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Identification of dissipative emissions for improved assessment of metal resources in life cycle assessment

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Abstract

Environmental dissipation is a novel approach to account for impacts from mineral resources. In contrast to all other resource-related life cycle impact assessment methods, which use data on extractions as input to calculation of indicator scores, environmental dissipation is characterized solely through emissions to the environment. Making environmental dissipation work as a viable resource use impact category in life cycle assessment requires, however, that truly dissipative emissions are differentiated from those anthropogenic releases which do not contribute to loss of accessibility of a given resource over the time frame considered. We present a new method that allows for this differentiation to be made for 65 metals and metalloids in a consistent way. It determines (1) whether an emission flow reported in a life cycle inventory actually contributes to loss of accessibility of a given element when environmental fate mechanisms are considered, and (2) whether the element comes from a source that would be considered as a mineral resource for any generation living between the present and the time frame of assessment. We apply the new method to four different emission inventories, and characterize the resulting list of truly dissipative emissions using recently proposed long-term environmental dissipation potentials (EDP). This highlights the need to differentiate dissipative emissions from other anthropogenic, potentially nondissipative emission flows of elements in metal resource impact assessment.

KEYWORDS

circular economy, industrial ecology, life cycle impact assessment (LCIA), material flow analysis (MFA), multimedia modeling, resource use indicator

1 | INTRODUCTION

Various approaches exist to account for impacts from mineral resource use in life cycle assessment (LCA). This includes methods relating resource consumption to resource reserves, methods which consider impacts from future resource extraction, or methods which take thermodynamic

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aspects into account (Rørbech et al., 2014). Recent developments of life cycle impact assessment (LCIA) methods for use in LCA and environmental footprint (EF) focus on resource dissipation (Beylot et al., 2020a; Schulze et al., 2020a, 2020b).

The SUPRIM project (SUstainable Management of PRIMary Raw Materials through a better approach in Life Cycle Assessment) has operationally classified dissipation of mineral resources (that is, decrease of resource accessibility understood as the ability to make use of a resource by humans), into three categories: (1) occupation in use, (2) technosphere hibernation, and (3) environmental dissipation. Dissipation through occupation in use addresses those resources which are embedded in a product that is in use (and hence not yet part of a waste flow or an emission to the environment), thereby making them unavailable for other concurrent uses. Technosphere hibernation concerns those product and waste flows which end up in stocks in the technosphere and are not recovered because of lacking economic drivers for this within the time horizon considered (van Oers et al., 2020b). The third category, environmental dissipation, concerns resource use impacts occurring solely through emissions of the elements to the environment. Occupation in use and technosphere hibernation are seen as relatively short-term phenomena (acting at time scales of less than a year to decades), while environmental dissipation is considered relevant for both short- and long-term time horizons (time scales of centuries).

Characterization factors for environmental dissipation have recently been made available for a large number of elements (van Oers et al., 2020b). However, in van Oers et al. (2020) all emissions of elements are considered dissipative, while their application in LCA might require that truly dissipative emissions are differentiated from those anthropogenic releases of metals which do not contribute to loss of accessibility of a given resource. An elementary flow reported as emission in the inventory may originate from a source of the element that would not be considered as a mineral resource now or in the future within the time frame considered, and its loss should thus not be considered dissipation of a resource. The emitted elementary flow may eventually contribute to building up a pool of that element in the environment that is or will, in the future be, accessible for extraction, and if this is the case, the emission also does not contribute to dissipation of the resource. There is thus a need for qualification of the inventory information in preparation for the application of the characterization factors proposed by Beylot et al. (2020b) or van Oers et al. (2020). We present a new method that supports this differentiation to be made.

The aim of this paper is to present a new method for identification of dissipative emissions in product life cycles based on consideration of the origin of an element reported in the emission flow and its environmental fate after emission. The method is an outcome of a collaborative research project with industry, ARP (Abiotic Resource Project). Dissipation criteria were developed and dissipation quotients calculated for a total of 23 metallic elements (i.e., metals and metalloids) taken as a starting point. Results for these 23 elements were then used to develop a stepwise approach to increase substance coverage and make the method applicable to a total of 65 metallic elements. The new method was applied to four different emission inventories, and impact scores computed using long-term EDP factors were compared to the outcome of using alternative approaches to investigate the significance of the new approach.

2 | METHODS

2.1 | Area of protection

We consider the definition of the Area of Protection (AoP), recently redefined by a UNEP/SETAC Life Cycle Initiative task force of the GLAM project (Global Guidance on Life Cycle Impact Assessment Indicators), as appropriate to use in the context of resource dissipation. It is given as: “the potential to make use of the value that mineral resources can hold for humans in the technosphere” (Berger et al., 2019). This potential is independent of the actual source or the resource of interest (i.e., primary or secondary). Damage on the area of protection is quantified as the reduction or loss of this potential caused by human activity. This definition was preferred because it has been developed specifically for use in LCA, integrates views of different stakeholders from industry and academia, and enjoys acceptance from the UNEP/SETAC Life Cycle Initiative. Note, that this AoP addresses resources only and does not consider other potential impacts of resource use and additional resource requirements (if resources are extracted from sources where concentrations of elements are low) (e.g., Schaubroeck and Rugani (2017)). The implication of this is that LCA practitioners may expect a tradeoff between environmental dissipation and other life cycle impacts which shall be considered in the LCA.

2.2 | Proposed framework

The proposed framework relies on the application of two criteria for dissipative emissions to an unfiltered (full) life cycle emission inventory, and calculation of life cycle impact scores using any set of matching characterization factors. The criteria are summarized in Table 1. Criterion A determines whether an emission flow actually contributes to loss of accessibility of the emitted element (rather than building up accessible stock of that element) when the environmental mechanisms governing the fate of the emitted element are considered over the relevant time frame. Criterion B determines whether the metal in a dissipative emission (according to criterion A) originates from a source that would be considered a mineral resource (i.e., would be accessible) for any generation living between the present and the relevant time frame. Criterion B thus excludes from

TABLE 1 Criteria for dissipative emissions and their reference concentration for the long-term time horizon

Criterion	Definition	Reference concentration
Criterion A	An elementary flow must represent an anthropogenic emission of an element for which the current annual rate of total anthropogenic emissions results in a steady state concentration in the receiving environment that is below a reference concentration reflecting what is accessible for humans within the considered time span	Average element concentration in the upper continental crust
Criterion B	An elementary flow must originate from a source with a concentration higher than a reference concentration reflecting what is accessible for humans within the considered time span	Average element concentration in the upper continental crust

Note: Both criteria must be met for an emission to be considered as dissipative. This case is illustrated in Figure 1a.

impact assessment those anthropogenic emissions, which originate from elements present as an incidental anthropogenic pollution (e.g., emission of a metal through combustion of fossil coal, where it is present in trace amounts in the coal and is not expected to be considered as a metal resource), which can be seen as potentially nondissipative. To acknowledge uncertainties, we apply the term “potentially nondissipative” rather than just “nondissipative” to those emissions which do not meet dissipation criteria.

The dissipation criteria were developed specifically for the very long-term (infinite) time horizon. The infinite time horizon was deemed most relevant to enable sustainability performance of products and technologies to be gauged in the context of the needs of both present and future generations (Brundtland, 1987), while keeping resource use a problem that is more relevant for the environmental dimension of sustainability rather than just the economic dimension (Verones et al., 2017). Note, that taking the infinite time horizon as the starting point does not mean that the time frame of the impact assessment is equal to true infinity. It depends on which environmental compartment is considered and on the residence time of an element in this environmental compartment, which is finite. For soils, metals’ residence time is typically in a range of centuries (Owsianiak et al., 2015). Further, adopting infinite (or very long) time horizon does not imply that short term impacts are ignored. Both short- and long-term impacts are captured by integrative approaches to characterization modeling, including the long-term environmental dissipation potential (EDP) factors of van Oers et al. (2020).

A reference value is needed to determine what is accessible for humans within the considered time span, and we propose that this reference is based on concentration of the element in the stock. The choice of concentration over alternative reference metrics is consistent with current exploration practice, where mineral exploration targets are generally identified where their concentration is considered anomalous (Filzmoser et al., 2005; Haldar, 2013; Joyce, 1988). Further, we propose to use concentration in the upper continental crust because upper continental crust has been the dominant source of mineral resources, and because soils are a potential source of metals for use by humans in the technosphere (Van Der Ent et al., 2015). Finally, we propose to use average concentration of an element in the upper continental crust as the reference for long-term accessibility. Today, most minerals are extracted from sites with concentrations that are 1 to 2 orders of magnitude higher than the average crustal concentration, except for gold and silver which are sometimes extracted from sources where their elemental concentration is comparable to the average crustal concentration (Singer et al., 1993). For the long-term perspective that is adopted in our approach, this suggests that average crustal concentration constitutes a threshold below which mineral resources can be considered inaccessible. This assumption is further discussed in Section S1 of Supporting Information S1.

As performance against criterion A depends on the emission compartment c , the equation for calculation of the impact score (IS) using substance-specific characterization factors, is (Equation 1):

$$IS = \sum_i \sum_c m_{\text{emitted},i,c} \cdot f_{\text{dissipative},i,c} \cdot CF_i \quad (1)$$

where $m_{\text{emitted},i,c}$ (in $\text{kg}_{\text{emitted}}$) is the total mass of element i emitted to environmental compartment c ; $f_{\text{dissipative},i,c}$ (in $\text{kg}/\text{kg}_{\text{emitted}}$) is the emission-compartment specific fraction of total emitted mass of element i to compartment c , which is dissipated; and CF_i is the characterization factor expressing the dissipation of substance i in category-specific units. Note, that Equation (1) does not require CFs to be specified in order to identify dissipative emissions using our framework. However, consistently with Beylot et al. (2020b) and van Oers et al. (2020) this CF_i is assumed to be independent of the emission compartment. The $f_{\text{dissipative},i,c}$ is emission-compartment specific because of differences in environmental fate mechanisms between environmental compartments. The product of $m_{\text{emitted},i,c}$ and $f_{\text{dissipative},i,c}$ defines the mass of element i emitted to environmental compartment c which is dissipated, $m_{\text{dissipated},i,c}$ (in $\text{kg}_{\text{dissipated}}$). Impact scores can be calculated taking this dissipated mass as input (2). Implementing this classification at the level of emitted mass, gives (2):

$$IS = \sum_i \sum_c m_{\text{dissipated},i,c} \cdot CF_i \quad (2)$$

The impact score will thus depend on multimedia fate mechanisms, determining which fraction of total emission ends up in the soil compartments, and on performance against criteria A and B in those environmental compartments. The fraction $f_{\text{dissipative},i,c}$ is equal to 1 in cases where total emission to compartment c is dissipative irrespective of the receiving compartment, and is lower than 1 in cases where a part of total emissions ends up in either natural or agricultural soil and does not meet dissipation criteria (3).

$$f_{\text{dissipative},i,c} = [1 - (f_{\text{natsoil},i,c} + f_{\text{agrsoil},i,c})] \quad (3)$$

where $f_{\text{natsoil},i,c}$ (kg/kg_{emitted}) is the fraction of total emitted mass of element i to compartment c which is transferred to natural soil and does not meet the dissipation criteria (i.e., either criterion A for natural soil, or criterion B or both), and $f_{\text{agrsoil},i,c}$ (kg/kg_{emitted}) is the fraction of total emitted mass of element i to compartment c which is transferred to agricultural soil and does not meet the dissipation criteria (i.e., either criterion A for agricultural soil, or criterion B or both).

The relevant parameters for the two criteria were derived for a total of 23 metallic elements: Ag, Al, As, Ba, Be, Cd, Co, Cr, Cs, Cu, Fe, Hg, Mn, Mo, Ni, Sb, Se, Pb, Sn, Sr, Tl, V and Zn. These elements were chosen as being covered by multimedia fate models that are needed to check compliance with criterion A.

2.3 | Calculation for criterion A

In LCA, emissions are scaled to the functional unit and very small masses are usually reported in the emission inventories (Bjørn et al., 2018). Thus, very few emissions resulting from fulfilling functional units can be expected to result in an environmental concentration higher than the reference concentration. Rather than applying to individual flows from a product system, this criterion therefore applies to all global anthropogenic emissions, occurring at present and in the future. When the resulting concentration of a given element from all these anthropogenic emissions at steady state is below the reference concentration, all elementary flows of that element, irrespective of the functional unit that they correspond to, may be considered to not contribute to a general building up of accessible environmental stock of that resource. These elementary flows may therefore be considered emissions that contribute to dissipation of the resource. By contrast, if the resulting environmental concentration from all anthropogenic emissions of an element at steady state in a given environmental (sub)compartment is above the reference concentration, any elementary flow, irrespective of the functional unit that it is attributed to, may be considered to contribute to a general building up of an accessible environmental stock of that resource. Consequently, these flows should not be classified as dissipative. In practice, this means that both background anthropogenic emissions and the potential of an element to accumulate in the environment, which in turn depends both on the ambient chemistry and physicochemical properties of the element, will determine whether an emission is dissipative according to criterion A.

Derivation of criterion A requires computation of the simulated concentration at steady state of element i emitted to compartment c in the environmental compartment of interest ($\Delta C_{\text{env},i,c}$ in ppm) and determination of the average crustal concentration ($C_{\text{ref},i}$ in ppm) for that element – the reference concentration. Criterion A is met (emission is dissipative) if the dissipation quotient ($DQ_{A,i,c}$), calculated as the ratio of change in steady concentration in the environmental compartment of interest relative to the reference concentration (Equation 4), is below 1.

$$DQ_{A,i,c} = \frac{\Delta C_{\text{env},i,c}}{C_{\text{ref},i}} \quad (4)$$

The average concentration of elements in upper continental crust were from Rudnick and Gao (2014), who curated data originating from various sources. Their recommended values were also employed to calculate the EDP characterization factors by van Oers et al. (2020).

The multimedia fate module of USEtox® 2.1 (LC-Impact version) was used to calculate steady state environmental concentrations resulting from global emissions corresponding to the current annual emission rate (Fantke et al., 2017; Verones et al., 2020). It considers metal speciation in solid and liquid phases of the soil and is applicable to elements emitted from anthropogenic sources (Owsianiak et al., 2013, 2015; Sydow et al., 2020). It allows for simulation of environmental concentrations in six environmental compartments (i.e., rural air, urban air, freshwater, sea water, natural soil, agricultural soil) resulting from emissions to any compartments. Natural and agricultural soils were chosen as the end compartments of interest because they are the top layer of the upper continental crust, and a more likely sources of any mineral resource that might re-accumulate in the long-term. However, other compartments (seawater, seawater sediment) could be considered in the same way as soil compartments are, as they contain minerals which are or could become accessible in the future (Wedding et al., 2015). Simulations were therefore made separately for emissions to rural air, freshwater, sea water, natural soil and agricultural soil. As input to the multimedia fate model, we used emission flows corresponding to the current annual total rate for emissions to each of these compartments, assuming that they are representative for future emissions as well. Future emissions may increase due to increasing extractions, but they may also decline due to stricter pollution control (Pacyna

et al., 2009). The implications of this assumption on dissipation quotients are discussed in section 4.2. Spatial differentiation in calculation of environmental concentration was not considered because (1) global, and geographically differentiated emission inventories are not available and it is not expected that they will become available in a foreseeable future; and (2) information about geography of global future emissions, that is also needed for calculation of dissipation quotients in Criterion A, is expected to be highly uncertain. Data underlying global anthropogenic emission inventories and fate simulations are presented in Supporting Information S1, Sections S2 and S3.

2.4 | Calculation for criterion B

Derivation of criterion B requires identification of the source of an element i reported in the emission inventory (a metal ore, coal, crude oil, etc.) and ultimately its origin. These sources can be of either mineral or fossil origin. This source is referred to with the subscript s . For each source-element combination, a dissipation quotient for criterion B ($DQ_{B,s,i}$) is calculated as the ratio of concentration in the source to the reference concentration (Equation 5). Criterion B is met (emission is considered dissipative) if the dissipation quotient is above 1.

$$DQ_{B,s,i} = \frac{C_{s,i}}{C_{ref,i}} \quad (5)$$

For the fossil origins, we have included those sources and those elements for which global average concentrations could be derived. We thus included crude oil and coal sources (disregarding natural gas as a potential source) based on data from Rauch and Pacyna (2009) who derived global average concentrations of Ag, Al, Cr, Cu, Fe, Ni, Pb and Zn by critically reviewing and curating available data. Although concentration of elements in different coals or crude oils can vary, global average concentration was preferred because information about which coal or crude oil an element reported in the inventory originates from, is hardly available. If this information were available, however, criterion B could be calculated for this specific source. Metals present in extracted ores as a primary metal are intrinsically accessible, and we furthermore assumed that any element present in any ore (defined as a mineral that contains metal that is valuable enough to be mined) may also be considered accessible in the long term irrespective of its concentration level in that ore, because it is not obvious which element will be the primary metal and co-produced metal in the long-term. Elements present in ores are expected to be accessible because reliable means of locating and extracting the ore are known and co-product elements can be reached with technological development if necessary in future (Cabri et al., 1989). Implications of this assumption on performance against criterion A are discussed in section 4.2.

Note, that criterion B does not differentiate between primary and secondary sources. For example, elements emitted during extractions of metals from those waste streams of fossil origin which contain high metal concentration (e.g., fly ash) are not expected to meet criterion B, because fossil coal is not expected to be mined just to recover metals from the fly ash (e.g., by burning the coal) if concentration of the element in the coal is below the reference reflecting what is accessible for humans in the very long term.

2.5 | Stepwise approach to increase substance coverage

In an ideal situation, information about masses that shall enter the characterization step is complete and known to an LCA practitioner. This, however, is not expected to be the case because life cycle emission inventories often include elements outside our core set of 23 elements. Thus, a stepwise approach was developed to increase substance coverage and to make the framework more broadly applicable to another 42 metallic elements (Figure 1). Details of the stepwise approach are presented in Supporting Information S1, Section S4.

2.6 | Case studies

The applicability of the new methods, including the stepwise approach, is demonstrated by applying it to four different life cycle emission inventories (Table 2). Characterized and normalized impact scores calculated using the proposed framework and the EDP characterization method of van Oers et al. (2020) were compared with EDP impact scores calculated for the unfiltered emission inventory where all emissions are assumed dissipative. Comparisons were made with impact scores calculated using three alternative characterization models recommended by the UNEP/SETAC Life Cycle Initiative GLAM project (Berger et al., 2019), representing different impact mechanisms. Abiotic Depletion Potential (ADP) was used as a depletion method (2020 updated as presented in van Oers et al., 2020a), Surplus Ore Potential (SOP) as a future efforts methods (as implemented in ReCiPe 2016 v1.1 based on Huijbregts et al., 2016), and Cumulative Exergy Extraction from the Natural Environment (CEENE) (2014 adapted for Ecoinvent v3.1 based on Dewulf et al., 2007) as a thermodynamic accounting method.

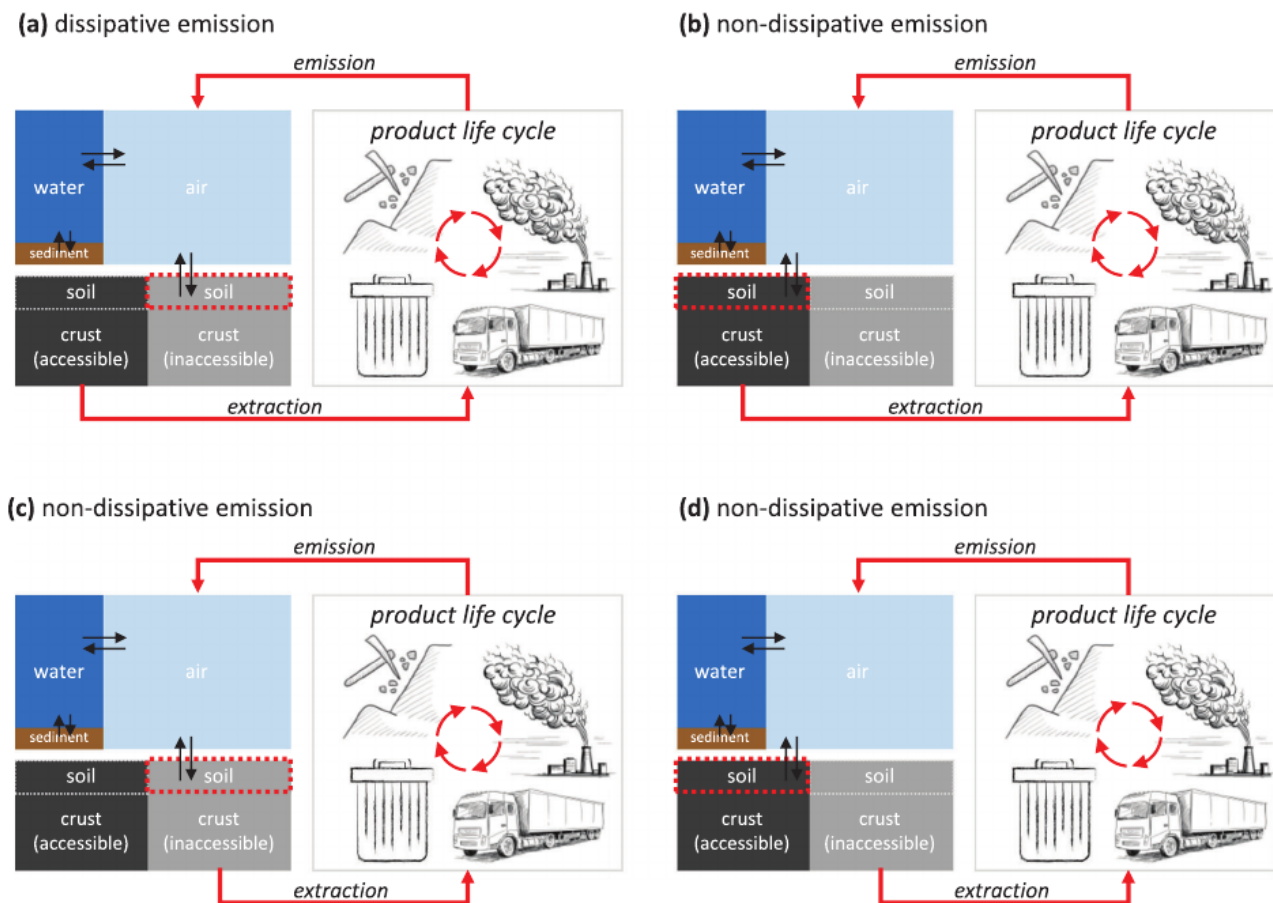


FIGURE 1 (a) Dissipative and (b–d) nondissipative emissions to air (chosen as exemplar emission compartment to make the figure legible) in product life cycles according to criteria given in Table 1. Only those emissions, which originate from a source that would be considered as a resource (here, accessible part of upper continental crust), and do not contribute to building up accessible stock in soil when environmental fate is considered, are dissipative (this is the case in (a) only). Black arrows represent environmental fate mechanisms (in illustrative only, noncomprehensive way), including transfer of emitted mass between environmental compartments. Air is chosen as exemplar emission compartment, but all compartments are relevant emission compartments and need to be modeled separately because emission compartment influences the size of $f_{\text{dissipative},i,c}$ (Equation 3)

3 | RESULTS

Dissipation quotients are first presented and interpreted for natural soils. Explanation of differences in dissipation quotients between natural and agricultural soils is presented next. Results for agricultural soils are presented in Supporting Information S1, Section S6.

3.1 | Influence of environmental fate

Dissipation quotients for criterion A in natural soils range from virtually 0 (for an emission of any element to either freshwater, or seawater or agricultural soil) to up to 1100 (emission of Ag to natural soil) across the 23 elements of the core set (Table 3). All waterborne emissions and all direct emissions to agricultural soils are therefore classified as dissipative. The majority of airborne emissions are also classified as dissipative. By contrast, direct emissions of metals to natural soil are classified as potentially nondissipative.

Strong dependency of the dissipation quotient A (DQ_A) in natural soil on the emission compartment is due to differences between environmental compartments in the mechanisms that govern the environmental fate of metals. A very small (in practice equal to 0) fraction of waterborne emissions ends up in natural soils, because there are no generic fate mechanisms that would transfer a metal from freshwater to natural soil (metals emitted to freshwater partly end up in the freshwater sediment, and partly flow out to coastal seawater). Likewise, there are no generic fate mechanisms that would transfer a metal from agricultural soil to natural soil. By contrast, a part of metal emitted to air deposits onto natural soils, explaining why DQ_A values for airborne emissions are significantly higher when compared to waterborne emissions. Within soils, metals undergo several fate processes like aging and weathering, leaching to deep soil layers and runoff to surface water (including through soil erosion). Yet, despite all these

TABLE 2 Main characteristics of the case studies used to demonstrate practical application of the proposed framework

Case and reference flow	Modeling platform and data format	Provider
Cradle-to-grave LCA on a fictive energy-using product (1 piece) ^a	SimaPro model with ecoinvent 3.5, disaggregated process	This study
Production of primary aluminium ingots (1 kg) ^b	SimaPro model with ecoinvent 3.5; disaggregated process	International Aluminium Institute
Lamp and lens materials in the ballasted compact fluorescent lamp, CFL (1 piece) ^c	SimaPro with ecoinvent 2.2; aggregated process	Four Elements Consulting, LLC
Hot and cold water supply hardware (five 100 m ² flats) ^d	Gabi model with its databases, aggregated process	The International Copper Association

^aThe product is made from Al, Cr, and Nb. It is assumed that energy required to produce and use the product during its life time comes from a fossil resource, causing emissions of Cr, Fe, Co, and Nb to the environment.

^bUnit process: Aluminium, primary, ingot [IAI Area, EU27 & EFTA] production | APOS, U, as embedded in ecoinvent, ver. 3.5 (Wernet et al., 2016).

^cInventory based on an LCA of ultra-efficient lamps (Navigant Consulting Europe, 2009).

^dUnpublished LCA commissioned by the International Copper Association.

generic fate mechanisms being active in soils, sorption of metal to soil constituents is strong enough for nearly all metals to allow for an increase in concentration in natural soils. Emissions of the majority of metals to natural soils are therefore potentially nondissipative in the long term.

Two factors explain differences in dissipation quotients between metals emitted to natural soils for criterion A. The first factor is the difference in the ability of the metals to bind to soil solids. For example, sorption of TI is the weakest among the 23 elements, resulting in relatively rapid leaching from soils and a modest concentration build up. This partly explains relatively low DQ_A values of TI when compared to other metals. The second factor is the difference in the magnitude of the anthropogenic emissions (driving the concentration build-up in competition with natural removal mechanisms), in relation to the reference concentration. For TI, V, Be and Sr the ratio of anthropogenic emissions to the reference concentrations is so low, that these metals do not witness a concentration build up above the reference concentration. Emissions of these four elements to natural soils are therefore classified as dissipative according to criterion A.

3.2 | Influence of the source

Dissipation quotients for criterion B (DQ_B) range from 0.000006 (emission of Al originating from crude oil) up to 7 (emission of Hg originating from coal) across the 23 elements of the core set (Table 3). They are generally below 1 for all elements originating from crude oil and for the majority of elements originating from coal. Thus, all emissions originating from crude oil and the majority, but not all, metal emissions originating from coal are classified as potentially nondissipative from a resource accessibility perspective (they may of course still drive impact scores for the toxicity related impact categories). Emission of an element originating from an ore that is a source of that element, is inherently dissipative. DQ_B values for elements that are present in ores as co-contaminants, could not be computed because of insufficient information about concentration of elements in various ores (and were assumed to be dissipative). DQ_B for fifteen out of 23 elements from the core set had to be derived using the stepwise procedure, and assumed to be dissipative because out of the eight metals for which data exist, only very few cases have been found to exert concentrations higher than the threshold, thus ruling out coal and oil as dissipative sources (see Supporting Information S1, Section S4).

Differences in DQ_B values between coal and crude oil are explained by differences in global average concentrations of elements, which are generally higher in coal than in crude oil. Although there is a positive relationship between average concentration in the crust and global average concentration of elements in coal or crude oil, As, Hg and Ag deviate most. Thus, emissions of As, Hg and Ag originating from coal are classified as potentially nondissipative, even if criterion A is met for these elements. Emissions of Cd originating from coal are also potentially nondissipative (again, even if criterion A is met), which is mainly because the crustal concentration of Cd is the lowest among all 23 elements considered.

3.3 | Application in LCA

Figure 2 shows characterized impacts calculated for the unfiltered emission inventories where all emissions are assumed dissipative (black bars), and for the emission inventories filtered by applying either criterion A (dashed bars), or both criteria A and B (white bars). The results show reductions in impact scores by up to a factor of 3 when truly dissipative emissions are differentiated from all other emissions which do not contribute to dissipation and resulting loss of accessibility. Table 4 presents calculation of total impact scores using the long-term EDP factors (for both filtered and unfiltered inventories), and using ADP, SOP and CEENE indicators. It shows that characterization of dissipative emissions only (as opposed to characterization of all emissions) either changes which substance is the dominant driver of impact, or changes the contribution of the

TABLE 3 Reference concentrations, estimated total anthropogenic emissions, and dissipation quotients for criteria A (in natural soils) and B

Element	Reference concentration (ppm) ^a	Estimated total anthropogenic emissions in 2010 (kg/yr)	Dissipation quotient for criterion A, DQ _A , in natural soil (dimensionless)					Dissipation quotient for criterion B, DQ _B (dimensionless)			
			Emissions to air	Emissions to freshwater	Emissions to seawater	Emissions to natural soil	Emissions to agricultural soil	Element in fossil coal	Element in crude oil	Element in an ore that is its source	Element in an ore that is not its source
Core set (basis for method development)											
Aluminium	82000	2.3 × 10 ¹¹ ^b	0.04	0	0	3.7	0	0.15	0.00001	>1 ^e	>1 ^e
Antimony	0.4	1.6 × 10 ⁸ ^c	0.15	0	0	76	0	<1 ^d	<1 ^d	>1 ^e	>1 ^e
Arsenic	4.8	6.8 × 10 ⁷ ^c	0.93	0	0	52	0	2.8	<1 ^d	>1 ^e	>1 ^e
Barium	630	4.2 × 10 ⁹ ^c	0.006	0	0	3	0	<1 ^d	<1 ^d	>1 ^e	>1 ^e
Beryllium	2.1	4.8 × 10 ⁵ ^c	0.002	0	0	0.89	0	<1 ^d	<1 ^d	>1 ^e	>1 ^e
Cadmium	0.09	3.1 × 10 ⁷ ^c	0.18	0	0	22	0	1.2	<1 ^d	>1 ^e	>1 ^e
Cesium	4.9	6.0 × 10 ⁷ ^c	0.014	0	0	7.1	0	<1 ^d	<1 ^d	>1 ^e	>1 ^e
Chromium	92	4.8 × 10 ⁹ ^b	0.097	0	0	120	0	0.17	0.002	>1 ^e	>1 ^e
Cobalt	17	5.7 × 10 ⁷ ^c	0.005	0	0	2.4	0	0.45	<1 ^d	>1 ^e	>1 ^e
Copper	28	1.3 × 10 ¹⁰ ^b	0.13	0	0	250	0	0.43	0.003	>1 ^e	>1 ^e
Iron	39000	7.2 × 10 ¹¹ ^b	0.016	0	0	6.4	0	0.23	0.0001	>1 ^e	>1 ^e
Lead	17	3.2 × 10 ⁹ ^b	0.48	0	0	330	0	0.82	0.018	>1 ^e	>1 ^e
Manganese	770	7.2 × 10 ⁹ ^c	0.039	0	0	20	0	0.18	<1 ^d	>1 ^e	>1 ^e
Mercury	0.05	2.5 × 10 ⁶ ^c	29	0	0	830	0	7	<1 ^d	>1 ^e	>1 ^e
Molybdenum	1.1	1.8 × 10 ⁸ ^c	0.3	0	0	160	0	<1 ^d	<1 ^d	>1 ^e	>1 ^e
Nickel	47	1.3 × 10 ⁹ ^b	0.022	0	0	13	0	0.3	0.21	>1 ^e	>1 ^e
Selenium	0.09	2.6 × 10 ⁶ ^c	23	0	0	650	0	<1 ^d	<1 ^d	>1 ^e	>1 ^e
Silver	0.05	1.8 × 10 ⁷ ^b	8.4	0	0	1100	0	4.3	0.002	>1 ^e	>1 ^e
Strontium	320	2.3 × 10 ⁸ ^c	0.00004	0	0	0.02	0	<1 ^d	<1 ^d	>1 ^e	>1 ^e
Thallium	0.9	3.7 × 10 ⁴ ^c	0.001	0	0	0.0001	0	0.88	<1 ^d	>1 ^e	>1 ^e
Tin	2.1	3.6 × 10 ⁸ ^c	0.45	0	0	120	0	<1 ^d	<1 ^d	>1 ^e	>1 ^e
Vanadium	97	6.0 × 10 ⁷ ^c	0.0002	0	0	0.1	0	<1 ^d	<1 ^d	>1 ^e	>1 ^e
Zinc	67	8.8 × 10 ⁹ ^b	0.028	0	0	23	0	0.43	0.005	>1 ^e	>1 ^e
Additional set (covered solely using the stepwise approach)											
Bismuth	0.16	6.4 × 10 ⁶ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Boron	17	1.6 × 10 ⁹ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Calcium	26000	1.0 × 10 ¹¹ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Cerium	63	7.4 × 10 ⁷ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Dysprosium	3.9	1.3 × 10 ⁶ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Erbium	2.3	7.1 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Europium	1	5.3 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Gadolinium	4	1.7 × 10 ⁶ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Gallium	18	2.0 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Germanium	1.4	1.6 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Gold	0.002	4.5 × 10 ⁶ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Hafnium	5.3	1.6 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Holmium	0.83	1.8 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Indium	0.1	6.7 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Iridium	0.00002	2.1 × 10 ² ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Lanthanum	31	3.4 × 10 ⁷ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Lithium	24	1.4 × 10 ⁷ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Lutetium	0.31	1.3 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Magnesium	15000	5.2 × 10 ¹⁰ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Neodymium	27	2.2 × 10 ⁷ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Niobium	12	3.7 × 10 ⁷ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Osmium	0.00003	1.6 × 10 ² ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Palladium	0.0005	5.3 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Platinum	0.0005	3.8 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Potassium	23000	7.6 × 10 ¹⁰ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Praseodymium	7.1	7.4 × 10 ⁶ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Rhenium	0.0002	8.3 × 10 ⁴ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Rhodium	0.001	8.2 ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Ruthenium	0.0003	6.5 × 10 ⁴ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Samarium	4.7	2.7 × 10 ⁶ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Scandium	14	2.5 × 10 ⁴ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Sodium	24000	7.6 × 10 ¹⁰ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Tantalum	0.9	2.0 × 10 ⁶ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Tellurium	0.001	2.4 × 10 ⁵ ^c	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f

(Continues)

TABLE 3 (Continued)

Terbium	0.7	2.5×10^{5c}	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Thulium	0.3	9.2×10^{4c}	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Titanium	3800	4.6×10^{9c}	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Tungsten	1.9	6.4×10^{7c}	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Uranium	2.7	5.1×10^{7c}	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Ytterbium	2	7.2×10^{5c}	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Yttrium	21	7.4×10^{6c}	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f
Zirconium	190	6.6×10^{8c}	<1 ^d	<1 ^d	<1 ^d	>1 ^d	<1 ^d	<1 ^e	<1 ^e	>1 ^f	>1 ^f

Note: Values in cells highlighted green indicate emissions which are potentially nondissipative. Results for agricultural soils are presented in Supporting Information S1, Section S6.

^aAverage concentration in upper continental crust, data retrieved from Table 3 in Rudnick & Gao, 2014.

^bDerived building on earlier reviews and inventories for developing normalization references for toxicity-related impact categories; see Supporting Information S1, Section S2 for details.

^cEstimated from data on extraction using a regression model; see Supporting Information S1, Section S2 for details.

^dAssumed (Step 1 of the stepwise approach); see Section 2.4 and Supporting Information S1, Section S3 for details.

^eAssumed; see Section 2.4 for details.

^fAssumed (Step 2 of the stepwise approach); see Section 2.4 for details.

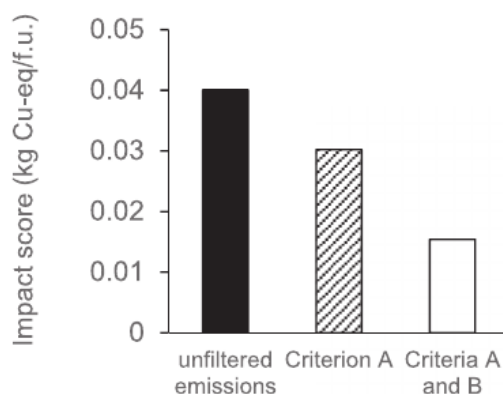
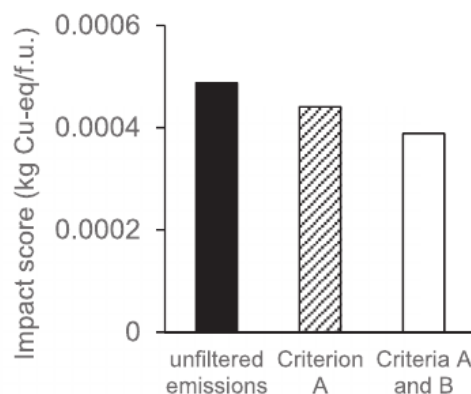
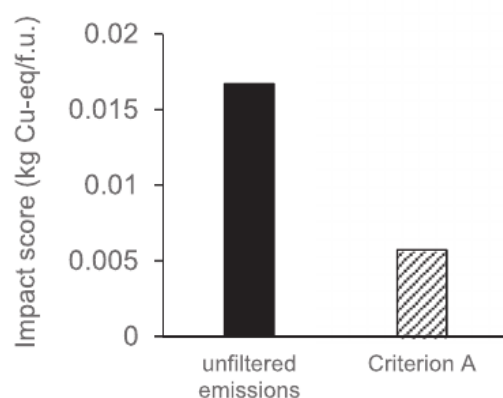
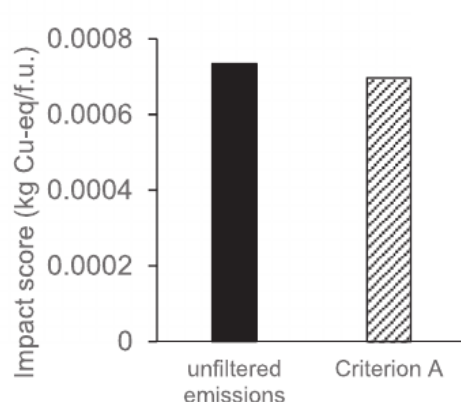
(a) an energy-using product (fictive case)**(b) primary aluminium ingots****(c) hardware for water supply****(d) CFL lamp and lens materials**

FIGURE 2 Characterized impact scores per functional unit (f.u.) in kg Cu-eq. calculated using environmental dissipation potentials (EDP) as indicators applied to unfiltered inventories where all emissions are assumed dissipative (black bars) and to filtered inventories, either applying criterion A only (dashed bars) or applying both criteria A and B (white bars). For product systems (c) and (d), criterion B could not be applied because emissions could not be tracked to their origin owing to inventories being available in aggregated form only. Data used to create the figure is provided in Supporting Information S1, Section S9

TABLE 4 Characterized and normalized impact scores and dominant contributors to total impacts calculated using EDP factors applied to dissipative emissions from filtered inventories

Product and metals intentionally used in the product	EDP indicators applied to dissipative emissions (filtered inventory) ^a	EDP indicators applied to all emissions (unfiltered inventory) ^a	ADP indicators applied to extractions	SOP indicators applied to extractions	CEENE indicators applied to extractions
An energy-using fictive product (Al, Cr, Nb)	<p>Characterized impact: 0.015 kg Cu-eq./f.u.</p> <p>Normalized impact: 0.001 pe/f.u.</p> <p>Dominant substance: Cr (56% of total impact) from emissions to air and fresh water during waste treatment (end-of-life stage)</p>	<p>Characterized impact: 0.04 kg Cu-eq./f.u.</p> <p>Normalized impact: 0.0008 pe/f.u.</p> <p>Dominant substance: Nb (62% of total impact) from emissions to air, freshwater water and natural soil during power generation (use stage)</p>	<p>Characterized impact: 0.0003 kg Sb-eq./f.u.</p> <p>Normalized impact: 0.006 pe/f.u.</p> <p>Dominant substance: Cr (99% of total impact) from extraction of Cr (raw material extraction stage)</p>	<p>Characterized impact: 0.76 kg Cu-eq./f.u.</p> <p>Normalized impact: 0.000006 pe/f.u.</p> <p>Dominant substance: Nb (75% of total impact) from extraction of Nb (raw material extraction stage)</p>	<p>Characterized impact: 1.48 MJex./f.u.</p> <p>Normalized impact: not available for CEENE indicators</p> <p>Dominant substance: Cr (77% of total impact) from extraction of Cr (raw material extraction stage)</p>
Primary aluminium ingots (Al)	<p>Characterized impact: 0.0004 kg Cu-eq./f.u.</p> <p>Normalized impact: 0.00003 pe/f.u.</p> <p>Dominant substance: Pt (38% of total impact) from emissions to air during catalytic oxidation of ammonia in nitric acid production (manufacturing stage)</p>	<p>Characterized impact: 0.0005 kg Cu-eq./f.u.</p> <p>Normalized impact: 0.00001 pe/f.u.</p> <p>Dominant substance: Pt (26% of total impact) from emissions to air during catalytic oxidation of ammonia in nitric acid production (manufacturing stage)</p>	<p>Characterized impact: 0.00008 kg Sb-eq./f.u.</p> <p>Normalized impact: 0.001 pe/f.u.</p> <p>Dominant substance: Cd (47% of total impact) from extraction of Cd during zinc-lead-mine operation (raw materials extraction stage)</p>	<p>Characterized impact: 0.17 kg Cu-eq./f.u.</p> <p>Normalized impact: 0.000001 pe/f.u.</p> <p>Dominant substance: Al (79% of total impact) from extraction of Al during bauxite mine operation (raw materials extraction stage)</p>	<p>Characterized impact: 0.53 MJex./f.u.</p> <p>Normalized impact: not available for CEENE indicators</p> <p>Dominant substance: Al (73% of total impact) from extraction of Al during bauxite mine operation (raw materials extraction stage)</p>

(Continues)

TABLE 4 (Continued)

Product and metals intentionally used in the product	EDP indicators applied to dissipative emissions (filtered inventory) ^a	EDP indicators applied to all emissions (unfiltered inventory) ^a	ADP indicators applied to extractions	SOP indicators applied to extractions	CEENE indicators applied to extractions
Hardware for water supply (Cu)	<p>Characterized impact: 0.007 kg Cu-eq./fu.</p> <p>Normalized impact: 0.0005 pe/f.u.</p> <p>Dominant substance: Cd (63% of total impact) from emissions to fresh water and air from untrackable process and life cycle stage</p>	<p>Characterized impact: 0.017 kg Cu-eq./fu.</p> <p>Normalized impact: 0.0004 pe/f.u.</p> <p>Dominant substance: Hg (63% of total impact) from emissions to air from untrackable process and life cycle stage</p>	<p>Characterized impact: 0.044 kg Sb-eq./fu.</p> <p>Normalized impact: 0.76 pe/f.u.</p> <p>Dominant substance: Mo (50% of total impact) from extraction of Mo (raw materials extraction stage)</p>	<p>Characterized impact: 48 kg Cu-eq./fu.</p> <p>Normalized impact: 0.0004 pe/f.u.</p> <p>Dominant substance: Mo (76% of total impact) from extraction of Mo (raw materials extraction stage)</p>	<p>CEENE factors are not available for Gabi</p>
CFL lamp and lens materials (Sn, Cu, Cr, Hg)	<p>Characterized impact: 0.00074 kg Cu-eq./fu.</p> <p>Normalized impact: 0.00005 pe/f.u.</p> <p>Dominant substance: Sb (96% of total impact) from emissions to fresh water from untrackable process and life cycle stage</p>	<p>Characterized impact: 0.0007 kg Cu-eq./fu.</p> <p>Normalized impact: 0.00002 pe/f.u.</p> <p>Dominant substance: Sb (91% of total impact) from emissions to fresh water from untrackable process and life cycle stage</p>	<p>Characterized impact: 0.0001 kg Sb-eq./fu.</p> <p>Normalized impact: 0.002 pe/f.u.</p> <p>Dominant substance: Sn (96% of total impact) from extraction of Sn (raw materials extraction stage)</p>	<p>Characterized impact: 0.033 kg Cu-eq./fu.</p> <p>Normalized impact: 0.0000003 pe/f.u.</p> <p>Dominant substance: Sn (91% of total impact) from extraction of Sn (raw materials extraction stage)</p>	<p>Characterized impact: 30 MJex/f.u.</p> <p>Normalized impact: available for CEENE indicators</p> <p>Dominant substance: Cu (76% of total impact) from extraction of Cu (raw materials extraction stage)</p>

Note: Comparison is made with EDP applied to unfiltered emission inventories; and ADP, SOP, and CEENE indicators applied to extraction inventories. Characterized impacts for the EDP indicators are the same as for Figure 2. Details of development of normalization references are presented in Supporting Information S1, Section S7.

^aNote that long-term emissions from landfills were not deemed relevant to consider. While long-term predictions of metal leaching from landfills can be made (Bakas et al., 2015), the fate of landfills over the long-term time frame is uncertain (Jain et al., 2014). Therefore, metals hibernating in landfills appear more relevant to characterize in the technosphere hibernation impact category.

dominant substance. It also shows that it is generally different substances that are identified as dominant drivers of impacts when dissipation-based approaches are compared to extraction-based methods. Details of application of criteria are presented in Section S5 of Supporting Information S1.

4 | DISCUSSION

4.1 | Improved assessment of mineral resources

Using the classification framework and analysis presented above we have demonstrated that not all emissions occurring in product life cycles are truly dissipative. This finding negates the default assumption that any elementary flow to the environment is dissipative (e.g., Helbig et al., 2020), and importantly should not be modeled as such in LCA. Only recently, van Oers et al. (2020) suggested that a distinction between dissipative and potentially nondissipative emissions could be made in LCA. Also Beylot et al. (2020b) indicated that a differentiation between dissipative and potentially nondissipative flows could be made based on general, shared, rules. The fact that impact scores are reduced by up to a factor of 3 for the four case studies included in the analysis, demonstrates that the consideration of criteria A and B is highly relevant to the LCIA process. Further, the fact that the extent of this reduction is dependent on the product being assessed, stresses the need to consider resource outputs from the product system, rather than resource inputs, when accounting for resource dissipation in LCA (Beylot et al., 2020b; Charpentier Poncelet et al., 2019).

The fundamental principles of a method must be scientifically solid in order to give it the stability that makes it a target for future method development. This has been attempted by careful consideration of what it is that the method must represent (i.e., dissipative use of resources and their future accessibility) and development of consistent approaches to fill data gaps through introduction of proxies and extrapolation procedures. We intentionally refrained from shaping the method according to the data that are readily available in inventory analysis today (e.g., resource input data) because this would not necessarily result in a method that addressed the issue of dissipation and accessibility. Indeed, impact scores depend more on dissipative flows within the product system rather than resource inputs, and the extent of dissipative losses can vary between different uses of the same metal (Ciacci et al., 2015). Future method development should focus on four main aspects: (1) increasing substance coverage; (2) increasing compartment coverage; (3) addressing short-term impact and other dissipation mechanisms; and (4) addressing fossil resources. These aspects are elaborated in Supporting Information S1, Section S8.

4.2 | Uncertainties and sensitivities

We propose a new approach for identification of dissipative emissions in product life cycles. It is, however, accompanied by uncertainties related to methodological choices and to the models and parameters used in calculating the quotients for the two criteria.

First, average concentration in the crust was used to distinguish what is or may become accessible to humans as a resource, but it is not known whether or not mineral resources will be extracted from sources where concentration is that low. This is generally not the case today, and there is no way to determine whether it will be in the future. Furthermore, the same reference value was used for metals extracted as primary metals and co-products. This presents a challenge, because although concentration of some co-produced metals, such as indium, can be found at the level of average crustal concentration (Werner et al., 2017), different factors may determine accessibility of indium and other co-produced elements. On the other hand, whether an element will be the primary metal and co-produced metal in the long-term will depend on future development in metal prices (Tilton et al., 2018). To explore sensitivity of our results to the choice of reference concentration, we increased it to lowest concentration in the ore that is mined as source of an element (ca. 1–2 orders of magnitude higher than average crustal concentration), and computed DQ_A values (data not shown). With this alternative reference concentration, 6 elements out of 16 for which DQ_A could be computed (Al, Ba, Cu, Pb, Mn, Ag) would now be classified as dissipative when emitted directly to natural soil (as opposed to 4 out of 16 with average crustal concentration as reference), and all elements (out of 16) except Ag would be classified as dissipative when emitted to air (as opposed to all elements except Ag, Hg, and Se when average crustal concentration is the reference). Therefore, the choice of reference concentration is expected to increase impact scores, particularly for those product systems dominated by airborne emissions of Hg and Se, and soil borne emissions of Al, Ba, Cu, Pb, Mn, or Ag. Future method development should therefore also include better specification of long-term accessibility as it depends on technological and economic factors.

Second, prediction of environmental fate of metals using multimedia fate models depends on several important model parameters. For example, the residence times of metals in the soil compartment and resulting steady-state concentrations are often determined by the rate of soil erosion, particularly so for metals which have strong affinity to soil constituents, like Cu or Pb (Owsianiak et al., 2015). The USEtox-default soil erosion rate of 0.03 mm/yr implies that about 3300 years are required for a 10-cm top soil layer to exchange completely as a result of erosion and run-off loss to surface water. Our analysis shows, however, that results are not very sensitive to the soil erosion rate. For example, Be emissions to natural soil would change status from dissipative to nondissipative in natural soil if the soil erosion rate were assumed equal to the ~50th percentile of rates found for areas under native vegetation (0.013 mm/yr) (Montgomery, 2007). Further, predicted environmental concentrations were modeled as global average, but it can be expected that in reality there will be sites where predicted concentrations are above (and below) global average. If

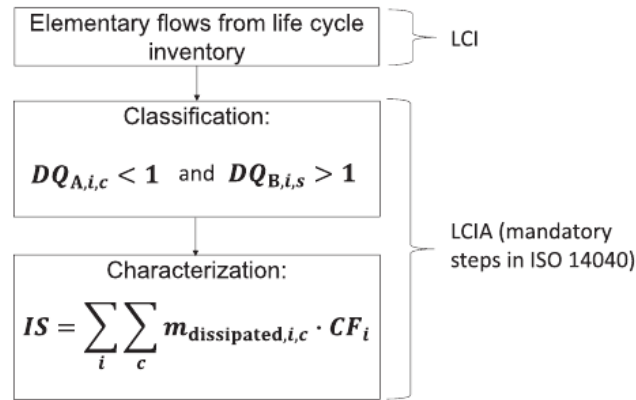


FIGURE 3 Identification of dissipative emissions as part of the classification step of the LCIA phase). The total mass of element i emitted to environmental compartment c which is dissipated, $m_{\text{dissipated},i,c}$ (in $\text{kg}_{\text{emitted}}$), is the product of $m_{\text{emitted},i,c}$ and $f_{\text{dissipative},i,c}$ (see Equation 2). The proposed classification procedure is applicable to any set of environmental dissipation-orientated long-term characterization factors which use emissions as input, like long-term EDP of van Oers et al. (2020). It is not relevant for short-term assessments of environmental dissipation and for those assessment methods which use extractions as input

average concentration is equal to the reference concentration, sites with environmental concentrations above the average concentration will be accessible. This supports the use of nonspatially differentiated multimedia fate model to account for long-term environmental dissipation impacts.

Third, global anthropogenic emission inventories of 15 elements were estimated using regression, but there is uncertainty about how applicable the regression is outside the domain of the derivation set of eight metals. Elements outside the derivation set are used in significantly lower quantities when compared to the common eight metals used to construct regressions, so it is realistic to expect that their emissions will be proportionally lower. On the other hand, emission patterns of some of these metals can be different from those of the common metals, which may introduce a bias. To explore the influence of this potential bias, we calculated dissipation quotients for criterion A assuming that total anthropogenic emissions to air and natural soil are either significantly higher (by a factor of 5) or significantly smaller (again, by a factor of 5) when compared to regression estimates. This was found to matter for four out of the fifteen elements only (Be, Ba, Cs and Co). Thus, these four elements should be given priority when refining their anthropogenic emission inventories in the context of environmental dissipation.

Finally, nonmetals (e.g., oxygen, hydrogen, carbon) do have EDP assigned in van Oers et al. (2020), but they were not considered because of challenges associated with modeling environmental fate, and because it is not yet obvious which factors determine whether a nonmetal element is accessible in the long term (see Supporting Information S1, Section S8 for discussion about factors which may influence accessibility of carbon).

4.3 | Practical implications

The main implication is the need to consider the two criteria when calculating impact scores using either long-term EDPs, or any other set of environmental dissipation oriented characterization factors calculated for the infinite time horizon that might become available in the future. We propose that the two criteria are embedded into a classification step in life cycle impact assessment (LCIA) of resource dissipation. This classification step provides the link between the life cycle inventory (LCI) phase, and the characterization step of the LCIA phase (ISO, 2006) (Figure 3).

The use of the proposed classification procedure could be seen as a challenge for the LCA practitioner. Yet, application of DQ_A is relatively straightforward because this criterion is substance- and emission-compartment specific, and can therefore be directly applied at the level of characterization factors. As shown in our case study, this allows LCA practitioners to use existing, unfiltered life cycle inventories as they occur in unit process databases, as input to life cycle impact assessment using characterization factors corrected for dissipation according to criterion A.

Application of criterion B may appear less attractive to the LCA practitioner due to a large number of processes that are typically included in an LCA model. This can be addressed using substance flow analysis. However, as demonstrated with case studies, a pragmatic (and sufficient) alternative to substance flow analysis is to identify those processes which are the hot spots and contribute most to total impact and apply criterion B to those processes only. The number of such processes is expected to be limited owing to large variability in EDP factors across different metals.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that supports the findings of this study are available in the supporting information of this article.

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