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DOI: 10.1111/jiec.13260

FORUM

JOURNAL OF INDUSTRIAL ECOLOGY WILEY

Life cycle assessment-based Absolute Environmental Sustainability Assessment is also relative

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Editor Managing Review: Annie Levasseur

Abstract

Over the past years, an increasing number of scholarly papers have used the planetary boundaries (PBs) within life cycle assessment (LCA) to determine if the life cycle impacts of a product system fit within those PBs and thereby establish the absolute sustainability of the product system. This type of LCA is nowadays coined as LCA-based Absolute Environmental Sustainability Assessment (AESA). “Absolute” thereby refers to methods enabling the comparison of environmental impacts of products, companies, nations, and so on, with an assigned share of environmental carrying capacity for various impact categories. A recent review of LCA-based AESA methods and their applications characterized 47 studies “according to their intended application, impact categories, basis of carrying capacity estimates, spatial differentiation of environmental model and principles for assigning carrying capacity.” However, the review and the majority of studies reviewed did not, or only to a limited extent, discuss potential temporal issues of assigning carrying capacity to product systems. Several of the carrying capacity estimates have a time dimension while LCA results lack a time dimension. In this article, we show that assigning PBs to product systems is only technically possible when adopting several fundamental though unrealistic assumptions, and conclude that even product LCA-based AESA is relative. This should not withhold scholars from developing approaches applying the PBs in LCA, but it should prevent them from claiming and using the term “absolute.”

KEYWORDS
Absolute Environmental Sustainability Assessment (AESA), industrial ecology, life cycle assessment (LCA), planetary boundaries, temporal issues

1 | INTRODUCTION

Following the launch of the planetary boundary (PB) concept (Rockström et al., 2009; Steffen et al., 2015), various authors have tried to establish a connection with existing environmental assessment methods. An important class of such methods is that of environmental life cycle assessment (LCA), a widely used method to assess the potential environmental impacts of product and service systems (Hellweg & Milà i Canals, 2014). Traditionally, LCA produces quantified impact indicators, informing the analyst that, for instance, polyethylene bottles have a potential climate impact of *x* kg CO_2-equivalent per bottle used, possibly with an indication of a range or standard deviation. Such impact indicators have been used so far to compare alternative product systems, or to find hotspots along the life cycle for improvement.

Bjørn and Hauschild (2015) introduced the term carrying capacity into the LCA-debate and defined it as "the maximum sustained environmental interference a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert." Bjørn, Chandrakumar, et al. (2020) later changed this definition into "the maximum persistent impact that the environment can sustain without suffering perceived unacceptable impairment of the functional integrity of its natural systems or, in the case of non-renewable resource use, that corresponds to the rate at which renewable substitutes can be developed." The PB concept provides a framework for defining estimates of carrying capacity as absolute limits to environmental impacts. For instance, for climate change, a critical value of 350 ppm CO_2 has been proposed (Rockström et al., 2009).

Establishing a connection between the frameworks of traditional, comparative LCA and absolute PBs has now become an active field of research (Bjørn et al., 2016; Bjørn, Sim, et al., 2019; Bjørn, Sim, Boulay, et al., 2020; Bjørn, Sim, King, 2020; Castellani et al., 2016; Chandrakumar & McLaren, 2018; Chandrakumar et al., 2019; Doka, 2016). The product level is not the only one where such research takes place. Far more research has been carried out on assigning the PBs to regions and countries (e.g., Cole et al., 2014), sectors (e.g., Algunaibet et al., 2019), or companies (e.g., Clift et al., 2017). In this article, the emphasis will be on assigning PBs to product systems.

Recently, Bjørn, Richardson, et al. (2019) proposed a framework for Absolute Environmental Sustainability Assessment (AESA) methods to study production and consumption activities. The framework consists of assessment steps and methodological choices needed to translate PBs to PB shares that can be allocated to an activity or to, in our case, a product system. One of the key choices to make concerns the quantitative assignment of PBs to product systems. This assignment is needed since PBs refer to the planet as a whole while product LCAs focus on individual product systems. Translating PBs to lower scales involves not just bio-physical dimensions (e.g., the relative sizes of countries), but may also involve differences in welfare and equity (Kates et al., 2001; Notter et al., 2013; Raworth, 2012; Steffen & Stafford Smith, 2013; UN, 2002; UNCED, 1992a, 1992b). Häyhä et al. (2016) developed a framework for a systematic translation of the PBs of Rockström et al. (2009) and Steffen et al. (2015) to boundaries for lower scales accounting for these three dimensions.

Coming back to the product level, over the past few years, scholarly publications attempting to use the PBs concept to define boundaries for product systems have gained momentum as referred to earlier. Each of these attempts—implicitly or explicitly—includes different aspects of the Häyhä et al. (2016) and Bjørn, Richardson, et al. (2019) frameworks including different assignment principles (Clift et al., 2017). Some of the publications use the term "LCA-based AESA," where the A of AESA stands for "Absolute." AESA refers to "an assessment that evaluates the absolute environmental sustainability of an anthropogenic system by comparing its estimated environmental impact to its assigned carrying capacity, taking a life cycle perspective and, ideally, having complete coverage of impact categories" (Bjørn, Chandrakumar, et al., 2020); see Box 1 for an illustration of this definition (Bjørn, Chandrakumar, et al., 2020).

Box 1 | Illustrative example of LCA-based AESA

"Instead of comparing the environmental impacts of different modes of transport to each other, an absolute assessment compares them to an external list of environmental carrying capacities. For example, the life-cycle climate impacts of a person's annual commuting generated by using a diesel car, an electric train and a bicycle could be compared to a share of a carrying capacity derived from the 1.5-degree climate goal of the Paris Agreement. This carrying capacity share could be calculated using one or more sharing principles. For example, an 'equal per capita'-principle could initially be used to assign a carrying capacity share to an individual, followed by another principle that captures the value of commuting relative to the other consumption activities in which that individual engages. It may then turn out that the bicycle is the only mode of transport whose climate impact does not exceed its assigned carrying capacity. Note that the ranking of performance between modes of transport is likely to be similar in a relative and absolute assessment (e.g., diesel car worst and bike best)."

Source: Bjørn et al. (2020a)

Adopting the term "absolute" in combination with LCA sounds as a "contradiction in terms," however. In LCA curricula, one of the first things a student learns is that LCA is a method for comparing alternative product systems on environmental aspects, implying that LCA is relative. For instance, the ISO 14044 standard (ISO, 2006) mentions that LCA results in "relative expressions and do not predict impacts on category endpoints, the exceeding of thresholds, safety margins or risks," The relative nature of LCA has been confirmed again and again (Guinée et al., 2017; Heijungs et al., 2019; Norris, 2001; Owens, 1997), so the label "absolute LCA" appears to be a contradiction in terms.

A recent review of LCA-based AESA methods and their applications characterized 47 studies "according to their intended application, impact categories, basis of carrying capacity estimates, spatial differentiation of environmental model and principles for assigning carrying capacity" (Bjørn, Chandrakumar, et al., 2020). As will be discussed in detail later, the review and also the majority of studies reviewed did not or only to a limited extent discuss potential temporal issues of assigning carrying capacity to product systems, while several of the carrying capacity estimates have a time dimension that might not be reconcilable with the time dimension present in LCA.

In this article, we address these temporal issues. We refrain from discussing issues such as the assignment of a share of the carrying capacity to a product system, country, or region and the differences in spatial representation of carrying capacity estimates and LCA results, topics that Bjørn, Chandrakumar, et al. (2020) already discussed comprehensively. We also do not discuss the incompatible metrics between LCIA methods and PB control variables in this article. Several publications (Doka, 2016; Ryberg, Owsianiak, Richardson, et al., 2018) have already excellently addressed this important issue. Here, we dive further into the following research question: *Is LCA-based AESA reconcilable with the temporal constraints of LCA*. First, we discuss some lessons from the past regarding the relativity of LCA and the temporal constraints behind that. Then, we discuss to what extent and how authors applying the PB concept to LCA have addressed and possibly solved these temporal issues. Finally, we discuss our findings and present our conclusions.

2 | WHY LCA IS RELATIVE

2.1 | A long-standing discussion

In the 1980s of the previous century, when LCAs were still called "ecobalances" (Bundesamt für Umweltschutz, 1984), Resource and Environmental Profile Analysis (Hunt & Franklin, 1996) or product line analyses (Grieshammer et al., 1991), the first methods for impact assessment were developed on the basis of so-called "units of polluted air" (Bundesamt für Umweltschutz, 1984). To understand how these methods work, we elaborate a simple example. Suppose a product system emits 2 kg of pollutant X and 10 kg of pollutant Y to air. How can we aggregate these into one number, acknowledging that the pollutant X and Y are not equally hazardous? The "trick" was found by using air quality standards. Suppose an authoritative body, such as WHO or US-EPA, has decided that the safe air quality limit for pollutant X is 10 g/m^3, and for pollutant Y it is 20 g/m^3. Then, you need 100 m^3 of air to dilute 1 kg X to the safe level, so 200 m^3 of air for the 2 kg of the product system example given earlier. Likewise, for 1 kg of Y, you need 50 m^3 air, so for the product system's 10 kg you need 500 m^3. Altogether, the product system needs 200 + 500 = 700 m^3 air to dilute its emissions just to the air quality limits. So, the air pollution score of this product system is 700 m^3 units of polluted air (UPA; this was the abbreviation used at that time). The UPA method was of course a crude method because it did not include degradation and multimedia transport and adopted air quality standards that may be a compromise between toxicology, economics, and technology. It was therefore soon replaced by multimedia models and predominantly toxicology-based thresholds (Guinée & Heijungs, 1993), which has today evolved in the well-known and broadly accepted USEtox model (Henderson et al., 2011; Rosenbaum et al., 2011).

The basic idea of "units of polluted air" is very instructive in the context of understanding the relativity of LCA vis-à-vis the absolute boundaries of the PB framework. In the oldest life cycle impact assessment (LCIA) methods, we do not claim that those 700 m^3 are available. Perhaps there are so many other polluting activities taking place, that all clean air is already fully exhausted. Or perhaps there is plenty of clean air. We only use the absolute air quality limits to relatively rank pollutants: Pollutant X is twice as hazardous as pollutant Y, because its air quality limit is half that of pollutant Y. We can also adopt a more precautionary approach and apply safety factor, for example, 10. Then the air quality limits change to 1 and 2 g/m^3, and the LCA score changes into 7000 m^3 of polluted air. But that value, 700 or 7000, has no meaning in itself. It has only a meaning in the context of a comparison of product systems, or in a contribution analysis seeking the dominant aspects. Note that it is also not possible to determine if a certain product system complies with the air quality guidelines. Concluding that a product system's impact is 700 m^3 polluted air cannot be coupled to a judgment if the air quality guidelines are respected or transgressed by that product system.

How does the earlier mentioned idea translate to the PB framework? According to Steffen et al. (2015), the proposed global phosphorous (P) boundary designed to avert widespread eutrophication of freshwater systems is 6.2 Tg/yr. Suppose product system A emits 10 kg P and product system B emits 50 kg P. Then A has, at least for phosphorous, a lower emission than B. But whether the emission of 10 kg P or 50 kg P fits within the PB of 6.2 Tg/yr and thus could be classified as environmentally sustainable in an absolute sense cannot be determined since LCA results miss a real-time dimension. To explain the latter, we need to dive further into some temporal constraints of LCA.

2.2 | Temporal constraints of LCA

The discussion on LCA missing a real-time dimension goes back to the 1990s of the previous century and was connected to a discussion on the irreconcilability of risk assessment (RA) and LCA (Heijungs & Guinée, 1993). The discussion on the irreconcilability of PB and LCA is very similar to the discussion on the irreconcilability of RA and LCA. The latter discussion was revived recently (Guinée et al., 2017; Linkov et al., 2017; Tsang et al., 2017) and it was argued among others that it is fundamentally impossible to perform an RA within the framework of LCA (Guinée et al., 2017). Inspired by this RA–LCA discussion, we illustrate similar problems between PB and LCA subsequently.

Consider a world with just two product systems, A and B, and five processes, generation of electricity, production of product A, production of product B, use of product A, and use of product B. We receive raw datasets on these processes from the producers and a statistics office based on their reporting for one year: 200 million MJ of electricity was produced emitting 1 billion kg of phosphorous (P), 40 million units of product

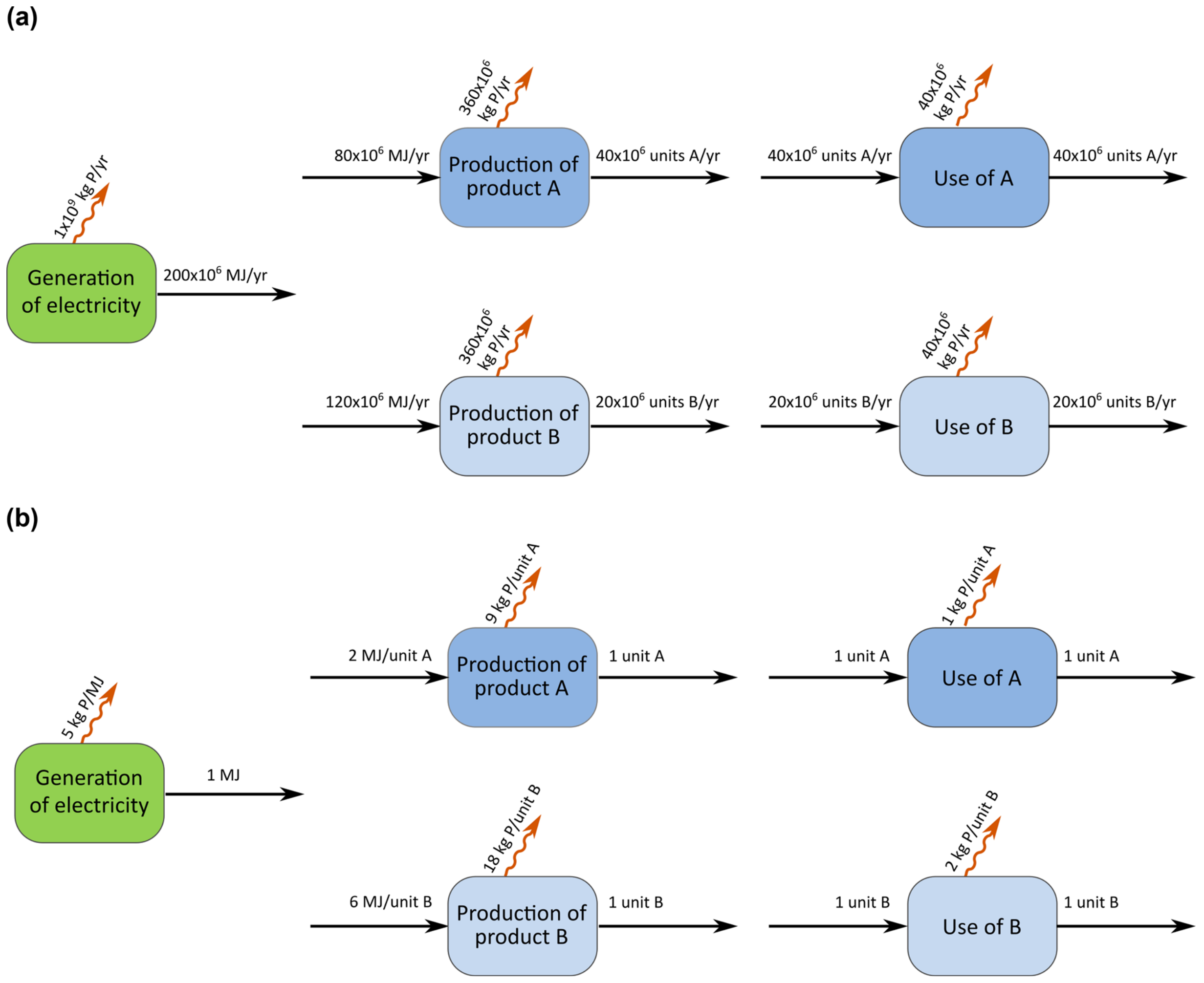

FIGURE 1 The hypothetical case study on products A and B. Orange arrows indicate the emission of phosphorus (P). Lines indicate unscaled in- and outputs for raw data of unit processes, for example, as reported per year by companies (Figure 1a: raw data, unscaled), or unscaled unit process data for use in LCA studies with no time dimension anymore (Figure 1b: unit process data for LCA, unscaled): Temporal issue 1.
Note: Underlying data for Figure 1 are hypothetical and explained in the main text

A were produced requiring 80 million MJ of electricity and emitting 360 million kg of P, 20 million units of product B were produced requiring 120 million MJ of electricity while also emitting 360 million kg of P, 40 million units of product A were consumed causing an emission of 40 million kg of P, and finally 20 million units of product B were consumed causing an emission of 40 million kg of P. These data are presented in Figure 1a. We can transform these datasets into data for use in LCA studies. We then rescale the original data to unit amounts of electricity generated, products produced, and products used, respectively. Then, the production of 1 unit of A emits 9 kg phosphorous (P), and the production of one unit of B emits 18 kg of P. Both production processes require electricity, respectively 2 and 6 MJ per unit of output. The electricity production process emits 5 kg of P per MJ. The use of 1 unit of A emits 1 kg P, and the use of 1 unit of B emits 2 kg of P (Figure 1b). Note that emissions of P from the different system processes can take place in the same, in subsequent years or can even be diluted over several years. For example, emissions of using A and B can take place years after the production of A or B.

We can now scale the process data of Figure 1b to the functional units in terms of the use of 1 unit of A and the use of 1 unit of B. Then, an LCA for product A would yield 1 kg + 9 kg + 5×2 kg = 20 kg of P per unit of product A; and an LCA for product B would yield 2 kg + 18 kg + 5×6 kg = 50 kg of P per unit of product B (Figure 2a).

Now suppose the global PB suggests that the safe level is 10^9 kg of P per year (= 1 Tg/yr). Then this world could accommodate 50 million units of A in one year, or 20 million units of B in one year, or any combination of A and B just matching the boundary. But the LCA only calculated impacts per unit of product, irrespective of the actual consumption level. Therefore, we cannot answer the question if product A is environmentally sustainable in an absolute sense. It depends on the number of consumed products per year.

To solve this, suppose that we decide to base the analysis on real consumption volumes, and that this consumption level is 40 million units of product A per year, and 20 million units of product B per year. Then the impact from all consumed products A is 800 million kg of P in one year,

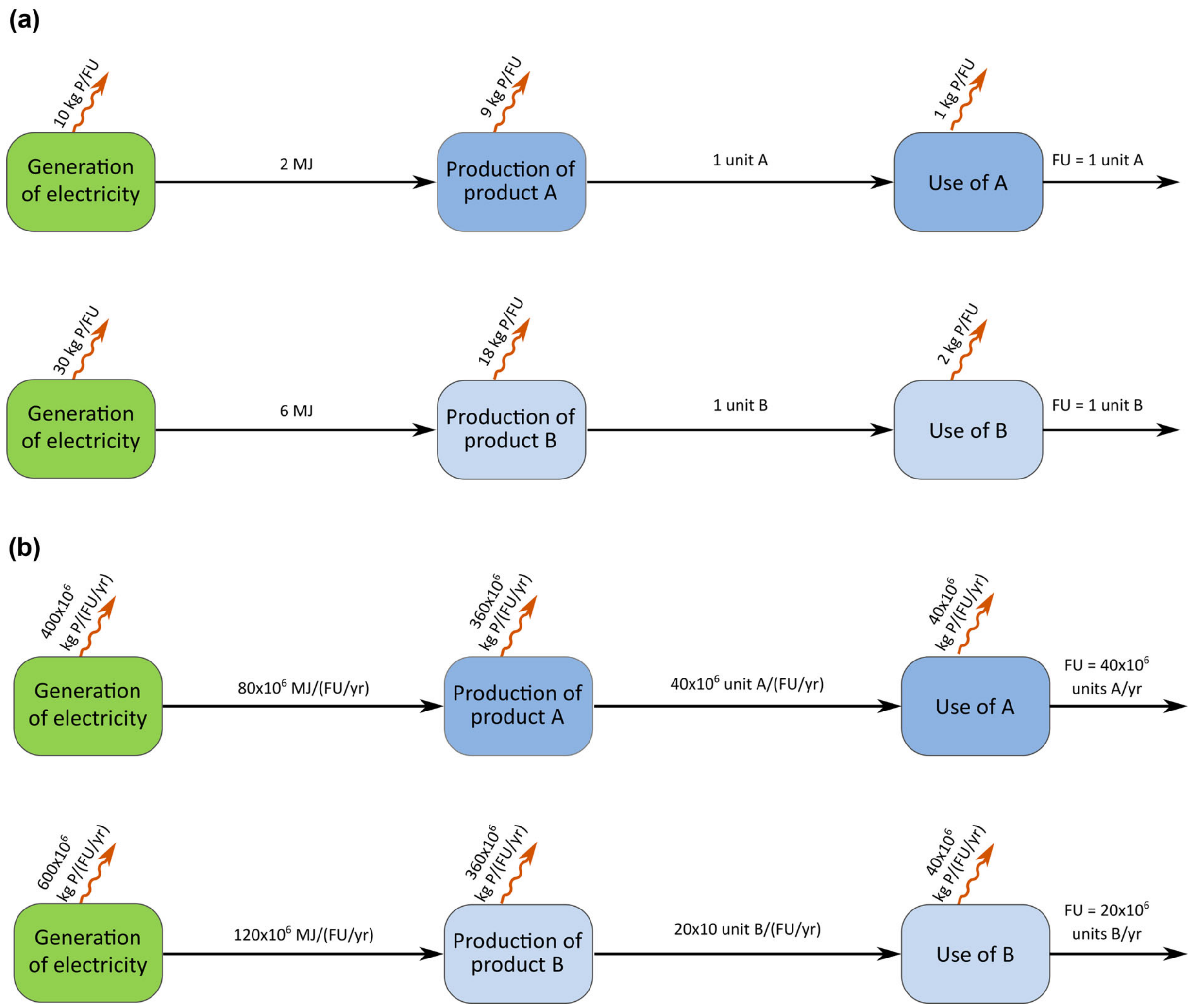

FIGURE 2 The hypothetical case study on products A and B scaled to a functional unit (FU) without a time dimension (Figure 2a: processes scaled to functional unit) and scaled to a functional unit based on known absolute size of product use A and B (Figure 2b: processes scaled to functional unit per year): Temporal issue 2. *Note*: Underlying data for Figure 2 are hypothetical and explained in the main text

and the impact from all consumed products B is 1 billion kg of P in one year (Figure 2b). For product A, this is less than the safe level of 10^9 kg per year, but product B alone is filling up the entire boundary. Together the products exceed the boundary. In order to determine if product A and B individually are environmentally sustainable, we need to assign a part of the global PB for P to product A and a part to product B. This is the issue of quantitative assignment of PBs to product systems that is part of the framework of Bjørn, Richardson, et al. (2019). Now suppose that there are ways to unambiguously assign the global PB for P to product A and product B. Then, theoretically, the assigned PB for product A and the assigned PB for product B would exactly add up to the global PB, since we defined our world as existing of just these two product systems.

So far so good, it seems. However, as already noted earlier, there is one more temporal issue: Processes and related emissions of P can take place in the same year, in subsequent years, or they can even be diluted over several years (see Figure 3). A life cycle of a product system can easily span several years or even decades. For example, emissions of using A and B can take place years after the production of A or B.

2.3 | Temporal constraints of LCA

When assigning proposed PBs to product and service systems, we thus need to account for the three temporal constraints of LCA:

1. Unit process data in LCA studies and databases lack a time dimension (*temporal issue 1*).
2. In most LCA studies, functional units are not based on annual consumption data and thus also lack a time dimension (*temporal issue 2*).

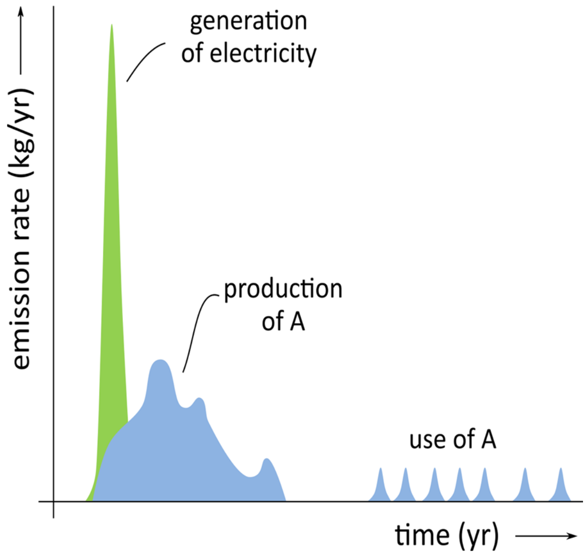

FIGURE 3 Hypothetical rates of emissions from different processes of product system A. Emissions may span multiple years or take place in different years: Temporal issue 3

3. Processes and related emissions over a life cycle of a product system can easily span several years or even decades, but almost never occur in the same year (*temporal issue 3*).

On the other hand, most, although not all,[1] proposed boundaries have a temporal dimension; they are expressed as an *annual* threshold level of a control variable, e.g., in Tg/year for P. When assigning proposed boundaries to product boundaries, one has to deal with the differences in time dimensions of LCA results (are expressed as quantities, e.g., kg) and annual PBs (expressed as flows, e.g., kg/yr), which poses similar problems as faced when introducing flux-based multimedia models into LCIA methods for toxicity impact categories (Guinée & Heijungs, 1993).

Temporal issue 1 (Figure 1) is a property of unit process LCA databases that cannot be changed by practitioners. Temporal issue 2 (Figure 2), on the other hand, can be addressed by basing functional units of LCA studies on real annual consumption data for a region, country, or the world. Using annual consumption data may reflect a realistic yearly emission for a consumption process, but it will not do so for any of its up- and downstream processes as we are not considering their total process flows but only the quantity needed for that specific annual consumption. A functional unit based on annual consumption data introduces a time dimension to these up- and downstream processes, but this reflects a virtual flow[2] rate rather than the real flow rate. However, assuming the perfect assignment of, for example, a global PB to all existing products systems in the world (assigning based on one consistent sharing principle and no double-counting, see Section 4), the assigned PBs to all globally existing product systems might exactly add up to the global PB (see Section 4 where we scrutinize the tenability of this assumption), despite having virtual flow rates for up-and downstream processes.[3] With this, we address time issue 2 without addressing issue 1. In addition, temporal issue 3 could be addressed by adopting a functional unit per year *and* assuming that it recurs in the same amount for each next year until a steady state has been reached.[4] Now we have technically addressed all-time issues, but we had to make two fundamental assumptions: *we assume perfect assignment of PBs to all existing product systems and a recurring of functional units in the same amount every year*.

3 | HOW DO LCA-BASED AESA METHODS ADDRESS THE TEMPORAL CONSTRAINTS OF LCA

Knowing the temporal constraints of LCA and the fundamental assumptions to be made, we can now address the question: How do existing studies establishing a connection between LCA and the PBs deal with them? For this, we selected all 47 lifecycle-based AESA studies reviewed by Bjørn,

[1] Several PBs as introduced by Rockström et al. (2009) have an explicit time dimension: Rate of biodiversity loss in number of species per million per year, nitrogen cycle in millions of tonnes per year, phosphorous cycle in millions of tonnes per year, and global freshwater use in cubic km per year. Other PBs have an implicit time dimension: Climate change in ppm is connected to emissions of GHGs in tonnes per year, stratospheric ozone depletion in Dobson units is connected to tonnes of CFC-emissions per year, and so on.
[2] Virtual refers to the fact that the time dimension added to up- and downstream processes in this way is not the actual rate at which these processes run, but the rate at which the functional unit runs.
[3] When assuming *perfect assignment and steady-state annual consumption*, the virtual flow rates of upstream and downstream processes of all products systems including those same processes should add up exactly to the total annual process flows of these up- and downstream processes.
[4] If each year the functional unit per year is the same (thus assuming a steady-state has been reached), total emissions related to one functional unit will equal the emissions in one year, which could solve the problem of product life cycle spanning multiple years while PBs only span one year, but it relies on the steady-state assumption.

TABLE 1 Summary of review results. Underlying data for Table 1 are available in Table S1 and S2 of Supporting Information S1

	Handling of temporal issues (problem addressed, solution proposed, assumptions mentioned?)		
Study reference	*Issue 1 (unit process data in LCA studies lack a time dimension)*	*Issue 2 (functional units lack a time dimension)*	*Issue 3 (LCA life cycles span multiple years and PBs only one year)*
Bendewald and Zhai (2013)	Not addressed.	Not addressed.	Not addressed.
Bjørn et al. (2016)	Addressed but not as part of assigning PBs to product systems, but rather as part of developing a PB-based characterization method.	Not addressed.	Not addressed.
Roos et al. (2016) Case study based on the method by Sandín et al. (2015)	Not addressed.	Not explicitly addressed. Solution implicitly proposed: "The service provided by one day's use of each studied garment."	Not explicitly addressed. Solution implicitly proposed: "where it can be reasonably assumed that the per capita demand for the FU is constant over time."
Ryberg, Owsianiak, Clavreul, et al. (2018) Case study based on the method by Ryberg, Owsianiak, Richardson, et al. (2018)	Not addressed.	Explicitly addressed. Solution: FU/yr.	Explicitly addressed, similar to Ryberg, Owsianiak, Richardson, et al. (2018).
Ryberg, Owsianiak, Richardson, et al. (2018)	Not addressed.	Explicitly addressed. Solution: FU/yr.	Explicitly addressed. "Continuous demand" (steady-state) assumption (see Section 4).
Sandín et al. (2015)	Not addressed.	Not explicitly addressed. Solution implicitly proposed: FU/day.	Not addressed.
Wolff et al. (2017)	Not addressed.	Not explicitly addressed. Solution implicitly proposed: FU/day.	Not addressed.

Chandrakumar, et al. (2020) (except for two studies that could not be identified in the public literature) and assessed the way they address the three temporal issues and:

1. Issue 1 (unit process data in LCA studies lack a time dimension): Does the study address the problem, is a solution proposed, and are underlying assumptions clearly mentioned?
2. Issue 2 (functional units lack a time dimension): Does the study include a functional unit based on a real annual consumption level?
3. Issue 3 (life cycles span multiple years and PBs only one year): Does the study address the problem, is a solution proposed, and are underlying assumptions clearly mentioned?

We also study the extent to which these studies mention the two assumptions:

1. Assumption 1 (assume perfect assignment of PBs to all existing product systems): Does the study assume that a proposed PB is allocated with a 100%-rule over all possible products?
2. Assumption 2 (recurring of functional units in the same amount every year): Does the study assume that we live in a steady-state world in which consumption levels of all products are constant?

Review results of all 45 studies are presented in Table S1 in the supporting information available on the Journal's website.

We found that most studies reviewed by Bjørn, Chandrakumar, et al. (2020) do not discuss the three temporal issues at all. Table 1 only shows seven studies that implicitly or explicitly discuss at least one of the three temporal issues. Only Bjørn et al. (2016) discuss issue 1 (unit process data in LCA studies lack a time dimension). These authors develop carrying capacity-based characterization factors for acidification for use in LCA. For calculating the indicator scores, the authors argue that the "[…] user should […] define the duration of environmental interventions (t) of each emission location," and in their case study they apply their approach to single processes, a coal-fired electricity plant, in 45 states of contiguous United States to annual electricity consumption by an average inhabitant. That however is not a full LCA and avoids the discussion on the problem that annualized consumption levels only introduce a virtual flow rate to up- and downstream processes and not a real flow rate.

Bendewald and Zhai (2013), Roos et al. (2016), Wolff et al. (2017), and Sandín et al. (2015) do not discuss issue 2 (functional units lack a time dimension) explicitly but they implicitly address it as their functional units included a time dimension (per year or per day). Wolff et al. (2017), for example, perform a cradle-to-farm-gate assessment of the food portfolio of products of the Casino company in France. Also in this case, the time dimension for the upstream processes represents a virtual flow rate since the food portfolio data may represent an exact year, but all upstream supply processes including farms and fertilizer companies will not be captured in this FU by their full annual production capacities. Sandín et al. (2015) write that their procedure also holds for other contexts than adopted in their paper "where it can be reasonably assumed that the per capita demand for the functional unit is constant over time." The latter suggests that the authors are aware of the "temporal" dichotomy between LCA and PBs, adopt the "virtual flow rate" solution, but this is merely based on our interpretation as it does not become fully clear from their publication.

The only method proposal that explicitly addresses both issue 2 and issue 3 (LCA life cycles span multiple years and PBs only one year) is Ryberg, Owsianiak, Richardson, et al. (2018). They recommend defining a functional unit per year to align the dimensions of PB and LCA results. In addition, they explicitly mention that they then assume a steady-state for that function, and consider it "to be in continuous demand" (see Figure 2 in Ryberg, Owsianiak, Richardson, et al. (2018)). They also acknowledge that "although it is likely that people will demand the same functions today and, in the future, [...] the way these functions are fulfilled and, thus, environmental profiles, will be very different in the future."

4 | DISCUSSION

According to Bjørn, Chandrakumar, et al. (2020), an LCA-based AESA study "evaluates the absolute environmental sustainability of an anthropogenic system by comparing its estimated environmental impact to its assigned carrying capacity, taking a life cycle perspective and, ideally, having complete coverage of impact categories," for example, the PBs of Rockström et al. (2009). The 45 studies concisely reviewed in this article on three temporal issues, all fit to this definition according to Bjørn, Chandrakumar, et al. (2020).

We identified three temporal issues for assigning PBs to product systems and analyzed that these can be technically overcome when adopting two fundamental assumptions. In our review, we found that only 7 out of the 45 LCA-based AESA studies reviewed in Bjørn, Chandrakumar, et al. (2020) implicitly or explicitly discuss at least one of these three temporal issues.

The most complete and explicit approach to these temporal issues so far is by Ryberg, Owsianiak, Richardson, et al. (2018) adopting annual consumption of products as a basis for functional units while assuming continuous recurring of functional units in the same amount every year. Their solution addresses both temporal issues 2 and 3.

As discussed in the beginning of this article, PBs have been used as absolute benchmark for a region's, a country's, a sector's, or a company's environmental performance. As Hauschild (2015) already stated: "Taking the approach further down to the product level to answer the question whether a product is sustainable in absolute terms is not meaningful unless the assessment is communicated together with fundamental conditions like:

- The assumed size of the safe operating space for each of the considered environmental impacts
- The number of people assumed to share these spaces
- The way in which the spaces have been divided between these people and the products that they consume."

We add to this that the temporal differences between LCA results and PBs can only be addressed satisfactorily if we adopt FUs based on annual consumption levels, while *assuming perfect assignment of PBs to all existing product systems **and** recurring of functional units in the same amount every year*. However, regarding the first assumption, Bjørn, Chandrakumar, et al. (2020) correctly argue that "multiple sharing principles are technically applicable" and that "there is generally a lack of consensus on the most appropriate principle to use" for assigning carrying capacity to, for example, product systems. Moreover, assigning PBs to all existing products will inevitably lead to double counting as some products include other products, like an LCA of a battery can be included in an LCA of a torchlight, and LCAs of car fuels (diesel versus gasoline) can also be part of an LCA of a truck (Finnveden et al., 2022). In addition, new products come to the market and outdated products disappear every day. Regarding the recurring functional unit assumption, Ryberg, Owsianiak, Richardson, et al. (2018) themselves argue that in practice demands continuously change and products are exchanged by other new products, and consequently assuming "continuous demand" is a rather gross and unrealistic assumption.

In conclusion, our answer to our research question "*is LCA-based AESA reconcilable with the temporal constraints of relative LCA"* for now is: *no, even LCA-based AESA is relative*. This should not withhold scholars from developing approaches applying the PBs in LCA, but it should prevent them from claiming and using the term "absolute" for such studies. Our take-home message is simply that there is currently no such thing as *absolute* LCA. We therefore argue that the use of this term should preferably be avoided, to minimize the risk of miscommunication.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

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How to cite this article: Guinée JB, de Koning A, Heijungs R. Life cycle assessment-based Absolute Environmental Sustainability Assessment is also relative. *J Ind Ecol*. 2022;*26*:673–682. https://doi.org/10.1111/jiec.13260