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Framework to define environmental sustainability boundaries and a review of current approaches

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Abstract

Environmental sustainability boundaries can help us navigate a sustainable development trajectory, by evaluating environmental performance of current actions in relation to such boundaries. However, current definitions of environmental sustainability boundaries have shortcomings when used in environmental assessments. The shortcomings include considerations of regional differentiation and transparency with respect to how uncertainty is addressed. This paper seeks to improve the definition and application of environmental sustainability boundaries in environmental assessments by reviewing existing approaches to set them and elaborating an analytical framework for defining, communicating and adopting environmental sustainability boundaries in assessments. 110 original environmental sustainability boundaries were identified from existing literature and grouped into 13 categories of boundary approaches. The framework addresses five components (objective, boundary principle, uncertainty principle, accepted levels of impacts, and scientific estimate), and recommends practices for each. The framework is recommended for defining, communicating and adopting environmental sustainability boundaries, to facilitate a consistent application of them in environmental assessments. The analysis of existing boundary approaches showed that they often handle value-based aspects (e.g. choice of uncertainty principle) differently. Thus, it is recommended that value-based aspects are communicated explicitly to enable a practitioner to consider how this resonates with his/her own values or the values of central stakeholders when adopting environmental sustainability boundaries in an environmental assessment.

1. Introduction

Despite a focus on sustainable development, society is far from solving the global environmental sustainability challenge, where climate change and biodiversity loss are of particular concern (UN 2019). This indicates that our current focus and efforts are insufficient and that we are on the wrong trajectory for sustainable development. We believe that operational methods to evaluate environmental achievements of current efforts in relation to absolute environmental sustainability boundaries are needed to help navigate a sustainable development trajectory. Thus, to achieve sustainable development, there is an urgent need for integrating environmental sustainability boundaries into environmental assessments (Muñoz and Gladek 2017).

The notion of environmental sustainability boundaries is not new. Global resource constraints were addressed as early as the late 18th century (Malthus 1798). Later, environmental degradation was related to boundaries in contributions such as The Limits to Growth (Meadows et al 1972) and critical loads in pollution regulation (UNECE 1979). Recently, several approaches to define environmental sustainability boundaries have been developed with the planetary boundaries (PBs) (Rockström et al 2009, Steffen et al 2015) as the most broadly studied.
and utilized approach (Muñoz and Gladek 2017). However, a comprehensive overview of existing sustainability boundary approaches is still lacking. While it is noted that some recent reviews address related aspects (Muñoz and Gladek 2017, Downing et al 2019; Bjørn et al 2020), the focus in those reviews was on applying environmental sustainability boundaries. Thus, those authors did not examine in depth the definitions of the boundaries themselves. Moreover, Bjørn et al 2020 concentrated only on methods related to life cycle assessment (LCA) and Downing et al (2019) focused only on studies related to the PBs. The present study directs specifically on boundary definitions and takes a comprehensive view on a variety of boundary approaches.

Environmental sustainability boundaries cannot be defined solely on the basis of biophysical or technical criteria, as their definition implies value-based decisions (Haines-Young et al 2006, Biermann 2012, Hoekstra and Wiedmann 2014, Van Vuuren et al 2016, Bjørn et al 2018). Acceptance of risk (Muñoz and Gladek 2017) and the definition and interpretation of the objective (i.e. what must be protected to achieve environmental sustainability) (Bjørn et al 2018), are some critical aspects that need to be considered when defining such boundaries. Nevertheless, boundaries are often communicated as objective and as scientific ‘facts’, underplaying the value-based aspects of their definition (Sayre 2008, Biermann 2012).

An increasing number of methods and studies relating environmental impacts to boundaries have emerged in the last years. For example, studies assessing environmental impacts or footprints in relation to boundaries at country or continental scales (e.g. Nykvist et al 2013, Dao et al 2018, Li et al 2019, EEA 2020) or at company or industry level (e.g. Wolff et al 2017, Chandrakumar et al 2019, Ryberg et al 2018). For a further overview of studies applying environmental sustainability boundaries refer to Downing et al (2019) and Bjørn et al 2020. Despite this increasing employment of environmental sustainability boundaries in assessments, criteria for defining them (i.e. which elements should be considered and communicated) and for their adoption (i.e. assessing and selecting the boundaries suitable for a specific study) have not yet been defined. Adopting boundaries without a thorough prior assessment may pose the risk that the value-based judgement underlying the boundary approach does not resonate with the values of the potential users of the approach. Moreover, when combining boundaries from different approaches in an environmental assessment and comparing across categories of environmental impact, it is essential that the boundaries are based on similar values, e.g. implying similar levels of precaution. At the very least, one should know if there is inconsistency in the accepted level of precaution implied by the different boundaries. For example, if unconsiously applying a boundary for acidification based on a high level of precaution and one for eutrophication with a low level of precaution may falsely indicate that acidification is a more pressing environmental problem than eutrophication. This may lead to misinformed decisions prioritizing the protection of one type of environmental impact over another.

The purpose of this study is to improve and facilitate definition and adoption of environmental sustainability boundaries for use in environmental assessments. This is achieved. (i) conducting a review of existing approaches for setting environmental sustainability boundaries, (ii) develop a framework for consistently defining environmental sustainability boundaries for use in environmental assessments, and (iii) analyse existing boundary setting approaches in relation to the framework and provide recommend practices for setting environmental sustainability boundaries.

2. Methods

2.1. Identification and categorization of environmental sustainability boundary approaches

The review focuses on approaches to define environmental sustainability boundaries, i.e. the way in which they are defined including, e.g. the coverage of environmental impacts, the origin of the boundary (scientific, political or expert judgment) and the objective of defining the boundary. The review does not aim to provide an exhaustive list of existing environmental sustainability boundaries as such but includes, based on a broad literature search, a comprehensive selection of boundaries from different fields that represent different approaches to define environmental sustainability boundaries (i.e. boundary approaches).

Terms such as environmental ‘limits’, ‘thresholds’ and ‘boundaries’ are not used consistently across different fields. In some areas the terms denote different sets of ideas, while in others they are used as synonyms (Haines-Young et al 2006). Table 1 describes how different terms related to environmental sustainability boundaries are understood and used in this work. Note that in this work we understand a boundary as a measure of something we do not want to surpass, i.e. as a guardrail, whereas we understand a target as a measure of something we aim to reach. For example, to limit the global temperature increase to 2 °C is a boundary, where a target, associated with this boundary, could be to reduce CO2 emissions with a specific amount by a specific year.

The review was scoped to include studies, environmental assessments, and policy and management frameworks, operating with boundaries for environmental pressures, impacts and constraints associated with human activities. The review was not limited
to a specific type of environmental pressures, and approaches for setting environmental sustainability boundaries at local, regional and global level were considered. However, only boundary approaches with a global coverage were included in the review. Hence, studies estimating boundaries for one specific location that is not scalable, transferable or relevant for other locations, were excluded (e.g. case study boundaries from the Regime Shift Database of Biggs et al (2018)). Boundaries related to human health aspects were not considered within the scope of the review (e.g. pollution limits in the World Health Organization’s air quality guidelines), as we focus on environmental boundary approaches. Approaches that are solely theoretical in proposing metrics for defining boundaries but not derive a boundary value (i.e. offering no practical and quantified example of the use of the boundary definition) were excluded (e.g. Sala and Goralczyk 2013, Mace et al 2014). Moreover, boundaries that are not original, i.e. are results of downscaling, copies or modifications of boundaries from other studies and approaches were not considered within the scope of this review (e.g. Häyhä et al 2016, Dao et al 2018, Li et al 2019, EEA 2020). While some of these studies propose minor modifications of boundaries (e.g. Dao et al (2018) translating the original PBs control variable for climate change from state to pressure point of the impact pathway) other studies provide more substantial modifications. For example, EEA (2020) focus on phosphorus releases from agriculture and wastewater in their modification of the phosphorus PB, resulting in a boundary 10 times lower than the original phosphorus PB.

With the goal of being as inclusive as possible and covering relevant results from the various disciplines that operate with the notion of environmental sustainability boundaries, boundary approaches were identified in three steps:

- Step 1: Examination of recent literature reviews concerned with environmental assessments involving environmental sustainability boundaries (Muñoz and Gladek 2017, Downing et al 2019,
Step 2: Screening of additional scientific literature using the literature search engine Scopus, applying strings of relevant key words such as ‘environmental boundary’, ‘sustainability’, ‘carrying capacity’ and ‘threshold’. See supplementary information (SI) table S1 (available online at https://stacks.iop.org/ERL/15/103003/mmedia) for applied search queries.

Step 3: Screening environmental strategies, regulations and agreements. This includes (1) environmental strategies and associated regulations represented by EU Environmental Action Programme (EEA 2018), (2) international environmental agreements (IEAs) represented by the United Nations Economic Commission for Europe (UNECE) conventions and associated protocols, (3) IEAs relevant for the PBs addressed in Nykvist et al (2013), and (4) the sustainable development goals (SDGs) related to the environment (12, 13, 14 and 15).

The identified boundaries were screened and analyzed according to the description of each boundary, environmental impacts addressed, the origin (scientific, political or expert judgment), the boundary principle applied (i.e. the basic idea that serves as the foundation for quantifying the boundary) and according to the environmental assessment method or framework to which they pertain and associated objective (where relevant). Based on this analysis, the boundaries were classified and bundled into boundary approaches (figure 1), assigning most weight to the last classification criteria (the method or framework of which the boundaries pertain and are applied and its objective).

The classified boundary approaches were then assessed according to criteria defined in an iterative approach outlined in section 2.2.

2.2. Development of the framework and assessment of boundary approaches

The framework was elaborated in an iterative process through five steps as illustrated in figure 2. The iterative process involved (1) identifying initial assessment criteria from Bjørn et al (2018), Muñoz and Gladek (2017) and Ryberg et al (2016), (2) screening a selection of the identified and classified boundary approaches according to the initial criteria, (3) refining the assessment criteria and revisiting step 2 (assessing approaches). After some rounds of iteration, the assessment criteria were arranged into the format of a framework (4) and based on the assessment of boundary approaches, recommendations for best practices were formulated (5).

3. Results

3.1. Environmental sustainability boundary approaches

3.1.1. Approach categories and applications

In the literature search we identified 110 original environmental sustainability boundaries applied in environmental assessments or policy and classified them into 13 boundary approach categories (table 2).

See table S2 (SI) for an overview of all identified boundaries including a short description, their application, coverage of environmental impact category, origin and assigned boundary approach category. Sixteen boundaries were considered not to relate to a specific boundary approach (listed as ‘not classified’ in table S2) and were, therefore, excluded from the further analysis of boundary approaches (section 3.3). Note that we did not find any new boundary approaches in the screening of the SDGs that were not already found from other literature.

As outlined in section 2.1, where applicable, boundaries were assigned into boundary approaches according to the method or framework in which they are applied and their associated objectives. However, exceptions were done where boundaries of methods applying similar principles for defining boundaries were merged into one boundary approach. This was the case of the biocapacity approach applied in the Ecological footprints method (Borucke et al 2013) and sustainable process index (SPI) (Krotscheck and Narodoslawsky 1996). Note that this classification resulted in approaches with different level of specificity. For example, the environmental flow requirements (EFR) approach covers one environmental impact category and has a higher level of specificity than e.g. the PBs approach covering several individual boundaries representing different environmental impact categories. Moreover, some approaches overlap. Specifically, the basin level freshwater use boundary and the biogeochemical flow boundary for phosphorous in the PB-framework are based on concepts of EFR and critical load approaches, respectively.

Four of the classified boundary approaches are applied in environmental footprint methods, which were originally designed to quantify the pressure on the environment of human activities (Laurent and Owsianiak 2017). The ecological, blue water, green water, chemical and gray water footprints address boundaries (Fang et al 2015), classified respectively as ‘biocapacity’, ‘EFR’, ‘green water availability’ and ‘gray water and chemical footprint boundaries’ in table 2. Note that carbon footprint has also been coupled with an boundary based on the 2 °C boundary (e.g. Fang et al 2015).

3.1.2. Environmental impacts

Figure 3 illustrates the coverage of environmental impact categories in the identified environmental...
Table 2. Identified and classified boundary approaches. See table S3 for a more in-depth description and an overview of their applications and coverage of environmental impacts. Where possible, the environmental impacts addressed by the boundary approach was aligned with the categories defined in EC-JRC (2011).

| Boundary approach | Environmental impacts | Description and applications |
|-------------------|------------------------|-----------------------------|
| Biocapacity       | Land use               | ‘Biocapacity’ is defined as the sum of biologically productive land of a given region. It is calculated based on area and land use specific yield factors and is provided for different land use types. The biocapacity boundary is applied in the Ecological footprints method (Borucke et al 2013) and sustainable process index (SPI) (Krotscheck and Narodoslawsky 1996). |
| Chemical critical concentration | Ecotoxicity | The chemical critical concentration approach covers boundaries expressed as critical concentrations of chemicals with toxic effects. The general principles in setting it are (1) collect data from toxicity tests, and (2) extrapolate the toxicity data based on deterministic (application of assessment factors) or probabilistic methods (species sensitivity distribution) (EC 2018). It is applied in policy (e.g. environmental risk assessments (ERA) and environmental quality standards (EQS)). |
| Critical loads and ecological critical concentrations | Eutrophication, acidification and ground-level ozone | A critical load is based on a quality criterion or an ecological critical concentration of a chemical set to protect a specified biological indicator for a chosen receptor ecosystem (e.g. for eutrophication as ‘a slight deviation from a reference condition with no, or only very minor, anthropogenic disturbance’ (EC 2019)). Critical loads are calculated by process-oriented models (dynamic simulation models or steady state models) and empirical models (Laane 2005). The approach is applied in policy (e.g. critical loads in the UNECE’s Convention of long-range transboundary air pollution (LRTAP)) and ecological critical concentrations in the EU Water Framework Directive (WFD) and scientific literature (e.g. Kerkhof et al 2015, Bjørn et al 2016). |
| Environmental flow requirement (EFR) | Resource depletion (water) | EFR considers the water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods depending on these ecosystems. Several EFR sub-approaches exist including site-specific (e.g. ecological limits of hydrologic alteration (ELOHA)) and more general approaches of setting standards across broad areas (e.g. minimum flow thresholds and ‘per cent-of-flow’ (POF) (Richter et al 2012)). EFR is applied in e.g. the blue water footprint (Hoekstra et al 2012) and in the boundary of water resource depletion of PBs. |
| Environmental space | Resource depletion, ecotoxicity and land use | The environmental space approach aims to calculate the amount of raw materials each person can consume in a sustainable, equitable future. This covers both the provision of materials and the capacity to absorb and degrade wastes (Spangenberg 1995). |
| Green water availability | Resource depletion (water) | Green water availability is the boundary applied in the green water footprint, which considers green water consumed for the production of resources for food, feed, fiber, timber and bioenergy. Green water is defined as the rainfall that infiltrates the soil or is contained by vegetation and which eventually flows back to the atmosphere as evapotranspiration (Schyns et al 2019). |
| Boundary approach | Environmental impacts | Description and applications |
|-------------------|-----------------------|------------------------------|
| Net Primary Productivity (NPP) | Other | NPP is the measure of solar energy, water and atmospheric CO2 captured by plants and photosynthetic organisms and converted to plant matter. As an environmental sustainability boundary, the NPP in a geographical area can be compared to the NPP lost and human appropriation of NPP (HANPP) in terms of food, fiber, and fuel (Haberl et al 2007). |
| Planetary Boundaries (PBs) | Ocean acidification, biodiversity loss, climate change, eutrophication, land use, ozone depletion, aerosol loading, resource depletion (water) | The PBs approach proposes boundaries within nine Earth system processes considered crucial for the functioning of the Earth system and to keep the planet in a Holocene-like state. The quantification of PBs combines scientific understanding of the Earth system and the precautionary principle and represents a 'safe operating space' for global human development (Steffen et al 2015). |
| Sustainable solution space | Climate change, biodiversity loss, acidification | The sustainable solution space is a tool to define a 'sustainability range' with maximum and minimum boundaries for environmental pressures per functional unit (as defined in an LCA). The sustainability range is defined based on (1) literature values and stakeholder interviews and (2) its relationship to other indicators (both economic and environmental) (Binder et al 2012). Details on how the ranges are defined are not provided. |
| System dynamics | Eutrophication, land use and biodiversity loss | The system dynamics boundary approach covers boundaries defined according to regime shifts, or thresholds, from a system dynamics perspective. A system dynamics perspective involves feedback loops and regime shifts, which occur when changes in the dominant feedbacks and drivers cause the system to reorganize into a different structure (Biggs et al 2018). A system dynamic boundary is often defined based on evidence in form of models, observations and experiments. |
| Tolerable soil erosion (TSE) | Land use | Tolerable soil erosion (TSE) is defined as ‘any actual soil erosion rate at which a deterioration or loss of one or more soil function does not occur’ (Verheijen et al 2009). |
| Gray water and chemical footprint boundaries | Ecotoxicity and eutrophication | Several environmental assessment methods such as the chemical footprint (Bjørn et al 2014, Zijp et al 2014, Tarasova and Makanova 2016) and gray water footprint (Liu et al 2012; Mekonnen and Hoekstra 2015) include boundaries based on similar approaches applying the assimilation capacity principle, considering the water volume needed to dilute pollutants to a level below a specified condition (e.g. a chemical critical concentration). |
| 2 °C boundary | Climate change | Several of the boundaries for climate change suggested and applied in literature and policies are expressed as a maximum atmospheric global average temperature increase compared to pre-industrial levels such as the IPCC’s 2 °C boundary. It is based on scientific projections of climate impacts to be expected at different temperature increases, value judgments on the (non-) acceptability of such impacts and political discussions on what is realistic (Gao et al 2017). |
sustainability boundary approaches. Note that we aimed to align the environmental impacts addressed by the boundary approach with the categories defined in EC-JRC (2011), although some approaches apply a different term for the environmental impact addressed and may have a slightly different or additional focus. For example, the nitrogen boundary of PBs is proposed not only to avoid eutrophication but also taking other potential environmental impacts into consideration such as radiative forcing by N$_2$O and atmospheric NH$_3$ concentrations. Moreover, such environmental impacts occurring at a regional/local scale (see section 3.3) are included in the PBs as they affect the capacity of the Earth system to stay in a Holocene-like state under changing conditions and can generate feedbacks to the processes that have large-scale thresholds. Hence, the focus here is on the large-scale consequences of these regional impacts (Steffen et al. 2015).

Some impact categories have a high coverage such as abiotic resource depletion represented by 17 boundaries (figure 3). This is mainly due to the environmental space approach (Spangenberg 1995, MST 1998) providing a boundary value for different types of resources (e.g. copper, limestone and oil) and even sets of boundary values for each resource (MST 1998). Land use, aquatic eutrophication, ecotoxicity and acidification are also strongly represented with respectively 17, 12, 11 and 8 defined boundaries for these impact categories across different approaches. Note that the acidification boundary proposed by PB concerns ocean acidification while the others concern terrestrial and freshwater acidification. The category ‘other’ includes a boundary related to nuclear energy from the environmental space approach and NPP where several aspects of environmental impacts are integrated such as land-use change, freshwater use, biodiversity loss, and global nitrogen and phosphorus cycles (Running 2008).

3.1.3. Origin
It is difficult to establish a clear division between political and scientific or expert opinion-based boundaries. Indeed, all boundary approaches set boundaries based on a mix of these considerations (with the exception of some individual boundaries that were not classified and some of the PBs (e.g. boundary for biodiversity loss), which are based on expert opinion only). For example, critical loads and gray water and chemical footprint boundary approaches are based on environmental models for their quantification. However, the input to the models are critical environmental concentrations (i.e. limits), which have been extracted from policy. These policy limits might again be based on scientific measurements and acceptable risk estimates. Similarly, the EFR defines the percentage change of natural flows based on reviews of case studies, where some of the case studies define the percentage change according

![Figure 1. Overview of the identification, classification and assessment of boundary setting approaches.](image1)

![Figure 2. Iterative process steps (1–5), for defining assessment criteria, framework components and recommendations for best practices.](image2)
Figure 3. Environmental impact categories covered in existing environmental sustainability boundary approaches indicating the number of boundaries identified for each environmental impact category. The environmental impact categories applied are in accordance with EC-JRC (2011) unless marked with *.

scientific peer-reviews of site specific studies, stakeholder involvement and expert opinions. Refer to table S3 in SI for a full overview of the origin of the assessed boundary setting approaches. In general, an environmental sustainability boundary cannot be based on fully objective scientific models, as value-based decisions will always be involved in its definition, for example, the objective of the approach and how to handle uncertainty and risk (further discussed in section 3.2).

3.2. The framework
The framework for defining, communicating and adopting environmental sustainability boundaries is presented in figure 4. It consists of five components; (I) objective (II) boundary principle, (III) impact and accepted levels (IV) scientific estimate and (V) uncertainty principle. The definition of environmental sustainability boundaries cannot be based solely on biophysical or technical criteria but requires several value-based decisions (Haines-Young et al 2006, Biermann 2012, Hoekstra and Wiedmann 2014, Van Vuuren et al 2016, Bjørn et al 2018). In fact, component I, II, III and V include value-based aspects.

Component I concerns the definition of the overall objective of the boundary approach. This involves considerations regarding what should be protected to achieve the related environmental goal, and this depends on which functions or services of nature are perceived as being critical (Bjørn et al 2018). For example, an overall objective could be to protect provisioning services of nature (i.e. material or energy outputs from ecosystems), or to protect the regulating and supporting functions of ecosystems (e.g. the ecosystem’s ability to regulate the quality of air and soil and maintain a diversity of plants and animals).

Inspired by the work by Holling (1985), boundary principles (component II) can be associated with the perception of the stability of nature and the Earth system and whether a conservative, moderate or optimistic basis for setting the boundary is adopted. For example, perceiving nature as unstable, responding with feedback amplification to even small disturbances, can channel a more protective or precautionary approach. This perception can be associated with boundary principles of zero tolerance and pre-industrial conditions where little change in the current state of the environment is allowed (i.e. conservative). The threshold boundary principle can be associated with a more moderate view of environmental protection, i.e. perceiving the Earth as a resilient system, capable of absorbing change in a way that its functions, structure and feedbacks are retained. However, this resilient system has boundaries, and when these boundaries are surpassed (i.e. the exceeding a threshold), the original system state may be pushed to a new state (Holling 1973). Although with the perception of the world as a more stable system, regeneration rate can also be associated with a moderate basis for setting a boundary where resource consumption is allowed as long as it does not exceed the natural rate of regeneration. Similar, the boundary principle of envelope of variability (i.e. a boundary at the point where the system moves outside of the long-term normal envelope of variability (Dearling et al 2014)), can be associated with a moderate view. Finally, a more optimistic view on nature’s tolerance to environmental impact might be applied. Here, nature is governed by negative feedback
mechanisms, which can cope with human pressure independent of the magnitude of the pressure. Environmental sustainability boundaries are not relevant when an optimistic view on nature's tolerance is applied.

Component III (impacts and acceptable levels), is only relevant for boundaries defined according to a threshold or assimilation capacity boundary principle where pressure-response relationships are considered. The other boundary principles are not associated with a response variable and consequently not related to a specific level of impact or environmental change. For example, boundaries defined according to the principle of regeneration rate are based on the idea that the use of resources should not exceed the rate of their regeneration (i.e. not associated with a specific level of impact or risk). The location of the response variable on the impact pathway should be considered as a part of component III to account for its relevance in relation to the type of impact it represents. An impact pathway describes the cause-effect chains linking human interventions to impacts on the environment. The drivers, pressures, state, impact and response (DPSIR) framework (EEA 1999) can be used to explain the cause-effect chains in an impact pathway linking the environmental drivers (e.g. energy consumption) to pressures (e.g. emission of CO$_2$) resulting in changes of environmental states (e.g. CO$_2$ concentration in the atmosphere) to impacts (e.g. increase in frequency of extreme weather or loss of biodiversity) (Hauschild and Huijbregts 2015). A response variable located at the impact point of the impact pathway, is more relevant for the environmental impact than a response variable located earlier on the impact pathway. Selecting the acceptable level of environmental degradation (i.e. the value of the response variable) is clearly value-based, and decision-makers and societies may have different perceptions of the levels of environmental impacts that are acceptable with respect to what is considered as harmful and valuable and the costs associated with avoiding them etc. For example, in the developed world, the society might aim to preserve the world as it is with a more risk-averse approach to environmental degradation, where developing countries might accept a higher risk or accept more damage to the environment, prioritizing economic development (Biermann 2012).

The scientific estimate of an environmental sustainability boundary (component IV), involves considerations regarding the biophysical properties of a natural system, suitable metrics and locations of the control variable on the impact pathway, spatial resolution, temporal dynamics and interactions between environmental impact categories. A control variable located close to the cause in the impact pathway is easier to control, but might be less relevant than one located farther from the cause (Ryberg et al 2016, Laurent and Owsianiak 2017). For example, in the PBs approach, the control variable N$_2$ fixation is not unambiguously related to eutrophication effects. Eutrophication effects are strongly affected by nitrogen management (e.g. emission control and release). On the other hand, N$_2$ fixation is more measurable and controllable than emissions of nitrogen to water since global data on N$_2$ fixation is available and can be easily translated to policy and management interventions (Ryberg et al 2016). Environmental sustainability boundaries and their associated carrying capacities may be dynamic due to natural seasonal and diurnal fluctuations (here referred to as temporal dynamics) (Bjørn and Hauschild 2015). For example, a boundary for water availability would be lower in dry seasons. Furthermore, a major challenge in the scientific estimate of environmental sustainability boundaries is how to handle the dynamic interactions between environmental impacts and the social systems in which they are embedded (Dearing et al 2015, Van Vuuren et al 2016, Muñoz and Gladek 2017).
The ‘uncertainty principle’ (component V) is overarching the boundary setting process and relates to how the unknowns involved in the different components of the framework are handled, e.g. component III and IV can draw on experiments or modelling of environmental pressures and responding impacts, where uncertainty might be present in input parameters, model structure or temporal or spatial variations (Mutel et al 2019). Here, one should consider if applying the precautionary principle or aiming for a best estimate.

3.3. Application of the framework—analysis of existing boundary approaches

We now demonstrate the application of the framework by analyzing the 13 categorized boundary approaches. The analysis of the approaches served as a basis for the formulation of the recommended practices presented in figure 4 and section 3.4. The details of the analysis are presented in SI, tables S4 and S5.

3.3.1. Component I: objective

An over-arching objective was stated for most of the boundary approaches except for biocapacity, green water availability, NPP and sustainable solution space. For example, the PBs approach has the objective of retaining Holocene-like conditions in order to preserve the biophysical habitability of the planet for humanity, while the 2 °C boundary approach aims to ‘prevent dangerous anthropogenic interference with the climate system’ (UNFCCC 1992).

The overall motivation of the reviewed approaches is, in general, to define environmental sustainability boundaries to protect nature in order to ensure human well-being and not the protection of nature as a goal in itself. Hence, the reviewed approaches are, as a starting point, defined according to an anthropocentric perspective (i.e. protecting nature due to the material or physical benefits it can provide for humans (Thompson and Barton 1994). However, some approaches include eco-centric elements (i.e. nature is valued for its own sake and deserves protection because of its intrinsic value (Thompson and Barton 1994)). Approaches with eco-centric elements include the green water availability, NPP defined in (Haberl et al 2004, Running 2008) and EFR. For example, the overall idea of the green water availability boundary is to estimate green water available to sustain human activities. However, a part of the green water is reserved for biodiversity protection. In contrast, the TSE approach, for example, considers solely the purpose that soil functions have for human activities and welfare such as production (e.g. providing food). Thus, there may be differences in the motivation behind the approaches, leading to different levels of ambition for environmental conservation (Thompson and Barton 1994), that one should be aware of when adopting and combining different boundary approaches.

3.3.2. Component II: boundary principle

As defined in section 3.2, the boundary principle provides the foundation for defining the boundary and is associated with the underlying perception of nature and the Earth system. It was found that the majority of the boundary approaches can be associated with a moderate view on environmental protection as they define the boundary as a threshold (the critical load and concentration approach, chemical critical concentration, maximum temperature increase, PBs and TSE). Four boundary approaches can be associated with a conservative view of environmental protection, i.e. applying regeneration rate as the basis for establishing boundaries (biocapacity approach, green water availability, NPP and TSE). For example, the TSE of Verheijen et al (2009) defines the boundary for soil erosion according to the ‘natural’ or geological erosion rate. The environmental space approach by MST (1998) provides several values for each boundary, reflecting different boundary principles. For example, the boundary for copper consumption is defined according to geological regeneration (conservative) and threshold (moderate). This is transparently communicated and enables the user of the environmental space approach to select a boundary according to their own preferred perspective.

3.3.3. Component III: impact and accepted levels

Assessing impacts and acceptable levels is only relevant for six boundary approaches (critical loads, chemical critical concentration, PBs, TSE, gray water and chemical footprint boundaries and 2 °C boundary), defined according to a threshold or assimilation capacity boundary principle where pressure-response relationships are considered (see section 3.2).

The chemical critical concentration, TSE and 2 °C boundary approaches define the response variable at the impact point of the impact pathway where the response variables in the PBs, critical loads and gray water and chemical footprint boundaries are at the state or impact point of the impact pathway, depending on the environmental impact category and ecosystem in question. For example, in the PBs approach, a concentration of phosphorus in the water body (i.e. state variable) is applied as response variable for the phosphorus eutrophication boundary (Carpenter and Bennett 2011). The phosphorus concentration is often used as an index for eutrophication, as it is closely correlated with the phytoplankton biomass, which is the most obvious sign of impaired water quality. However, new studies have shown that the correlation is not straightforward linear, as several location specific factors influence the relationship between phosphorus concentration and biomass (e.g. temperature and variations in depth) (Carpenter and Bennett 2011). A response variable closer to the actual
effect (increase in phytoplankton biomass), considering location specific factors, should thus preferably be used.

Boundaries defined according to the notion of thresholds, generally accept some level of environmental impact, although this is usually not explicitly stated. For example, in the latest assessment report of IPCC (ARS), several impacts were associated with the 2 °C temperature increase boundary (e.g. species extinction and negative impact on average crop yields). Each impact was associated with a risk at a scale from very low to very high, also considering the level of adaptation. For example, risk of species extinction was associated with a medium-high level of risk (IPCC 2014). Note that the risks associated with different temperature increases have changed with every IPCC assessment report. For example, in 2001, the risk for exceeding tipping points was considered likely if global warming exceeded 5 °C above pre-industrial levels, and in the IPCC Report published in 2018, between 1 and 2 °C (Lenton et al 2019). Hence, the point where the threshold of abrupt irreversible changes in our climate system is expected to occur has evolved over time, in pace with increasing knowledge of the climate system.

The objective of the critical load approach is to assure that ‘significant harmful effects on specified sensitive elements of the environment do not occur, according to present knowledge’ (UNECE 1979). Hence, boundaries defined by the critical load approach ultimately depend on judgments of what are ‘significant harmful effects’, i.e. which ecosystem functions should be protected and at what level. Should critical loads, for example, be calculated to protect the forest productivity or biodiversity, and is there a maximum level of damage that can be accepted? (Laane 2005). There is a lack of transparency and consistency with respect to how critical loads are operationalized and which level of environmental impact is accepted. For example, ecological critical concentrations (i.e. the input to critical loads) are, in the EU policy framework Water Framework Directive (WFD), associated with ‘good ecological status’ and defined as a slight deviation from a reference condition with no, or only very minor, anthropogenic disturbance (EC 2019). However, the EU countries are responsible to specify this ecological critical concentration for different water bodies themselves. In some cases, these show more than a 10-fold difference in concentrations across European countries (Poikane et al 2019). This great variation illustrates that it is important to consider local differences when defining a boundary (i.e. differences in reference situations, in this case) and that qualitative described conditions (in this case, the ‘very minor’ deviation from the reference situation) can be interpreted differently. Similarly, the accepted environmental impact in the gray water and chemical footprint boundary approaches depends on the protection level that the selected critical concentration represents.

3.3.4. Component IV: scientific estimate
Component IV concerns the scientific estimate of an environmental sustainability boundary and considerations regarding the biophysical properties of the natural system in question. Here, elements including location of the control variable on the impact pathway, spatial resolution, temporal dynamics and interactions between environmental impact categories are included. How the analyzed approaches address these elements are summarized in figure 5. See SI table S4 for a full overview of the data.

3.3.4.1. Control variable and location on impact pathway
Seven approaches were found to apply control variables at the pressure point (biocapacity, critical loads, environmental space, green water availability, NPP, gray water and chemical footprint boundaries and three of the boundaries of the PBs approach) and seven approaches at the state point (chemical critical concentration, ecological critical concentrations, EFR, 2 °C boundary, TSE, system dynamics and seven of the boundaries of the PBs approach). One PB applies a control variable at the impact point (functional biodiversity) (figure 6a). Note that the PBs concerning biodiversity loss are interim boundaries until more pertinent ones are developed (Steffen et al 2015). Boundaries with the control variable at the pressure point are easier to control and to measure. However, they might be less environmentally relevant. Specifically, one can question a boundary’s ability to represent environmental concerns when it is not related to a response variable (i.e. impact). For example, the biocapacity approach expresses the boundary early in the impact pathway as the amount of productive land, considering the regenerative and absorptive capacity of the biosphere, with no further connection to environmental impacts. In contrast, e.g. the 2 °C boundary, is expressed at the state point of the impact pathway and, therefore, has stronger environmental relevance. It is, on the other hand, less controllable or measurable. Note that none of the boundary approaches have their control variables located at the drivers and response points of the impact pathway (figure 6), as such boundary approaches (e.g. zero-growth and population boundaries) were out of the scope of this review since they are considered to be socio-economic and not environmental sustainability boundaries.

3.3.4.2. Spatial resolution
Three boundary approaches (chemical critical concentration, 2 °C boundary and NPP), propose boundaries at a global scale (figure 5). For the chemical critical concentration, as applied in ERA,
the boundary for a chemical (the predicted no-effect concentration (PNEC)) is determined for specific environmental compartments (e.g. freshwater, marine water, soil etc.) (Gustavsson et al 2017). However, it does not account for environmental variability of local environments, unlike the predicted environmental concentration (PEC)) (Goussen et al 2016). Nonetheless, studies accounting for local ecological factors when estimating the boundary for chemicals in ERA exist. For example, Goussen et al (2016) use environmental scenarios to assess the effect of a combination of stressors (e.g. species composition, temperature or food availability and mixing of chemicals) by integrating exposure and ecological modelling.

Seven approaches define boundaries on a sub-global scale, either at country/continental or even smaller scales (labeled as ‘regional’ and ‘local’, respectively, in figure 5). The level of detail varies between the approaches in terms of parameters applied to account for spatial differences. For example, the biocapacity approach applies region-specific yield factors to reflect spatial differences. Yield factors at finer scales can also be applied (Borucke et al 2013). The critical loads approach is more comprehensive when accounting for spatial differences, either through an empirical or a modelling based method. For example, a critical load for nitrogen deposition can be based on empirical field experiments for different types of forests (e.g. calcareous forest or mesotrophic fens) (Groffman et al 2006) combined with modifying factors taking location specific information into account. For example, for stable dune grasslands, the pH of the dune is the
modifying factor defining whether the lower or higher level of the critical load range from field experiments should be used (Bobbink and Hettelingh 2010). For a modelling based approach, algorithms allow to estimate critical loads based on internal ecosystem characteristics such as soil properties, vegetation type and climatic data (Bashkin and Demidova 2008).

The PBs and environmental space approaches determine some of their proposed boundaries at global scale, while others at a regional scale (hence a ‘mixed’ spatial resolution in figure 5). Three out of the nine Earth system processes of the PBs are considered to have a global threshold (climate change, ocean acidification and stratospheric ozone depletion) and their boundaries are defined accordingly at global scale. It is recognized that the other processes (biosphere integrity (biodiversity loss), biogeochemical flows (eutrophication), land-system change, freshwater use and atmospheric aerosol loading) operate at regional scales. However, the regional nitrogen and phosphorus flows boundaries are derived from the global boundary assuming a uniform rate of addition. Hence, these proposed boundaries may deviate significantly from local and regional pollution limits (Steffen et al 2015).

3.3.4.3. Temporal dynamics

Only two boundary approaches (EFR and PBs) consider temporal and natural variations and how these may affect boundary values (figure 5). The EFR as applied in the blue water footprint (Hoekstra et al 2012), considers seasonal variation in water availability by comparing water use and availability on a monthly, rather than an annual, basis. Furthermore, Richter et al (2012) suggest adjusting the percentage of allowable depletion according to seasons. For setting the freshwater use boundary in the PBs approach, which is defined according to EFR principles, an intra-annual variability is reflected by applying a maximum monthly withdraw (as a percentage of mean monthly river flow). This is specified for three different flow regimes into high-, intermediate- and low-flow months. For ocean acidification, atmospheric aerosols loadings and ozone depletion the seasonal variability is incorporated into the boundary (Steffen et al 2015).

3.3.4.4. Indicator interaction

Despite the complexity of interactions between environmental processes, there are studies that consider interactions between systems of environmental pressures when estimating environmental sustainability boundary values. For example, the system dynamics approach aims to create dynamic modelling tools and Lenzen et al (2007) explored how past trends and human interactions with the biosphere might shape future biocapacity. Specifically, dynamic interactions between biodiversity and productivity and the impacts on future biocapacity were estimated. Lade et al (2019) considered interactions between the environmental pressures constituting the PBs. Two types of interactions are considered where changes in a control variable either leads to (1) changes in the control variable of another PB or (2) changes in the boundary value for another PB. As an example of the first type of interaction, land-system change can lead to carbon emissions that increase the atmospheric concentration of carbon dioxide, i.e. the control variable for the climate change PB changes, but the boundary value for climate change remains the same. As for the second type of interaction, climate change may affect the amount of freshwater that can be safely extracted. Thus, climate change can alter the boundary value of freshwater use. In general, the model and estimates of interactions of Lade et al (2019) are highly simplified hence they do not recommend this model for decision making. Rather, the provisional estimates for interactions and the new PBs should be seen as a summary of current scientific knowledge and a call for future research to better understand interactions (Lade et al 2019). Another recent study (Bjørn et al 2019), considered how interactions between the PBs for land-system change, climate change and freshwater use should be handled. Bjørn et al (2019) suggested to fix the positions of the boundaries and that interactions should be handled by calculating changes in control variable values (i.e. similar to the first type of interaction suggested in Lade et al (2019)).

3.3.5. Component V: uncertainty principle

In general, there is a lack of transparency with respect to how uncertainty is handled in the different approaches. Nevertheless, for most approaches, examples were found that indicate which uncertainty principle was applied. It was found that the majority of the boundary approaches apply a precautionary approach when dealing with data uncertainty. Exceptions are biocapacity and green water availability that tend to use the average value when handling a data range. The basis for defining the TSE (i.e. the boundary principle) is the natural regeneration rate of soils. Hence, the approach is, as a starting point, conservative. However, when defining the boundary based on a range of natural soil formation rates in EU, the boundary is set above the average (Verheijen et al 2009), i.e. not in accordance with the precautionary principle.

The PNEC, applied in ERA, is an example of a boundary defined according to the precautionary principle. Data used to predict chronic ecosystem effects and PNEC are limited for most substances as, in general, only short-term toxicity data are available. To account for missing data, empirically derived assessment factors can be used to translate from laboratory tests to the field. The assessment factors range from 1–1000, where the highest assessment factor is applied if the PNEC is based
on few toxicological studies (European Commission 2003). It is shown that, as intended, the initial assessment factor of 1000 comprises a measure of conservatism where access to additional data (requiring use of a lower assessment factor), therefore, generally generates higher PNECs (Gustavsson et al 2017).

This resembles the practice for handling uncertainty for some of the boundaries in the PBs approach. For example, for the interim boundary for functional biodiversity loss, a large uncertainty range reflects the considerable knowledge gaps about the BII-Earth system functioning and the placement of the boundary. However, for some of the PBs, the precautionary principle is manifested differently, with the uncertainty range mainly being based on actual data (not the lack of data as for the PNEC). For example, data exist to quantify the global threshold for ocean acidification. However, to account for seasonal variability, the acidification boundary is chosen well below the threshold (Rockström et al 2009).

Nonetheless, whether the uncertainty range stems from temporal or spatial variations, input parameters or model structure is not always clear. For example, the rationale behind the uncertainty ranges of the boundaries for regional land use and ozone depletion is not explicitly stated. Moreover, the rationale is not always consistent across the boundaries. For instance, for the freshwater use PB, it is transparent that the uncertainty range originates from an assessment of the variability in environmental water flow estimates when applying different calculation methods (hence model uncertainty), while seasonal variability is already considered by providing specific boundaries for each flow regime (i.e. low, intermediate and high flow). In contrast, for ocean acidification, the zone of uncertainty mainly reflects variabilities. Despite the fact that the rationale behind the uncertainty range is not always consistent and transparent, the uncertainty range for each boundary is transparently reported. Moreover, the PBs approach consistently err on the side of caution by setting a trade-off between environmental relevance and uncertainty range.

3.4. Recommended practices

In addition to provide a structured way to analyze and adopt environmental sustainability boundaries, the proposed framework outline recommended practices, based on the analysis and discussion of existing approaches (figure 4). Firstly, it is recommended that developers of boundary approaches explicitly and transparently state value-based considerations (component I, II, III and V) involved in the boundary definition. This will allow potential users of the approach to consider if and how they resonate with their own values or values of central stakeholders and allow them to potentially adapt them to their values. For example, in the analysis in section 3.3, it was discussed that the level of accepted impacts (component III) of the critical loads approach is only qualitatively stated, leaving the practitioner to quantify the level. This led to a high difference in the levels applied across EU. Ideally, these levels should be aligned by explicitly stating a common level of accepted impacts. At least, this high variation stresses the importance of explicitly and transparently report the level of accepted impacts and its rationale. As discussed in section 3.3, most approaches report their overall objective transparently. However, in many cases, the objective is not very specific, e.g. they do not explicitly report a time perspective, i.e. what is the temporal scope of the defined boundary in terms of its objective and what is sought to be protected. For example, do they aim to protect the current and/or future human generations, or to avoid destabilization of the Earth system using an, in principle, infinite time perspective? The objective, being the backbone and motivation of the approach, should be transparently reported with not only what we want to protect, but also where and when, i.e. specifying the geographical and temporal scope of the objective and, thus, the boundary.

When adopting several environmental sustainability boundaries from different approaches in an environmental assessment, value-based components should be consistent across impact categories or the discrepancies should be clearly stated. For example, if adopting boundaries in an environmental assessment and comparing across impact categories with a quantified uncertainty range, the boundaries should be placed on the uncertainty range based on the same values, e.g. applying a precautionary or a 'best estimate' principle across all impact categories. It should be clearly documented if such consistency is not applied. It is important to make users aware of such inconsistencies to avoid misunderstanding and potential misuse of the methods, since this difference can introduce a bias towards some impact categories in the assessment.

For component III (impacts and acceptable levels), it is recommended that response variables are located at the impact point of the impact pathway, describing directly the effect or change of state (e.g. damage to biodiversity), to ensure their relevance in relation to the type of impact they represent. However, it can be difficult to model the impact for several impact categories and the resulting level of uncertainty might be high. Therefore, a trade-off between environmental relevance and uncertainty must be considered.

In component IV (scientific estimate), boundaries should, as a starting point, be defined with an appropriate spatial resolution according to the environmental impact pathway of the impact category. For example, for global processes such as climate change, ocean acidification and stratospheric ozone depletion, boundaries should be defined at
global level. For non-global processes, such as freshwater use and land-system change, simple global average boundaries might hide important exceedance of boundaries at the non-global scale (Steffen et al 2015, Ryberg et al 2016). Thus, we recommend that non-global boundaries are defined at the relevant level of the process. In case that an approach has a global perspective but include non-global processes, like in the PBs approach, non-global processes can be sought aggregated to global level by, e.g. applying weighting factors that consider the relative importance of potential exceedances of regional boundaries in terms of impact on the global Earth system function, as outlined for the water PB in Gleeson et al (2020). In addition, dynamic interactions (i.e. temporal variations and interactions between environmental pressures) and the entailed consequences on boundary positions should be considered taking inspiration from the work of, e.g. Bjørn et al (2019) and Lade et al (2019), discussed in section 3.3.

Ideally in component V (uncertainty principle), any numerical value involved in the boundary quantification should be presented together with statistical information on the uncertainty and variability, e.g. as a confidence interval or a probability distribution. This would allow the user to apply an uncertainty principle in accordance with its respective perspective or the purpose of the assessment. For example, in decision support contexts such as the design of emission standards, a precautionary approach would often been applied when dealing with uncertainties. In contrast, in an LCA context, where the purpose is to compare indicator scores across assessed product systems and impact categories, the ‘best estimate’ expresses by the median or average value a probability range is normally used to avoid bias when comparing across impact categories (Bjørn and Hauschild 2015, Hauschild and Huijbregts 2015).

Finally, it is recommended to consider and align all components of the framework when developing, communicating and using an environmental sustainability boundary approach. However, the sequence of the components should not necessarily be followed. Indeed, the elaboration of a boundary is often an iterative process. For example, the boundary principle (component II) might be reconsidered if the uncertainty in the scientific estimate turns out to be too high. Taking the example of the 2 °C boundary, a political formulation of the objective (component I) was done by a political organ (the UN Framework Convention on Climate Change, UNFCCC) in 1992, followed by a scientific estimate by the IPCC, providing comprehensive and transparent assessment of impacts and risk entailed to different boundary values (component IV). Subsequently, the objective was translated to a numerical global temperature boundary (2 °C) with legal effects in a political process considering acceptable risks and impacts in the 2015 Paris Agreement (Gao et al 2017) (component III and V). Note that it was explicitly stated in the Paris Agreement, that efforts should be pursued to limit the temperature increase to 1.5 °C above pre-industrial level, which have later been emphasized in the IPCC Special Report published in 2018 (IPCC 2018) and adopted in recent studies (e.g. Chandrakumar et al 2020). This new value of the global temperature boundary exemplifies the iterative nature of setting environmental sustainability boundaries. Moreover, as discussed in section 3.3, the risks associated with different temperature increases have increased with every IPCC assessment report. This emphasize the importance of considering the definition and estimation of environmental sustainability boundaries as an iterative process, where the boundary must be continuously updated as more knowledge is obtained.

4. Conclusion

In this study, we identified, categorized and analyzed environmental sustainability boundary approaches and presented a framework for defining and adopting them in environmental assessments. Apart from providing the basis for developing the framework, the identification and analysis of existing boundary approaches may be used by potential practitioners applying environmental sustainability boundary approaches in an assessment to find boundaries suitable for their study and understand their differences and similarities.

We encourage potential developers and users of environmental sustainability boundary approaches to apply the framework presented here to enable a structured and consistent way to define, communicate, critically analyze and select boundaries and, thereby, to improve and facilitate application of them in sustainability assessments. The ‘recommended practices’ of the framework should be followed and further research should concentrate on the shortcomings of existing boundary approaches discussed in this study, such as further advancing the quantification of dynamic interactions between boundaries and increase the spatial resolution where relevant.

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Data availability statement

All data that support the findings of this study are included within the article (and any supplementary information files).
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