Development of a life-cycle impact assessment methodology linked to the Planetary Boundaries framework

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A R T I C L E   I N F O

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A B S T R A C T

To enable quantifying environmental performance of products and technologies in relation to Planetary Boundaries, there is a need for life-cycle impact assessment (LCIA) methods which allow for expressing indicators of environmental impact in metrics corresponding to those of the control variables in the Planetary Boundaries framework. In this study, we present such a methodology, referred to as PB-LCIA. Characterization factors for direct use in the LCIA phase of a life cycle assessment, or other life-cycle based assessment, were developed for a total of 85 elementary flows recognized as dominant contributors to transgressing specific Planetary Boundaries. Exception was made for “biosphere integrity” and “introduction of novel entities” where a Planetary Boundary is yet to be defined for the latter and characterization models are considered immature for the former. The PB-LCIA can be used to quantify the share of the “safe operating space” that human activities occupy, as was illustrated by calculating indicator scores for about 10,600 products, technologies and services exemplifying several sectors, including materials, energy, transport, and processing. The PB-LCIA can be used by companies interested in gauging their activities against the Planetary Boundaries to support decisions that help to reduce the risk of human activities moving the Earth System out of the Holocene state.

1. Introduction

It has become evident that depletion of Earth’s natural resources and services, through human activities, can lead to undesirable conditions on Earth (Daily and Ehrlich, 1992; Vitousek, 1997). In an attempt to reduce the risk of human activities inadvertently leading to a change in Earth System state towards conditions less conducive to humanity, the Planetary Boundaries (PB) framework (Rockström et al., 2009a; Steffen et al., 2015) identified nine key Earth System processes and defined quantitative ‘Planetary boundaries’ which delimit a “safe operating space” for humanity to act within. The metric of the PB and the state of the Earth System process is expressed by a control variable defined as either an environmental state or flow rate (e.g. stratospheric ozone concentration measured in Dobson units and anthropogenic nitrogen fixation in Tg N per year). Although none of the PBs, in principle, should be transgressed in order to minimize the risk of human activities pushing the Earth System out of its Holocene-like state, anthropogenic pressures have already led to exceedance or near exceedance of several PBs (Steffen et al., 2015). The PB-framework has been adopted by different societal actors such as governmental organizations and industries who have an interest in expressing sustainability in relation to the PBs (Galaz et al., 2012; Stockholm Resilience Centre, 2012; Sim et al., 2016; Bjørn et al., 2016; Clift et al., 2017). Despite this interest, however, consistent and operational methods for quantifying human activities (including the creation of products and technologies) in relation to the PBs are lacking.

1.1. Planetary Boundaries and life-cycle assessment

Life-cycle assessment (LCA) is a decision support tool (ISO 2006a,b; EC-JRC, 2010) for quantifying impacts of human activities on environment, resources, and humans. LCA involves construction of a life-cycle inventory (LCI) comprising all elementary flows (i.e. emissions and resource uses) arising throughout the life-cycle of the assessed activity. The elementary flows in the LCI are, in the life-cycle impact assessment (LCIA) phase, characterized into potential impacts by multiplication with characterization factors (CFs). LCA has been identified as a useful tool for quantifying human activities relative to the safe operating space because LCA is based on the holistic principles of assessing the full life-cycle and including all relevant environmental
impacts (Bjørn et al., 2015; Ryberg et al., 2016).

Approaches for including the PB-framework in LCA have already been put forward. This was initially seen through development of weighting factors for existing impact categories based on the distance between the PBs and their control variable value (Tuomisto et al., 2012) and as development of normalization references, based on carrying capacities and matched with existing impact categories in LCA (Bjørn and Hauschild, 2015; Sandin et al., 2015). The conversion of the metrics in the PB-framework to existing impact categories in LCA is not straightforward as there is general misalignment within the impact pathways (cause-effect chain mechanisms leading from pollutant emission or resource use to impacts) between the position of conventional LCA indicators and control variables of the PB-framework (Ryberg et al., 2016; Laurent and Owsianiak, 2017). In addition, there is a fundamental difference between conventional LCA indicators and control variables of the PB-framework with regard to the area of protection (resources, ecosystems and human health in LCA, versus Holocene state of the Earth System in the PB-framework) (Ryberg et al., 2016). These important differences pose a challenge with respect to the communication of results to decision makers who may not be familiar with conventional LCA indicators. Communication to decision makers in industries and governments could potentially be eased by expressing impacts in metrics of the PBs which are already known to decision makers. Doka (2015, 2016) presented an LCIA method where impacts of activities were related to a global annual per capita PB allowance, generally expressed in the metrics of the PBs. However, the method’s indicators are pre-allocated into an annual equal per capita personal budget which limits the method’s applicability to assessments regarding the share of a personal budget occupied by an activity. Thus, the method is suited for consumer-citizen guidance in terms of developing a sustainable lifestyle but is not suited for assessments focusing on how production oriented activities impact the PBs.

Therefore, methods that allow for expressing impacts in the metrics of the PBs and which can be scaled to the scale of the assessed activity, incl. both consumption and production oriented activities, are still required. Having, such method could also aid quantifying and evaluating progress with relation to the 12th sustainable development goal on ensuring sustainable consumption and production patterns (UN, 2015). As a first step towards the creation of such method, Ryberg et al. (2016) identified six key challenges for development and implementation of an LCIA that could fully express impact scores in the metrics of the Planetary Boundaries (referred to as PB-LCIA). The identified challenges were:

1. Introduction of a new area of protection: The Holocene state of the Earth System;
2. Calculation of characterization factors for the Earth System processes’ control variables for use in Life-Cycle Impact Assessment
3. Identifying and dealing with Earth System processes where the impacts overlap;
4. Facilitating spatial differentiation of control variables at sub-global level;
5. Applying the precautionary principle instead of best-estimates for defining the safe operating space;
6. Inclusion of environmental constraints in Life-Cycle Assessment and how to assign shares of the ‘safe operating space’ in an operational way for sustainability assessments (Ryberg et al., 2016).

In this study, we address challenge 2, 3, and 4 i.e. the quantification and expression of impact scores for human activities in metrics consistent with the PB control variables. Challenge 2 is addressed through the development of CFs. Challenges 3 and 4 are part of the CF development and are described for the relevant PBs where overlaps are identified (i.e. “change in biosphere integrity”, “ocean acidification”, and “flow of phosphorus from freshwater to oceans”) or are spatially differentiated (i.e. “freshwater use”, “land-system change”, and “atmospheric aerosol loading”). A discussion regarding how challenges 3 and 4 are resolved and the resulting implications are provided in Section 4.2.

Challenges related to interpretation of results using a PB-LCIA and assigning shares of the safe operating space (challenge 1, 5, and 6), and the requirements for applying the LCIA methodology are described and discussed in Section 4.1. These are, however, not fully explored in this study where the main focus is on the technical challenges of developing CFs which can be used in LCAs, thus establishing the groundwork for applying a PB-LCIA methodology. To what degree the PB-LCIA yields similar or different conclusions in comparison with conventional LCIA methodology (ILCD 2011; EC-JRC, 2011; Hauschild et al., 2013) is evaluated by calculating impact scores for 10,687 unit processes in the life cycle database ecoinvent which is the most established and comprehensive database of unit processes for LCA. The overall outcome of this study is a PB-LCIA methodology that can be used for assessing impacts of human activities relative to the PBs.

2. Methods

2.1. Current characterisation modelling practice

The current LCIA framework is designed to estimate time integrated exposure $[γ; \text{mass.time}]$ from a pulse emission of elementary flow $[Δm; \text{mass}]$ superimposed on a background level (Heijungs, 1995) (Eq. (1)).

$$γ = \int_0^T (e^{At}Δm)dt$$

(1)

where $t$ is time after emission, $T$ is time duration over which exposure is considered, and $A$ is a matrix of coefficients expressing a substance’s fate in the environment and exposure of humans and ecosystems. $T$ can either be finite (e.g. 100 yrs as used in the GWP100) or infinite ($T → ∞$) to capture full exposure. The analytical solution to Eq. (1) for $T → ∞$ and with negative coefficients in $A$, i.e. inputs are removed and not generated in the environment, is given in Eq. (2). This gives the conventional framework for characterization modeling in LCA where $γ$ expresses the time integrated exposure from the emitted elementary flow. $γ$ can be multiplied with an effect factor to express the potential impacts on humans or ecosystems from exposure.

$$γ = -A^{-1}Δm$$

(2)

2.2. Proposed framework for characterization modeling in PB-LCIA

Control variables in the PB-framework are expressed as environmental states or environmental flow rates where emissions and resource uses from human activities should not lead to exceedance of the PBs. Indeed single occurring pulse emissions do not generally lead to exceedance of PBs. On the other hand, long-term exceedance of PBs can be caused by human activities putting continuous pressure on the environment and this, over time, erodes resilience (Goodland, 1995; Scheffer et al., 2001). An LCA intended for relating impact scores to the PBs should include this aspect and, therefore, express impact scores as changes in environmental states or flow rates as a result of continuous pressures (i.e. continuous emission and resource uses).

Because environmental impacts in conventional LCIA are integrated over time and do not relate to a specific point in time, these can only be used for comparative purposes (Heijungs, 1995) and not for expressing changes in environmental states or flow rates. An exception is Global Temperature change Potential from a pulse emission (GTPp) (Shine et al., 2005; Levasseur et al., 2016) which express change in surface temperature at a point time as a result of a pulse GHG emission. The magnitude of the GTPp indicator is, however, highly sensitive to the specific time point and the indicator will approach zero after sufficiently long time due to removal of the GHG from the atmosphere. Generally, time-integrated impacts are not suitable for expressing...
<table>
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<th>Impact category (Earth System)</th>
<th>Impact category (Control variable in PB-framework)</th>
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<th>No. of elementary flows covered</th>
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<tr>
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<td>1</td>
<td>( CF_{\text{W,RF,}\Delta} = \frac{\Delta RF}{\text{GW,RF,}x} )</td>
<td>18</td>
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<td>Atmospheric CO(_2) concentration [ppm CO(_2)]</td>
<td>350</td>
<td>( CF_{\text{GW,CO}_{2}\text{-em} ,x} = \frac{\text{GW,CO}<em>2\text{,x}}{\Delta S\text{G}</em>{\text{GHG}}^{\text{x}}} )</td>
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<tr>
<td>Stratospheric ozone depletion</td>
<td>Stratospheric O(_3) concentration in Dobson Units [DU]</td>
<td>5% reduction relative to a-pre-industrial level of 290 DU (( = 75 \text{ DU} ))</td>
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<td>Ocean acidification</td>
<td>Carbonate ion concentration, with respect to aragonite saturation state ([\Omega_{\text{arag}}])</td>
<td>2.75</td>
<td>( CF_{\text{OC},x} \frac{\text{mol}^{2} \text{m}^{-3}}{\text{mol} \text{m}^{-3}} \frac{\Delta \text{CO}<em>2\text{,atmosphere}}{\Delta S\text{G}</em>{\text{O}_{3}}} )</td>
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<tr>
<td>Biogeo-chemical flows</td>
<td>Global: Phosphorus flow from freshwater system into ocean [Tg P yr(^{-1})]</td>
<td>11</td>
<td>( CF_{\text{P},\text{global}} = \frac{\Delta P_{\text{marine}}}{\Delta P_{\text{compound-emitted}}} \times \frac{1 \times 10^{-9}}{\text{P}<em>{\text{outflow}}} \times \frac{1 \times 10^{-9}}{\text{P}</em>{\text{outflow}}} )</td>
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<td>26.2</td>
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<td>1</td>
</tr>
<tr>
<td></td>
<td>Global: Industrial and intentional biological fixation of nitrogen [Tg N yr(^{-1})]</td>
<td>62</td>
<td>( CF_{\text{N},\text{global}} = \frac{\Delta N_{\text{hr}}}{\Delta N_{\text{compound-emitted}}} \times \frac{1 \times 10^{-9}}{\text{N}<em>{\text{emitted}}} \times \frac{1 \times 10^{-9}}{\text{N}</em>{\text{emitted}}} )</td>
<td>1</td>
</tr>
<tr>
<td>Land-system change</td>
<td>Global: area of forested land as% of original forest cover [%]</td>
<td>75%</td>
<td>( CF_{\text{forest,global}} = \frac{\Delta A_{\text{forest}}}{\Delta A_{\text{forest}}^{\text{potential}}} \times 100% )</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Biome: area of forested land as% of potential forest [%]</td>
<td>Tropical: 85%</td>
<td>( CF_{\text{forest,biome}} = \frac{\Delta A_{\text{forest}}^{\text{potential}}}{\Delta A_{\text{forest}}^{\text{potential}}} )</td>
<td>3</td>
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</tbody>
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impacts in metrics of the control variables in the PB-framework. Instead, to express impact scores as environmental states or flow rates, either at the time of equilibrium (steady state) or other points in time, the presented PB-LCIA method requires that the LCI provides flows as constant inputs [mass time$^{-1}$] instead of pulses [mass]. To accommodate for this, the functional unit (FU) in the LCA, on which the assessment is based, must be defined with a continuous constant time duration, i.e., as annual fulfilment of the function in the FU in order for the LCI to express the elementary flows that will occur annually in order to continuously fulfil the FU. The definition of the FU and the specifics surrounding it are further discussed in Section 4.1.2, where examples of how to define the FU in accordance with the new approach are presented. Mathematically, this means that instead of an emission pulse ($\Delta m$ in Eq. (2)), the input for the characterization model is a constant annual flow ($S$; mass yr$^{-1}$). Thus, the framework for the characterization modelling in the PB-LCIA follows a first-order differential equation which can be solved for steady state (Eq. (3)) to express the steady state mass ($m_{ss}$) in the environment from a constant annual input or mass at a specific point in time, where $m$ is a vector of differential quotients $dm/dt$ expressing change of mass with time.

$$m = -A \cdot m(t) + S \Rightarrow \lim_{t \to \infty} m = 0 = -A \cdot m(t) + S \Rightarrow m_{ss} = A^{-1}S \tag{3}$$

### 2.3. Characterization models

The most mature PBs defined in Steffen et al. (2015) were used as a basis for the PB-LCIA and CFs were developed for all PBs except “biosphere integrity” and the “introduction of novel entities” because a PB is yet to be defined for the latter and characterization models are considered immature for the former (Ryberg et al., 2016). Models used for developing CFs draws on existing research and experiences within the field of life-cycle impact assessment and environmental modelling in general. Models were selected based on the following requirements: Models should previously have been used in research (preferable for LCIA or other impact assessment purposes), be evaluated by peer-review, and be documented with detailed description of modelled mechanism and equations. Moreover, the complexity of the models should be consistent with the spatial and temporal scale of the control variable, e.g., a highly spatially differentiated model will not give better results than a non-spatially differentiated model if the indicator is defined at the global level. A general principle for the PB-LCIA method was to use the average approach for derivation of CFs (Huijbregts et al., 2011). The average approach is in line with the principles of the PB-framework, i.e., the adoption of a long-term perspective and giving equal importance to all elementary flows regardless of when they occur as the CFs express the average change in distance between current and preferred environmental state per unit change in elementary flow. For the control variables: “ocean acidification”, “stratospheric ozone depletion” and “land-system change”, increased values indicate an improved state and, correspondingly, a reduction leads to a worsening. For these impact categories, the characterization model was developed to follow general LCA conventions where increased impact scores are considered negative and decreased scores considered positive. How impact scores are converted to metrics directly comparable to the control variables is presented under the description of these impact categories.

The general characterization framework was based on models requiring constant inputs, which are provided from the LCI that expresses the elementary flows occurring annually in order to continuously fulfil the FU. However, there are specific cases of emissions and resource uses that in practice occur only once. This can, nevertheless, be problematic as they can lead to the exceedance of the Earth System’s recovery time and, thus potential exceedance of PBs, due to a long residence time in the environment. This can be exemplified with the example of the removal and transformation of natural forest land to agricultural land.
this case, the carbon stored in the soil and vegetation will be released once, i.e., during the transformation. Nevertheless, this single pulse emission of carbon as CO₂ may be problematic with respect to climate change due to the long atmospheric residence time of CO₂. In this case, specific CFs for emissions of CO₂ from land-transformation were estimated where the unit was expressed as “per kg” instead of “per kg per year” to indicate that this is a single occurring pulse emission. This special approach does not affect the ability to compare impacts from continuous constant emissions and pulse emissions as the indicator metric remains the same. CFs were calculated for each of the PB-LCIA impact categories listed in Table 1 using the framework and approaches described above. A brief method description for each impact category is given in this section while specific details are provided in Supporting Info. The governing equations used for characterizing the impact of human activities relative to the Planetary Boundary are presented in Table 1.

2.3.1. Climate change
Climate change affects the climate system which results not only in changes in physical conditions (temperature, rainfall, storm activity, sea level, etc.) on Earth but also the biosphere (Rockström et al., 2009a; Steffen et al., 2015). The climate change PB is described by two different control variables, i.e., energy imbalance at top-of-atmosphere expressed by radiative forcing (RF; Wm⁻²) and atmospheric CO₂ concentration [ppm CO₂] (Rockström et al., 2009a; Steffen et al., 2015). The control variable for energy imbalance at top-of-atmosphere takes into account all greenhouse gases (GHGs), aerosols, and other factors that affect radiative forcing and is considered more stringent than the boundary for atmospheric CO₂ concentration (Steffen et al., 2015). The boundary for atmospheric CO₂ concentration only includes CO₂ and CO₂-precursors and is chosen because of the large anthropogenic CO₂ emissions (Steffen et al., 2015) and because it is a proxy for radiative forcing from all GHGs because the current cooling effect from aerosols approximately counteracts the warming effect of non-CO₂ GHGs (Rockström et al., 2009a).

2.3.1.1. Energy imbalance at top-of-atmosphere. CFs for energy imbalance at top-of-atmosphere express change in RF per unit change in continuous GHG emissions. The change in RF as a result of a change in GHG emissions was based on the absolute global temperature change potential (AGTP) for a continuous emission (Shine et al., 2005) divided by the equilibrium climate sensitivity parameter which links radiative forcing and temperature (Myhre et al., 2013). AGTP was not valid for CO₂ due to its long life time and complex response function (Shine et al., 2005). Therefore, the CF for CO₂ was estimated based on the average change in RF per unit change in CO₂ concentration and average change in CO₂ concentration per unit change in annual CO₂ emissions. The latter was calculated based on an assessment of the RCP2.6 scenario (Meinshausen et al., 2011; van Vuuren et al., 2011) and the former was based on how a change in CO₂ concentration affects RF (Myhre et al., 1998; Myhre et al., 2013). For the land transformation, specific CFs, the average change in CO₂ concentration was estimated as average change in atmospheric CO₂ concentration per kg of the total tolerable mass of CO₂ emitted between 2000 and 2300 based on the RCP2.6 scenario in Meinshausen et al. (2011) and averaged over the 300 yr time horizon to express the pulse emission as an average emission flux over 300 yrs. The 300 yr time horizon was selected because the atmospheric CO₂ concentration was found to start stabilizing and to approach steady state after this period (see Fig. S1).

2.3.1.2. Atmospheric CO₂ concentration. CFs for atmospheric CO₂ concentration express the average change in atmospheric CO₂ concentration from a change in continuous annual emissions of CO₂ or CO₂-precursors and was estimated based on an assessment of the RCP2.6 scenario (Meinshausen et al., 2011). CO₂-precursors which are converted to CO₂ after reaching the troposphere were included via the mass of the emitted substance which is converted to mass of CO₂ in the atmosphere, i.e. 1.57, 2.75 (Gillenwater et al., 2006), 3, 2, and 1 (Wenzel and Hauschild, 1998) for CO, CH₄, NMVOC as hydrocarbons, NMVOC as partly oxidized hydrocarbons, and NMVOC as partly chlorinated hydrocarbons, respectively. Again, emissions that occur as one off pulse emissions, such as land transformation, were treated as special cases by estimating land transformation specific CFs where the unit was expressed in “per kg” instead of “per kg per year”.

2.3.2. Stratospheric ozone depletion
Stratospheric ozone depletion is problematic as the ozone layer protects humans and other organisms by absorbing harmful ultraviolet-B (UV-B) radiation from the sun (Fahey and Hegglin, 2011). Stratospheric ozone is removed by reactions with chlorine and bromine which are present in the stratosphere due to emissions of long-lived ozone depleting substances (ODS) (Fahey and Hegglin, 2011). CFs for stratospheric ozone depletion [Dobson units: DU yr⁻¹] were calculated via a number of steps starting from the continuous emission of ODS [kg yr⁻¹], over a change in tropospheric chlorine loading through a change in equivalent stratospheric chlorine level ending with the change in stratospheric ozone concentration [DU]. To comply with general LCA conventions where increased impact scores are considered negative, impact scores calculated with the CF express the absolute change in stratospheric ozone concentration as a result of the assessed activity ([DU]). Thus, an increasing distance indicates a lowering of stratospheric ozone concentration. The impact scores can be converted to an absolute stratospheric ozone concentration value by deducting the absolute difference from the pre-industrial stratospheric ozone concentration level to get the stratospheric ozone concentration in response to the assessed activity.

2.3.3. Ocean acidification
Ocean acidification occurs through accumulation of carbonic acid in the oceans created as the  \( \mathrm{CO}_2 \) in surface ocean waters increases in response to increased atmospheric concentrations of \( \mathrm{CO}_2 \) (Wanninkhof et al., 2009). The gradual acidification affects calcium carbonate producing organisms as CaCO₃ dissolves under acidified conditions (Orr et al., 2005; Feely et al., 2009b). Ocean acidification is expressed as the CaCO₃ saturation state (Ω) expressing the concentration of the mineral forming ions (Ca²⁺ and CO₃⁻²) relative to the minerals’ equilibrium concentration. The Ω for the CaCO₃ mineral aragonite (Ωarag) is used as control variable for the PB and becomes critical at 2.9, i.e., ca. 380 ppm (Guinotte and Fabry, 2008). To comply with general LCA conventions where increased impact scores are considered negative, the CF for ocean acidification expresses the change in absolute difference between pre-industrial Ωarag levels and Ωarag as a result of a changed emission flux of CO₂ and CO₂-precursors. Impact scores for ocean acidification can be converted to an absolute Ωarag level by deducting the absolute difference from the pre-industrial Ωarag level (=3.44) to show what the absolute Ωarag level will be as a result of the assessed activity. The CF takes into account the change in atmospheric CO₂ concentration from a change in continuous emission of CO₂ or CO₂-precursors and the change in ocean chemistry as changing Ωarag from changing atmospheric CO₂ concentration. The latter was based on estimates of seawater system parameters (such as ocean CO₂ concentration, pH and Ωarag) at different atmospheric CO₂ concentrations (Guinotte and Fabry, 2008; Feely et al., 2009a). The former was estimated based on an assessment of the RCP2.6 scenario (Meinshausen et al., 2011) with CO₂-precursors included via the mass fraction of the precursor that is converted to CO₂ (Wenzel and Hauschild, 1998; Gillenwater et al., 2006).

2.3.4. Biogeochemical flows – nitrogen
Human induced emissions of reactive nitrogen (N) compounds to the environment have a number of potential impacts, i.e., increased
atmospheric NH$_3$ concentrations, radiative forcing by N$_2$O, ground-
water contamination by NO$_3^-$, eutrophication of aquatic ecosystems
and biodiversity changes (De Vries et al., 2013). CFs for human induced
nitrogen fixation were calculated for emissions of N-compounds to
freshwater, groundwater and air. Emissions of N-compounds to the
environment, as reported in the LCI, were via inverse modelling
transformed to equivalent amounts of fixed N. It is important only to
select one of the emission compartments for the inverse modelling to
avoid double-counting because emissions of N to different environ-
mental compartments can stem from the same amount of N fixed. For
instance, for agricultural processes, we recommend to use the CFs for
emissions to freshwater via runoff or leaching through drains as this is
the dominating N loss route for this process category (Brentrup et al.,
2000; Langevin et al., 2010).

2.3.5. Biogeochemical flows – phosphorus
The problem of excessive phosphorus (P) release to the environment
is primarily related to eutrophication of lakes and freshwater reservoirs,
the risk of ocean anoxia from massive P outflows to marine waters and
changes in biodiversity (Röckström et al., 2009b; Steffen et al., 2015).
Hence, the PB for P is defined as two control variables: (i) annual mass
of P applied to soil and (ii) annual mass of P outflow to marine waters.

CFs for phosphorus applied to soil were estimated for substances that
are directly lost to surface freshwater as the primary concern was
eutrophication of rivers and lakes. Similar to the CF for the N-cycle,
annual emissions of P-compounds to the environment were via inverse
modelling transformed to equivalents of annual mass of P applied to
soil. The relationship between application of P to soil and emissions of P
to the environment was estimated with the Annual Phosphorus Loss
Estimator v2.4 (Vadas et al., 2009, 2012; Vadas, 2013).

The CFs for global phosphorus outflow to marine waters were es-
timated as annual outflow of P to marine ecosystems per unit change in
annual emission of P to freshwater systems. Fate of P in freshwater and
outflow rate to marine waters was based on transfer coefficients for
sedimentation of P in lakes and rivers and P outflow to marine waters
(Carpenter and Bennett, 2011).

2.3.6. Land-system change
The PB for Land-system change focuses on processes in land systems
that regulate exchange of energy, water and momentum between the
land surface and the atmosphere and expresses the minimum required
percentage of forest land area relative to original forest cover (Steffen
et al., 2015). Global CFs reflect changes in percentage of potential forest
area per unit area of forest transformed and were calculated as the
inverse of potential global forest area multiplied with 100%. Thereby,
the impact score shows the percentage of potential forest area trans-
formed as a result of the assessed activity, which complies with LCA
conventions about considering increasing impact scores as negative. By
deducting the impact score from 100%, it is converted to percentage of
potential forest area remaining which is directly comparable to the PB
control value. The CFs at biome level were defined in the same way,
expressing change in percentage of potential forest biome area per
change in forest biome area transformed. CFs for the global PB and CFs
for the biome level PBs are kept separate because the relative impor-
tance of the forest biome in the PB-framework, as illustrated by the
difference in required percentage of forest remaining, cannot be ade-
quately aggregated into a single score expressed in the metric of the
global forest PB. It is, therefore, recommended to show impact scores
for land-system change at both global and biome level to get a detailed
assessment at both levels.

2.3.7. Freshwater use
Freshwater flow is important for sustaining biodiversity and the
functions required for maintaining the state of the ecosystem (Poff
et al., 2010) and excessive human induced freshwater withdrawal can
affect freshwater ecosystems (Smakhtin, 2008). Average global
freshwater availability is sufficient to remain within the PB but the
large spatial and temporal variability in freshwater availability means
that PBs at river basin level are exceeded (Hoekstra and Mekonnen,
2011). The CF for global freshwater use was defined as 10$^{-9}$ km$^3$ m$^{-3}$
to convert the metric of common LCI data on water use [i.e. m$^3$ yr$^{-1}$]
to the metric of the control variable for the Planetary Boundary on global
freshwater use [i.e. km$^3$ yr$^{-1}$].

The CFs for freshwater use at the basin level express the change in
share of freshwater flow available for human activities per change in
human induced freshwater withdrawal. The calculation of freshwater
flow available for human withdrawal was made according to the vari-
able monthly flow (VMF) method (Pastor et al., 2014) using spatially
differentiated river basin data on mean monthly natural runoff
(Hoekstra and Mekonnen, 2011). The VMF method estimates the en-
vironmental flow requirements and, thus, the share of freshwater flow
which is unavailable for human withdrawal (Pastor et al., 2014). A
challenge in development of CFs for freshwater use at river basin level
is to have spatial differentiation while also allowing for aggregating
spatially differentiated impacts to a single indicator. For freshwater use,
the challenge is that averaging over regions is required to estimate a
global total single score. Here, a global average may hide important
exceedances of PBs at river basin level (Ryberg et al., 2016). To opera-
tionalize spatial differentiation of CFs, the river basins were classified
into aridity based archetypes, i.e. arid (includes hyper arid), semi-arid
and humid (includes sub-humid) based on the Aridity Index (UNEP,
1997).

Thus, CFs for water withdrawal in arid regions are higher than CFs
for water withdrawal in humid regions because water availability is
lower in arid regions. The archetype classification allowed for ag-
grating river basins within the same archetype ensuring that exceed-
ceedance in water withdrawal in arid regions is not “hidden” by water
abundance in humid regions. CFs were also calculated based on the
period of water withdrawal, i.e. if water is withdrawn during low, in-
termediate or high flow months where CFs are larger during low-flow
months where water availability is lower. The spatial and temporally
differentiated CFs allow for assessing impacts at different spatial and
temporal resolutions but cannot be aggregated to a single global score
without averaging across spatial and temporal scales and, thereby,
losing specific details. It is, therefore, suggested to evaluate impact
scores at different spatial and temporal scales, e.g. at global average
and at aridity archetype level and, if required, also at river basin level,
to get a more detailed basis for decision-making.

2.3.8. Atmospheric aerosol loading
Atmospheric aerosol loading stems from emissions of primary and
secondary aerosols which can impact global and regional climate sys-
tems (Ramanathan et al., 2001; Ramanathan et al., 2005; Boucher et al.,
2013). The control variable for atmospheric aerosol loading is the
Aerosol Optical Depth (AOD; dimensionless) which expresses the ab-
sorption or scattering of solar radiation. Atmospheric aerosol loading is
spatially and temporally differentiated because aerosols’ life time is
normally a few days (Ramanathan et al., 2001; Boucher et al., 2013).
CFs were estimated as global averages and for different regions and
express the change in AOD per annual mass emission of aerosols.
Change in AOD per unit change in mass load was expressed by the
aerosols’ specific extinction efficiency and mass load was estimated as a
function of aerosol life time and the area the aerosol distributes over.
It was assumed that aerosols are fully distributed over the regions in
which they are emitted and that aerosol mass transfer does not occur
between regions. This assumption aligns with the TM5-FAST source-
receptor model (EC-JRC, 2016) and the study by Yu et al. (2013) where
interregional transport was found to be limited, and at least 86% and
64% of changes in aerosol concentration stems from emissions within
the assessed regions, respectively. Global average CFs indicate how
emissions at different locations affect global AOD over global terrestrial
area while regional CFs (Table S9) express regional AOD from emissions
within the region. Impact scores for regional CFs cannot be aggregated to a total AOD without averaging across regions because regional AOD depends on the region’s area. The regional CFs should, therefore, be evaluated as separate impact scores to evaluate exceedance of any local PBs.

2.4. Comparison of PB-LCIA with ILCD-LCIA

An evaluation of whether the PB-LCIA would give similar or different conclusion compared to a conventional LCIA methodology was done by comparing results of the PB-LCIA with results calculated with the ‘ILCD 2011’ LCIA methodology (referred to as ILCD-LCIA). The comparison was done by comparing impact scores for impact categories in PB-LCIA with scores for similar categories in ILCD-LCIA, based on similarities in the environmental pressure and elementary flows covered in the impact category (Table 2). For instance, ocean acidification in PB-LCIA was compared with climate change in ILCD-LCIA because ocean acidification is driven by emissions of CO2 and CO2-precursors. We stress that this exercise was done only to compare results of the two methods when employed in regular life cycle assessments of products and systems. The reader should note, however, that the impact categories and their CFs should otherwise not be matched 1:1 as they differ in several areas (e.g. different units, position in impact pathway, and area of protection).

Impact scores for the PB-LCIA and the compared impact categories in ILCD-LCIA were calculated and ranked for 10,687 unit processes in the ecoinvent v. 3.1 consequential life cycle unit process database (Weidema et al., 2013). The unit processes in ecoinvent give information on the emissions and resource uses per unit output from the process. For use with the PB-LCIA, the process output was interpreted as emissions and resource uses per annual unit output to comply with the PB-LCIA’s requirement for annual flows. Pearson correlation and Spearman rank correlation analyses were performed on the absolute impact scores for regional CFs cannot be aggregated to a total AOD without averaging across regions because regional AOD depends on the region’s area. The regional CFs should, therefore, be evaluated as separate impact scores to evaluate exceedance of any local PBs.

3. Results

3.1. Characterization factors in PB-LCIA

The derived CFs for the PB-LCIA are listed in Table 3 and include CFs for 85 elementary flows covering the most prominent contributors to each of the included impact categories in Table 1. Moreover, the calculated impact scores for the unit processes in ecoinvent generally span over 10–13 orders of magnitude depending on the impact category which is comparable to the span in conventional LCIA methods (Laurent et al., 2012).

3.2. Comparison of PB-LCIA with ILCD-LCIA

The comparison of the PB-LCIA with ILCD-LCIA showed that Impact categories generally correlated well in terms of ranking unit processes with Spearman rank correlation coefficients above 0.85 for all impact categories except “biogeochemical flows – Regional P” with 0.61. The correlation between magnitude of impact scores was also found to be reasonable with Pearson correlation coefficients above 0.97 for all impact categories except “Land-system change” and “biogeochemical flows – Regional P” with 0.27 and 0.51, respectively. This indicates that the overall ranking of processes is similar for the two methods. Hence, in an LCA, both methods would generally give a similar estimate of which processes contribute most to environmental burdens. Though the two methods are found to give similar estimate of process contributions the added value of using the PB-LCIA relative to ILCD-LCIA is presented in Section 4.1.3. Fig. 1 shows the comparison of ranking of impact categories using the PB-LCIA and ILCD-LCIA for three impact categories representing the general trends in rank correlation. The climate driven impact categories and ozone depletion generally showed good correlation in ranking of the unit processes. This was due to similarities in

<table>
<thead>
<tr>
<th>ILCD-LCIA impact category</th>
<th>PB-LCIA impact category</th>
<th>Rationale for comparing impact categories</th>
</tr>
</thead>
<tbody>
<tr>
<td>Climate change</td>
<td>Climate change – CO2 concentration</td>
<td>Both express climate change</td>
</tr>
<tr>
<td>Climate change</td>
<td>Climate change – Energy imbalance</td>
<td>Both express climate change</td>
</tr>
<tr>
<td>Climate change</td>
<td>Ocean acidification</td>
<td>Ocean acidification is linked to the drivers of climate changes because ocean acidification is a consequence of anthropogenic CO2 emissions (Feely et al., 2004; Doney et al., 2009).</td>
</tr>
<tr>
<td>Ozone depletion</td>
<td>Stratospheric ozone depletion</td>
<td>Both express ozone depletion.</td>
</tr>
<tr>
<td>Land use</td>
<td>Land-system change – Global</td>
<td>Both express land use.</td>
</tr>
<tr>
<td>Water resource depletion</td>
<td>Freshwater use – River basins</td>
<td>Both express freshwater use.</td>
</tr>
<tr>
<td>Photochemical ozone formation</td>
<td>Atmospheric aerosol loading</td>
<td>The PB was compared with “Photochemical ozone formation” because both include emissions of aerosols to the atmosphere. However, the area of concern for the two indicators differ slightly, where “Photochemical ozone formation” is about ground level ozone formation (and concentration) and how this affects humans and ecosystems (EC-JRC, 2010b; Hauschild and Huijbregts, 2015; van Zelm et al., 2016), while “Atmospheric aerosol loading” is about aerosols in the atmosphere and how the increased loading may lead to undesired effects due to changes in solar radiation and regional ocean-atmosphere circulation (Steffen et al., 2015). Hence, the two impact categories differ in their area of concern; however, they have been compared in this study due to their similarities in impact pathway and to allow a comparison of results for aerosols between PB-LCIA and ILCD-LCIA.</td>
</tr>
<tr>
<td>Freshwater eutrophication</td>
<td>Biogeochemical flows – Regional P</td>
<td>The PB was compared with “Freshwater eutrophication” because in majority of LCIA methods, phosphorus is considered the primary contributor to freshwater eutrophication (EC-JRC, 2011; Goedkoop et al., 2013). This is because phosphorus is the predominant growth-limiting nutrient for freshwater ecosystems (Schindler, 1977; Carpenter et al., 1998; Smith, 2003) and thus most problematic in terms of freshwater eutrophication.</td>
</tr>
<tr>
<td>Marine eutrophication</td>
<td>Biogeochemical flows – N</td>
<td>The PB was compared with “Marine eutrophication” because in majority of LCIA methods, nitrogen is considered the primary contributor to marine eutrophication (EC-JRC, 2011; Goedkoop et al., 2013; Conne et al., 2015). This is because nitrogen is, in many cases, the predominant growth-limiting nutrient for marine ecosystems (i.e. estuaries and coastal systems) (Carpenter et al., 1998; Vitousek et al., 2002; Howarth and Marino, 2006) and thus most problematic in terms of marine eutrophication.</td>
</tr>
</tbody>
</table>
Table 3
List of estimated characterization factors for the Planetary Boundaries included in the PB-LCIA methodology.

<table>
<thead>
<tr>
<th>Earth System process</th>
<th>Control variable</th>
<th>Environmental flow</th>
<th>Emission compartment</th>
<th>Characterization factor</th>
<th>Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Climate change</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CO₂</td>
<td></td>
<td></td>
<td></td>
<td>3.53 × 10⁻¹³</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>CO₂, land transformation</td>
<td>Air</td>
<td></td>
<td></td>
<td>1.18 × 10⁻¹⁵</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>CH₄</td>
<td></td>
<td></td>
<td></td>
<td>1.59 × 10⁻¹²</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>N₂O</td>
<td>Air</td>
<td></td>
<td></td>
<td>4.64 × 10⁻¹¹</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>CO</td>
<td>Air</td>
<td></td>
<td></td>
<td>2.74 × 10⁻¹³</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>NMVOC, hydrocarbons</td>
<td>Air</td>
<td></td>
<td></td>
<td>1.06 × 10⁻¹²</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>NMVOC, partly oxidized hydrocarbons</td>
<td>Air</td>
<td></td>
<td></td>
<td>7.07 × 10⁻¹³</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>NMVOC, partly chlorinated hydrocarbons</td>
<td>Air</td>
<td></td>
<td></td>
<td>3.53 × 10⁻¹³</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>CFC-11</td>
<td>Air</td>
<td></td>
<td></td>
<td>4.79 × 10⁻¹⁰</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>CFC-12</td>
<td>Air</td>
<td></td>
<td></td>
<td>1.49 × 10⁻⁹</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>CFC-13</td>
<td>Air</td>
<td></td>
<td></td>
<td>8.61 × 10⁻⁹</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>CFC-113</td>
<td>Air</td>
<td></td>
<td></td>
<td>1.94 × 10⁻⁹</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>CFC-114</td>
<td>Air</td>
<td></td>
<td></td>
<td>7.43 × 10⁻⁹</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>NF₃</td>
<td>Air</td>
<td></td>
<td></td>
<td>7.92 × 10⁻⁹</td>
<td>W·m⁻²·kg⁻¹</td>
</tr>
<tr>
<td>SF₆</td>
<td>Air</td>
<td></td>
<td></td>
<td>6.71 × 10⁻⁹</td>
<td>W·m⁻²·kg⁻¹</td>
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<tr>
<td>HCFC-21</td>
<td>Air</td>
<td></td>
<td></td>
<td>1.39 × 10⁻¹⁰</td>
<td>W·m⁻²·kg⁻¹</td>
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<tr>
<td>HCFC-22</td>
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<td></td>
<td></td>
<td>1.63 × 10⁻¹⁰</td>
<td>W·m⁻²·kg⁻¹</td>
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<tr>
<td><strong>Atmospheric CO₂ concentration</strong></td>
<td>Air</td>
<td></td>
<td></td>
<td>2.69 × 10⁻¹¹</td>
<td>ppm·yr⁻¹·kg⁻¹</td>
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<tr>
<td>CO₂</td>
<td>Air</td>
<td></td>
<td></td>
<td>4.23 × 10⁻¹¹</td>
<td>ppm·yr⁻¹·kg⁻¹</td>
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<tr>
<td>CO</td>
<td>Air</td>
<td></td>
<td></td>
<td>7.40 × 10⁻¹¹</td>
<td>ppm·yr⁻¹·kg⁻¹</td>
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<tr>
<td>NMVOC, hydrocarbons</td>
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<td></td>
<td>8.07 × 10⁻¹¹</td>
<td>ppm·yr⁻¹·kg⁻¹</td>
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<tr>
<td>NMVOC, partly oxidized hydrocarbons</td>
<td>Air</td>
<td></td>
<td></td>
<td>5.38 × 10⁻¹¹</td>
<td>ppm·yr⁻¹·kg⁻¹</td>
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<tr>
<td>NMVOC, partly chlorinated hydrocarbons</td>
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<td></td>
<td></td>
<td>2.69 × 10⁻¹¹</td>
<td>ppm·yr⁻¹·kg⁻¹</td>
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<tr>
<td>CO₂, land transformation</td>
<td>Air</td>
<td></td>
<td></td>
<td>8.97 × 10⁻¹⁴</td>
<td>ppm·kg⁻¹</td>
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<td><strong>Stratospheric ozone depletion</strong></td>
<td>Air</td>
<td></td>
<td></td>
<td>7.85 × 10⁻⁹</td>
<td>DU·yr⁻¹</td>
</tr>
<tr>
<td>CO₂</td>
<td>Air</td>
<td></td>
<td></td>
<td>9.10 × 10⁻⁹</td>
<td>DU·yr⁻¹</td>
</tr>
<tr>
<td>CH₄</td>
<td>Air</td>
<td></td>
<td></td>
<td>1.15 × 10⁻⁸</td>
<td>DU·yr⁻¹</td>
</tr>
<tr>
<td>CFCl₂</td>
<td>Air</td>
<td></td>
<td></td>
<td>5.74 × 10⁻⁸</td>
<td>DU·yr⁻¹</td>
</tr>
<tr>
<td>HCFC-22</td>
<td>Air</td>
<td></td>
<td></td>
<td>4.83 × 10⁻⁸</td>
<td>DU·yr⁻¹</td>
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<tr>
<td>HCFC-141b</td>
<td>Air</td>
<td></td>
<td></td>
<td>1.07 × 10⁻⁸</td>
<td>DU·yr⁻¹</td>
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<tr>
<td>HCFC-142b</td>
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<td></td>
<td></td>
<td>6.76 × 10⁻⁹</td>
<td>DU·yr⁻¹</td>
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<tr>
<td>Halon-1001, Methyl bromide</td>
<td>Air</td>
<td></td>
<td></td>
<td>7.46 × 10⁻⁹</td>
<td>DU·yr⁻¹</td>
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<tr>
<td>CFC-114</td>
<td>Air</td>
<td></td>
<td></td>
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<td>DU·yr⁻¹</td>
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<tr>
<td>Halon-2402</td>
<td>Air</td>
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<td></td>
<td>5.39 × 10⁻⁸</td>
<td>DU·yr⁻¹</td>
</tr>
<tr>
<td>R-40, methyl chloride</td>
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<td></td>
<td>4.03 × 10⁻¹⁰</td>
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<tr>
<td><strong>Ocean acidification</strong></td>
<td>Air</td>
<td></td>
<td></td>
<td>8.22 × 10⁻¹⁴</td>
<td>mol·yr⁻¹·kg⁻¹</td>
</tr>
<tr>
<td>CO₂</td>
<td>Air</td>
<td></td>
<td></td>
<td>1.29 × 10⁻¹³</td>
<td>mol·yr⁻¹·kg⁻¹</td>
</tr>
<tr>
<td>CH₄</td>
<td>Air</td>
<td></td>
<td></td>
<td>2.26 × 10⁻¹⁰</td>
<td>mol·yr⁻¹·kg⁻¹</td>
</tr>
<tr>
<td>NMVOC, hydrocarbons</td>
<td>Air</td>
<td></td>
<td></td>
<td>2.47 × 10⁻¹⁰</td>
<td>mol·yr⁻¹·kg⁻¹</td>
</tr>
<tr>
<td>NMVOC, partly oxidized hydrocarbons</td>
<td>Air</td>
<td></td>
<td></td>
<td>1.64 × 10⁻¹³</td>
<td>mol·yr⁻¹·kg⁻¹</td>
</tr>
<tr>
<td>NMVOC, partly chlorinated hydrocarbons</td>
<td>Air</td>
<td></td>
<td></td>
<td>8.22 × 10⁻¹⁴</td>
<td>mol·yr⁻¹·kg⁻¹</td>
</tr>
<tr>
<td>CO₂, land transformation</td>
<td>Air</td>
<td></td>
<td></td>
<td>2.74 × 10⁻¹⁶</td>
<td>mol·kg⁻¹</td>
</tr>
<tr>
<td><strong>Biogeochemical flows</strong></td>
<td>N cycle: Global: Industrial and intentional biological fixation of N</td>
<td>Air</td>
<td></td>
<td>3.04 × 10⁻¹⁰</td>
<td>Tg·yr⁻¹·kg⁻¹·yr⁻¹</td>
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<tr>
<td>NH₃</td>
<td>Air</td>
<td></td>
<td></td>
<td>8.22 × 10⁻¹⁰</td>
<td>Tg·yr⁻¹·kg⁻¹·yr⁻¹</td>
</tr>
<tr>
<td>N-total</td>
<td>Freshwater</td>
<td></td>
<td></td>
<td>2.44 × 10⁻⁸</td>
<td>Tg·yr⁻¹·kg⁻¹·yr⁻¹</td>
</tr>
<tr>
<td>NO₂⁻</td>
<td>Freshwater</td>
<td></td>
<td></td>
<td>5.51 × 10⁻⁷</td>
<td>Tg·yr⁻¹·kg⁻¹·yr⁻¹</td>
</tr>
<tr>
<td>NO₃⁻</td>
<td>Groundwater</td>
<td></td>
<td></td>
<td>6.45 × 10⁻⁷</td>
<td>Tg·yr⁻¹·kg⁻¹·yr⁻¹</td>
</tr>
<tr>
<td>Phosphorus</td>
<td>Freshwater</td>
<td></td>
<td></td>
<td>3.68 × 10⁻⁷</td>
<td>Tg·yr⁻¹·kg⁻¹·yr⁻¹</td>
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<tr>
<td>Phosphorus</td>
<td>Freshwater</td>
<td></td>
<td></td>
<td>8.61 × 10⁻¹⁰</td>
<td>Tg·yr⁻¹·kg⁻¹·yr⁻¹</td>
</tr>
<tr>
<td><strong>P flow from fertilizers to erodible soils</strong></td>
<td>Global: P flow from freshwater systems into the ocean</td>
<td>Air</td>
<td></td>
<td>1.56 × 10⁻¹²</td>
<td>%·m⁻²</td>
</tr>
<tr>
<td>Forest transformation</td>
<td>Resource</td>
<td></td>
<td></td>
<td>4.44 × 10⁻¹²</td>
<td>%·m⁻²</td>
</tr>
<tr>
<td>Boreal forest transformation</td>
<td>Resource</td>
<td></td>
<td></td>
<td>5.26 × 10⁻¹²</td>
<td>%·m⁻²</td>
</tr>
<tr>
<td>Temperate forest transformation</td>
<td>Resource</td>
<td></td>
<td></td>
<td>4.41 × 10⁻¹³</td>
<td>%·m⁻²</td>
</tr>
<tr>
<td>Tropical forest transformation</td>
<td>Resource</td>
<td></td>
<td></td>
<td>4.41 × 10⁻¹³</td>
<td>%·m⁻²</td>
</tr>
</tbody>
</table>

(continued on next page)
Environmental flow

<table>
<thead>
<tr>
<th>Earth System process</th>
<th>Control variable</th>
<th>Environmental flow</th>
<th>Emission compartment</th>
<th>Characterization factor</th>
<th>Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Freshwater use</td>
<td>Global: maximum amount of consumptive blue water</td>
<td>Global</td>
<td>Resource</td>
<td>$1.00 \times 10^{-9}$</td>
<td>km$^3$ m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td>River basin: blue water withdrawal</td>
<td>Arid - Low flow month</td>
<td>Resource</td>
<td>$1.21 \times 10^{-9}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Arid - Intermediary flow month</td>
<td>Resource</td>
<td>$1.87 \times 10^{-10}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Arid - High flow month</td>
<td>Resource</td>
<td>$1.67 \times 10^{-11}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Arid – Average annual flow month</td>
<td>Resource</td>
<td>$1.51 \times 10^{-11}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Semi-arid - Low flow month</td>
<td>Resource</td>
<td>$4.51 \times 10^{-11}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Semi-arid - Intermediary flow month</td>
<td>Resource</td>
<td>$7.92 \times 10^{-12}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Semi-arid - High flow month</td>
<td>Resource</td>
<td>$1.17 \times 10^{-12}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Semi-arid – Average annual flow month</td>
<td>Resource</td>
<td>$9.97 \times 10^{-13}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Humid - Low flow month</td>
<td>Resource</td>
<td>$4.54 \times 10^{-12}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Humid - Intermediary flow month</td>
<td>Resource</td>
<td>$6.30 \times 10^{-13}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Humid - High flow month</td>
<td>Resource</td>
<td>$8.82 \times 10^{-14}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Humid – Average annual flow month</td>
<td>Resource</td>
<td>$7.61 \times 10^{-14}$</td>
<td>yr m$^{-3}$</td>
</tr>
<tr>
<td>Atmospheric aerosol loading</td>
<td>AOD (Global CFs shown)</td>
<td>SO$_2$</td>
<td>Air</td>
<td>$1.67 \times 10^{-13}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>SO$_3$</td>
<td>Air</td>
<td>$6.84 \times 10^{-14}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Dimethyl sulfide</td>
<td>Air</td>
<td>$2.49 \times 10^{-14}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>PM$_{2.5}$</td>
<td>Air</td>
<td>$5.20 \times 10^{-15}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>PM$_{10}$</td>
<td>Air</td>
<td>$1.08 \times 10^{-15}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Generic Carbon aerosols (e.g. organic carbon)</td>
<td>Air</td>
<td>$9.69 \times 10^{-15}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Black carbon (e.g. soot)</td>
<td>Air</td>
<td>$1.11 \times 10^{-13}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>NO$_x$, Nitrate</td>
<td>Air</td>
<td>$9.73 \times 10^{-14}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>NMVOC, urban</td>
<td>Air</td>
<td>$4.83 \times 10^{-15}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>NMVOC, rural</td>
<td>Air</td>
<td>$1.93 \times 10^{-14}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>NO$_2$</td>
<td>Air</td>
<td>$3.67 \times 10^{-14}$</td>
<td>yr kg$^{-1}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>NO$_x$</td>
<td>Air</td>
<td>$3.67 \times 10^{-14}$</td>
<td>yr kg$^{-1}$</td>
</tr>
</tbody>
</table>

derivation of characterization models where fate models for both LCIA methods are based on recommendations by IPCC and WMO for climate change and ozone depletion, respectively. The correlation between “photochemical ozone formation” and “atmospheric aerosol loading” showed lower correlation, primarily for material production (primarily wood and biofuel related processes) and processing (mainly waste treatment). This was primarily due to a difference in coverage of environmental flows. For example, “atmospheric aerosol loading” includes particulate matter as this contributes to aerosol loading, while particulate matter is not included for “photochemical ozone formation” as particulate matter does not contribute to the formation of photochemical ozone. The low Pearson correlation coefficient and Spearman rank correlation coefficient for “biogeochemical flows – Regional P” was almost independent of the unit process category, although the lowest Spearman rank correlation was found for unit processes relating to material production (mainly fuel and wood production) processing of waste, and energy production (primarily from coal and lignite). Indeed, the low correlations were primarily due to differences in coverage of elementary flows where PB-LCIA only includes emission of P compounds to freshwater while ILCD-LCIA includes other emissions routes, such as phosphate to groundwater. For instance, this was the case for the difference in ranking of electricity from coal and lignite where P emitted to groundwater is not considered in the PB-LCIA as the concern is emissions to freshwater compartments.

4. Discussion

4.1. Applicability

A PB-LCIA method with a new set of CFs for life cycle impact categories corresponding to control variables in the PB-framework were developed in this study. As a consequence, new requirements for the application of the PB-LCIA arise because the goal of an LCA using the PB-LCIA, and the proposed characterization modelling framework differs from conventional LCAs. Moreover, the new requirements and the use of the PB-framework introduce a number of new aspects with

Fig. 1. Comparison of PB-LCIA and the ILCD-LCIA in ranking of impact scores between compared impact categories for unit processes from the ecoinvent life cycle inventory database.
regards to interpretation of results from an LCA using the PB-LCIA.

4.1.1. Goal definition

The PB-LCIA was developed to allow for relating impacts of activities to the Planetary Boundaries. Thus, the goal of an LCA using the PB-LCIA is to evaluate how the assessed activity contributes or may contribute to occupying the safe operating space and to what extent the share of the operating space occupied by the assessed function can be considered environmentally sustainable. Additional goals from the assessment may be to assess the level of environmental improvements required for the assessed function to be environmentally sustainable and to test whether planned or potential environmental improvements are sufficient or if further and more ambitious improvements are required for the activity to be environmentally sustainable. This is in contrast to conventional LCAs, where the goal normally is to compare potential impacts of different ways to fulfill the same FU or to identify where in a product life-cycle impacts occur (EC-JRC, 2010a).

4.1.2. Scope definition and life cycle inventory modelling

A requirement for assessments where absolute values are important is that the scope of the assessment should cover all processes linked to fulfilling the FU to provide a comprehensive account of the assessed activity’s impacts. This requirement is already given for LCAs where the goal is to provide a descriptive environmental account of a product, e.g. Product Environmental Footprints (EC-JRC, 2010a). However, this comprehensiveness is not required for comparative LCAs, where identical processes and life-cycle stages can be excluded as only the differences between the compared systems are relevant for comparing the environmental performance (EC-JRC, 2010a). Thus, the PB-LCIA can be used for comparative LCAs but still requires a comprehensive LCI if the goal is to also provide a comprehensive account of the assessed activity’s environmental performance relative to the PBs.

To express impact scores in the metrics of the PB control variables, the new characterization models require a continuous constant input for the LCI. This aspect must be re-considered for the ILCD-LCA. Indeed, where a continuous constant input for the LCI is provided, its quantity, the duration, and to what quality the function is provided. Indeed, where finite time duration is often applied when defining the FU in conventional LCAs, when applying the PB-LCIA, it is a requirement that the duration specified in the FU should be to provide the function continuously over time. Consequently, elementary flows should be expressed per time duration (e.g. mass time$^{-1}$), i.e. as continuous flows that allow modelling changes in environmental states and flow rates. For application with existing LCA software and LCI databases this means that the FU should be scaled to one year to give results that show potential impacts associated with continuous annual fulfillment of the FU. For instance, a conventional FU for mobility could be defined as traveling 25,000 km per year for 10 years while the FU using the PB-LCIA should be defined as traveling 25,000 km per year which implies that the function is continuous and has to be provided every year as people will continue to require mobility. In fact, the PB-LCIA should be used for assessing activities where the goal is to assess a function that can be considered to be in continuous demand and, thus, will be continuously provided. This is schematically shown in Fig. 2, where life-cycle stages related to the discrete life-cycles of a continuously fulfilled function are shown. As shown in Fig. 2, a conventional LCI includes all elementary flows stemming from a finite set of discrete life-cycles; while the inventory for the PB-LCIA includes the annually occurring elementary flows stemming from overlapping discrete life-cycles which are required for continuously fulfilling the FU.

Although it is likely that people will demand the same functions today and in the future, it is acknowledged that the way these functions are fulfilled and, thus, environmental profiles, will be very different in the future. For example, there has always been a demand for mobility but economic and technological development has meant that mobility providing functions have developed over time e.g. from horse-riding to cars and airplanes. Such technological changes (e.g. effects of learning, upscaling, change in processes or technologies) are also not commonly considered in traditional LCA but can be assessed using ‘what if’ analyses. Along the same lines of thinking, and depending on whether an attributional or consequential perspective is applied, the PB-LCIA can be used to indicate either the share of the safe operating space that a particular function occupies (attributional LCA); or the consequential changes in occupation of the safe operating space from a change in how to provide a function in a certain way (e.g. through technological changes).

4.1.3. Interpretation

As illustrated by the comparison of results using the PB-LCIA and ILCD-LCIA, conclusions about the best performing product from an LCA using the PB-LCIA are likely to be similar to the ILCD-LCA for the impact categories which were closely correlated while conclusions may differ for the impact categories with a weaker correlation. This difference in results and thus interpretation can be attributed to the fact that the PB-LCIA and the underlying principles differ from conventional LCA in several aspects where the main differences are presented and discussed in the following section.

Although the primary goal of an assessment using the PB-LCIA is to evaluate the environmental sustainability of an activity, impact scores cannot directly be used for assessing this. Indeed, this requires determination of the share of the safe operating space the activity can be considered entitled, where impact scores of the activity should be within the assigned share of the safe operating space (Ryberg et al., 2016). The safe operating space can be assigned in numerous ways, depending on normative values and perceptions and on standards currently exist for best practice when it comes to assigning a share of the space. Because the safe operating space can be assigned in numerous ways, it is suggested to test how robust any conclusions about an activity’s sustainability are to the choice of sharing principle (Ryberg et al., 2016).

A key aspect of the PB-framework is that none of the PBs should be exceeded and transgression of just one PB may lead to an unacceptable risk of a state change in the Earth System due to the interlinkages between the Earth Systems processes. Hence, a ‘strong’ sustainability perspective (Dobson, 1996) should be applied when interpreting results of the PB-LCIA and only activities that do not exceed the assigned safe operating space for any of the PBs can be considered absolutely sustainable. Hence, exceedance of one PB cannot be compensated for by performing well within the assigned space for other PBs. This is in contrast to conventional LCAs, where substitutability between impacts is acceptable and where large impacts within one impact category can be compensated by impact reductions in another impact category (Ryberg et al., 2016).

The CFs for the PB-LCIA in Table 3 primarily express the fate of elementary flows in the environment and exposure of and effect on ecosystems is not part of the CFs. In that sense, the method can be considered relatively immature compared to conventional LCIA methods where exposure and effect on ecosystems is often included through damage modelling. However, further damage modelling of the impact categories in the PB-LCIA e.g., to damage on the Holocene-state would likely be a very difficult endeavor and perhaps not beneficial as this might not adequately represent the complex interactions between the Earth System processes which should be included when trying to estimate the combined effects of impacts on each Earth System process in terms of damage on the Holocene state. Moreover, the PB-LCIA only includes few impact categories relative to conventional LCIA which is primarily due to a difference in the ‘area of protection’ (AoP) for the two approaches. Conventional LCAs are concerned with protecting the three AoPs: human health, biotic natural environment and abiotic natural environment (Jolliet et al., 2004). Thus, impact categories in a conventional LCA covers all impacts that contribute to any of the AoPs.
On the other hand, the PB-framework (and hence the PB-LCIA) focuses on protecting the Earth System from moving out of the Holocene state which is considered a functional value for protecting humanity (Rockström et al., 2009a). Therefore, impact categories are only included for the Earth System components identified in the PB-framework. These are substantially fewer than those encompassed by a conventional LCA, mainly because abiotic resources and human health are not included as part of the PB-framework. That they are not included is because they have not been identified as potentially being able in their own right to change the state of the Earth System away from Holocene-like conditions. This also means that an LCA using the PB-LCIA will only provide information concerning how the assessed activity contributes or may contribute to occupying the safe operating space, as given from the goal of the study, and will not provide information about an activity’s potential impacts on resources and human health. It is important to be aware of these differences and, depending on the goal of the study, it may be beneficial to apply both a conventional LCIA and the PB-LCIA to obtain a comprehensive assessment of how the activity performs relative to the assigned safe operating space identified in the PB-framework and with regards to the conventional AoPs in LCA in order to obtain a more comprehensive basis for decision-making.

By applying the carrying capacity based normalization factors developed by Bjørn and Hauschild (2015), it is already possible to relate impacts scores estimated with ILCD-LCIA to environmental carrying capacities expressed in the default indicators of ILCD-LCIA. However, the added value of using the PB-LCIA compared to e.g. ILCD-LCIA is that PB-LCIA allows for expressing the results for indicators which are consistent with the control variables of the PB-framework. This is a starting point for absolute sustainability assessments using the PB-framework. In fact, this eases communication of LCA results to decision makers already familiar with the PB-concept, thereby, providing a better basis for decision makers to make environmentally conscious and sustainable decisions. Moreover, if a share of the safe operating space is assigned to the activity, then the method can be used for assessing the absolute environmental sustainability of that activity (both production and consumption oriented activities) by evaluating whether or not the activity exceeds its assigned share of the safe operating space and, thus, whether or not it can be considered sustainable in absolute terms.

4.2. Dealing with overlapping or spatially differentiated boundaries

For the PB-LCIA, overlapping impact categories were accepted as the overall goal was to comply with the PB-framework and being able to express impacts of activities in metrics of the control variables in the PB-framework. The acceptance of overlapping impacts, such as climate change and ocean acidification which both are the result of atmospheric CO₂ concentration is not problematic because the PB-LCIA method is not intended for comparisons across impact categories. Instead, the focus is on relating each particular impact category either to the PB or an assigned share of the safe operating space. This differs from conventional LCIA which strives to have mutually exclusive and collectively exhaustive (MECE) impact coverage. This is important because comparisons across impact categories are often done in conventional LCA after normalization and weighting. This may lead to ‘double counting’ of the same pressures for more than one impact category which may bias results.

Aggregation of the spatially differentiated PBs, i.e. freshwater use, land-system change and atmospheric aerosol loading, to a single score for each impact category was not feasible in the PB-LCIA, as aggregation would require averaging over regions potentially leading to the overlooking of important regional exceedances of the PBs. The reason for this being that control variables of spatially differentiated PBs in the PB-framework are expressed in metrics that are not additive across systems because each depends on the specific spatial system. For instance, for freshwater use at the river basin level; if 50% and 75% of the water flow in two river basins is withdrawn for human use, the two shares cannot be added to give the total share of water withdrawn.

Instead, the additive values flow volume and withdrawn volume have to be known for each river basin to estimate the average share of water withdrawn for human use across the river basins. This issue was resolved by providing results at different spatial scales; i.e. showing global average results and also spatially differentiated average results to allow assessing impact scores at different spatial resolution. Further improving spatial aggregation would require modification of impact indicators from the PB control variables to additive metrics that are independent of the spatial system. Similar approaches have been used for conventional LCIA methods, where spatially differentiated impact categories are expressed in additive metrics, such as mass equivalents which can be aggregated across spatial systems. The advantage of modifying impact indicators is that regional impacts can be aggregated but the consequence is that a direct link to the PB-framework and the ability to relate impact scores to the safe operating space disappears.

5. Conclusions

This work presents the first steps in developing an operational method for quantifying impacts of human activities and expressing
these in the metrics of the PB control variables. CFs for a total of 85 elementary flows recognized as dominant contributors to transgressing specific PBs have been calculated. The method can be used for assessing the share of the safe operating space that human activities occupy or for conducting assessments of absolute sustainability after assigning a share of the safe operating space. The current version of the method is considered as a proof-of-concept but further testing and validation is required before the method can be considered mature. Nevertheless, the method can already be used by companies interested in gauging their activities against the PBs, and be used to complement conventional LCAs to aid informed decision making in which the finite nature of Earth’s natural capital is taken into account.

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Appendix A. Supplementary data

Supplementary data (detailed description of characterization methods for each impact category and complementary result figures) associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoind.2017.12.065.

References


