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From cumulated energy demand to cumulated raw material demand: the material footprint as a sum parameter in life cycle assessment

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Abstract

Background: Global targets for reducing resource use have been set by organizations such as the International Resource Panel and the European Commission. However, these targets exist only at the macro level, e.g., for individual countries. When conducting an environmental analysis at the micro level, resource use is often neglected as an indicator. No sum parameter indicating all abiotic and biotic raw materials has been considered for life cycle assessment, as yet. In fact, life cycle assessment databases even lack some of the specific input flows required to calculate all abiotic and biotic raw materials. In contrast, the cumulative energy demand, an input-based indicator assessing the use of energy resources, is commonly used, particularly when analyzing energy-intensive product systems.

Methods: In view of this, we analyze the environmental relevance of the sum parameter abiotic and biotic raw material demand, which we call the material footprint. First, we show how abiotic and biotic raw material demand can be implemented in the Ecoinvent life cycle assessment database. Employing the adapted database, the material footprint is calculated for 12 individual datasets of chosen materials and crops. The results are compared to those of the cumulated energy demand and four selected impact categories: climate change, ozone depletion, acidification, and terrestrial eutrophication.

Results: The material footprint is generally high in the case of extracted metals and other materials where extraction is associated with a large amount of overburden. This fact can lead to different conclusions being drawn compared to common impact categories or the cumulative energy demand. However, the results show that both the range between the impacts of the different materials and the trends can be similar.

Conclusions: The material footprint is very easy to apply and calculate. It can be implemented in life cycle assessment databases with a few adaptions. Furthermore, an initial comparison with common impact indicators suggests that the material footprint can be used as an input-based indicator to evaluate the environmental burden, without the uncertainty associated with the assessment of emission-based impacts.

Keywords: Abiotic resources, Biotic resources, Cumulative energy demand, MIPS, Material footprint, Ecoinvent database

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Background

Decoupling, which aims at disconnecting natural resource use from economic growth, can be seen as one of the key strategies for sustainable development [1]. It has furthermore become clear that an absolute global reduction in raw material use is needed: Bringezu [2] recently suggested three targets for global raw material use (societal perspective), which are in line with the reflections from the International Resource Panel on the establishment of Sustainable Development Goals [3]. The "10-2-5 target" triplet, introduced for the purpose of policy guidance, has target values with a suggested resource reduction factor ranging from four to ten (10 t per person per year of total abiotic raw materials and 2 t per person per year of total biotic raw materials used, where direct raw material consumption accounts for 5 t per person per year). Furthermore, Lettenmeier et al. [4] suggest a sustainable resource cap target from an end user perspective of 8 t per person per year for Finnish households, which would mean a reduction by a factor of 5 compared to the current state. In spite of these existing targets, resource use is usually neglected as an indicator during environmental analysis at the micro level, particularly in terms of raw materials.

The authors of this article believe there are several reasons for the need to consider raw material demand in addition to the measurement of specific impact categories. One of the main reasons for this need is that all anthropogenic emissions are based on the extraction of natural resources. As a consequence, reducing the amount of natural resources extracted can also lead to less environmental degradation [5].

Furthermore, although considerable progress has been made in impact assessment within life cycle assessment (LCA), it is unlikely that the environmental categories proposed based on current knowledge, such as in the product environmental footprint (PEF) [6], actually cover all environmental interventions. This fact, pointed out by Klöpffer as early as 1997 [7], is still valid to this day. For instance, LCA does not allow a reliable assessment to be made of all environmental impacts such as biodiversity [8] and impacts on land use [9]. In particular, biotic raw materials are addressed inadequately in LCA. As Bringezu [2] points out, impacts due to stressing topsoil, forest biomass, and fish stocks are hardly reflected.

In view of this complexity, companies and institutions can benefit from a simple mass-based indicator that allows them to measure the environmental performance of products without the uncertainties associated with ex-ante approaches, with which LCA impact assessment commonly goes along.

So far, however, resource indicators have been the subject of controversial debate. One of the main arguments against a sum parameter without a characterization of impacts is that some small-scale materials with a large environmental impact can be overlooked, while materials that are dominant in weight dominate the result. For this reason, De Bruyn and colleagues (2003) [10] concluded, for instance, that mass-based indicators are insufficient as a measure for reducing environmental pressure. Today, LCA mainly considers raw material use, and the problems associated with it, from the perspective of the criticality and depletion of resources in an economic sense. Methods to assess resource use in terms of minerals and biomass cover only a small number of raw materials. Biotic raw materials in particular are often neglected [11, 12]. For example, the common method abiotic depletion potential, which is part of the life cycle assessment method CML 2002, only covers 48 abiotic minerals and 5 biotic resources [12]. The method does not focus on the impact on the environment but on the scarcity of minerals, as raw materials are characterized by the assumed extractable reserves. As a consequence, depletion methods provide incentives to use renewable raw materials. In view of the current problems associated with land use changes, e.g., as a result of biofuel production [13, 14], this incentive has become questionable.

The recent draft of the environmental footprint on resources, water, and land [15] addresses this problem. The authors emphasize the importance of considering not only the criticality of resource provisions but also the environmental impact caused by resource depletion, energy carriers, and the total of materials extracted from the nature. They propose measuring the cumulated energy demand as well as abiotic and biotic raw materials in addition to LCIAMs, since the trade-off between stressing abiotic raw materials and impacts on land use cannot be assessed using current methods.

Whereas the assessment of the cumulative energy demand (CED) can be referred to as good practice today, and has been implemented in most LCA databases [16], calculation of the raw material demand is not yet widely accepted. Moreover, LCA databases lack a number of specific input flows required to calculate all abiotic materials (minerals) and all biotic raw materials used in a product system.

Against this background, this paper presents a methodology for how to calculate the amount of abiotic and biotic raw materials in LCA using the Ecoinvent database. The adapted database is tested, analyzing the environmental relevance of the material footprint indicator by comparing material footprint results to results of the CED and other selected impact categories.

In the first part of the paper, the material footprint is introduced, followed by a detailed description of the adaptions to the life cycle database. The second part of

the paper contains the results of the material footprint calculations, which are compared to the CED and selected impact categories. Finally, conclusions are drawn concerning the similarities and differences between resource use and the selected impact categories, and the need for further research is addressed.

Methods

The following steps show how raw material use was determined and compared to environmental impacts in this study:

1. Selection of an indicator to measure abiotic and biotic resources
2. Selection of further impact categories for comparison
3. Selection of an LCA database
4. Adaption of the database and creation of a life cycle inventory analysis (LCIA) scheme to calculate the raw material indicator
5. Comparison of raw material demand with other impact categories

Each step is described in detail in this section.

Selection of a raw material indicator - the material footprint

Based on the proposition made in [15] to measure abiotic and biotic raw materials, we chose the material footprint as an indicator. It is based on the MIPS (material input per service unit) concept [17–20]. The material footprint covers two of the five categories of the MIPS concept: abiotic raw materials and the biotic raw materials, which can either be added together and used as one indicator or considered separately [17]. The MIPS concept applies the same system boundaries as material flow analysis [21], i.e., products from the agricultural system are regarded as input by fresh weight. Unlike many other approaches that address resources, this approach does not address criticality, nor does it consider the available resource stock, or extraction and replenishing rates. As such, it is an easy-to-grasp concept, and the results generated do not become invalid owing changing evaluations of different resources, extraction rates, or policies.

The abiotic raw material category of the material footprint considers all mineral resources. Different resources are not weighted, i.e., all materials are considered with 1 kg/kg. The category includes resources exploited economically as well as resources that are extracted but not further processed, such as overburden from mining and excavated soil from infrastructure construction. Although no weighting factors are used, small-scale materials can influence the results significantly. Generally, high amounts of ore and rock extraction and high amounts of overburden are necessary to extract some of these small-scale materials. This is the reason why for example extracted gold will show a dramatically higher abiotic raw material demand per kilogram than sand. This way the scarcity of a metal, which is mostly linked to a low ore grade or high amounts of overburden, indirectly has an impact on the results. At the same time, the major impact caused by mining activities is also reflected.

The biotic raw material category contains all plant biomass from cultivated areas as well as plant and animal biomass from uncultivated areas. Animals from cultivated areas (e.g., cattle breeding) are accounted for by the plant biomass input for their feed. Biomass is considered with the moisture content at the time of harvest [20]. As is the case with abiotic material, not only the used extraction of biotic material is considered but also all organic material that is taken from nature (including biotic material taken from agricultural systems). Hence, plant waste taken from the ecosystem during trimming or harvest is also considered, even if it is not further processed.

Since the moisture content of a plant species can vary significantly, the specific weight can also vary depending on the cultivation conditions. A possible way to achieve more consistent results could be to standardize the moisture content [22].

Topsoil erosion is not considered within the abiotic raw material category or in the biotic raw material category. In the MIPS concept, the "earth movement" category, which includes the mechanical movement of earth (e.g., due to ploughing) and erosion, is regarded separately. If data is available, this category can be incorporated into the material footprint as well [17]. So far, however, LCA databases have not included such data because this data is very region-specific, depending for instance, on the cultivation, local weather conditions, and the gradient.

Selection of impact categories

Since both energy use and resource use can have a high impact on the environment, it is interesting to see how these two different approaches compare to each other. Resource and energy use are often considered to be linked, but this is less and less the case as the supply with renewable energies—particularly wind and photovoltaic—increases. Comparing the material footprint to the CED can show how energy demand and raw material demand interact, and whether one of them could be suitable to assess the overall environmental impact. The CED is a sum parameter [7], just like the material footprint, and does not express an actual impact on humans or the environment. Nonetheless, we refer to the CED as an impact category here, since it is available in

LCIA models. Although there is no specific international standard [16], the method can currently be described as good practice. It has been considered for standards such as EN 15978 [16], and a German industry standard is also available [23]. In addition, CED has been broadly implemented in most LCA databases. We calculated the CED as implemented in the Ecoinvent database [24]. The CED was calculated including all given energy categories (non-renewable including fossil, nuclear, primary forest) as well as renewables (including biomass, geothermal, solar, water, wind) without any further weighting. Table 1 provides an overview of the inputs of the CED considered, compared to the material footprint. In addition to the CED, we selected further impact categories from the environmental footprint (EF) [25] to reflect environmental impacts. It seems unreasonable to compare the material footprint to very specific impact categories concerning matters such as human toxicity or ecotoxicity (e.g., cancer effects, non-cancer effects, particulate matter). After all, these categories are not expected to be dependent on the overall quantity of extracted material, but are mainly linked to the use of specific chemicals or materials. Impact categories concerning water (e.g., aquatic fresh water ecotoxicity, aquatic eutrophication, or water depletion) should also result in fundamentally different impacts, as the material footprint does not consider water.

The LCIA methods pack 1.5.4 supplied by Openlca was used [26, 27] to determine the impact categories, as well as the CED. As a result, four out of all 14 default EF impact categories were selected for comparison to the material footprint. These four impact categories are climate change (according to the Intergovernmental Panel of Climate Change 2007 [28]); ozone depletion (according to the World Meteorological Organization [29]); acidification (according to [30]); and terrestrial eutrophication (according to [30]). Table 2 provides an overview of all 14 default impact categories of the Product Environmental Footprint. The table also indicates, whether correlations to the material footprint may be possible or can be ruled out directly.

Selection of an LCA database—the Ecoinvent database

A reliable comparison of environmental impacts and resource use is only possible when using the same inventory data for all calculations. For this reason, the most convenient solution is to use one LCA database for the determination of all indicators. At present, there is no database that considers all of the input flows required to calculate the material footprint. Two of the biggest LCA databases, in terms of the number of processes, are Gabi [31] and Ecoinvent [32].

One important criterion is the structure of the life cycle database. Gabi is based upon system processes that consider the entire life cycle inventory (LCI) related to a product system from cradle-to-gate or cradle-to-grave in the form of elementary flows in one process, directly linking to resources taken from nature or emissions to nature. In this way, inventories in Gabi do not enable users to retrace from which life cycle step flows originate, making it difficult to include further aspects such as soil extraction during infrastructure construction.

Due to these shortcomings, the Ecoinvent database was chosen, where almost all processes (apart from some datasets such as plastics) are available as unit processes. As described by Wiesen et al. [22], there are several challenges involved when adapting the database to calculate both abiotic and the biotic raw materials. Our adaptions to the database are described in the following section. Calculations are based on version 2.2., which still is widely used in the LCA community. The datasets selected remained the same in the following Ecoinvent version. The adaptions described in this paper can be carried out for newer versions, e.g. 3.3, in a similar way. However, the effort required to adapt the database will be slightly higher as newer versions have a new data structure (Ecospold 2) and a larger number of processes.

Adaption of the Ecoinvent database and creating an LCIA scheme

Regarding the calculation of abiotic raw material, the Ecoinvent database provides only elementary flows from

Table 1 Overview of the inputs considered for the material footprint and CED approach

Input category	Material footprint Inputs considered	Cumulative energy demand Inputs considered
Fossil fuels	Mass, fossil fuels	Energy resources, fossil fuels
Biomass	Mass, wood and other biomass	Energy resources, primary forest and other biomass
Inorganic material	Mass, minerals mass, metal ores mass, overburden and soil excavation	Energy resources, nuclear (uranium)
Converted renewable energy	–	Energy resources, solar, converted energy resources, kinetic (in wind), converted geothermal energy, converted energy resources, potential in barrage storage, converted

Table 2 Default categories chosen from the product environmental footprint for a comparison with the material footprint

Product environmental footprint—default EF category	Is correlation to the material footprint expected?
Climate change	Correlation possible, as it concerns environmental impact
Ozone depletion	Correlation possible, as it concerns environmental impact
Ecotoxicity for aquatic fresh water	Not expected, as it is related to water
Human toxicity—cancer effects	Not expected, as it is related to human toxicity
Human toxicity—non-cancer effects	Not expected, as it is related to human toxicity
Particulate matter/respiratory inorganics	Not expected, as it is very specific
Ionizing radiation—human health effects	Not expected, as it is related to human toxicity
Photochemical ozone formation	Not expected, as it is very specific
Acidification	Correlation possible, as it concerns environmental impact
Eutrophication—terrestrial	Correlation possible, as it concerns environmental impact
Eutrophication—aquatic	Not expected, as it is related to water
Resource depletion—water	Not expected, as it is related to water
Resource depletion—mineral, fossil	Not regarded as it is an input-based indicator
Land transformation	Not regarded, as it is an input-based indicator

nature that are exploited economically. For abiotic resources, this means that

- in metal mining processes, only the weight of specific metal ores without tailings is considered
- overburden is not considered in any of the mining processes
- soil excavation, e.g., for construction processes in road or building infrastructure is not available.

Saurat and Ritthoff [33] describe how tailings and overburden can be considered in Ecoinvent using so called unused extraction factors. These factors, relating to elementary flows from nature, are embedded in a characterization method implemented in the LCA software. The extraction factors are based on data published in [34]. As described by Wiesen et al. [22], this approach does not entirely meet the needs of the material footprint with regard to the following aspects:

(1) Ecoinvent only provides location-specific elementary flows for metals, such as nickel, copper, and silver. In the case of hard coal and lignite, there is only one elementary flow for each material. Taking the example of hard coal, Table 3 shows that overburden may vary greatly depending on the country. The data exhibits a wide range from 0.75 kg/kg in China up to 17.6 kg/kg in Australia, with a world average of 4.28 kg of unused extraction per kilogram of hard coal [34]. Hence, it is necessary to add region-specific values for overburden to the coal mining processes.

(2) In addition to the overburden, the material footprint also considers excavated soil [22], which can have a significant impact, particularly on the abiotic resource use of infrastructure, e.g., for the construction of railway tracks, roads, airports, landfills, and gas pipelines. Since there is no elementary flow for soil in the current version of Ecoinvent, it cannot be considered in a characterization scheme.

(3) In general, the approach involving the application of unused extraction factors in a characterization scheme results in incomplete inventories. To reach more detailed conclusions, especially when overburden and tailings dominate the results, one possibility may be to break down results into used and unused extraction, as described in [17], which necessitates the consideration of overburden and tailings as elementary flows.

To address problem (1), rather than considering overburden in coal mining processes with a factor in the characterization scheme, a new elementary flow "soil, overburden" was defined. With this approach, differences in mining operation resulting from the accessibility of the coal and the amount of overburden generated in different regions can be taken into account. The flow was included in the ten existing hard coal mining

Table 3 Unused extraction factors of selected countries for hard coal extraction, taken from [34]

Country (as given in Ecoinvent)	Unused extraction factor for hard coal (kg/kg)
Australia	17.6
China	0.75
Colombia	11.99
Germany	0.95
India	5.3
Russian Federation	7.3
South Africa	7.56
USA	5.5
World average	4.28

processes for the following geographical regions: Australia, Latin America and the Caribbean, Northern America, Centrally planned Asia and China, China, Central and Eastern Europe, Russian Federation, Western Europe, and South Africa (geographical term and definition as used in Ecoinvent).

For all lignite mining processes, only one process "lignite, at mine" was originally available in the database, even though the processes are partly regionalized. To be able to differentiate the abiotic raw material demand here, the process "lignite, at mine" was used to create regionalized mining processes, adapting the amount of overburden and scaling diesel consumption accordingly. In this way, it is possible to consider differences in the overburden of lignite mining for Austria, Germany, Spain, France, Greece, Hungary, the former Yugoslav Republic of Macedonia, Poland, Slovenia, and Slovakia.

Regarding problem (2.), we defined the elementary flow "soil, excavated" and added it to excavation processes included in the database. Since the processes assess only excavation in cubic meter an average soil density of 1.8 t/m³ was used for assessment.

Addressing aspect (3.), we did not manage to include elementary flows for tailings and overburden (apart from excavated soil and overburden for the aforementioned processes) in all processes, but used the characterization scheme from Saurat and Ritthoff [33]. However, we recommend that this should be changed in one of the upcoming Ecoinvent versions.

While the accounting for abiotic raw material lies within the system boundaries of the International Reference Life Cycle Data System ILCD and ISO [35], the method of accounting for biotic raw materials differs from the LCA perspective. In LCA, the system boundaries for agricultural processes include the crop harvested so that crops and seeds are considered to be part of the technosphere (economy) because they are based on economically controlled processes. The material flow analysis, on the other hand, which is the basis of the material footprint, considers all biotic materials at harvest to be part of the ecosphere (nature) [21]. This means that the biotic input is considered using the

weight of the harvested good at the time of harvest, and not only considering dry matter or energy content, for example.

In the case of Ecoinvent, the database provides only biotic elementary flows for some wood types, given in cubic meter [33]. These elementary flows are differentiated according to wood type (softwood, hardwood). To achieve more reliable results, we assumed that hardwood (e.g. beech) and softwood (e.g. spruce) have approximate densities of 1000 and 800 kg/m³, respectively. As an estimation, the resulting factors were added to the LCIA method.

All further additional biotic flows, mainly crops, were added to the specific processes and to the LCIA scheme, taking into consideration the unused extraction factors according to [34], the moisture content of the plant at harvest, and, if necessary, the allocation factor and yields for side products. The biotic raw material input MI_{biot} in kilogram per kilogram is calculated for an agricultural product P1 according to the following equation:

$$MI_{\text{biot},P1} = \frac{Y_{P1} + Y_{P2}}{Y_{P1}} \cdot F_{\text{alloc},P_1} \cdot (1 + UUE) \cdot \frac{1 - w_{\text{reference}}}{1 - w_{\text{at·harvest}}}$$

where Y_{P1}: yield of product 1 in tons per hectare; Y_{P2}: yield of product 2 (side product) in tons per hectare; $F_{\text{alloc},P1}$: allocation factor to product 1; UUE: unused extraction factor for the plant in kilogram per kilogram; $w_{\text{reference}}$: moisture content of the product at time of reference; $w_{\text{at harvest}}$: moisture content of the product at time of harvest.

Table 4 shows the values generated for a number of examples. Yields and moisture content were taken from [36, 37] and [38]; allocation factors were taken directly from the Ecoinvent database; unused extraction factors were taken from [34].

Comparison of the material footprint with selected impact categories

For a comparison of the results obtained using Ecoinvent, 12 exemplary materials and crops were chosen. These consist of economically important and

Table 4 Examples of the calculation of characterization factors for biotic resource use

Process in Ecoinvent 3.1/2.2	Yield Y in t/ha	Moisture content (w) of reference product	Moisture content w at harvest	Unused extraction factor (UUE) in kg/kg	Allocation factor F (as used in Ecoinvent)	Material footprint in kg/kg
Barley production, organic (grains)/barley grains organic	4.15	15%	16%	0.237	91.3% to grains	1.947
Barley production, organic (straw)/barley straw organic	2.92	15%	16%	0.237	8.7% to straw	0.264
Grass silage production, organic/grass silage organic	8.10	0%	65%	0.1	100%	3.143
Soybean production, organic/soybeans organic	2.81	11%	16%	0.36	100%	1.441

often-used materials and products covering metals (chromium steel, low-alloyed steel, aluminum, copper), plastics (PET, HDPE), paper and crops (wheat, corn, cotton), and a number of other materials (glass, concrete). In the case of metals, only metals from primary production were chosen, since results for recycling materials also address the challenge of system-wide allocations. An overview of the materials and processes used from the Ecoinvent 2.2 database is given in Table 5.

Results and discussion

The concept of the material input should usually refer to a "service" like such as material input for nutrition per day, for transportation per kilometer, or the use of a personal computer for a year. However, specific materials or crops per kilogram were chosen here as functional unit to make it easier to grasp the concept of abiotic and biotic material input. Figures 1, 2, 3, 4, and 5 show a comparison of the material footprint to CED, climate change, ozone depletion, acidification, and terrestrial eutrophication for a number of selected materials and crops. Since different impact categories have different units, it is not possible to compare them to each other directly. As a workaround for the comparison, the process with the highest impact in each category was scaled to 100%, enabling the relation of the impacts of the different processes to each other to be shown. In addition, the processes were arranged, resulting in a score based on the order of the material footprint, from

Table 5 Processes chosen from the Ecoinvent database 2.2 for materials and crops for the comparison of impact categories

Metals	
Primary chromium steel	Steel, converter, chromium steel 18/8, at plant
Primary low-alloyed steel	Steel, converter, low-alloyed, at plant
Primary aluminum	Aluminum, primary, at plant
Primary copper	Copper, primary, at refinery
Plastics	
PET	Polyethylene terephthalate, granulate, bottle grade, at plant
High density PE	Polyethylene, HDPE, granulate, at plant
Paper and crops	
Paper	Paper, woodfree, uncoated, at regional storage
Wheat	Wheat grains IP, at farm
Corn	Grain maize IP, at farm
Cotton	Cotton fibers, at farm
Other materials	
Glass	Concrete, normal, at plant
Concrete	Flat glass, uncoated, at plant

primary copper (which has the highest material footprint) to concrete (which exhibits the lowest material footprint). All impacts and the specific results are shown in Table 6.

The differences between the material footprint and CED are shown in Fig. 1. Not only the energy demand in the production process is considered in the CED but also the incorporated energy of the products themselves. For this reason, plastics and agricultural products in particular are evaluated as having a higher impact compared to the material footprint. Since the production of primary aluminum requires large amounts of energy, it comes as no surprise that this process has the highest CED. The values for the material footprint, on the other hand, are high for materials such as copper, because the extraction of copper involves large amounts of tailings and overburden. All in all, the CED and the material footprint exhibit different results: within the top five of the material footprints, only aluminum, chromium steel and cotton also exhibit high impacts for CED. Although renewable energies can be considered in a separate category in CED, they are considered with the same energy value as fossil energies, meaning that their benefits and the reduced impact they have on the environment are not always clearly visible. When renewable energies are considered using abiotic and biotic raw materials, however, the differences of the impact on the environment are clearly shown; this is especially important regarding the future increase of renewable energies.

Compared to the material footprint, the impact category for global warming potential particularly assesses the environmental burden to be higher for processes with a high direct energy and fuel demand, as these are normally associated with high carbon emissions as well. This is especially true for aluminum and plastic processes (see Fig. 2). Due to the high energy input needed for primary aluminum production, it comes as no surprise that this process also has the highest global warming potential. One of the reasons for the high impact of cotton fibers is the emission of nitrous oxide (laughing gas) due to the use of inorganic fertilizers, which has a much higher impact on climate change than carbon dioxide. Aluminum has the highest global warming potential, whereas copper has the highest material footprint. However, steel, paper, wheat, corn, glass, and concrete are assessed similarly with regard to the material footprint and global warming potential, also considering the order of the processes.

According to the indicator for ozone depletion, aluminum is the material with the highest impact, as shown in Fig. 3. Since ozone depletion is closely linked to the use of a number of specific chemicals such as chlorofluorocarbons (CFCs), almost no similarities can be detected between ozone depletion and the material
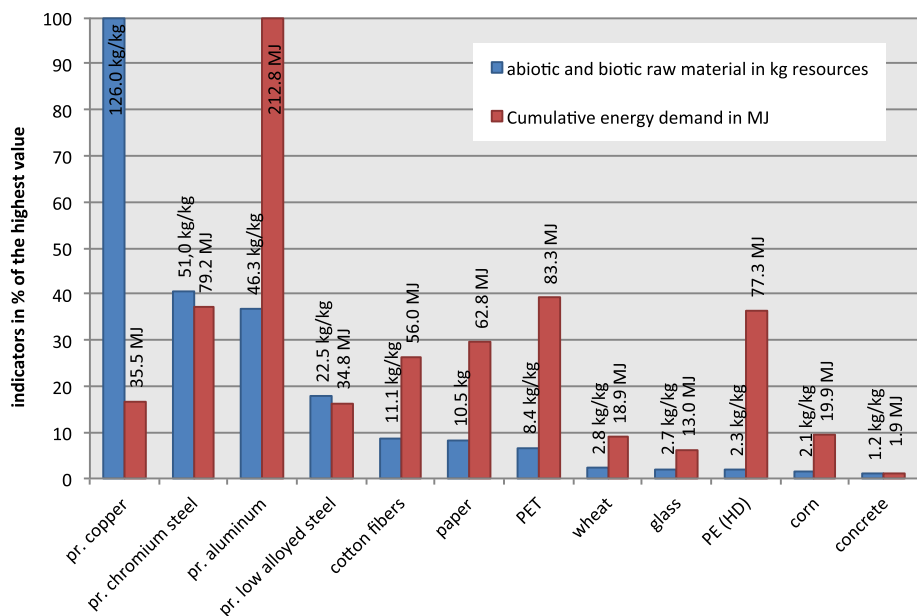
Fig. 1 Comparison of the material footprint to the cumulative energy demand for selected materials and crops

footprint. Both the order and the extent of the impacts exhibit differences. Apart from aluminum, which is the process with the highest impact, copper, low-alloyed steel, cotton fibers, and plastics are also evaluated differently for these two indicators. However, the three materials with the highest impact on ozone depletion: aluminum, chromium steel, and cotton fibers—are nonetheless within the top five materials of the material footprint.

The main drivers for acidification are combustion of fossil fuels, combustion of biomass, the deployment of fertilizers [39], and mining activities [40]. Hence, it comes as no surprise that the two indicators exhibit similar impacts for most of the selected materials, as is shown in Fig. 4. Both the material footprint and the indicator for acidification assess primary copper as the material with the highest impact of the chosen materials. Nonetheless, the impact of cotton fibers and aluminum
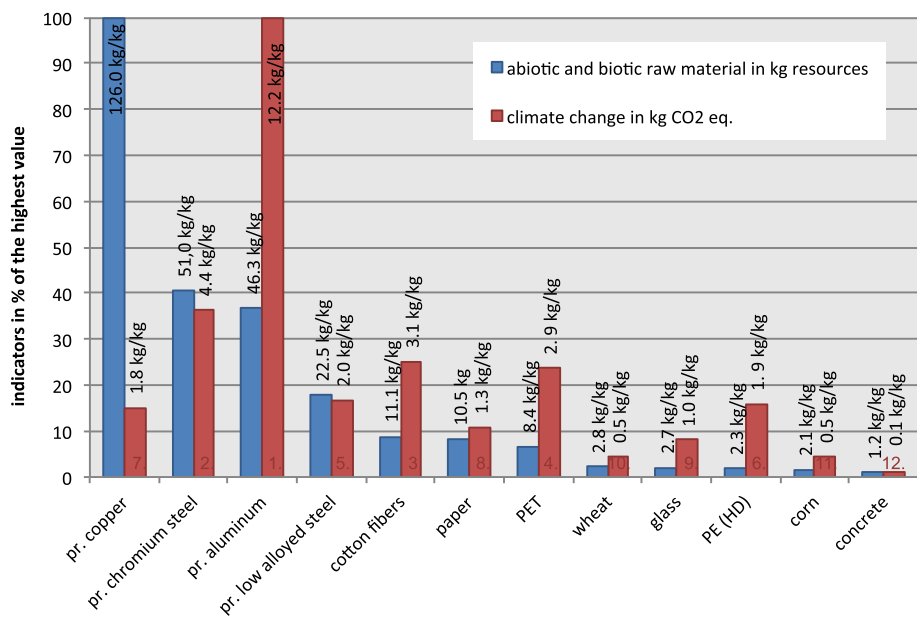

Fig. 2 Comparison of the material footprint to climate change for selected materials and crops
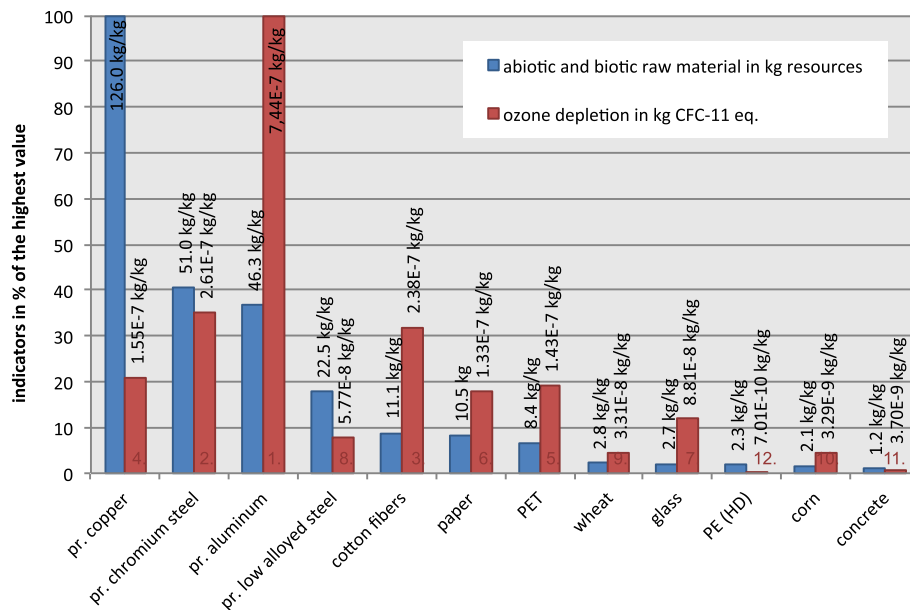
Fig. 3 Comparison of the material footprint to ozone depletion for selected materials and crops

on acidification in particular is higher compared to that on the material footprint.

Cotton fibers have the highest impact on terrestrial eutrophication (as shown in Fig. 5), since their production requires enormous amounts of fertilizers, one of the main sources of nitrogen [41]. In particular, the impact on eutrophication of the agriculturally based materials cotton, paper, wheat, and corn is shown to be higher than their impact on resource use. Nonetheless, of the

three processes with the highest impact on terrestrial eutrophication (cotton fibers, copper, and aluminum), two are not agriculturally based and all three are also evaluated with a high material footprint (top five).

For the agriculturally based processes, biotic raw materials account for a large proportion of the material footprint: 76% for corn, 72% for wheat, 33% for paper, and 24% for cotton fibers. For the other processes, on the other hand, the contribution of biotic raw materials to
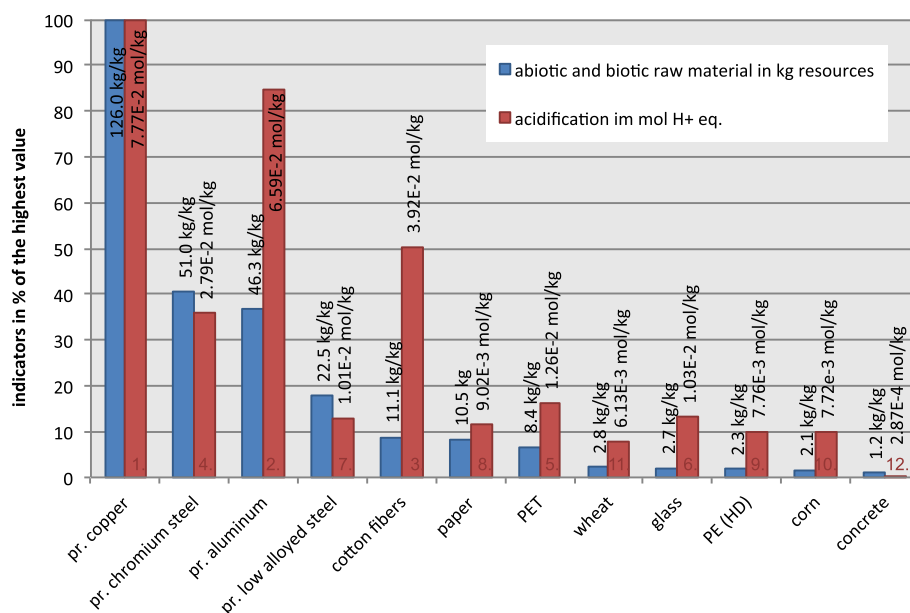

Fig. 4 Comparison of the material footprint to acidification for selected materials and crops
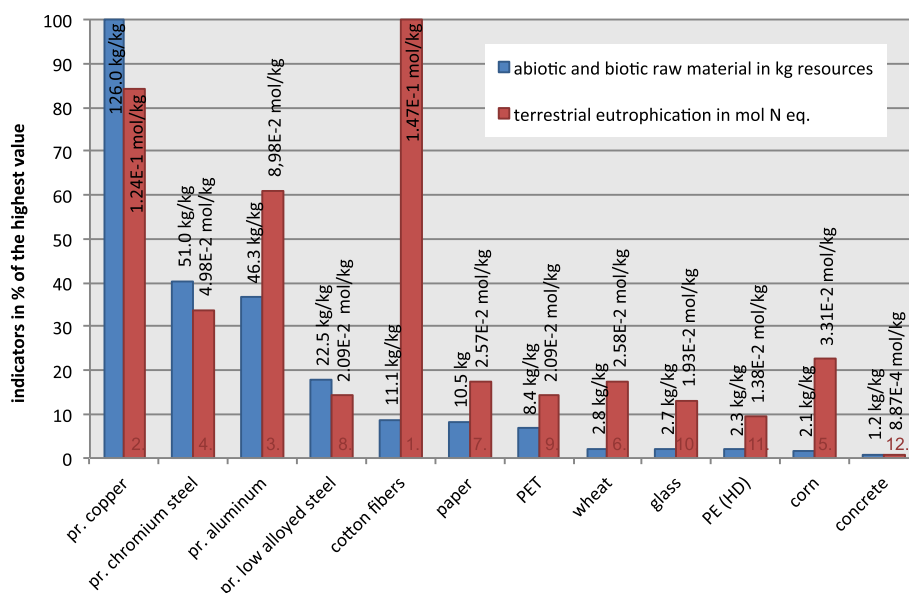
Fig. 5 Comparison of the material footprint to terrestrial eutrophication for selected materials and crops

the overall material footprint is lower than 1%. It can therefore generally be said that biotic raw material demand only has a major impact on agricultural products and other materials that are based on organic input, while abiotic raw material demand significantly contributes to rock, ore, and fossil fuel-based materials.

Conclusions
Adaption of the Ecoinvent database
To determine the material footprint using the Ecoinvent life cycle database, several adaptions to the database were necessary: soil excavation related to infrastructure construction as well as inputs for tailings and overburden from mining activities have been taken into account. Furthermore, new elementary flows for crops have been added. Future updates of the database should, if possible, include these additions defining new elementary flows for abiotic unused extraction (e.g., "soil, overburden," "rock, tailings," "soil, excavated") and for plant species (e.g., "potato, at harvest"). These inputs should, of course, also be considered on the output side, providing complete and comprehensible inventories.

There are some limitations and challenges remaining, which are addressed in the following:

Regarding the flows for overburden and tailings, ores with a high ore grade are less and less available, meaning that the average ore grade decreases over time. Hence, the related input flows need to be updated regularly, and cannot be viewed as set values.

To improve the accuracy of the results for the material footprint, the number of regionalized processes has to be increased since, e.g., the amount of overburden in

hard coal extraction in Germany differs from that in China. This would improve the data quality for other indicators as well, because other flows such as energy use are also influenced by the amount of overburden.

Since there is no differentiation between wood types in the database (only softwood and hardwood), only a rough assumption regarding the density of wood can be made. We highly recommend that elementary flows should be linked to multiple wood types relating to fresh weight in kilograms (water content and calorific value should be noted in the documentation). In addition, further biomass should also be included in the process inventories to enable the easy calculation of all biotic raw materials.

Regarding unused biotic extraction, values are not generally readily available for all agricultural products in literature. Since such data is seldom collected, it is usually difficult to undertake a detailed consideration of unused biotic extraction. However, this is the case for all biotic raw material calculations, not just those involving the use of an LCA database. Furthermore, as biotic input is considered using the weight at the time of harvest, the impact on the environment may not always be reflected accurately for agricultural products with different moisture contents. An adaption of the MIPS calculation method for the biotic raw materials could be considered (e.g., use of average moisture contents) in the future to rectify this shortcoming.

All in all, it can be concluded that the calculation of the material footprint including biotic and abiotic raw material can be implemented and executed using existing LCA databases. Abiotic raw material demand is

Table 6 Overview and rank of results for all impact categories investigated (lowest score is the best)

Impact category	Material footprint	Cumulative energy demand	Climate change	Ozone depletion	Acidification	Terrestrial eutrophication
Unit	kg resources	MJ	kg CO_2 eq	kg CFC-11 eq	mol H^+ eq	mol N eq
Per kg material/crop						
Copper	*1*	7	7	4	1	2
	126.0 kg	35.5 MJ	1.8 kg	1.55E−7 kg	7.77E−2 mol	1.24E−1 mol
Chr. steel	*2*	3	2	2	4.	4
	51.0 kg	79.2 MJ	4.4 kg	2.61E−7 kg	2.79E−2 mol	4.98E−2 mol
Aluminum	*3*	1	1	1	2	3
	46.3 kg	212.8 MJ	12.2 kg	7.44E−7 kg	6.59E−2 mol	8.98E−2 mol
Low-all. steel	*4*	8	5	8	7	8
	22.5 kg	34.82 MJ	2.0 kg	5.77E−8 kg	1.01E−2 mol	2.09E−2 mol
Cotton fibers	*5*	6	3	3	3	1
	11.1 kg	56.0 MJ	3.1 kg	2.38E−7 kg	3.92E−2 mol	1.47E−1 mol
Paper	*6*	5	8	6	8	7
	10.5 kg	62.8 MJ	1.3 kg	1.33E−7 kg	9.02E−3 mol	2.57E−2 mol
PET	*7*	2	4	5	5	9
	8.4 kg	83.3 MJ	2.9 kg	1.43E−7 kg	1.26E−2 mol	2.09E−2 mol
Wheat	*8*	10	10	9	11	6
	2.8 kg	18.9 MJ	0.5 kg	3.31E−8 kg	6.13E−3 mol	2.58E−2 mol
Glass	*9*	11	9	7	6	10
	2.7 kg	13.0 MJ	1.0 kg	8.81E−8 kg	1.03E−2 mol	1.93E−2 mol
HDPE	*10*	4	6	12	9	11
	2.3 kg	77.3 MJ	1.9 kg	7.01E−10 kg	7.76E−3 mol	1.38E−2 mol
Corn	*11*	9	11	10	10	5
	2.1 kg	19.9 MJ	0.5 kg	3.29E−8 kg	7.72E−3 mol	3.31E−2 mol
Concrete	*12*	12	12	12	12	12
	1.2 kg	1.9 MJ	0.1	3.70E−9 kg	2.87E−4 mol	8.87E−4 mol

relatively easy to implement with a few minor changes to the database. Biotic raw material demand can also be implemented in the existing LCA databases. However, since no mass-based flows connected to biotic inputs exist in the Ecoinvent database, each process in which an agricultural product is considered needs to be adapted separately.

The authors strongly recommend considering mass-based biotic flows and mass-based flows for top soil since all natural resources have a function irrespective of their impact on humans and ecosystems. In particular, biotic raw materials should not be overlooked, as mentioned at the beginning of the paper [42].

Calculation results

As a second part of the analysis, the adapted database was tested with regard to the environmental relevance of the material footprint by comparing the results of the material footprint with those from other selected impact categories. Although this comparison is not very

extensive, it allows some initial conclusions to be drawn regarding the environmental relevance of the material footprint: the top five material footprint materials include all of the top three materials of ozone depletion, acidification, eutrophication, and climate change and the top two of the CED (shown in Table 6). This shows that the material footprint is able to indicate an environmental relevance. The biggest differences are visible when comparing the material footprint to CED. One reason for this could be that incorporated energy, renewable enrgy, and fossil energy are evaluated in the same way, meaning that different impacts due to the different energy forms, especially regarding incorporated energy, are not shown when using the CED in an aggregated form. The material footprint, on the other hand, aggregates materials—in a similar way how CED aggregates energy—and does not evaluate metals, rock, or organic materials themselves differently. However, as mentioned in the method section, the extraction of metal, for example includes the extraction of rock and ore, so that

the material footprint for 1 kg of extracted metal will be calculated as much higher than that for 1 kg of easy-to-extract rock or soil.

In addition, some specific conclusions can be drawn:

- Metals with a low ore content and materials associated with a high amount of overburden exhibit a high relevance for the material footprint.
- Compared to the material footprint, CED, which regards energy resources, shows higher values, in particular, for plastic and agricultural products, since the incorporated energy is also considered.
- Regarding the material footprint fossil, energy-intensive materials are evaluated with a slightly lower environmental relevance compared to climate change. This is especially true for mineral oil, while energy from coal is also very raw material-intensive due to the large amount of overburden in the extraction phase.
- Compared to terrestrial eutrophication, the material footprint exhibits lower impacts for materials and processes linked to agriculture.
- In general, acidification and the material footprint exhibit a similar trend because mining, raw material extraction for fossil fuels, and the use of biomass have a high influence on both raw material demand and acidification.
- Since ozone depletion is closely linked to the use of a number of specific chemicals such as CFCs, similarities are not immediately obvious.

The next step in order to extensively analyze the environmental relevance of the material footprint should be to conduct a correlation analysis for all Ecoinvent processes and impact categories. This could be done using software tools such as Brightway [43], which allows extended graphical visualizations and can be used to analyze correlations with the material footprint and impact categories for all datasets in Ecoinvent.

Abbreviations
CED: Cumulated energy demand; CFC: Chlorofluorocarbons; HDPE: High density polyethylene; LCA: Life cycle assessment; LCI: Life cycle inventory; LCIA: Life cycle inventory analysis; MIPS: Material input per service unit; PET: Polyethylene terephthalate

Authors' contributions
MW drafted the manuscript. KW participated in its design and coordination and drafted the manuscript. Both authors read and approved the final manuscript.

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Competing interests
The authors declare that they have no competing interests.

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