	880305	0,0128018	-0,01645112	0,02800448	0,00103617	0,00021227
PRI	0	0	0	0	0	0	0	0	0	0
PRK	396560,681	0	0	322198,754	74362	0,00322816	0	0	0,00262283	0,00060534
PRT	59964463,4	20603762,6	32323003,5	4297747,21	3692824	0,03505198	0,01204384	0,01833728	0,00251223	0,00215862
PRY	15782702,9	15782702,9	0	0	0	0,05535539	0,05535539	0	0	0

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
PSE	1048432,19	0	1080278,52	0	0	0,00912372	0	0,00912372	0	0
QAT	700796303	688003084	13181815	0	0	0,54142925	0,53154532	0,00988393	0	0
ROU	21050070,5	-12756918,8	34833882,8	0	0	0,01234835	-0,00748344	0,01983179	0	0
RUS	-257559631	-508481310	244191591	9114964,74	4813826	-0,02185073	-0,0431383	0,02010589	0,00077329	0,00040839
RWA	7301082,6	7301082,6	0	0	0	0,10732181	0,10732181	0	0	0
ESH	0	0	0	0	0	0	0	0	0	0
SAU	2224326044	2160907931	65053815	4214,41302	277853	0,40057411	0,38915327	0,01137004	7,59E-07	5,00E-05
SDN	170489544	170489507	0	18,3624731	19	0,72893958	0,72893942	0	7,85E-08	7,97E-08
SSD	0	0	0	0	0	0	0	0	0	0
SEN	18492346,4	18492346,4	0	0	0	0,11272645	0,11272645	0	0	0
SLB	858203,587	858203,587	0	0	0	0,0762618	0,0762618	0	0	0
SLE	4080711,43	4033310,51	0	48000,2035	-599	0,14149038	0,13984685	0	0,00166431	-2,08E-05
SLV	12593626,8	12593626,8	0	0	0	0,06829929	0,06829929	0	0	0
SOL	0	0	0	0	0	0	0	0	0	0
SOM	145460,977	145460,977	0	0	0	0,00436423	0,00436423	0	0	0

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
SRB	4321368,58	-4730952,43	8993704,47	297250,569	26498	0,01208697	-0,01323258	0,02441402	0,00083142	7,41E-05
SUR	3592088,44	3343467,62	0	194443,214	54178	0,14685276	0,13668858	0	0,00794928	0,0022149
SVK	23495298,2	-12005459,3	36579088,8	6,64929556	4	0,03148423	-0,01608759	0,0475718	8,91E-09	5,11E-09
SVN	5402780,05	-5753104,23	11494746,3	0	0	0,01412931	-0,01504547	0,02917478	0	0
SWE	1084883,41	-91531569,7	90455999,5	5620240,23	-793168	0,00027665	-0,02334089	0,02238661	0,00143318	-0,00020226
SWZ	1420579,53	1420579,53	0	0	0	0,04312903	0,04312903	0	0	0
SYR	36870790,3	36870790,3	0	0	0	0,08034602	0,08034602	0	0	0
TCD	21322487,3	21322487,3	0	0	0	0,26871941	0,26871941	0	0	0
TGO	4064181,26	4064181,26	0	0	0	0,10731235	0,10731235	0	0	0
THA	494783790	347031121	146331372	5010922,59	724185	0,13836279	0,0970448	0,03971421	0,00140127	0,00020251
TJK	611498,588	163244,02	460647,061	345,084606	842	0,01151338	0,00307358	0,00841744	6,50E-06	1,59E-05
TKM	12155591,1	12155591,1	0	0	0	0,04224011	0,04224011	0	0	0
TLS	0	0	0	0	0	0	0	0	0	0
TTO	13206560,2	13206560,2	0	0	0	0,07857048	0,07857048	0	0	0

iso_a3	All costs [€]	Climate costs [€]	Abiotic Resources[€]	Land [€]	Water [€]	All costs [%] GDP	Climate costs [%] GDP	Minerals [%] GDP	Land [%] GDP	Water [%] GDP
TUN	36732614,7	29724731,9	7218379,12	2051,42031	248	0,13082427	0,10586549	0,02495059	7,31E-06	8,84E-07
TUR	209309358	40642156,1	169183464	2301726,36	2169495	0,03808826	0,0073957	0,02987893	0,00041885	0,00039479
TWN	213814378	0	220309029	0	0	0,04981139	0	0,04981139	0	0
TZA	49080947,1	47619141,5	1505673,78	381,0	138	0,11985917	0,11628933	0,00356857	9,31E-07	3,36E-07
UGA	38239680,5	38239679,1	0	1,5	0	0,16455101	0,16455101	0	6,72E-09	-7,66E-10
UKR	15071326,1	-24206773,3	29496494,2	7862666,41	2788488	0,01630799	-0,02619304	0,03097591	0,00850783	0,00301729
URY	11089446,7	5941611,73	5304201,46	0	0	0,02635613	0,01412134	0,01223479	0	0
USA	4475247945	2182489449	2323216424	17980780,8	20049091	0,03080086	0,01502097	0,01551815	0,00012375	0,00013799
UZB	21914723,4	10476181,7	11783804,7	239,656429	1881	0,06159769	0,02944635	0,03214538	6,74E-07	5,29E-06
VEN	334795094	331079852	0	3770602,24	-55360	0,50448299	0,49888471	0	0,0056817	-8,34E-05
VNM	205975192	127997826	80243152,7	94133,5163	5626	0,11897753	0,07393543	0,04498447	5,44E-05	3,25E-06
VUT	237533,401	237533,401	0	0	0	0,03678248	0,03678248	0	0	0
YEM	47988706,1	47988706,1	0	0	0	0,28924541	0,28924541	0	0	0
ZAF	321009323	151135281	59694967,3	21022353,6	90916513	0,12345941	0,05812626	0,02228173	0,00808515	0,03496627
ZMB	20221254,5	17785341,3	2184785,98	281859,204	33675	0,10885471	0,09574174	0,01141439	0,0015173	0,00018128
ZWE	3737077,8	1103978,41	1509207,26	184630,74	983752	0,02177282	0,00643196	0,00853367	0,00107569	0,0057315