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ASEA Method Framework

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A framework for development and communication of absolute environmental sustainability assessment methods

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Summary
An absolute environmental sustainability assessment (AESA) addresses whether a production or consumption activity can be considered environmentally sustainable in an absolute sense. This involves a comparison of its environmental pressure to its allocated share of environmental carrying capacity. AESA methods have been developed in multiple academic fields, each using their own set of concepts and terms with little communication across the fields. A recent growing interest in using AESA methods for decision support calls for a better common understanding of the constituents of an AESA method and how it can be communicated to scientific peers and to potential users.

With this aim, we develop a framework for AESA methods, composed of a succession of four assessment steps and involving six methodological choices that must be made by the method developer or the user. We then use the framework to analyze and compare five selected AESA methods that focus on the release of phosphorus and nitrogen to the environment. In this manner, we show that the framework is able to systematically differentiate AESA methods that initially appear to be similar. Intended users of the framework include 1) method developers communicating new AESA methods to academic peers or potential method users, and 2) researchers comparing a group of existing AESA methods and communicating their differences to other researchers and to potential users looking for guidance on method selection.

<heading level 1> Introduction
An absolute environmental sustainability assessment (AESA) can be used to study production or consumption activities of different types of entities, such as nations, companies or individuals, and is designed to answer the question “is the environmental pressure of this activity sufficiently low for it to be considered environmentally sustainable, and if not, how much lower should the pressure be?” The answer is based on comparing an activity’s estimated environmental pressure to the environment’s carrying capacity, which may be understood as an environment’s maximum persistently supportable anthropogenic pressure (Fang et al. 2015; Rees 1996). An activity that does not exceed its allocated share of carrying capacity can be considered environmentally sustainable.

When used in sustainability assessments, an environment’s carrying capacity serves to guide the protection of the natural capital that is judged to be “critical” for human well-being, meaning that it cannot be substituted by man-made capital (Daly 1995; Ekins et al. 2003). This criticality assumption is a defining characteristic of the “strong” sustainability school and stands in contrast to the fundamental assumption of the “weak” sustainability school that natural and man-made capital are substitutable in their generation of the material foundation for human well-being (Neumayer 2013). Their affiliation to the strong sustainability school makes AESA methods different from the large number of sustainability assessment methods in which the performance of an assessed activity depends on that of a reference activity and in which different types of environmental impacts are seen as substitutable (Bjørn et al. 2015; Moldan et al. 2012). For example, life cycle assessment (LCA) commonly compares two or more functionally equivalent products or services with the aim of identifying the one with the best overall performance, based on an aggregation of indicator scores across space, time and environmental issues (ISO 2006a, 2006b). Although AESA methods are unique in their affiliation with the strong sustainability school, they do resemble methods using policy targets or standards as performance reference. These include some distance-to-target methods in LCA (e.g., Wenzel et al. (1997)) and risk assessment methods used to analyze, for example, whether a site or scenario fulfills the policy goal of “good ecological quality”, as laid out in the Water Framework Directive of the European Commission (EC 2011). Policy targets or standards may be inspired by carrying capacities but are often less ambitious, as exemplified by the discrepancy between international policy targets for climate change and the stricter targets proposed by climate scientists (Rockström et al. 2009; Hansen et al. 2008; Lenton et al. 2008).

While AESA methods are still few within the total pool of sustainability assessment methods, their numbers have recently increased, coinciding with a growing interest in their use exhibited by a variety of decision-makers. More than 400, primarily large, companies have committed to defining “science based targets” (SBT) for greenhouse gas (GHG) emissions, i.e. targets that align with a global emission reduction pathway designed to fulfill the 2-degree climate goal (SBT 2018a; Krabbe et al. 2015). The World Wildlife Fund (WWF) is encouraging the scientific and business communities to develop methodologies that enable the definition of corporate level SBT for other environmental issues than climate change (OPT 2018; Muñoz and Gladek 2017). The Global Reporting Initiative recommends that companies report pollution loads “in relation to the capacity of the regional ecosystem to absorb the pollutant” (GRI 2016a). The operationalization of planetary boundaries in corporate decision-making has been explored (Clift et al. 2017; Whiteman et al. 2013). In addition, national governments have shown interest in applying the planetary boundaries framework to governance (Nykvist et al. 2013; Cole et al. 2014; Dao et al. 2018).

The growing interest in AESA methods calls for a solid, common understanding of what an AESA method is composed of and how it can be communicated to scientific peers and potential users. As noted by Fang et al. (2015), this understanding is currently limited, owing to the fact that AESA methods have been developed in multiple scientific fields, each using its own concepts and terms when communicating AESA methods. There is, therefore, a need for a common framework to improve communication of AESA methods, thereby fostering an increased understanding of methods amongst users and stimulating methodological improvements. Furthermore, to accommodate criticism of the strong sustainability school (Mark Sagoff 1995), the planetary boundaries concept (Leach 2014; Nordhaus et al. 2012; Rayner 2013) and AESA methods for, e.g., “presenting human values as facts of nature” (Weidema and Brandão 2015), normative choices in AESA methods should receive special attention. The purpose of this paper is to develop and present a framework for AESA methods and explain how it can guide communication within the scientific community and with external stakeholders. To demonstrate its applications, the framework is used to analyze and compare five recently developed AESA methods designed to study the anthropogenic release of phosphorus and nitrogen to the environment.

<heading level 1> Development of the framework

The AESA method framework comprises 1) a number of assessment steps that must be followed in carrying out an AESA and 2) a number of choices that must be made before the user can advance through the assessment steps. The normative aspects among these receive special attention.

The framework was developed in an iterative process involving 1) a review of steps and methodological choices in sustainability assessment methods in general and 2) a screening of a selection of AESA methods. The first step ensured the inclusion of steps and choices that are involved in all sustainability assessment methods, while the second step ensured capture of those steps and choices that are unique to AESA methods. Due to the multifaceted nature of sustainability and the vast and increasing number of sustainability assessment methods, there exists no universal and definitive list of their steps and choices. To the best of our knowledge, the list of Zijp et al. (2015), based on a review of 27 studies focusing on method selection, is the most recent and extensive analysis of the components of sustainability assessment method. We, therefore, reviewed this list in the first step of the framework development. In the second step, we screened a selection of AESA methods that we judge to be influential in academia and decision support and that, furthermore, originate in a wide range of academic fields:

- Context Based Sustainability (McElroy and van Engelen 2012).
- Ecological footprint (Borucke et al. 2013).
- Human Appropriation of Net Primary Production (HANPP) (Vitousek et al. 1986; Haberl et al. 2004).
- Methods originating in LCA, e.g. Bjørn and Hauschild (2015), Doka 2016 and Ryberg et al. (2018).
- Planetary boundaries (Rockström et al. 2009; Steffen et al. 2015).
- Science based targets (SBT 2018b).
- Water footprint (Hoekstra et al. 2011).

In addition to the screening of these AESA methods, we took into account a recent review of "One Planet Approaches" (OPAs) (another term for AESA methods) which was commissioned by WWF and the Swiss Federal Office for the Environment (Muñoz and Gladek 2017). The OPAs review focused on the AESA methods applicable to companies and was based on interviews with 19 researchers from many of the scientific fields in which the seven listed AESA methods originate. We consider the terminology of the OPAs review neutral (not biased towards any specific field), and therefore largely adopted it. Table 1 can be used as a "dictionary" to translate the key AESA terms used in this paper to terms commonly used in specific scientific fields.

Table 1: Key AESA terms used in this paper (first column) and their common synonyms in other scientific fields and literature sources. In cases where there is no synonym, the reason for this is explained.

This study	Context Based Sustainability (McElroy and van Engelen 2012)	Ecological footprint (Borucke et al. 2013)	HANPP (Vitousek et al. 1986; Haberl et al. 2004)	LCA (e.g., ISO (2006a, 2006b), Bjørn and Hauschild (2015), Doka 2016 and Ryberg et al. (2018))	One Planet Approaches review (Muñoz and Gladek 2017)	Planetary boundaries (Rockström et al. 2009; Steffen et al. 2015)	Science based targets (SBT 2018b)	Water footprint (Hoekstra et al. 2011)
Absolute environmental sustainability assessment (AESA)	Context-based sustainability assessment	Ecological footprint assessment or overshoot assessment	HANPP assessment	Absolute environmental sustainability assessment (AESA)	One Planet Approach	Planetary boundary assessment	No synonym, since the focus is on calculating allocated carrying capacity	Water Footprint Sustainability Assessment
Environmental sustainability objective	No synonym, since there is no explicit sustainability objective	No synonym, since the protection of ecological capital is an inherent objective	No synonym, since the protection of biodiversity is an inherent objective	Area of Protection	Sustainability objective	No synonym, since the protection of the Holocene state is an inherent objective	No synonym, since the meeting of the 2-degree climate goal is an inherent objective	No synonym, since the sustaining of freshwater and estuarine ecosystems is an inherent objective
Carrying capacity	Carrying capacity	Biocapacity	Net primary production (NPP)	Carrying capacity	Safe operating space	Safe operating space	Carbon budget	Water availability
Environmental pressure	Impact	Ecological footprint	HANPP	Impact potential	(Environmental) impact	(Change to) control variable value	GHG emissions	Water footprint
Allocation principle	Allocation principle	No synonym, since the equal per capita principle is default	No synonym, since all anthropogenic activities affecting a specific land are studied in aggregation	Assignment/allocation/sharing/ entitlement principle	Allocation principle	No synonym, since all anthropogenic activities globally are studied	Allocation principle	Allocation principle

<heading level 1> Presentation of framework

Figure 1 shows the resulting AESA method framework.

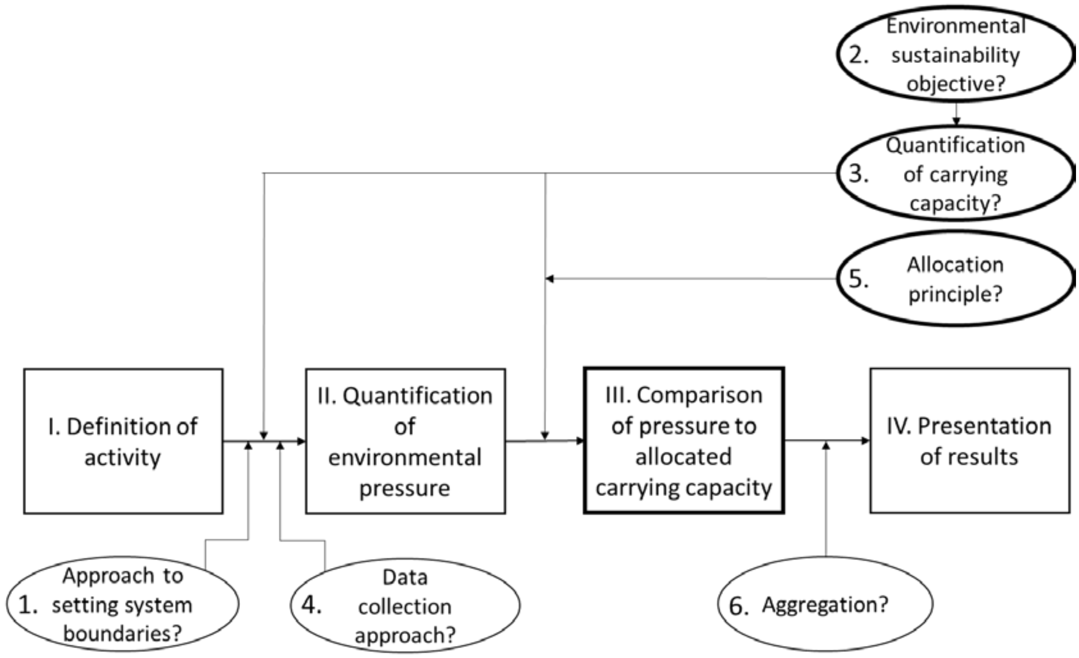

Figure 1: AESA method framework comprising four assessment steps (boxes) and six methodological choices (ellipses). Thick borders indicate that an element is unique to AESA methods, as opposed to taking part in all quantitative sustainability assessment methods.

The framework is comprised of four assessment steps and six methodological choices. Three of the four assessment steps are found in all sustainability assessment methods, while step III, the comparison of pressures to allocated carrying capacity, is unique to AESA methods and, therefore, highlighted with a thick border in Figure 1. Likewise, three of the six methodological choices are found in all sustainability assessment methods, while Choices 2, 3 and 5 are unique to AESA methods.

Below, we elaborate on each of the six choices, after which we propose how a method developer may deal with the normative aspects of the choices in method design and communication.

<heading level 2> Choice 1: Territorial or consumption-based approach to setting system boundaries?
Assessment step 1 defines the activity to be assessed, for example, the consumption of a city or the production of a company. In order to progress to assessment step 2, a choice must be made between two approaches for setting the system boundaries around the defined activity. In the *territorial approach*, system boundaries follow the territorial extension of an activity. In the examples above, this corresponds to the boundaries of the city and of the plot of land owned or leased by the company. By contrast, in the *consumption-based approach*, system boundaries are set so they enclose all the production processes that are needed by the assessed activity, regardless of where these processes occur (Muñoz and Gladek 2017). For a company, these production processes constitute its value chain, while they extend over, what is often called, a city's "hinterland" (Lenzen and Peters 2010).

The choice between the two principles for setting system boundaries boils down to which production activities an entity can be considered responsible for. In a complex socio-economic system, however, actions can often not be attributed fully to a single entity. Accordingly, the responsibility for an action may be argued to belong to the entity directly causing an action (such as a Chinese steel company), i.e., in accordance with a territorial approach. Alternatively, it can be argued to belong to the entity indirectly causing the action by purchasing a good or service (such as a steel consumer in Europe), i.e., in accordance with the consumption-based approach. There is no consensus on the most appropriate approach amongst institutions that are globally influential in the management of environmental issues. For example, the UNFCCC treaty on climate change has predominately been taking a territorial approach when accounting for GHG emissions of nations and setting national emission targets (UN 1992). In contrast, within the sphere of corporate social responsibility, a consumption-based approach is often taken, as is evidenced by the existence of several guidelines on the reporting of supply chain issues and management from the Global Reporting Initiative (e.g. GRI (2016b)) and United Nations Global Compact (e.g. UNGC (2015)). The perception of responsibility can also change with time. Shortly after the 2013 collapse of a textile factory in Bangladesh that claimed the lives of around 1,100 workers, a number of big fashion brands sourcing textiles from that region committed to financing a scheme for improving safety at factories, even though they had previously been reluctant to take such responsibilities (Ek and Kane 2013).

<heading level 2> Choice 2: Environmental sustainability objective?
An environmental sustainability objective defines that which must be protected to achieve environmental sustainability. As explained in the introduction, all AESA methods are associated with the strong sustainability school, meaning that a fundamental assumption is that human well-being depends on some minimum level of environmental protection to safeguard critical natural capital. Environmental science has a key role in creating knowledge on the complex interactions between elements within ecosystems and their responses to anthropogenic pressure. However, even in the hypothetical situation where knowledge of an ecosystem is perfect, environmental scientists cannot objectively decide what is to be considered critical natural capital by an AESA method. That decision depends on which functions, or services, of nature are seen as critical to humans (Cornell 2012) and this is, to some extent, a normative question (Ekins et al. 2003). In the ecosystem services framework (Millennium Ecosystem Assessment 2005), a large number of services are mapped and categorized as provisioning, regulating, cultural and supporting services. Many AESA methods, such as the ecological footprint and HANPP (Haberl et al. 2004; Borucke et al. 2013), are based on the environmental sustainability objective of protecting provisioning functions or services. By comparison, AESA methods related to the planetary boundaries concept focus on protecting the regulating and supporting functions or services supplied by the "Earth System" (Rockström et al. 2009; Steffen et al. 2015).

The ecosystem services concept sees nature protection as a means to ensure human well-being and not as a goal in itself. Still, it has been argued that there is room for eco-centrism in sustainability assessments, which means valuing nature for its intrinsic properties (Muñoz and Gladek 2017). This is because the recognition of nature's intrinsic value is known to be part of human moral psychology through humans' ability to empathize with other biological organisms and through the sacredness humans may attribute to natural sites on a spiritual level (Rottman et al. 2015). Acting in a way that does not respect nature's intrinsic value could thus conflict with our moral psychology and thereby reduce human well-being in a non-material sense. An eco-centric environmental sustainability objective translates to more ambitious sustainability objectives than an anthropocentric objective, for example, the protection of all species in an ecosystem versus the protection of species that are materially important for human well-being only.

An environmental sustainability objective does not have to cover all types of environmental issues. For example, several environmental issues that are commonly covered in LCA are not covered by the planetary boundaries framework (Steffen et al. 2015), as they are not relevant for protecting the functional integrity of the Earth System (Ryberg et al. 2018b; Chandrakumar and McLaren 2018). Examples are, non-renewable resource scarcity and direct impacts on human health. If an AESA method does not cover all environmental issues that are perceived as important by the method users, there is a risk of "burden shifting" to uncovered issues and this risk should clearly be communicated to users (see *Different applications of AESA framework*, below). An alternative approach is to base an AESA method on a set of existing environmental issues. For example, Bjørn and Hauschild (2015) identified carrying capacity estimates in the environmental science literature that can be compared to environmental pressures expressed with indicators commonly used in LCA. The benefit of this approach is that the resulting AESA methods cover all environmental issues typically covered in LCA, which limits the risk of burden shifting (see *Choice 2: Environmental sustainability objective*). The disadvantage is that the chosen carrying capacities are not necessarily related to a single, coherent, environmental sustainability objective.

<heading level 2> Choice 3: Quantification of carrying capacity?

Having defined an environmental sustainability objective, a choice must be made on how to translate this into one or more quantified environmental carrying capacities. In the context of AESA methods, carrying capacity may be defined as "the maximum persistent anthropogenic pressure that the environment can tolerate without suffering impairment of the functional integrity of its ecosystems". This definition is closely related to the definition offered by Fang et al. (2015), who drew on Rees (1996) and (Catton 1986).

<heading level 3> Impact pathway

In order to quantify carrying capacity, one or more indicators must first be identified. This is done by mapping the causal relationship between threats to an environmental sustainability objective and anthropogenic pressures (resource use and emissions) in an *impact pathway*. Figure 2 shows an illustrative impact pathway beginning with the environmental sustainability objective "prevent dangerous anthropogenic interference with the climate system", as defined by UNFCCC (UN 1992)), and ending with the identification of a list of pressures that threatens this objective.

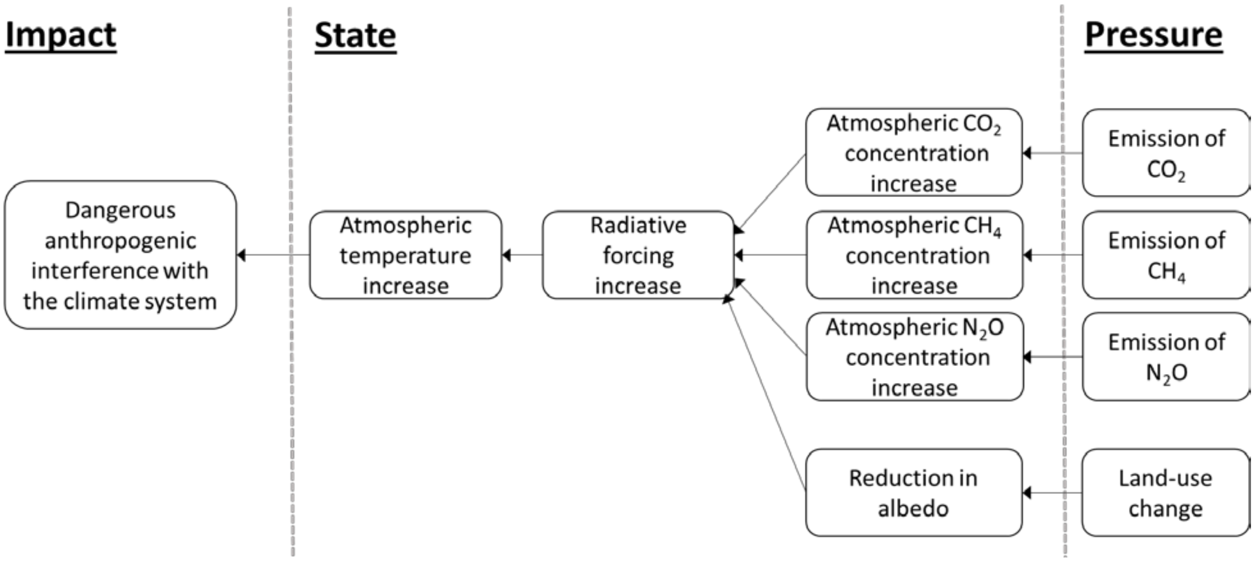

Figure 2: Illustrative impact pathway for an environmental sustainability objective related to climate change.

For many environmental sustainability objectives such a mapping leads to the identification of more than one pressure. In the case of Figure 2, for example, multiple emissions as well as land-use change were identified as pressures, in which case a carrying capacity must be calculated for a chosen reference pressure, such as emissions of CO_2 in the example of Figure 2.

<heading level 3> Quantifications

The calculation of a carrying capacity value typically involves a normative interpretation of ambiguous terms used in the environmental sustainability objective, such as “dangerous anthropogenic interference” in the objective of UNFCCC (Hansen et al. 2008; Mann 2009; Petschel-Held et al. 1999; van Zalinge and Dijkstra 2017; UN 1992). It also typically involves considerations of thresholds or “tipping points” in the response of an impact indicator to increasing pressure, as simulated with *system modelling* (Dearing et al. 2014; Lenton 2013). In some cases, the critical parameter value derived from an environmental sustainability objective is expressed in a state indicator and not a pressure indicator. For example, the planetary boundary for climate change is expressed in radiative forcing increase (and in atmoshperic CO_2 concentration) (Steffen et al. 2015), rather than in emission of CO_2 (see Figure 2). In such cases additional environmental modelling is needed to calculate a carrying capacity value. Alternatively, the environmental indicators to be calculated in assessment step II may be expressed at the state level to enable a direct comparison to boundaries, in which case carrying capacities need not be calculated (option not shown in Figure 1). In both approaches, life cycle impact assessment methods may be used to translate values between different indicators in an impact pathway (Hauschild and Huijbregts 2015).

When system modelling is not possible or practical, *expert judgement* may also be used to quantify carrying capacity (Muñoz and Gladek 2017). In the case where threats to fulfilling an environmental sustainability objective increase (roughly) linearly with anthropogenic pressure, carrying capacities should be based on *societal acceptability* (Dearing et al. 2014), i.e. reflect the answer to the normative question “how degraded do we accept this ecosystem to be, considering the value of products and services, whose production contributes to its degradation?”.

<heading level 3> Uncertainties

The quantification of carrying capacity usually involves uncertainty due to imperfect knowledge of the quantitative relationships between indicators in an impact pathway. For example, the quantitative relationship between radiative forcing increase and atmospheric temperature increase in the impact pathway for climate change (see Figure 2), determined by the so-called climate sensitivity parameter, is relatively uncertain, despite decades of intense climate change research (IPCC 2013). In the face of such uncertainty, a normative choice lies in which parameter value to use, ranging from conservative to optimistic. The planetary boundaries framework is based on consistently making conservative choices in response to uncertainty (Steffen et al. 2015). In comparison, the carrying capacity values derived in Bjørn and Hauschild (2015) are based on average or median parameter values as is the common approach in life cycle impact assessment (Hauschild and Huijbregts, 2015).

<heading level 3> Consequence for quantification of environmental pressure

The identification of carrying capacity defines the environmental pressures that require quantification in assessment step II (see example in Figure 2). The impact assessment method used to calculate a carrying capacity value can also be used to translate environmental pressures to a single reference pressure. In the example of Figure 2, different climate forcers (right hand) can be expressed in CO_2 equivalents, based on their radiative forcing relative to CO_2, following life cycle impact assessment methods (Cherubini et al. 2016). The identification of carrying capacity also decides the ideal temporal and spatial resolution for the quantification of pressures. For example, the AESA for terrestrial acidification and eutrophication of Bjørn et al. (2016) was based on the quantification of carrying capacity in 99,515 spatial units, and this informs the ideal spatial resolution of environmental pressures, in the form of SO_X, NO_X and NH_X emissions to air, to be quantified.

<heading level 2> Choice 4: Data collection approach?

In an assessment with a territorial scope (see Choice 1) data can be straightforwardly collected when the boundaries of a studied activity (e.g. total production at an industrial site) matches the boundaries of environmental data reported by the entity directly responsible for those activities (e.g. through corporate responsibility reporting).

Data collection in consumption-based assessments (see Choice 1) is more complicated, since it involves compiling data from multiple sources. For this, one of two approaches can be adopted. *The process approach* involves the mapping of production processes within the system boundary by tracking the physical flows linking them. The mapping is followed by a quantification of environmental pressures for each process. In practice, the mapping of production processes is typically based on a combination of first-hand data, often provided by the entity responsible of the studied activities and generic data from a database of production processes, such as ecoinvent (Association Ecoinvent 2018). By comparison, the *environmentally extended input-output (EEIO) approach* uses a combination of average environmental data of industrial sectors from specific nations and data from national accounts on the trade between nation-specific industrial sectors (Wiedmann 2009). In this way, the sectors involved in the provisioning of a product or service are linked by economic flows.

The EEIO approach can lead to a poorer representation of products and services from economic sectors of a heterogeneous nature (such as “chemicals not elsewhere classified” in the Exiobase EEIO database (Stadler et al. 2018)) than the process approach. Also, the EEIO approach currently covers a much shorter list of environmental pressures than the process approach, which can be a serious limitation in an AESA, depending on the environmental pressures that need quantification, as per Choice 3. On the other hand, the process approach inevitably leads to an incomplete coverage of production processes. This is because of the practice of “cutting off” from the system boundaries the parts of the supply chains of products and services, which are judged to, individually, have negligible effects on the total environmental pressure and for which process data are difficult to obtain (Bjørn et al. 2018). By contrast, the EEIO approach, in principle, leads to a 100% coverage of processes.

When choosing between a process and EEIO approach, requirements to spatial and temporal resolutions of environmental pressures, as per Choice 3, should be considered. The EEIO approach is typically spatially resolved at a national level (a few large nations may be resolved at the provincial or state level) and temporally resolved at a yearly level (Stadler et al. 2018). By comparison, the process approach typically leads to quantified environmental pressures of varying spatial and temporal resolutions, from very high for processes covered by first-hand data (e.g., geolocated) to generic (e.g., a global average) for some of the processes modelled using a database, such as ecoinvent (Association Ecoinvent 2018).

Regardless of the approach taken, multifunctional processes pose a modelling challenge. For example, a milk production system has the additional function of producing, amongst other things, meat and cowhide. The total environmental pressures of the milk production system must therefore somehow be allocated to the production of milk, meat, hide and other co-products, and this is essentially a normative exercise (Dalgaard et al. 2014). AESA is consistent with “attributional” modelling approaches in LCA and multifunctional processes can therefore be handled by, what Majeau-Bettez et al. (2018) refer to as, partition allocation and, with some restrictions, alternate-activity allocation.

Common for both approaches is also uncertainties, due to imperfect knowledge of the production processes associated with the assessed activity and the environmental pressures occurring per unit of production from each process. Again, a normative choice lies in which parameter value, ranging from conservative to optimistic, to use in an AESA.

<heading level 2> Choice 5: Principle for allocating carrying capacity to activity?

Comparing the environmental pressure of an assessed activity directly to a carrying capacity is only meaningful if there are no other activities occupying a part of the carrying capacity. This is generally not the case and, therefore, a part of the quantified carrying capacity must be allocated to the assessed activity.

Different allocation principles for distributing scarce resources exist (Muñoz and Gladek 2017; Ryberg et al. 2018a): The *contribution to value-added principle* holds that the allocated carrying capacity is proportional to the economic value produced by an activity (Randers 2012). According to the *grandfathering principle*, an activity’s allocated carrying capacity is “inherited” from its share of total environmental pressure in a past reference year (Knight 2014). The *equal per capita principle* allocates carrying capacity equally among all individuals in a territory (Starkey 2008). More principles exist and each has conceptual and practical advantages and disadvantages. For example, the principle of equal per capita allocation is consistent with the ideal of intra- and intergenerational equity embedded in the sustainable development concept (WCED 1987; Holden et al. 2014), but it is difficult to apply it for production activities, when it is unclear how these contribute to the total consumption of individuals. This is especially the case for the production of goods to be sold for further industrial processing. The contribution to value-added and grandfathering principles, on the other hand, are convenient to apply, as they rely on economic data or environmental pressures in the past that are generally easy to obtain. Nevertheless, the principles are somewhat at odds with equity ideals because they may lead to current inequalities being passed on into the future.

<heading level 2> Choice 6: Aggregation?

Aggregation is a common technique for facilitating the interpretation of assessment results and can be useful in environmental assessments where several activities are compared with the goal of identifying the one with the best overall performance, across space, time and types of carrying capacities. For example, ecological footprint assessments generally involve aggregation across space and types of footprints (cropland, grazing land, “carbon uptake land”, etc.) (Borucke et al. 2013). One aggregation technique is to count the number of spatial and temporal units in which environmental pressure exceed a carrying capacity. Another technique is to add the exceedance of carrying capacities across space and time, which resembles the approach taken in “accumulated exceedance”-based life cycle impact assessment indicators, e.g. Seppälä et al. (2006). Inevitably, any aggregation of pressures compared to allocated carrying capacity leads to a loss of information. It is especially problematic if the aggregation techniques allows hiding exceedance of allocated carrying capacity in one place by “unused” carrying capacity elsewhere (Bjørn and Røpke 2018). Whether the benefits of aggregation outweighs this disadvantage depends on the assessment context.

<heading level 2> Normative aspects in AESA

Table 2 summarizes the normative aspects involved in the six methodological choices described above.

Table 2: Normative aspects in the six choices of the AESA framework. The normative aspect of Choice 2 is underlined to indicate that it is of a non-modular nature. Choices that can be covered in a perspective archetype approach are in italic (see explanations in the text below the table).

Choice	Normative aspect
1: Approach to setting system boundaries?	• What activities are an entity responsible for?
2: Environmental Sustainability objective?	• *What should be protected to enable human well-being?*
3: Quantification of carrying capacity?	• *How strictly should the environmental sustainability objective be interpreted?* • *What parameter value should be used when knowledge of a natural system is incomplete?* In the case of (close to) linear cause/effect relationship: • *What level of ecosystem degradation is acceptable?*
4: Data collection approach?	• *What parameter value should be used when knowledge of an anthropogenic system is incomplete?* • How to deal with multi-functional processes in the attribution of environmental pressure to an activity?
5: Allocation principle?	• How to allocate carrying capacity among activities?
6: Aggregation?	In comparative assessments of multiple activities: • How to aggregate information about environmental pressures on carrying capacity across space, time and, potentially, different types of carrying capacities?

The six choices should ideally reflect the values of an assessment’s stakeholders. For example, in the assessment of the activities of a company, stakeholders typically comprise customers, suppliers, civil society (e.g. environmental non-governmental organizations), public opinion, investors, potentially affected parties (e.g. people living close to the company’ industrial sites) and policy-makers (Penna and Geels 2012). As stakeholders vary between assessments, a method developer is rarely capable of making choices that are consistent with all use scenarios of an AESA method. Therefore, some choices are often left to the user of an AESA. Such choices must be of a modular nature, meaning that a change in one choice does not affect the solution space of other choices. All choices in the AESA framework are modular, except Choice 2, since the solution spaces of Choice 3 and, to some extent, 6 depend on it. The method developer must, therefore, perform Choice 2 and communicate clearly how this was done to method users, who can then take this into account in the process of selecting a method amongst alternatives (see also *Different applications of AESA framework*, below).

The remaining five choices (1 and 3-6) can be left to AESA method users. To facilitate this, options for handling choices may be classified according to “perspective archetypes”, which is a common approach to handling value judgement in life cycle impact assessment (Hofstetter 1998; Huijbregts et al. 2017). Perspective archetypes are located on a spectrum from optimism (with respect to technological innovation and ecosystem resilience) to conservatism (being highly precautionary). In the field of LCA, three archetypes labelled “individualist”, “hierarchist” and “egalitarian” have commonly been used (Hofstetter 1998; Goedkoop et al. 2009), but this is a convention and different numbers of archetypes and alternative labels are possible. The benefit of a perspective archetype approach is that it allows method users to make several choices, some of which may be of a highly technical nature, simultaneously, simply by deciding which perspective archetype best represents the values of an assessment’s stakeholders. However, only some of the normative aspects under Choice 3 and 4 fit neatly on an optimism/conservatism scale (Italicized in Table 2). The remaining aspects must, therefore, be left explicitly for the user to handle.

<heading level 1> Applications of the framework

We demonstrate the framework’s applications by using it to analyze and compare five selected AESA methods.

<heading level 2> Analysis and comparison of five AESA methods

For the analysis, we selected five AESA methods that all focus on the management of the release of reactive phosphorus and nitrogen to the environment: 1) The original planetary boundary method for phosphorus and nitrogen (Rockström et al. 2009). 2) The new planetary boundary method for nitrogen (de Vries et al. 2013). 3) The new planetary boundary method for phosphorus (Carpenter and Bennett, 2011). 4) The grey water footprint method for nitrogen and phosphorus (Liu et al. 2012). 5) The grey water footprint for nitrogen (Mekonnen and Hoekstra, 2015).

Three of the AESA methods have been developed to support planetary boundaries science (de Vries et al. 2013; Carpenter and Bennett 2011; Rockström et al. 2009). While the method of de Vries et al. (2013) covers four different types of carrying capacities associated with reactive nitrogen, only the two carrying capacities related to terrestrial and aquatic eutrophication and acidification are considered here, to be consistent with the focus of the other four AESA methods. The two AESA methods related to phosphorus and nitrogen flows that were developed in the second key publication on planetary boundaries of Steffen et al. (2015) are not considered here, because they are adaptations with minor refinement of the AESA methods of Carpenter and Bennett (2011) and de Vries et al. (2013), already included here. The remaining two AESA methods (Liu et al. 2012; Mekonnen and Hoekstra 2015) have been developed in the water footprinting community as grey water footprint methods.

The analysis of the five AESA methods is based on their application to the recent global release of reactive phosphorus and/or nitrogen, according to the methods’ documentation. This common activity of analysis means that any difference between assessment results logically must reflect different handling of the six choices (see Figure 1) by the method developers. Note that the five AESA methods can also be used to assess various activities at a sub-global level. For example, the planetary boundaries related methods have been used to support decision-making at a national scale (Dao et al. 2018; Nykvist et al. 2013; Cole et al. 2014) and at the scale of industrial sectors (Sandin et al. 2015; Roos et al. 2016). Likewise, the grey water footprinting methods have been widely used for decision support on different scales (Hoekstra et al. 2011). Table 3 shows the analysis of the five AESA methods according to the six methodological choices. In cases where method developers handled normative aspects in alternative ways, these are in bold

, in the last row, summarizes their assessment results.

Table 3: Analysis of five selected AESA methods according to the six methodological choices of the AESA method framework. The bottom row shows AESA result, expressed as a default value, if reported, and an uncertainty interval. In cases where method developers handled normative aspects in alternative ways, these are in bold.

Methodological choices	Original planetary boundaries for biogeochemical flows (Rockström et al. 2009)		New planetary boundary for nitrogen (de Vries et al. 2013)		New planetary boundary for phosphorus (Carpenter and Bennett, 2011)	Grey water footprint for nitrogen and phosphorus (Liu et al. 2012)	New grey water footprint for nitrogen (Mekonnen and Hoekstra, 2015)
1. Approach to setting system boundaries?	Territorial/ consumption-based.		Territorial/ consumption-based.		Territorial/ consumption-based.	Territorial/ consumption-based.	Territorial/ consumption-based.
2. Environmental sustainability objective?	"Avoid unacceptable global environmental change" → Avoid pushing "the planet out of the desired Holocene state"		None explicitly stated. "the environmental footprint of agricultural nitrogen use on water quality, biodiversity and climate has to be reduced".		Avoid "freshwater eutrophication".	None explicitly stated.	"Protection of aquatic life".
	→ Avoid critical reduction in "overall resilience of ecosystems via acidification of terrestrial ecosystems and eutrophication of coastal and freshwater systems".	→ "Avoid a major oceanic anoxic event".	→ "Prevent aquatic ecosystems from developing eutrophication or acidification".	→ Avoid "adverse biodiversity impacts" on terrestrial ecosystems.			
3. Quantification of carrying capacity?	Global N_2 fixation, compared to current fixation (reference state) based on tentative expert judgement. **Conservative and optimistic** estimate given.	Global phosphorus flow to ocean compared to pristine weathering (reference state), based on system modelling using geological model of coupled phosphorus and oxygen cycles (Handoh and Lenton 2003). **Conservative and optimistic** estimate given.	**Nitrogen load to runoff water from agricultural fields or N_2 fixation** at a 0.5^0 by 0.5^0 spatial resolution. Calculated using the IMAGE model (Bouwman et al., 2006) and a **conservative and optimistic critical nitrogen concentration in runoff water** identified in a literature review. No reference state explicitly defined.	**NH_3 emissions to air or N_2 fixation** at a 0.5^0 by 0.5^0 spatial resolution. Calculated using the TM5 model and **a conservative and optimistic critical atmospheric NH_3 concentration** identified for lichens and higher plants, respectively, in a literature review. No reference state explicitly defined.	**Global flow of phosphorus from terrestrial ecosystems to freshwater or flow of phosphorus as fertilizer to "erodible" soil**. Calculated using a simple, dedicated linear mass balance model, based on a pre-industrial reference, and **two alternative values of critical phosphorus concentration**, reflecting, respectively, a common policy target for lakes and reservoirs and the approximate	Virtual freshwater flows, at a 0.5^0 by 0.5^0 spatial resolution, containing **eight different phosphorus and nitrogen compounds** in concentrations not exceeding **default, conservative or optimistic critical values**, identified through a review of concentration targets in policy documents. The carrying capacity value is the actual water discharge at a basin level. The	Virtual freshwater flow, at a 0.5^0 by 0.5^0 spatial resolution, containing nitrogen at a critical concentration, proposed by the Canadian Council of Ministers of the Environment. The carrying capacity value is the actual water discharge at a basin level. The quantity of pollution representing one unit of virtual water flow is a function of the critical

					pre-industrial phosphorus concentration in the world's rivers.	quantity of pollution representing one unit of virtual water flow is a function of the critical minus **default, conservative or optimistic estimates of "natural" concentrations**.	minus the pristine concentration.
4. Data collection approach?	Literature review.	Literature review.	Use of the IMAGE model, which estimates agricultural emissions of nitrogen to runoff water and NH_3 to air as a function of agricultural output (Bouwman et al., 2006).		Use of simple, dedicated linear mass balance model relying on literature estimates of parameters in the global phosphorus cycle, involving **three different estimates of current phosphorus flow to the sea.**	Use of the NEWS model, which estimates river exports of different forms of nitrogen and phosphorus as a function of anthropogenic activities (Mayorga et al., 2010).	Use of a detailed, dedicated mass balance, based on estimated inputs of nitrogen to the economy and fate modelling of different routes of nitrogen loads on aquatic ecosystems.
5. Allocation principle?	Not needed, since all global activities are studied		Not needed, since all global activities that are significant for nitrogen loads to runoff water and NH_3 to air are studied		Not needed, since all global activities are studied	Not needed, since all global activities are studied	Not needed, since all global activities are studied
6. Aggregation?	No aggregation performed.		Ratio of environmental pressure to carrying capacity is averaged across spatial units, while adjusting ratios below 1 to a value of 1.		No aggregation performed.	The share of spatial units with pressure exceeding carrying capacity is calculated for each compound and for all compounds, collectively.	The share of global river discharge with pressure exceeding carrying capacity is calculated.
AESA result	Pressure share of carrying capacity is 400% (286-400%).	Pressure share of carrying capacity is 94% (9.4-94%).	Pressure share of carrying capacity is 149-198%.	Pressure share of carrying capacity is 105-136%.	Pressure share of carrying capacity is 115-594%.	Carrying capacity exceeded in 24% (1.7-89%) of spatial units.	Carrying capacity exceeded in 9% (6-14%*) of spatial units.

* The uncertainty interval around the share of spatial units having their carrying capacity exceeded was not reported by Mekonnen and Hoekstra (2015). It was approximated here from the reported uncertainty range of −33% to +60% for the global estimated environmental pressure (grey water footprint) (Mekonnen and Hoekstra 2015).

The last row of Table 3 shows that the AESA results, all based on the study of global anthropogenic activities around the year 2000, differ substantially, both in format and in numerical values. Also, the alternative ways of handling normative aspects (see bold text in Table 3) result in quite wide uncertainty intervals, especially in the cases of Liu et al. (2012) and Carpenter and Bennett (2011). The reason for differences in AESA results, within and between methods, can be found in different handling of Choices 2, 3, 4 and 6 by the method developers. In contrast, Choices 1 and 5 are handled the same way in all five AESA methods and, therefore, cannot explain differences in assessment results. Moreover, given that all anthropogenic activities globally are analyzed in the methods' application studied here, these two choices are actually not relevant. This is because, for Choice 1, taking a territorial approach leads to the exact same system boundary as taking a consumption-based approach and because, for Choice 5, allocation of carrying capacity to the assessed activities is, logically, 100%.

The environmental sustainability objective (Choice 2) of Rockström et al. (2009) is the broadest, initially formulated as "Avoid unacceptable global environmental change", meaning protecting the Holocene state of the Earth system. From this, two specific objectives were developed that refer to environmental consequences from releasing nitrogen and phosphorus, respectively (see Table 3). de Vries et al. (2013) challenged the sustainability objective of Rockström et al. (2009) by arguing that "nitrogen availability in the Holocene was too limited to feed the current world population" but did not explicitly state a sustainability objective on their own. Carpenter and Bennett (2011) complemented the ocean-focused sustainability objective of Rockström et al. (2009) by a sustainability objective focusing on eutrophication of freshwater from the release of reactive phosphorus. Liu et al. (2012) did not explicitly state an objective, while Mekonnen and Hoekstra (2015) focused broadly on "protection of aquatic life".

The chosen carrying capacity indicators (Choice 3) vary from the input of nitrogen as a resource (N_2 fixed from the atmosphere) to emissions of nitrogen and phosphorus to different environmental compartments (soil, freshwater, ocean and air). The two grey footprint indicators (Liu et al. 2012; Mekonnen and Hoekstra 2015) express carrying capacity as freshwater flows that are used, in a virtual sense, as transporters of released pollutants in critical concentrations. Some AESA methods propose more than one carrying capacity indicator, varying according to their location in an impact pathway (de Vries et al. 2013; Carpenter and Bennett 2011) or chemical form of nitrogen and phosphorus (Liu et al. 2012). While three AESA methods propose spatially resolved carrying capacities, the methods of Rockström et al. (2009) and Carpenter and Bennett (2011) propose global carrying capacities. While Rockström et al. (2009) used tentative expert judgement to quantify the nitrogen-related carrying capacity, all other methods relied on system- or environmental models for the its quantification. The inputs to these models were, in most cases, a critical aquatic concentration of nitrogen or phosphorus, which had been obtained from a review of literature covering environmental science or policy targets (as mentioned in the *Introduction*, the latter is often inspired by the former).

The approach to data collection (Choice 4) was for the method of Rockström et al. (2009) a literature review of environmental pressures. The other AESA methods estimated pressures for different economic sectors (agriculture, wastewater treatment, etc.) by use of existing environmental models, such as IMAGE (Bouwman et al. 2006), or by use of models developed specifically for the AESA methods.

All the three AESA methods that involve spatially resolved carrying capacities presented aggregations of pressures compared to carrying capacity (assessment step III), in addition to disaggregated results (Choice 6). The aggregations across spatial units were either done by calculating the share of spatial units in which environmental pressure exceed carrying capacity (Liu et al. 2012; Mekonnen and Hoekstra 2015) or by averaging ratios of environmental pressure to carrying capacity, while adjusting ratios below 1 to a value of 1 (de Vries et al. 2013).

<heading level 2> Different applications of AESA framework

In the previous section, it was demonstrated that the AESA framework can be used to systematically dissect AESA methods and differentiate them based on the six methodological choices made by their developers. In this section we propose four non-exhaustive applications for such an analysis, see Figure 3.

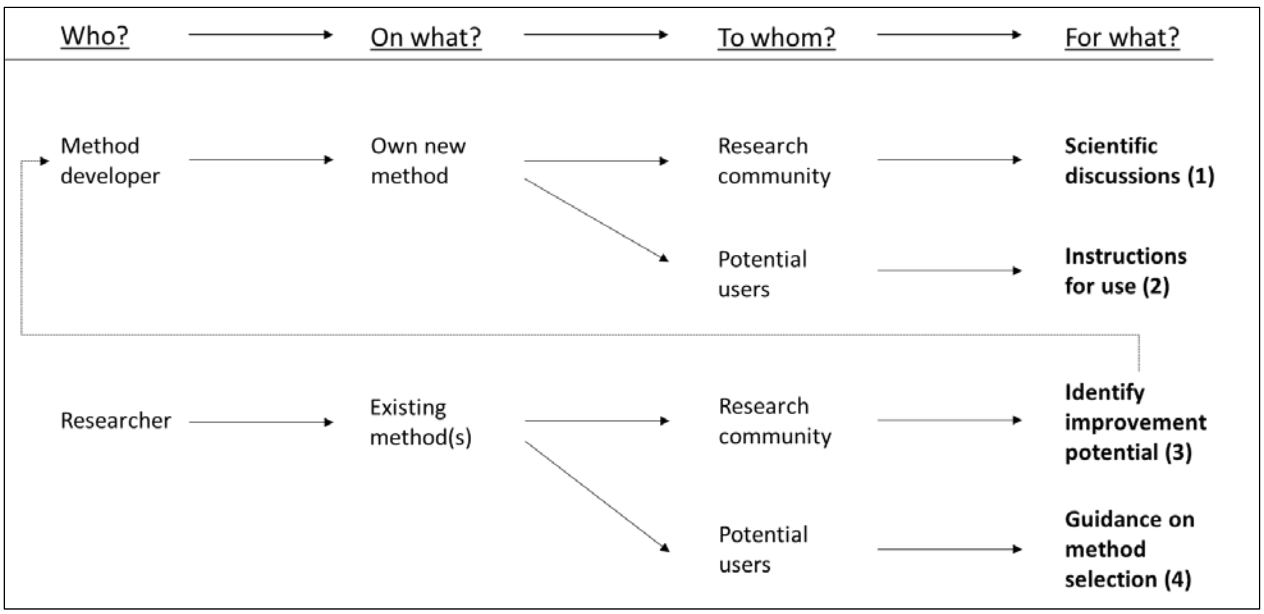

Figure 3: Decision tree demonstrating four different applications of the AESA framework.

The first application of the AESA framework is to contribute to qualifying the scientific discussion. This happens when a method developer uses the framework to communicate on a new AESA method to the research community. Concretely, the documentation of an AESA method, e.g. in the form of a scientific paper, could be structured according to the six choices and four assessment steps of the framework (see Figure 1 and Table 3). It is important to clearly communicate all normative aspects, either handled by the method developer or left for users to handle (see Table 2). This reduces the risk that scientific peers overlook that there is actually normativity involved, which could lead to (unintendedly) “presenting human values as facts of nature” (Weidema and Brandão 2015). For example, corporate sustainability reports were found to rarely justify the choice of a given carrying capacity allocation principle (Bjørn et al. 2017). This obscures the fact that corporate environmental targets calculated using AESA methods are not entirely “science-based”, albeit more so than existing corporate target setting practices. Among the five cases of AESA methods analyzed here, we observed that the handling of normative aspects related to environmental sustainability objective (Choice 2) was not always clearly communicated. Another use of the framework, when communicating on a new AESA method to scientific peers is for the method developer to structure a systematic sensitivity analysis. This can identify the methodological choices to which the results of the AESA method are most sensitive, i.e., those choices that can be seen as "low hanging fruits" in future efforts aimed at reducing uncertainties of results. Among the five case AESA methods, Carpenter and Bennett (2011) present the most comprehensive sensitivity analysis. They report environmental pressure and carrying capacity values for all combinations of the three sets of normative options left for users to choose from (see Table 3) and also report carrying capacity values for all combinations of perturbations (+/- 5%) of six parameter values. From this analysis, it can be seen that reducing the uncertainty of the critical aquatic concentration of phosphorus (currently 24-160mg/m^3) should be a focus for reducing uncertainties of assessment results.

The framework's second application is for the developer of a new AESA method to develop instructions to potential users of that method. Instructions should be given as to how to carry out each of the four steps. This involves presenting to the user the different options for handling normative aspects. When doing so, method developers could consider using a perspective archetype approach to group those options for handling normative aspects that can be located on an optimism/conservatism scale. In the case of Liu et al. (2012), for example, the options for choosing critical concentrations and natural concentrations of nitrogen and phosphorus compounds can be classified to a number of perspective archetypes. By comparison, the normative aspects handled entirely by the method developer should be explicitly explained. Specifically regarding Choice 2 on environmental sustainability objective, which is always made by the method developer, users must be explained the potential risk of burden shifting, which occurs if the chosen sustainability objective precludes some environmental issues that are perceived important to users. In such cases, users should be advised to complement the AESA by a conventional LCA.

The third application of the framework is for a researcher to identify improvement potentials of one or more existing AESA methods and to communicate these to scientific peers. This may happen by using the framework to compare a group of AESA methods, as was done in Table 3, in order to structure the identification of "state of the art" for the handling of choices of a technical (i.e. non-normative) nature. This identification may then inspire the design of new AESA methods or the improvement of existing methods. Technical choices relate to impact assessment modelling, including the choice of an indicator for carrying capacity and environmental pressure (Choice 3), and approaches to quantifying environmental pressures (Choice 4) (see *Presentation of framework*). It falls outside the scope of this paper to identify best practice for all such choices amongst the AESA methods of Table 3. Yet, it can be noted that the quantification of emissions (Choice 4) in the two grey water footprint methods (Mekonnen and Hoekstra 2015; Liu et al. 2012) was based on well-established environmental models covering many emission pathways and, therefore, is likely to be superior to the more crude emission estimations of Rockström et al. (2009) and Carpenter and Bennett (2011). This observation is consistent with the recommendation of Fang et al. (2015) that planetary boundaries researchers should use approaches developed in the LCA and footprinting communities when estimating environmental pressures.

The AESA framework's fourth application proposed here is to provide guidance to potential users on how to select a method amongst a pool of candidates. Such guidance should be based on an analysis of a pool of method candidates with respect to the six methodological choices, much like the analysis of Table 3, but, potentially, using layman's terms when appropriate for the recipient. As in Table 3, the normative aspects that method users can manage, through their choices between a number of predefined options, should be highlighted. Such a “dissection” of AESA methods can allow users to make an informed choice regarding the method that best aligns with their needs, including the values of the stakeholders of a planned assessment.

In all of the four applications presented above, users of the framework may adopt the neutral terminology used throughout this paper. However, an alternative terminology may be more suitable when the aim is to address scientific peers within a specific field (see Table 1).

<heading level 1> Outlook

Given the increasing interest from decision-makers in AESA, it is important that the method developers from different scientific fields can use a common language when communicating internally as well as to potential users of their methods. The framework developed in this paper contributes in this direction and can thereby help to increase the transparency, credibility and robustness of AESA methods.

Our framework focuses on environmental sustainability and. A social aspect of sustainability is only present in the question of how to allocate carrying capacity to different activities (Choice 5), including personal consumption. The “isolation” of environmental sustainability from the sustainability concept is a helpful form of reductionism for many analytical purposes. However, it is also artificial and there is a risk of burden shifting if decisions made in striving for environmental sustainability make the goal of social sustainability (regardless of how that is understood) harder to achieve. Therefore, a promising avenue of further exploration is the deeper integration of social and environmental concerns in absolute sustainability assessment. This could happen by linking in the assessment of an activity its social impacts to what may be termed social sustainability objectives. For example, a product life cycle’s impact on the income of various stakeholders (e.g. employees at a manufacturing process) may be linked to the first sustainable development goal (SDG) of “no poverty” (UN 2015), as also argued by Schaubroeck and Rugani (2017) and Weidema (2017). Conceptual frameworks for combining absolute social sustainability assessment (whether drawing on SDGs or other sets of social sustainability objectives) and AESA have already been sketched out by Griggs et al. (2013), and discussed in greater detail by McElroy and van Engelen (2012) and Raworth (2017). The primary challenge at hand is then to develop assessment methods that link social impacts (whether negative or positive) of an assessed activity to social sustainability objectives. Such social sustainability assessment methods should involve setting the same system boundaries around an activity as in an AESA, to allow integration of the two types of assessments. The developments in social life cycle assessment (Benoît-Norris et al. 2011; Russo Garrido et al. 2018; Zanchi et al. 2018), and its integration with (environmental) LCA in life cycle sustainability assessment (Sala et al. 2012; Kloepffer 2008) may serve as inspiration for such an integrative approach to absolute sustainability assessment.

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