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Better, but good enough?

Indicators for absolute environmental sustainability in a life cycle perspective

PhD thesis 8.2015

DTU Management Engineering

Anders Bjørn
November 2015

DTU Management Engineering
Department of Management Engineering

I have amazing news for you. Man is not alone on this planet. He is part of a community, upon which he depends absolutely.
Daniel Quinn, Ishmael (1992)

I understand now that boundaries between noise and sound are conventions. All boundaries are conventions, waiting to be transcended.
David Mitchell, Cloud Atlas (2004)

PREFACE

This PhD thesis presents the outcome of the PhD project "Better, but good enough? Indicators for absolute environmental sustainability in a life cycle perspective." The project was carried out at the Division for Quantitative Sustainability Assessment of the Department of Management Engineering at the Technical University of Denmark. The project was supervised by Professor Michael Zwicky Hauschild (main supervisor) and Professor Inge Røpke, Aalborg University, and Professor Katherine Richardson, Copenhagen University.

The PhD project was carried out from December 2011 to April 2015 and included an exchange stay at CIRAIG (Centre international de référence sur le cycle de vie des produits, procédés et services) in Montreal, Canada, under supervision of Professor Manuele Margni and Professor Cécile Bulle.

The backbone of this thesis is six scientific articles. These articles are included as appendices and will be referred to by the numbers given below:

I. Bjørn, A., Diamond, M., Birkved, M., & Hauschild, M. Z. (2014). Chemical footprint method for improved communication of freshwater ecotoxicity impacts in the context of ecological limits. *Environmental Science and Technology*, 48(22), 13253-13262.

II. Bjørn, A., & Hauschild, M. Z (2015). Introducing Carrying Capacity Based Normalisation in LCA: Framework and Development of References at Midpoint Level. *International Journal of Life cycle assessment*, 20(7), 1005–1018.

III. Bjørn, A., Bey, N., Georg, S., Røpke, I., & Hauschild, M. Z. (2015). Is Earth recognized as a finite system in corporate responsibility reporting? *Journal of Cleaner Production*. Accepted with minor revisions.

IV. Bjørn, A., Margni, M., Roy, P. O., Bulle, C., & Hauschild, M. Z. (2015). Modifying life cycle assessment to measure absolute environmental sustainability. *Ecological indicators*. Accepted with minor revisions.

V. Bjørn, A., Richardson, K., & Hauschild, M. Z. (2015). Environmentally sustainable or not? Managing and reducing indicator uncertainties. *Ecological indicators*. To be resubmitted.

VI. Bjørn, A., Diamond, D., Owsianiak, M., Verzat, B., & Hauschild, M. Z (2015) Strengthening the link between life cycle assessment and indicators for absolute sustainability to support development within planetary boundaries. *Environmental Science and Technology*, 49(11), 6370-6371.

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These three years have passed incredibly fast and, yet, I feel that I have learned a lot and hopefully produced a bit of knowledge for others to learn from. This could not have happened without, in particular, my main supervisor Michael Zwicky Hauschild or co-supervisors Katherine Richardson and Inge Røpke. Being immersed in the supportive, energetic and pleasantly geeky environment of the QSA Division at DTU has also been invaluable for my journey. The inspiring research stay at CIRAIG, over a long wonderful summer, in addition, gave me a unique opportunity to test and develop my ideas with people having new and refreshing perspectives. I would also like to thank the co-authors of the articles that was written during the PhD project for countless times gently pushing me in the right direction and helping me to express my thought better than I could do so myself. My friends and family also deserve my gratitude for coping with my frequent physical and mental absence during, especially, the last months of the project. Finally, this project would never have happened had it not been for the Danish tax payers funding it. I will try to make it worthwhile.

SUMMARY

An increasing focus on sustainability has led to proliferation of the use of environmental indicators to guide various types of decisions, from individual consumer choices to policy making at the national, regional and global scale. Most environmental indicators are relative, meaning that quantified environmental interferences of a studied anthropogenic system (a product, a company, a city, etc.) are compared to those of chosen anthropogenic systems of reference. The use of relative indicators can give the impression that societies are moving towards environmental sustainability when decisions are being made which favour solutions with lower environmental interferences than alternative solutions. This impression is very problematic considering that monitoring repeatedly shows that many environments are highly degraded and that degradation often increases over time. This shows that society-nature interactions in many cases are environmentally unsustainable and that the level of unsustainability may be increasing over time. A clear rationale therefore exists for developing and using absolute environmental sustainability indicators (AESI) that not only can identify the anthropogenic system with the lowest environmental interferences in a comparison of systems, but also can evaluate whether any of the compared systems can be considered environmentally sustainable, and if not, can quantify the decrease in environmental interferences required for environmental sustainability. The purpose of this PhD thesis is to improve AESI using life cycle assessment (LCA) and to deepen the understanding of drivers and obstacles for increasing the use of AESI in decision-support. The thesis summarizes in three core chapters the work of five peer reviewed scientific articles and one scientific viewpoint article.

The first chapter is concerned with operationalizing the concept of carrying capacity as reference value of environmental sustainability in environmental indicators in general and in LCA indicators specifically. LCA is a tool that quantifies environmental stressors (resource use and emissions) occurring over the life cycles ("cradle to grave") of anthropogenic systems and translates these stressors into metrics of environmental interferences for a number of mutually exclusive and collectively exhaustive "impact categories", such as climate change, eutrophication and ecotoxicity. Carrying capacity is in this thesis defined as "the maximum sustained environmental interference a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert." In the design of AESI a choice needs to be made for each of 12 identified concerns. Existing AESI are found to be based on different choices for concerns, such as "threshold value", "quantifying environmental in-

terferences of studied system" and "modelling of carrying capacity." This difference in choices across AESI can lead to high uncertainties in indicator scores, potentially 3 orders of magnitude, and should thus be reduced where possible. Existing AESI are also found to only partially cover all impact categories. LCA indicators can potentially contribute to increasing the coverage of impact categories in AESI and to reducing indicator uncertainties, due to the consistent choices made for LCA indicators for many of the 12 indicator concerns. LCA indicators are relative and must be modified with carrying capacity references to become AESI. This modification can either happen in the normalisation of indicator scores or by developing new characterisation factors (CFs) used to translate environmental stressors to metrics of environmental interferences in LCA. Operational global and European carrying capacity based normalisation references are developed for 11 LCA impact categories and can be used to translate indicator scores from metrics specific to each impact category (such as Global Warming Potential for the impact category climate change) to a common metric of carrying capacity occupation, expressed in person years. To improve the representation of spatial variations, a generic mathematical equation for integrating carrying capacity in CFs is developed. Such CFs express indicator scores as hectare years, i.e. occupation of carrying capacity integrated over space and time. CFs for the impact category terrestrial acidification are developed and show strong local and regional variations (e.g. ranging above a factor of 5 across contiguous United States). The high spatial variation is an argument for using carrying capacity modified CFs, as opposed to modified normalisation references, when the locations of stressors of a studied anthropogenic system are known.

The second chapter is concerned with calculating carrying capacity entitlement of individual anthropogenic systems, with analysing the applicability of different valuation principles in calculating entitlements and with how sensitive calculated entitlements are to choice of valuation principle. Entitlements must be calculated to evaluate whether an anthropogenic system can be considered environmentally sustainable, which is the case when carrying capacity occupation does not exceed entitlement. Calculation of entitlement must consider the perceived value of a studied system relative to systems that compete for the same carrying capacity for their functioning. An ideal and a simplified method for identifying competing systems in a spatial assessment are outlined. A list of valuation principles is presented and includes contribution to Gross domestic product (GDP) and contribution to meeting human needs. The applicability of the valuation principles on different types of anthropogenic systems (territorial or lifecycle-based from micro- to macro scale) is analysed. Case studies are used to illustrate that the choice of valuation principle has a potentially large influence on the carrying capacity entitled to an anthropogenic system.

The third chapter is concerned with characterising companies' use of AESI in stakeholder communication and with how to increase this use. Companies have recently been encouraged by various initiatives to adopt AESI to define targets with deadlines for environmental sustainability at company level. A screening and context analysis of the largest global database of corporate responsibility reports found that only 23 out of 9,000 companies were following this advice. Explanations for the low share may be that the use of AESI is (still) not being sufficiently demanded by critical stakeholders and that operational AESI for impact categories other than climate change are either not available or not compatible with the tools with which companies express their environmental interferences. Two strategies for increasing the use of AESI by companies are proposed: 1) AESI based on LCA indicators should be further developed and made available to companies, since many companies already use LCA to reporting environmental interferences. 2) The awareness of AESI must be increased amongst critical stakeholders so that they can pressure companies to adopt AESI.

Following the three core chapters, a final chapter with recommendations is provided. This chapter outlines future research needs on AESI related to indicator development and refinement, inventory data, social sustainability references and consensus needs. Practical measures for increasing the use of AESI in decision-making are also proposed.

RESUMÉ

En øget fokus på bæredygtighed har ført til en stigende brug af miljøindikatorer som støtte til beslutninger fra individ- til samfundsniveau. De fleste miljøindikatorer er relative, hvilket vil sige at de sammenligner kvantificerede miljøpåvirkninger fra et studeret menneskeskabt system (et produkt, en virksomhed, en by, mm.) med miljøpåvirkninger fra et referencesystem. Brugen af relative indikatorer kan give indtryk af at samfundet bevæger sig mod miljømæssig bæredygtighed når træffede beslutninger tilgodeser løsninger som har lavere miljøpåvirkninger end alternative løsninger. Det indtryk er meget problematisk, set i lyset af at studier af miljøets tilstand gang på gang viser at den i mange henseender er dårlig og at der ofte observeres en forværring over tid. Dette viser at menneskers interaktion med miljøet i mange tilfælde er miljømæssigt ubæredygtig og at graden af ubæredygtighed kan være voksende. Denne problemstilling udgør en klar motivation for at udvikle og bruge indikatorer for absolut miljømæssig bæredygtighed (IAMB), som ikke blot kan identificere det menneskeskabte system der har lavets miljøpåvirkning blandt en række alternativer, men også kan evaluere hvorvidt nogle af systemerne overhovedet kan betragtes som miljømæssigt bæredygtige, og hvis ikke, kan kvantificere den reduktion i miljøpåvirkning der er nødvendige for miljømæssig bæredygtighed. Formålet med denne ph.d.-afhandling er at videreudvikle IAMB ved hjælp af livscyklusvurdering (LCA) og at undersøge drivkræfter og hindringer for at øge brugen af IAMB til beslutningsstøtte. Afhandlingen sammenfatter i tre hovedkapitler forskningsarbejdet fra fem fagfællebedømte videnskabelige publikationer og én videnskabelig ”viewpoint”-publikation.

Det første kapitel omhandler operationalisering af konceptet bæreevne som referenceværdi for miljømæssig bæredygtighed i miljøindikatorer generelt og i LCA-indikatorer specifikt. LCA er et værktøj der kvantificerer de miljøinteraktioner (ressourceforbrug og emissioner) som finder sted gennem livscyklussen af menneskeskabte systemer (”vugge til grav”) og oversætter disse miljøinteraktioner til mål for miljøpåvirkning for en række gensidigt ekskluderende og kollektivt udtømmende ”påvirkningskategorier”, så som klimaforandring, eutrofiering og økotoksicitet. Bæreevne defineres i denne afhandling som ”den maksimalt vedholdte miljøpåvirkning et naturligt system kan modstå uden at undergå negative forandringer i struktur eller funktion som er vanskelige eller umulige at reversere.” Når IAMB designes skal der træffes valg for hvert af 12 identificerede karakteristika. Eksisterende IAMB har vist sig at være baseret på forskellige

valg for karakteristika så som ”tærskelværdi”, ”kvantificering af et studerets systems miljøpåvirkning” og ”modellering af bæreevne.” Denne forskellighed i valg kan føre til høje usikkerheder i indikatorresultater, potentielt 3 størrelsesordener, og bør derfor reduceres hvor muligt. Desuden har eksisterende IAMB vist sig kun delvist at dække alle påvirkningskategorier. LCA-indikatorer kan potentielt bidrage til at øge dækningen af påvirkningskategorier i IAMB og til at reducere usikkerheder, pga. ensartetheden i valg truffet for LCA-indikatorer for mange af de 12 karakteristika. LCA-indikatorer er relative og skal derfor modificeres med bæreevnereferencer for at blive IAMB. Denne modifikation kan enten ske via normaliseringen af indikatorresultater eller via udviklingen af nye karakteriseringsfaktor (CF), som anvendes til at oversætte miljøinteraktioner til mål for miljøpåvirkning i LCA. Operationelle globale og Europæiske normaliseringsreferencer baseret på bæreevne udvikles for 11 LCA påvirkningskategorier og kan benyttes til at oversætte indikatorresultater for specifikke mål knyttet til hver påvirkningskategori (så som global opvarmningspotentiale for kategorien klimaforandringer) til et fælles mål for beslaglæggelse af bæreevne, udtrykt i person-år. En generisk matematisk formel til integrering af bæreevne i CF udvikles med henblik på at forbedre modelleringen af stedslig variation. Sådanne CF kan bruges til at udtrykke indikatorresultater i hektar-år, dvs. beslaglagt bæreevne integreret over rum og tid. CF for påvirkningskategorien terrestrisk forsuring udvikles og udviser stærk lokal og regional variation (f.eks. spænder CF over mindst en faktor 5 på tværs af den geografisk sammenhængende del af USA). Den høje variation er et argument for at bruge bæreevne-modificerede CF, i stedet for modificerede normaliseringsreferencer, når den geografiske beliggenhed af miljøinteraktioner kendes.

Det andet kapitel omhandler udregning af individuelle menneskabte systemers berettigelse til bæreevne, analyse af anvendeligheden af forskellige prissætningsprincipper samt følsomhed af den udregnede berettigelses overfor valg af prissætningsprincip. Berettigelse skal udregnes for at muliggøre evaluering af hvorvidt et menneskeskabt system kan betragtes som miljømæssigt bæredygtigt, hvilket er tilfældet når beslaglæggelsen af bæreevne ikke overstiger berettigelsen til bæreevne. Udregningen af berettigelse skal medtage den opfattede værdi af det studerede system, relativt til systemer som konkurrerer om at beslaglægge dele af den samme bæreevne. En ideel og en forenklet metode til identifikation af konkurrerende systemer i rummeligt opløste studier opridses. En liste med prissætningsprincipper præsenteres og indeholder, bl.a., bidrag til bruttonationalproduktet (BNP) og bidrag til at imødekomme menneskers behov. Anvendeligheden af prissætningsprincipperne overfor forskellige typer af menneskeskabte systemer (territorielle eller livscyklusbaserede fra mikro- til makroskala) analyseres. Casestudier bruges til at illustrere at valg af prissætningsprincip har potentiel høj indflydelse på berettigelsen af bæreevne til et menneskeskabt system.

Det tredje kapitel omhandler virksomheders brug af IAMB i kommunikation til interessenter og hvordan denne brug kan øges. Virksomheder er på det seneste blevet opfordret til at benytte IAMB til at definere målsætninger med deadlines for miljømæssig bæredygtighed på virksomhedsniveau. Via en screening og kontekstanalyse af den største globale database af virksomheders ansvarlighedsrapporter findes det at kun 23 af 9.000 virksomheder har fulgt det råd. Mulige forklaringer på den lave andel er at kritiske interessenter (stadigt) ikke stiller krav til brugen af IAMB og at operationelle IAMB for andre påvirkningskategorier end klimaforandring enten er utilgængelige eller ikke er kompatible med de værktøjer virksomheder bruger til at udregne og rapportere deres miljøpåvirkninger. Der forslås to strategier for at øge virksomheders brug af IAMB: 1) IAMB baseret på LCA-indikatorer bør videreudvikles og gøres tilgængelig for virksomheder, siden mange virksomheder er vant til at udtrykke deres miljøpåvirkninger ved brug af LCA. 2) Bevidstheden om IAMB må øges hos kritiske interessenter så de kan presse virksomheder til at bruge IAMB.

Efter de tre hovedkapitler præsenteres et kapitel med anbefalinger. Kapitlet opridser nye behov for forskning i IAMB relateret til indikatorudvikling- og forbedring, opgørelser over miljøinteraktioner, referencer for social bæredygtighed og behov for konsensus. Til sidst forslås praktiske måder at øge brugen af AESI som beslutningsstøtte.

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1 INTRODUCTION

Over the past decades increasing priority has been given to sustainability in policy and many companies reputedly perceive their sustainability performance and commitments as crucial for achieving or maintaining a "social license" (Gunningham et al., 2004). Sustainable development is commonly defined as "…development that meets the needs of the present without compromising the ability of future generations to meet their own needs." (WCED, 1987). Due to the complexity of sustainable development and sustainability, many indicators have been developed and used to quantify the sustainability performance of anthropogenic systems, such as products, companies and nations, with the aim of supporting decisions where sustainability is part of the decision criteria.

In this thesis the focus is indicators of environmental sustainability, which may be defined as "…seek[ing] to improve human welfare by protecting the sources of raw materials used for human needs and ensuring that the sinks for human wastes are not exceeded, in order to prevent harm to humans" (Goodland, 1995). Hence social and economic sustainability indicators will not be dealt with in the core research work. Environmental indicators (here used synonymously with environmental sustainability indicators) generally reflect a systems analysis view of the relations between the natural system and the anthropogenic system in physical, biological or chemical terms (Smeets and Weterings, 1999). Environmental indicators can be expressed at different points, from underlying cause to ultimate effect, in a so-called impact pathway. The DPSIR framework in Figure 1.1 defines Driving forces, Pressures, State of the environment, Impact on natural systems and societal Response in an impact pathway and characterises the links between these points.

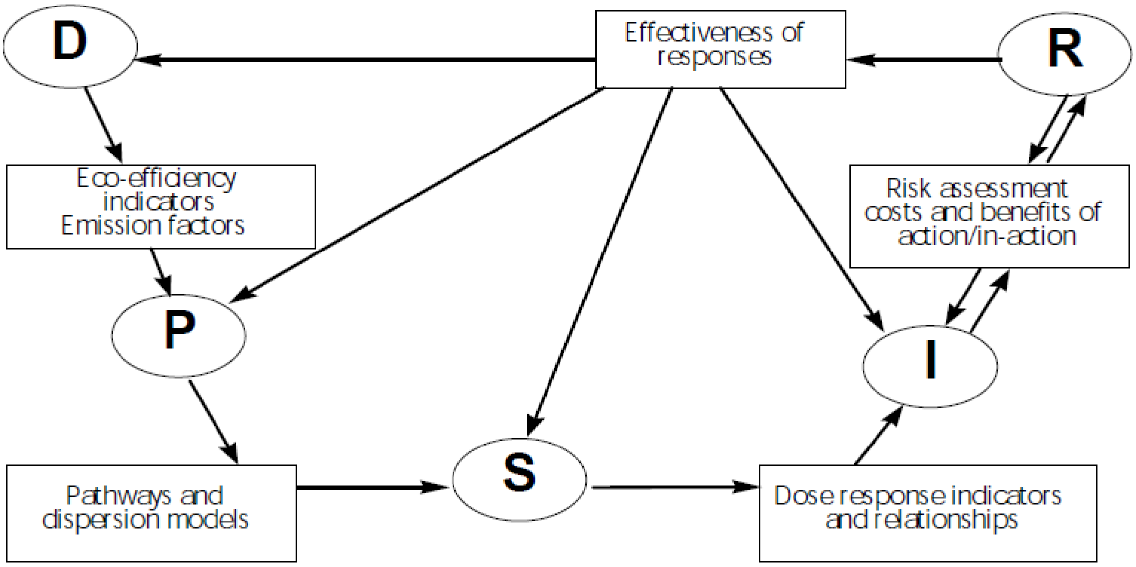

Figure 1.1: Links between driving forces (D) pressures (P), state of the environment (S), impacts on natural systems (I) and societal responses (R) in the DPSIR framework. Adapted from Smeets and Weterings (1999)

Indicators of driving forces describe the social, demographic and economic developments in societies. An example is the development in numbers of private cars in a city. Pressure indicators describe developments in emissions and the use of resources. An example is the development of annual national greenhouse gas emissions. State indicators describe the development in the quantity and quality of physical phenomena (such as temperature), biological phenomena (such as fish stocks) and chemical phenomena (such as atmospheric CO_2-concentrations) in a natural system. Impact indicators describe negative changes in of natural systems with respect to structure (e.g. the species present) and functioning (e.g. the provision of biological resources or climate regulation). Response indicators describe responses by groups and individuals in society to prevent, compensate, ameliorate or adapt to changes in the driver, pressure, state or response. Responses thus span from targeting consumption patterns (driver) to compensating for or adapting to impacts (Smeets and Weterings, 1999). In this thesis the term "environmental interference" will be used generically to describe the object of indication by pressure, state and impact indicators. Response indicators will not be dealt with directly.

The anthropogenic systems evaluated by environmental indicators can generally be either of territorial or life cycle character. Territorial systems are delimited by the geographical boundaries of a territory, such as a property, a city or a nation, and defined by their physical production of goods or services (e.g. electricity supplied to grid). Life cycle systems are defined by the functions they deliver (such as mobility) and all environmental interferences resulting from the supply of this function from “cradle” (raw materials) to “grave” (waste management) are included in the system boundary, no matter where these environmental interferences occur. A territorial perspective is generally taken in studies of environmental interferences of production (e.g. for a specific production plant or sector within a nation), while a life cycle perspective is taken in studies of the environmental interferences of consumption. Territorial and life cycle systems can both vary on a continuum from the micro to the macro level. Figure 1.2 illustrates the diversity of systems that can be evaluated by environmental indicators.

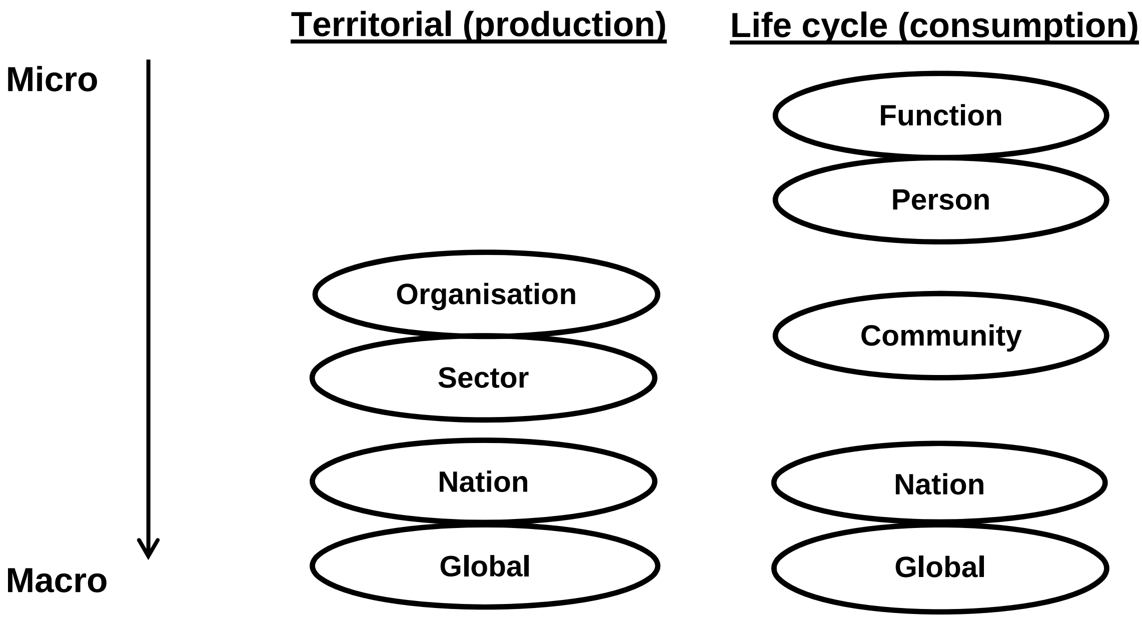

Figure 1.2: Classification of selected anthropogenic systems that may be evaluated by environmental indicators by the territorial and life cycle categories and micro-macro scale continuum.

Existing environmental indicators are predominantly relative. Such indicators compare the sustainability performances of, for instance, a group of functionally equivalent products to identify the product with the overall lowest environmental interferences (Moldan et al., 2012). In that way, most systems can in principle appear environmentally sustainable when they can be compared to other systems that perform worse (see, for example, Waechter et al. (2015)). This relativistic approach can give the impression that the world is moving towards environmental sustainability seeing as how many decisions favour solutions with a relatively good environmental performance compared to alternatives (Ehrenfeld, 2005). This impression of (moving towards) environmental sustainability is very problematic considering that environmental monitoring repeatedly shows that the opposite is the case: The Millennium Ecosystem Assessment found that 15 of 24 ecosystem services are generally being degraded or used unsustainably (WRI, 2005), global biodiversity continues to decline (UNEP, 2012) and the concentration of CO_2 in the atmosphere is not only increasing, but increasing at an accelerating rate (IPCC, 2013).

Relative environmental sustainability indicators (RESI) therefore needs to be supplemented by absolute environmental sustainability indicators (AESI) where sustainability reference values, that are external to studied anthropogenic systems, represent the absolute element (Moldan et al., 2012). The concept of carrying capacity (Sayre, 2008) can be applied to operationalize and quantify the sustainability reference values needed in AESI. Environmental interferences, affecting a given natural system, would then be considered environmentally sustainable if their levels are below the carrying capacity of that natural system. The planetary boundaries concept and various footprint methods can be characterised as AESI because they express environmental interferences as occupations of carrying capacity (Borucke et al., 2013; Hoekstra et al., 2012; Rockström et al., 2009). Current AESI all have shortcomings, related to e.g. data quality and incomplete coverage of environmental issues, that can potentially be overcome by the use of life cycle assessment (LCA) (Galli et al., 2012; Huijbregts et al., 2008).

LCA aims to quantify all relevant environmental interferences over the life cycle (from raw materials to waste management) of an anthropogenic system (typically a product system) (EC, 2010b; ISO, 2006a, 2006b). Environmental interferences are expressed in indicator scores for a number of so-called impact categories, such as climate change, eutrophication and ecotoxicity. Indicator scores reflect the potentials of emissions and resource uses (collectively referred to as stressors) to create a small change in the level of environmental interferences. LCA indicators therefore generally do not include carrying capacity as sustainability reference values (Castellani and Sala, 2012). To harness the benefits of LCA in AESI, LCA indicators need to be modified to quantify occupations of carrying capacity. This modification can either happen by applying a carrying capacity reference after the calculation of indicator scores or by integrating such a reference directly into LCA indicator models, in which case spatial variations of carrying capacity may be captured. In both cases, carrying capacities must be derived from threshold values and expressed in metrics similar to those of LCA indicators. Essential for the derivation of numerical values of carrying capacities is therefore the choice of threshold and its translation to carrying capacity.

There are generally no natural mechanisms distributing carrying capacity between its users. Therefore a key aspect in AESI is the share of a natural system's carrying capacity that a studied anthropogenic system can be considered entitled to. Entitlement is a normative concept because it inherently involves value judgement of anthropogenic systems that are competing for the occupation of the same finite carrying capacity. Since the sustainability criterion of any scenario or system is a function of entitlement, it is important to explore the range of possible approaches to calculating the carrying capacity entitled to any type of anthropogenic system.

When designing indicators for AESI the users of AESI also need to be considered. Since the primary decision support of LCA happens in industry, it is of special importance to characterise companies' use of AESI. There are many reasons why companies may use environmental indicators. These reasons are to a large extend related to companies' social license, defined by Gunningham et al. (2004) as "the demands on and expectations for a business enterprise that emerge from neighbourhoods, environmental groups, community members, and other elements of the surrounding civil society." An important mean for companies to communicate to these stakeholders is corporate responsibility (CR) reporting. CR reports therefore offer potentially valuable insight to companies' use of environmental indicators, including AESI. This insight can be used to develop AESI that are more likely to be used by companies. AESI allows for companies to quantitatively define absolute environmental sustainability targets at the company level and communicate these to stakeholders. This practice can be seen as essential to improve the current situation where RESI are used to associate many companies, undeserved, with environmental sustainability.

1.1 RESEARCH QUESTIONS

Three research questions were defined to study the considerations above:

1. How can the carrying capacity concept be operationalized as sustainability reference value in environmental indicators in general and in LCA indicators specifically?
2. How can the carrying capacity entitlement of individual anthropogenic systems be calculated, how applicable are different valuation principles to calculating entitlements and how sensitive is this calculation to choice of valuation principle?
3. What characterises companies' use of AESI in CR reports and how may answers to research question 1 and 2 contribute to an increase in companies' use of AESI?

The structure of the thesis is aligned with the three research question. Chapters 2, 3 and 4 are devoted to answering each of the questions. Each chapter draws upon one or more of the five scientific **articles** (**I-V**), which are attached as appendices: Chapter 2 is supported by **articles I**, **II**, **IV** and **V**; Chapter 3 is supported by **articles I**, **III**, **IV** and **V**; Chapter 4 is supported mainly by **article III**. Chapter 5 provides recommendations based on the conclusions of the previous chapters. A list of major findings is subsequently provided. The viewpoint (**article VI**) can be considered a popular summary of the entire thesis.

2 CARRYING CAPACITY IN ENVIRONMENTAL INDICATORS AND LCA

In this chapter the first main research question is answered: “How can the carrying capacity concept be operationalized as sustainability reference value in environmental indicators in general and in LCA indicators specifically?”

2.1 INTRODUCTION TO CARRYING CAPACITY AND ITS CURRENT USE

2.1.1 DEFINITION

Carrying capacity generally refers to a certain quantity of X that some encompassing Y is able to carry (Sayre 2008). X and Y can refer to different entities, depending on the discipline in which carrying capacity is applied.[1] In all applications carrying capacity has always aspired to idealism, stasis, and numerical expression (Sayre 2008). In ecology, for example, carrying capacity describes the maximum equilibrium number of organisms of a species (X) that a given environment (Y) in theory can support indefinitely (Odum, 1971). Carrying capacity is included in the common definition of eco-efficiency (WBCSD, 2000): “Eco-efficiency is achieved through the delivery of competitively priced goods and services that satisfy human needs and bring quality of life while progressively reducing environmental impacts of goods and resource intensity throughout the entire life-cycle to a level at least in line with the Earth's estimated carrying capacity.” In this use of the carrying capacity concept X is unspecified environmental interferences and Y is the planet and carrying capacity thus acts as the boundary between global environmental sustainability and unsustainability (Goodland, 1995).

Motivated by this use, carrying capacity is here defined as: “the maximum sustained environmental interference a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert” (**article II** and **IV**). Here, natural system may refer to ecosystems or, more broadly, Earth´s interacting physical, chemical, and biological processes, which for instance make up the climate system. By considering both functioning and structure the carrying capacity definition aims for a balanced approach: Whereas the concept of ecosystem functioning may have an anthropocentric bias, in that it tends to focus on functions valuable to humans, the concept

[1] Wildlife management, chemistry, medicine, economics, anthropology, engineering, and population biology are listed as examples by Sayre (2008).

of eco-system structure is eco-centric because no judgement is made on the relative inherent value of organisms.[2]

2.1.2 Operationalization and use in existing AESI

When operationalising carrying capacity, two additional terms need introduction: 1) control variable: "a numerical indicator of the structure or functioning of a natural system." 2) threshold: "the maximum value of a control variable a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert." Figure 2.1 shows that thresholds can be derived from different responses of a natural system to changes in control variable (Dearing et al., 2014).

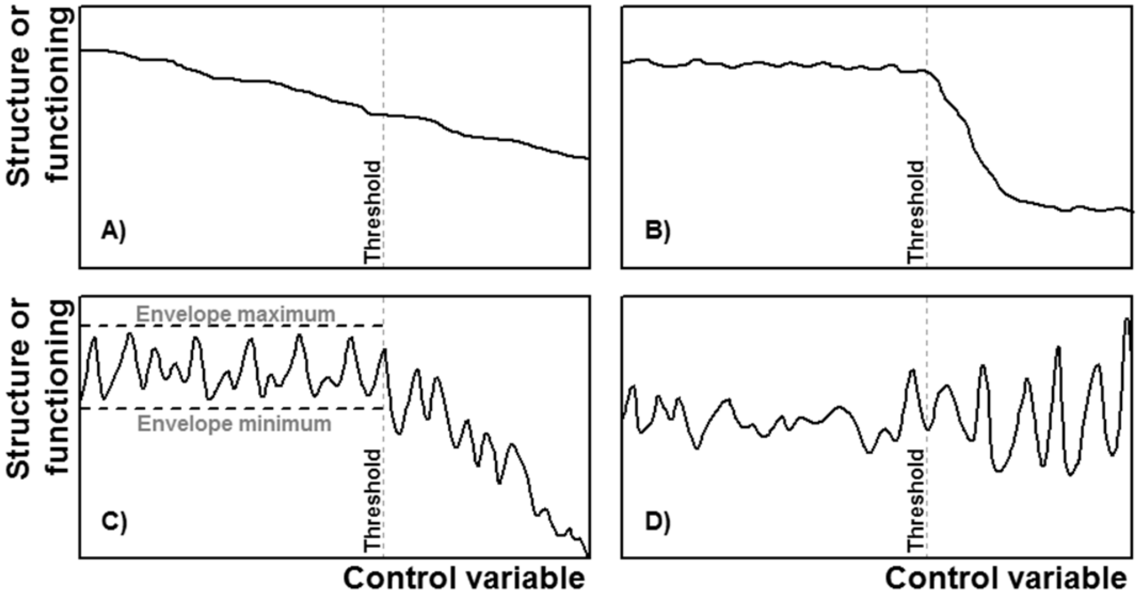

Figure 2.1: Classification of thresholds depending on a natural system's response to environmental interferences: A) gradual degradation, B) true threshold, C) envelope of variability, D) early warning signals. Adapted from Dearing et al. (2014).

Type B, C and D thresholds are all located at a point where increases in control variable starts to create different types of responses in the natural systems structure and/or functioning: For type B the response can be characterised as a "true threshold" because the response to changes in control variable at the threshold value is much larger than the response at lower levels. For type C the threshold

[2] The concept of resilience may offer a bridge between anthropocentric and eco-centric approaches to environmental management, since studies generally show that ecosystems with high genotype- and species diversity has a high resilience, meaning in general terms, that they are better at adapting to sudden changes in conditions than ecosystems with lower diversity (Carpenter et al., 2001; Scheffer et al., 2001). Thus the protection of ecosystem structure can be seen both as eco-centric and as being in the enlightened self-interest of man.

value marks the point where the response falls outside the “envelope” of natural variability. For type D the threshold value is where the magnitude or frequency of deviations from average condition of the response starts to increase. In contrast, Type A thresholds are associated with close to linear response function (“gradual degradation”). Deriving a type A threshold can thus involve expert-judgement and the opinion of stakeholders on an optimum between the benefits of anthropogenic systems causing the control variable to increase and the disadvantage of the gradual degradation of natural systems caused by this increase. Due to the complexity of natural systems and limitations in scientific understanding it is often not clear which of the four threshold types that best characterise a system.

Thresholds act as primary sustainability reference values and carrying capacity as derived sustainability references expressed in a metric identical to the metric of modelled environmental interferences for use in AESI (**article II and IV**). The occupation of carrying capacity is typically expressed as a ratio of current environmental interferences over carrying capacity, sometimes aggregated over spatial units. The operationalization of carrying capacity and other features of AESI involve a number of concerns, for each of which a choice needs to be made. Based on **article V,** Table 2.1 presents 12 such mutually exclusive and collectively exhaustive concerns. The list of concerns comprises an analytical tool to characterise AESI. Table 2.1 provides examples of this characterisation for three well known AESI: the ecological footprint (Borucke et al., 2013), the blue water footprint (Hoekstra et al., 2012) and the planetary boundary for climate change (Rockström et al., 2009; Steffen et al., 2015).

Table 2.1: Identification of 12 universal concerns in the design of AESI (based on article V) and characterisation of three AESI examples. Concerns in italic are relevant in the characterisation of LCA indicators.

Concern	Explanation	Ecological footprint (Borucke et al., 2013)	Blue water footprint (Hoekstra et al., 2012)	Planetary boundary for climate change (Rockström et al., 2009; Steffen et al., 2015)
1. Natural system of focus	As a stressor can affect several natural systems, a natural system of focus needs to be chosen.	All within Earth's biologically productive areas.	Freshwater ecosystems.	The climate system.
2. Goal	Natural systems are complex and can prevent harm to humans in many ways. A goal specifying the structure or functioning that should be protected as a precondition for environmental sustainability must be stated.	Protecting provisioning and regulating services (inferred).	Avoid "moderate to major changes in natural structure and ecosystem functions" due to alteration of natural flows.	Maintain the climate system in a state characterised by the Holocene.
3. Control variable	To measure the degree to which the goal is met a relevant control variable must be chosen.	Net depletion of biological stock (kg carbon/year).	Alteration of natural flows (%).	Increase of radiative forcing (W/m^2) relative to the pre-industrial level.
4. Basis for the threshold (classification according Figure 2.1)	A threshold is a value of the control variable that indicates whether the goal is met. Thresholds can be derived in different ways and a choice must therefore be made.	A net depletion of biological stock that does not lead to a decrease in biological productivity (Type B, true threshold).	A presumptive standard restricting hydrologic alterations to within a percentage-based range around natural flow variability (Richter et al., 2012). (Type C, envelope of variability.)	Studies of natural variability within the Holocene combined with models of hysteresis type abrupt thresholds. (Type C, envelope of variability and type B, true threshold.)

5. Threshold value	A single value must be chosen from the range of numerical threshold values derived.	Maximum 0 kg carbon/year. The threshold value was derived mechanistically (no range).	Maximum 20% alteration of natural flows. The used threshold value is "conservative and precautionary", according to the authors.	Maximum 1 W/m^2. The lower value in a range of threshold values is used.
6. Location of carrying capacity in impact pathway (see Figure 1.1)	To facilitate the comparison of a threshold to the environmental interference of a studied system, the threshold is commonly translated to a metric of carrying capacity. Carrying capacity is generally expressed at a chosen earlier point in the impact pathway than the threshold.	Pressure point: "biocapacity" (global hectare years)	Pressure point: Water consumption (m^3/month).	State point: Increase of radioactive forcing (identical to threshold)
7. Modelling of carrying capacity	In the translation of a threshold to carrying capacity an impact pathway model is needed. A choice needs to be made between available models which may vary in structure and parameters used.	Modelled for each nation based on the area of productive land, normalised to global hectare years in a calculation involving: 1) Yield factors: the relationship between national average and global average yields for different agricultural products and land use type (cropland, pasture, etc.), and 2) Equivalence factors: the relationship between the average yield of different land use types.	Water balance of Fekete et al. (2002) is used to calculate river specific flows. When multiplying with the threshold value (20%) a river specific carrying capacity expressed as maximum alteration of natural flows (m^3/month) is then calculated.	No modelling needed as threshold and carrying capacity are identical.

8. Quantifying environmental interferences of studied system	An approach related to monitoring, modelling or a combination of the two must be chosen to quantify environmental interferences of the system studied by the indicator.	Production, import and export of different types of biomass and embedded CO_2 emissions from a nation's annual consumption are expressed in global hectares by use of Yield and Equivalent factors.	Agricultural consumption is based on spatially resolved water balance model for soils. Industry and domestic consumption are based on water withdrawal data, which are spatially adjusted based on population densities (Hoekstra et al., 2011). Consumption (m^3/month) is directly comparable to carrying capacity, expressed in maximum alteration of natural flows (m^3/month).	Global average of radioactive forcing measurements supplemented by modelling.
9. Spatial coverage and resolution	Spatial variations in impact pathway and carrying capacity within chosen geographical boundaries can be captured to a varying extent depending on the choice of spatial resolution.	Global coverage with a nation level resolution.	Global coverage with a five by five arc minutes resolution.	Global coverage with a spatially generic resolution, since climate stressors have high dispersion and since the threshold is expressed as global average.
10. Temporal coverage and resolution	Some carrying capacities vary within the chosen time frame, due to natural dynamics, and so do environmental interferences. These variations can be captured to a varying extent depending on the choice of temporal resolution.	Footprints and carrying capacities are estimated annually.	Environmental interferences and carrying capacities are estimated at a monthly basis. Monthly carrying capacities are averaged over 10 years to take into account inter-annual climatic variability.	Carrying capacity is static. Environmental interferences are typically estimated annually.

11. Aggregation of indicator scores	A choice must be made as to how to aggregate estimated degrees of carrying capacity exceedance across spatial and temporal units to a single indicator score for the entire ecosystem within the geographical boundary.	Footprints (global hectares) are added no matter where and when they take within the year of study.	The share of global basins where carrying capacity is exceeded for each month is given.	Not relevant because indicator is spatially generic.
12. Normative basis for carrying capacity entitlement	When different anthropogenic systems are causing environmental interferences within a geographical boundary, the entitlement to carrying capacity of each system must be decided.	Personal entitlement either based on global per capita carrying capacity or nation-specific per capita carrying capacity.	None. The environmental sustainability of all anthropogenic systems is evaluated collectively.	None. The environmental sustainability of all anthropogenic systems is evaluated collectively.
Type of anthropogenic system analysed (see Figure 1.1)		Life cycle, typically applied to scale of nations.	Territorial, typically applied to scale of sectors.	Potentially all.

The three AESI examples exhibit a quite high variation in choices made for the 12 concerns, with the choice of global coverage for concern 9 being the only common choice for all three indicators. For some concerns, variation in choices between indicators can be considered positive. For example, indicators should collectively cover different “Natural systems of focus” (concern 1) because a single anthropogenic systems can affect more than one natural system. Also, due to the different interpretations of environmental sustainability, indicators based on different choices of “goal” (concern 2) may complement each other, when such choices are made transparent to indicator users (**article V**). Diversity in choices across AESI can also be justified for concerns, for which the ranges of possible choices are, to some extent, restricted by the choice made for concern 1 or 2. For example, the choice of control variable (concern 3) is largely restricted by the choices of natural system and goal (concern 1 and 2). However, for many concerns, diversity in choices is problematic because it inhibits comparison of indicator scores for a studied anthropogenic system across AESI and thus creates uncertainty in the evaluation of whether a system is environmentally sustainable (see **article V**). For example diversity in choices for “Quantifying environmental interferences of studied system” (concern 8) and “Aggregation of indicator scores” (concern 11) is unwanted.

2.1.3 Improvement potentials of existing AESI

Existing AESI, including the three presented in Table 1, have made important contributions to evaluating the absolute environmental sustainability of anthropogenic systems. Collectively they do, however, suffer from a number of important shortcomings:

1. Their choice for concern 1 and 2 do not cover all relevant environmental issues. For example, toxic impacts from chemical pollutants are poorly represented (Galli et al., 2012; Persson et al., 2013).
2. They show large variations in choices made for concern 5 to 12, which contribute to uncertainties in indicator scores.
3. It is impractical for users to use different AESI complementarily, as these are typically made available through different software and rely on different data sources for quantifying environmental interferences (concern 8), some of which are very crude (Huijbregts et al., 2008; Kitzes et al., 2009).

As demonstrated below, these shortcomings can potentially be overcome by integrating carrying capacity as reference value for environmental sustainability in LCA.

2.2 INTRODUCTION TO LCA

2.2.1 BASIC CHARACTERISTICS

LCA is designed to quantify and compare environmental impacts from products and systems in a life cycle perspective (i.e. cradle to grave), typically covering abstraction of raw materials (water, timber, minerals, fossil fuels, etc.), processing, assembly, distribution, use and disposal (recycling, incineration, landfilling, etc.). The anthropogenic system(s) of study is in an LCA defined by a so-called functional unit, which specifies the function that the system is required to fulfil. A functional unit for an LCA comparing different paints could for instance be: "Coat and cover 1 m^2 outdoor wooden wall according to existing building standards in Sweden a red colour (RAL code 3020) for 10 years." A valid functional unit is important to ensure that compared systems are in fact fulfilling the same functions. After the functional unit has been defined, the life cycle inventory (LCI) of the studied system is modelled by linking a series of processes, each responsible of converting one or more product inputs (e.g. aluminium ingot) into one or more product outputs (e.g. aluminium packaging). Each process contains an inventory of stressors[3] (resource uses and emissions), generally expressed in kg. An LCI is usually constructed in dedicated software, such as GaBi, SimaPro or openLCA (GreenDelta, 2015; PRé, 2015; Thinkstep, 2015). This is done by combination of "foreground" inventory data, often in the form of product-specific data that are supplied by the commissioner of an LCA study, and "background" inventory data, e.g. related to the electricity grid mix of a specific nation, supplied by an LCI database such as EcoInvent (AE, 2015). An LCI covers an (ideally) complete list of stressors that take place over the product life cycle to fulfil the functional unit. The stressors are classified into mutually exclusive and collectively exhaustive impact categories as shown in Figure 2.2.

[3] Stressors are sometimes termed "elementary flows" in LCA.

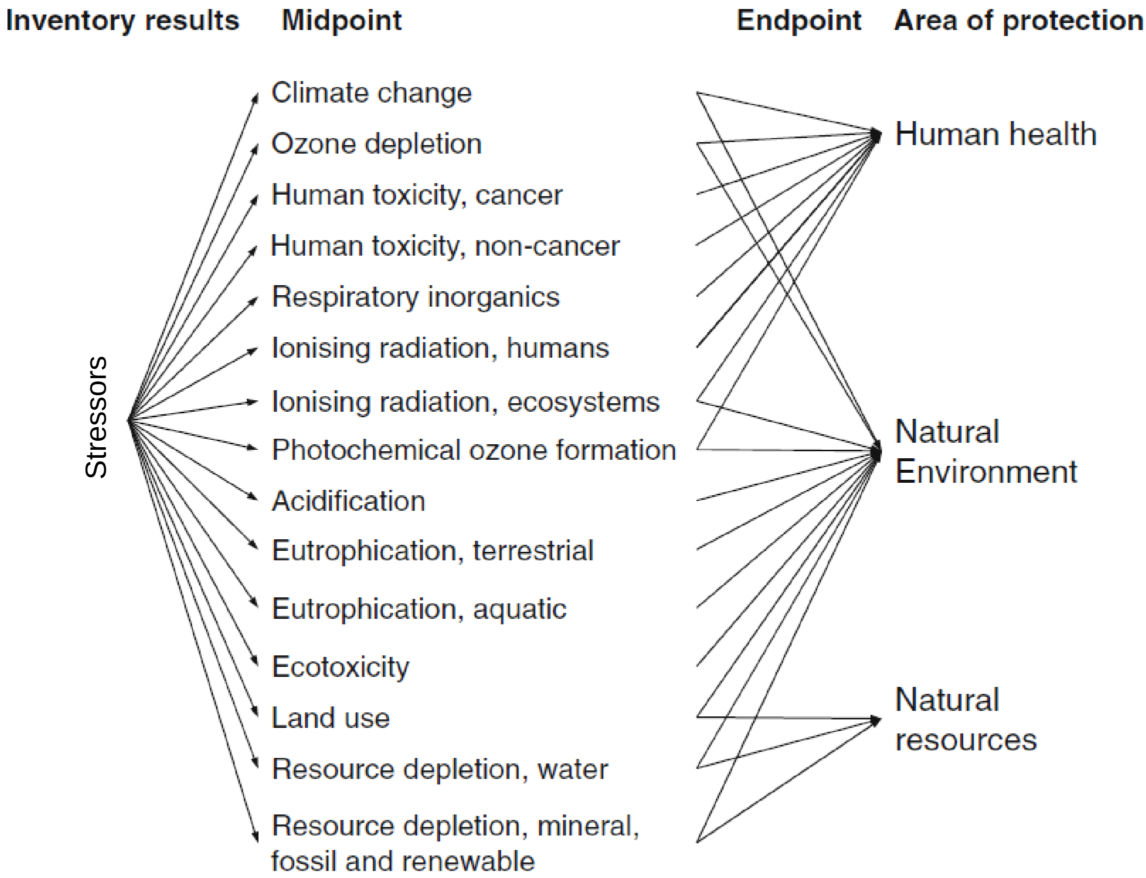

Figure 2.2: Framework of life cycle impact assessment linking stressors from the inventory results to indicator scores at midpoint level and endpoint level, here for 15 midpoint and endpoint impact categories belonging to 3 areas of protection. Adapted from EC (2010a).

2.2.2 LCA INDICATORS

LCA indicators can either be located at the "middle" (midpoint) or end (endpoint) of the impact pathway (elaborated below). Each impact category pertains to one or more of the three Areas of Protection "Human Health", "Natural environment", and "Natural resources." Several LCA indicators have been developed for each impact category (Hauschild et al., 2013). Indicator scores are calculated by multiplication of stressor quantities (e.g. kg or m^3) with characterisation factors (CF). CFs are derived from mathematical impact assessment models and are specific for each combination of stressor and emission compartment (air, water, soil and possibly sub-compartments) and sometimes also spatially explicit, e.g. for a continent, nation, water shed or grid cell. Spatially derived CFs for impact categories related to emissions can be mathematically expressed in a generic equation:

$$CF_{x,i,k} = \sum_j FF_{x,i,k,j} \cdot XF_{i,j} \cdot EF_{i,j} \quad (2.1)$$

Here CF is the characterisation factor for substance x emitted within spatial unit i into environmental compartment k (air, soil or water). The CF expresses the indicator score (e.g. in kg CO_2-eq. for climate change) per stressor (e.g. in kg). FF is a fate factor linking an emission of pollutant x within i into k to its fate, which is expressed as a change in substance concentration (a state indicator, see Figure 1.1) in the receiving spatial unit j. XF is an exposure factor which accounts for the fraction of pollutant x that species of concern in j are exposed to. EF is an effect factor, which calculates the effect increase on these species in j from an increased exposure of x. FF, XF and EF can be calculated using a marginal, linear or average approach (Hauschild and Huijbregts, 2015). In a marginal approach to calculating EF the marginal increase in effect from raising the state indicator marginally is calculated as the derivate in the point on the response curve (the shape of which varies between mechanisms) that correspond to the current effect level. In a linear approach to calculating EF, linearity is assumed between 0 and a chosen effect level. In an average approach to calculating EF, linearity is assumed between 0 effect (or a preferred maximum effect) and the current effect level. The average approach is rarely used and will not be referred to in the remainder of this thesis.

Figure 2.3a shows the elements of an LCA that are used as indicators for and mechanistic translators between the points of an impact pathway and shows conceptual response curves for the translation between points. Indicator scores can be expressed both at the so-called midpoint and endpoint levels. As illustrated for the impact category “freshwater eutrophication” in Figure 2.3b, the midpoint is ideally characterised by the earliest point of convergence of individual stressors in the impact pathway. By contrast, the endpoint is characterised by the end of the impact pathway and expresses damage to the Area of Protection, such as Natural Environment (Hauschild et al., 2013). Many midpoint indicators are located at the state point in the impact pathway and for these, no XF and EF are required to calculate characterisation factors, which therefore simplifies to FF, according to equation 2.1.

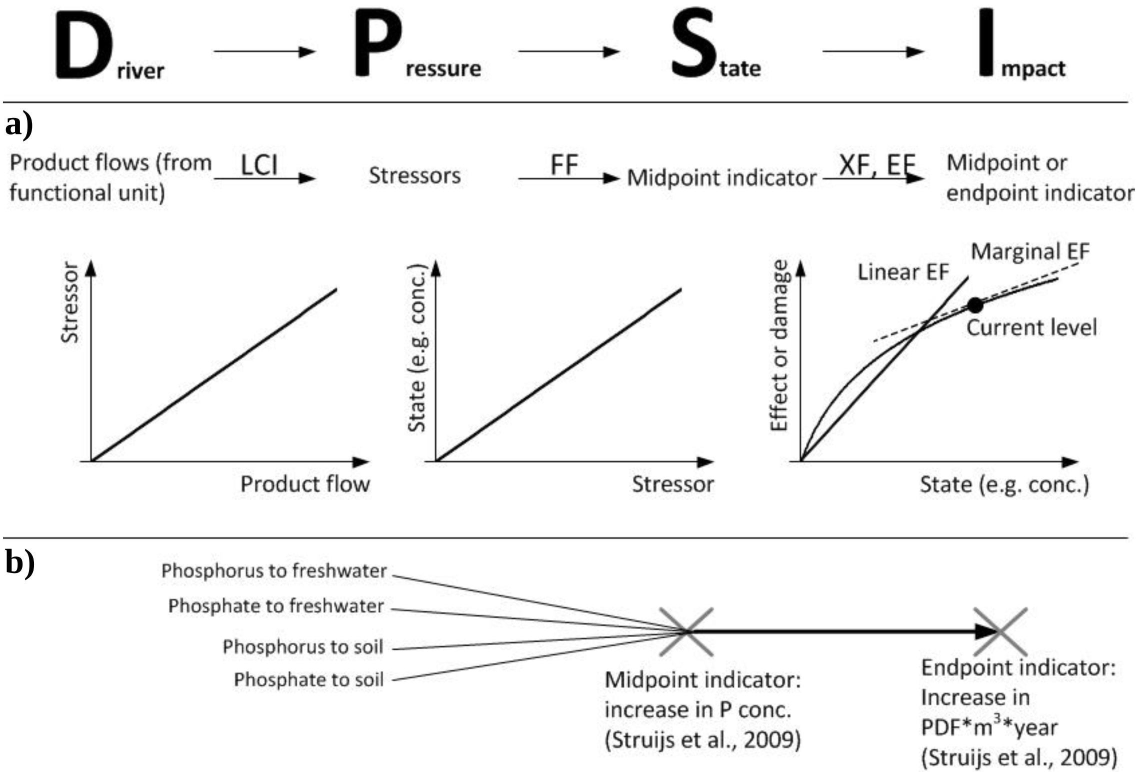

Figure 2.3: Elements of LCA placed in the DPSIR impact pathway framework (Smeets and Weterings, 1999) (response point not included). Figure 2.3a maps elements of LCA and their interactions. Figure 2.3b locates the midpoint and endpoint for the example of freshwater eutrophication. Based on article II.

To facilitate the interpretation of indicator scores, which are typically expressed for between 10 and 20 impact categories in different metrics, indicator scores are often normalised into person years (sometimes termed person equivalents) of environmental interference, by using normalisation references for the average annual environmental interference of a person in a specified region and time. Since the normalisation step does not reflect the severity of environmental interferences, a weighting step can subsequently be applied in which different weights are assigned to indicator scores of different impact categories. An ISO standard (14040 and 14044 (ISO, 2006a, 2006b)) exists for LCA and the European Commission has published recommendations on the different steps of an LCA (EC, 2010b).

2.3 Potential use of LCA in AESI

LCA indicators, including normalisation references and weighting factors, may be characterised using the concerns of Table 2.1 that are unrelated to thresholds or carrying capacity (concern 1, 2, 3, 8, 9, 10 and 11). As for AESI, variations in choices made for different LCA indicators are for some concerns desirable when such variation ensures that different indicators complement each other in representing the complexity of anthropogenic interferences with the environment. For other concerns, variations in choices made for different LCA indicators are unwanted. The LCA research community is having a strong focus on harmonization and consensus building with the aim of reducing uncertainties by means of reducing the variations of choices made for different LCA indicators. This has lead to firm standards and guidelines for calculation of LCIs and design of impact pathway models (concern 8) and aggregation of indicator scores (concern 11) (see e.g. (Hauschild et al., 2011).

From the characteristics of LCA outlined above, it can be argued that LCA can be used to potentially overcome the three shortcomings listed in Chapter 2.1.3:

1. LCA indicators collectively cover all relevant environmental issues.
2. Concerns 8 to 11 of Table 2.1 are dealt with in much the same way for all LCA indicators, because these have been developed to fit a single environmental assessment framework (LCA) with associated standards and guidelines. This characteristic means that uncertainties in indicator scores are minimized.
3. The existence of LCA software and LCI databases covering all impact categories means that indicator scores can be calculated for all impact categories simultaneously.

2.4 Integrating carrying capacity in LCA

Two principal approaches to integrating carrying capacity in LCA as sustainability reference value are here proposed. The first approach is to design carrying capacity based normalisation references to normalise LCA indicator scores to express occupation of carrying capacity. The second approach is to develop new characterisation factors for all impact categories which allows expressing spatially resolved occupations of carrying capacity directly in the indicator score.

2.4.1 Normalisation references based on thresholds at midpoint

Article II presents the development of capacity based normalisation references (NR) for Europe and a global average for the LCA midpoint indicators that link to the Area of Protection “Natural environment” (see Figure 2.2). These are climate change, ozone depletion, photochemical ozone formation, terrestrial acidification, terrestrial eutrophication, freshwater eutrophication, marine eutrophication, ecotoxicity, land use and water depletion.[4] Carrying capacities were based on thresholds at midpoint, identified from the literature for each impact category. The thresholds all reflected a goal (see concern 2 in Table 2.1) of protecting structure or functioning and pertained to one of the four threshold types, or variants of these, of Figure 2.1. The thresholds were in **article II** translated through the impact pathway to carrying capacities using components of the underlying life cycle impact assessment models of LCA indicators (see Figure 2.3a). For example, fate factors were used to translate thresholds at the state point to carrying capacities at the pressure point to be used in NRs compatible with LCA indicators of pressure.

Several LCIA models exist for the characterisation within each impact category. When possible, the recommendations for best existing practice by Hauschild et al. (2013) was followed when choosing the characterisation model and factors with which NR should be compatible. Exceptions were made for recommended models based on a marginal approach, which were replaced by models using a linear approach because the calculation of carrying capacity, as interpreted in this thesis, should not depend on current levels of environmental interferences (see **article II**). This procedure led to the replacement of ILCD recommended models for terrestrial acidification, terrestrial eutrophication, land use and water depletion by models using a linear approach.

NR is calculated as the carrying capacity (CC, indicator score per year) for impact category i in region j, divided by the population in the region (P):

$$NR_{i,j} = \frac{CC_{i,j}}{P_j} \tag{2.2}$$

When dividing characterised LCIA results by NR they are converted into normalised results expressed in units of person years (sometimes termed person equivalents). Here 1 person year can be interpreted as the environmental interference corresponding to the annual personal share of the carrying capacity for impact category i. Table 2.2 presents the developed NR for Europe and the global average along with corresponding “traditional” normalisation references based on society’s background environmental interferences (NR’).

[4] The impact category accounting for ionizing radiation effects on the natural environment was excluded since the recommended LCIA model was classified as interim by Hauschild et al. (2013).

Table 2.2: Developed global and European normalisation references based on carrying capacity (NR), comparison with traditional normalisation references (NR') and across scale. Bold values indicate that NR'/NR fractions are above 1. Italics CF references mean compatibility with characterisation methods recommended by Hauschild et al. (2013). Note that NR_{Europe} denotes the European continent, while NR'_{Europe} denotes nations of EU27. Based on article II.

Impact category	**NR_{Global}** (per person year)	$\frac{NR'_{Global}}{NR_{Global}}$	**NR_{Europe}** (per person year)	$\frac{NR'_{Europe}}{NR_{Europe}}$	$\frac{NR_{Global}}{NR_{Europe}}$	**CF compatibility**	**Threshold**
Climate change	985 kg CO_2-eq.	**8.2**	985 kg CO_2-eq.	**9.4**	1	*GWP100 (CO_2-eq)* (Forster et al., 2007).	Temperature increase of 2°.
	522 kg CO_2-eq.	**15**	522 kg CO_2-eq.	**18**			Radioactive forcing increase of $1 W \cdot m^{-2}$.
Ozone depletion	0.078 kg CFC-11-eq.	0.53	0.078 kg CFC-11-eq.	0.28	1	*ODP (Montzka and Fraser, 1999).*	7.5% decrease in average ozone conc.
Photo-chemical ozone formation	3.8 kg NMVOC-eq.	**15**	2.5 kg NMVOC-eq.	**13**	1.6	*Tropospheric ozone concentration Increase (van Zelm et al., 2008).*	Tropospheric ozone concentration of 3 ppm·hour AOT40.
Terrestrial acidification	$2.3 \cdot 10^3$ mole H+-eq.	0.34	$1.4 \cdot 10^3$ mole H+-eq.	0.53	1.7	OT method of (Posch et al., 2008).	Deposition of 1170 and 1100 mole H^+ $eq \cdot ha^{-1-} \cdot year^{-1}$ globally and for the EU.
Terrestrial eutrophication	$2.8 \cdot 10^3$ mole N-eq.	0.13	$1.8 \cdot 10^3$ mole N-eq.	0.30	1.5	OT method of (Posch et al., 2008)	Deposition of 1340 and 1390 mole N $eq \cdot ha^{-1} \cdot year^{-1}$ globally and for the EU.
Freshwater eutrophication	0.84 kg P-eq.	0.74	0.46 kg P-eq.	**3.22**	1.8	*P concentration increase (Struijs et al., 2009).*	P concentration of 0.3mg/L.
Marine eutrophication	29 kg N-eq.	0.32	31 kg N-eq.	0.55	0.95	*N concentration increase (Struijs et al., 2009).*	N concentration of 1.75 mg/L.

Freshwater ecotoxicity	$1.9 \cdot 10^4$ [PAF]$\cdot$m$^3\cdot$day.	0.036	$1.0 \cdot 10^4$ [PAF]$\cdot$m$^3\cdot$day.	0.85	1.8	*CTU (Rosenbaum et al., 2008).*	HC5(NOEC).
Land use, soil erosion	1.8 tons eroded soil.	**4.9**	1.2 tons eroded soil.	**9.3**	1.6	Soil erosion (Saad et al., 2013), land occupation CFs only.	Tolerable soil erosion of 0.85 tons$\cdot$ha$^{-1}\cdot$year^{-1}.
Land use, biodiversity	$1.5 \cdot 10^4$ m$^2\cdot$year.	0.42	$9.5 \cdot 10^3$ m$^2\cdot$year.	0.79	1.6	LCI data, land occupation only.	31% conserved land area.
Water depletion	306 m^3.	**1.3**	490 m^3.	0.52	0.63	LCI data classified as blue water consumption.	Conservation of 57% of river flows for aquatic and 30% for terrestrial ecosystems.

NR'/NR ratios above 1 mean that current levels of environmental interferences exceed carrying capacity and therefore that normalised indicator scores will become higher when a traditional normalisation reference is replaced by a carrying capacity-based one. This is the case for climate change (both thresholds), photochemical ozone formation and land use (soil erosion) both at the global and European scale, for freshwater eutrophication at the European scale and for water depletion at the global scale. The NR'/NR ratios for the remaining impact categories are all below 1 and normalised indicator scores of these categories thus become smaller when replacing traditional normalisation references with carrying capacity based ones. When comparing across scale (column 6 in Table 1) it can be seen that for all impact categories, except water depletion and marine eutrophication, NR_{Europe} is smaller than NR_{Global}, which is mainly due to Europe's relatively high population density.

The interpretation of results for climate change, photochemical ozone formation, land use and water depletion is that humanity is globally environmentally unsustainable according to the calculated carrying capacities (and thus chosen thresholds). Global degrees of environmental unsustainability are seemingly greatest for climate change (when carrying capacity is based on the 1 W/m^2 threshold) and photochemical ozone formation for which environmental interferences need to decrease by a factor of 15, compared to environmental interferences in the year 2010 and 2000 respectively, to reach sustainable levels. For the remaining impact categories current environmental interferences appear environmentally sustainable globally because NR'_{Global}/NR_{Global} is below 1. From this, it can appear that human societies are generally environmentally sustainable with respect to these impact categories. However, due to the spatially heterogenic nature of many environmental interferences, regional and local carrying capacities may well be exceeded although global carrying capacities are not. The coarse spatial representation of normalisation references should therefore be kept in mind when using them.

2.4.2 NORMALISATION REFERENCES BASED ON THRESHOLDS AT ENDPOINT

The midpoint references presented in Chapter 2.4.1 were based on carrying capacities calculated from science-based thresholds at midpoint. These thresholds are generally accepted as quality targets for environmental management and have been adopted in policies and regulations. However, there is a risk that the thresholds represent different levels of ecosystem protection, because of differences in choices of aim (concern 2, i.e. what structure or functioning to protect?), control variable (concern 3, i.e. how to measure the level of protection?) threshold basis (concern 4, e.g. which of the four threshold types to assume when the response mechanism is poorly understood?) and threshold value (concern 5, e.g. lower, medium or higher value in uncertainty range?). An alternative strategy to calculating carrying capacities is to start with a common goal (concern 2) for all impact categories, from which to choose a common control variable at endpoint (concern 3), followed by the choice of a common threshold basis and value (concern 4 and 5). Carrying capacity based normalisation references (e.g. global or European averages) could then be calculated at either midpoint or endpoint from such a common threshold value.

There seem to be no obvious candidates for a common goal. In other words there is little agreement on the environmental structure and functioning that should be protected as a precondition for environmental sustainability (defined by Goodland (1995)) as ”…seek[ing] to improve human welfare by protecting the sources of raw materials used for human needs and ensuring that the sinks for human wastes are not exceeded, in order to prevent harm to humans.”) Examples of environmental management goals are 1) ”good ecological status”, as e.g. adopted by the EU Water Framework Directive (EC, 2000), 2) protection of ecosystem services, as encouraged by the Millennium Ecosystem Assessment (WRI, 2005), and 3) resilient socio-ecological systems, which is a goal that has recently risen on the political agenda (UNDP, 2014). Common for these goals is that there is no obvious single comprehensive control variable for measuring achievements towards them.

Despite of the difficulties of defining a common goal for all impact categories, there is really only one candidate for a control variable at endpoint, since all LCA endpoint indicators for the Natural environment Area of Protection quantify impacts on ecosystems in a PDF-related metric, having units of [PDF]m^2·year or [PDF]m^3·years (Hauschild et al., 2013). PDF is an acronym for Potentially Disappeared Fraction of species. The existence of a common endpoint indicator is one of the strengths of endpoint modelling, since it allows for a direct comparison of indicator scores, contrary to midpoint modelling. The modelling of damage to the Natural environment in LCA is thus based on changes in ecosystem structure, i.e. damage to the species of an ecosystem.

With regards to a common threshold basis and value (concern 4 and 5), there seems to be no scientific consensus on an environmentally sustainable PDF across all types of impacts and natural systems (Mace et al., 2014). One reason for this is the lack of consensus on the goal of protecting natural system structure or functioning (see above). A pragmatic approach for, nevertheless, obtaining a common endpoint threshold value in a PDF-related metric is to base the endpoint threshold on the impact category specific midpoint threshold applied in Chapter 2.4.1 that is closest to endpoint. Proximity to endpoint is desirable to have the smallest possible uncertainties in the translation from midpoint to endpoint. Such a threshold close to endpoint can be inferred from the HC5(NOEC) concept, which was applied to calculate the carrying capacity of freshwater ecosystems with respect to the impact category ecotoxicity (**article I** and **II**). HC5(NOEC) is the concentration (e.g. in µg/l) of a substance at which maximum 5% of species are affected above their NOEC (no observable effect concentration), which is the highest concentration tested where no statistically significant chronic effects are observed (EC, 2003) (**article I**). HC5(NOEC) is commonly used as an indicator of “good ecological status”, for example within the EU Water Framework Directive (EC, 2000). From this ecotoxicity threshold a common threshold in a PDF related metric may be calculated in two steps: 1) By using the translation approach presented in **article I,** the concentration threshold HC5(NOEC) can be expressed as the fraction of species that is potentially affected above the concentration at which 50% of a species’ population displays an acute effect (PAF($EC50_{acute}$), in short). 2) PDF can be assumed to be equal to PAF($EC50_{acute}$), as is often assumed in LCA impact assessment models of ecotoxicity (EC, 2010a).[5]

Once this overarching threshold value, expressed in PDF, has been calculated, carrying capacities for the different impact categories needs to be expressed in units compatible with a PDF-related endpoint metric. Carrying capacities in [PDF]m^2 or [PDF]m^3 may easily be calculated for each impact category by multiplying the overarching threshold value in PDF by the environmental area or volume considered by the relevant impact assessment model. Normalisation references at endpoint can then be calculated following equation 2.2 (i.e. division with population). Normalisation references at midpoint based on the overarching threshold at endpoint can also be calculated by applying the substance generic conversion factor between midpoint and endpoint for each impact category.[6]

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[5] This assumption is based on the empirical observation that species tend to eventually disappear when half of their population displays an acute effect (Snell and Serra, 2000). For a review of alternative conversion approaches see Larsen and Hauschild (2007).

[6] Note that this conversion factor may depend on the spatial coverage of the normalisation references and therefore e.g. vary between European and global average carrying capacities.

2.4.3 Spatially differentiated characterisation factors

In **article IV** a generic mathematical equation for integrating carrying capacity in any spatially resolved emissions based characterisation model is presented. This equation is simply the content within the summation of equation 2.1 divided by spatially resolved carrying capacity:

$$CF_{x,i,k} = \sum_j \frac{FF_{x,i,k,j} \cdot XF_{i,j} \cdot EF_{i,j}}{CC_j} \tag{2.3}$$

Here CF (ha·year·$kg_{emitted}^{-1}$) is the characterisation factor for substance x emitted within spatial unit i into environmental compartment k (air, soil or water). CC is the carrying capacity in j expressed in a metric that must be aligned with the metrics of FF, XF and EF, because the metric of the CF must be ha·year·$kg_{emitted}^{-1}$. Note that equation 2.3 applies to indicators expressed at the damage point in the impact pathway. If indicator scores are expressed at the exposure point or the state point, the denominator should only contain FF and XF or only FF. When multiplying CFs with an LCI emission the indicator score is expressing the carrying capacity occupation in a unit of ha·year, which resembles that of the ecological footprint method (Borucke et al., 2013) and is designed to be compared to the availability of land or water. Note that for some impact categories it may be more convenient to express the occupation of carrying capacity in a unit of m^3·year, as was done in the proposal of a chemical footprint indicator for freshwater ecotoxicity in **article I**. For reasons given in Chapter 2.4.1, the proposed method is only compatible with indicators for which FF, XF or EF are of a linear nature.

In **article IV** equation 2.3 is demonstrated on the impact category terrestrial acidification, for which the spatial derivation was based on the only existing global deposition model presented in Roy et al. (2012) and having a 2.0°·2.5° resolution (i.e. composed of 13,104 grid cells). Two complementary pH-related thresholds are chosen, inspired by the critical loads concept (Spranger et al., 2004): a pH decrease of 0.25 compared to natural levels and an absolute pH value of 4.2. These thresholds reflected, respectively, the two complementary goals (concern 2) of not weakening soil buffer capacities and avoiding the mobilization of aluminium (III).The carrying capacity is expressed as a critical deposition of acidifying compounds (eq.·ha^{-1}·$year^{-1}$, where 1eq. refers to 1 mol H^+-eq.). The carrying capacity was derived for 99,515 spatial units, covering the global terrestrial area, from pH simulations in the geochemical steady-state model PROFILE (Warfvinge and Sverdrup, 1992). For each spatial unit the lower of the two calculated carrying capacities, which reflected the two complementary pH-related thresholds, was chosen. An average carrying capacity was then calculated for each grid cell of the deposition model of Roy et al. (2012) (13,104 grid cells), weighted by the area of the each of the spatial units within. CFs were then calculated according to the following simplification of equation 1 (excluding XF and EF because CC is expressed at the state point) using atmospheric fate factors (FF, $keq_{deposited} \cdot kg_{emitted}^{-1}$) of Roy et al. (2012):

$$CF_{x,i} = \sum_j \frac{FF_{x,i,j}}{CC_j} \tag{2.4}$$

Figure 2.4 shows the distribution of CFs for all global locations of SO_X (a common reference substance for terrestrial acidification).

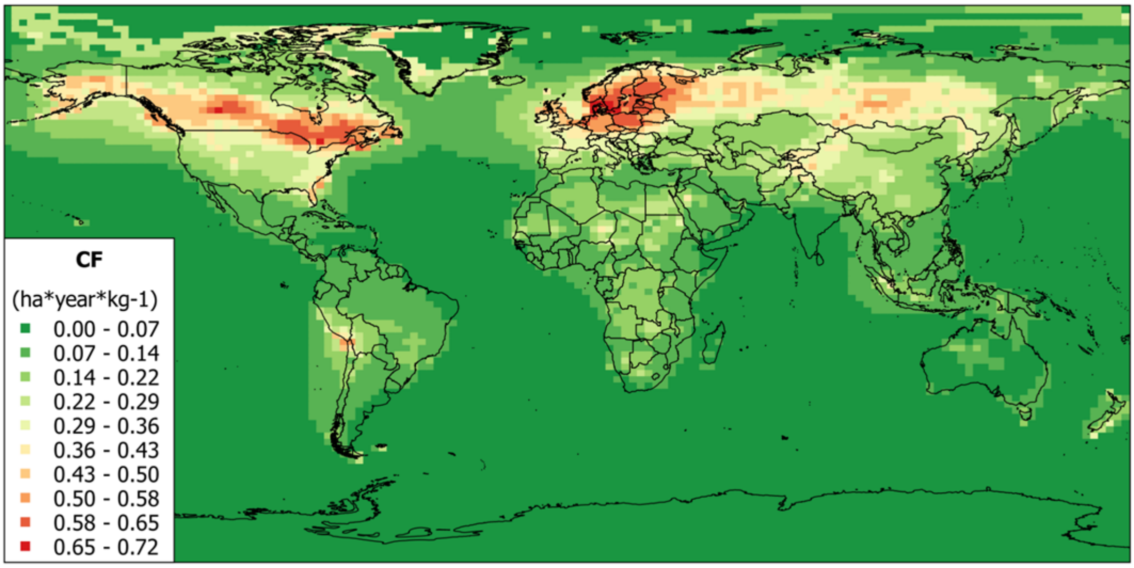

Figure 2.4: Global distribution of CFs for SO_X. Based on article IV.

CFs are high for emissions locations from which SO_X deposits on grid cells with low carrying capacity. Such emissions locations include parts of Russia, Northern Europe, Canada and Alaska. CFs for SO_X range from less than 0.0054 $ha \cdot year \cdot kg^{-1}$ (10^{th} percentile) to more than 0.41 $ha \cdot year \cdot kg^{-1}$ (90^{th} percentile) with a median value of 0.16 $ha \cdot year \cdot kg^{-1}$ (disregarding CFs for locations in the open sea). This wide range shows the importance of spatial differentiation when integrating carrying capacity in LCA.

2.5 CONCLUSION AND OUTLOOK

In this chapter the first main research question was answered: “How can the carrying capacity concept be operationalized as sustainability reference value in environmental indicators in general and in LCA indicators specifically?”

It was demonstrated how the carrying capacity concept can be used as environmental sustainability reference in AESI to express environmental interferences as occupation of carrying capacity. It was argued that existing AESI suffers from a number of weaknesses. Weaknesses that LCA can potentially reduce when modified with carrying capacity as environmental sustainability reference. Carrying capacity can either be integrated in normalisation references or characterisation factors. In both of these integrations carrying capacities may be derived either from impact category specific thresholds at midpoint or from a common threshold at endpoint. Advantages and disadvantages of each approach are summarized in Table 2.

Table 2.3: Advantages and disadvantages of different means of carrying capacity integration in LCA and of LCA-based AESI compared to standard LCA.

	Advantages	**Disadvantages**
Normalisation references	• Relatively few resources required, as only one reference must be calculated per impact category. • Potential high acceptance by users, because references are compatible with well-known and accepted characterisation models.	• Spatial variations in stressors, fate, exposure, effect, thresholds and, hence, carrying capacity can only be covered to a very limited extent, because normalisation references are, by nature, global or regional. • Relatively few resources required, as only one reference must be calculated per impact category.
Characterisation factors	• Potentially captures spatial variations in stressors, fate, exposure, effect, thresholds and, hence, carrying capacity.	• New sets of CFs must be calculated for every impact category, which is time demanding. • The user acceptance of new sets of CFs may take a long time.
Thresholds at midpoint	• Carrying capacities based on accepted scientific thresholds. • Relatively low uncertainty in translation of thresholds to carrying capacities.	• Thresholds may correspond to different levels of ecosystem protection.
Common threshold at endpoint	• Carrying capacities based on a common threshold that reflects a consistent level of species protection (a proxy for ecosystem protection). • Carrying capacities for all impact categories easy to calculate from a common threshold.	• Relatively high uncertainty associated with endpoint modelling. • Calculation of common threshold affected by uncertainties in conversion from HC5(NOEC) to PDF.
LCA-based AESI versus standard LCA	• Allows for evaluating the environmental sustainability of anthropogenic systems in absolute terms. • Occupation of carrying capacity may be an easier metric to communicate to non-experts than metrics of current LCA indicators.	• Calculated carrying capacity values are potentially sensitive to normative choices of (environmental sustainability) goal, threshold basis and threshold value. • Incompatible with marginal LCA indicators. • Only compatible with impact categories linked to the Area of Protection Natural environment.

It can be seen that the advantages and disadvantages of integrating carrying capacity in normalisation references versus doing it in characterisation factors largely mirrors each other. There is thus a trade-off between ease of implementation and expected acceptance by users on one hand and accurate representation of spatial variations on the other. Note that the representation of spatial variation will also depend on the spatial details of LCIs of studied anthropogenic systems (see Chapter 5 for an elaboration on this point). A similar trade-off exists between the approaches of calculating carrying capacities from impact category specific science-based thresholds at midpoint versus calculating carrying capacities from a common threshold at endpoint. Calculated carrying capacities of the former approach have relatively low uncertainties, but risk corresponding to different levels of species protection, while calculated carrying capacities of the latter approach reflect a common (although somewhat arbitrarily chosen) level of species protection, but have relatively high uncertainties.

When comparing the LCA indicators modified to AESI, proposed in this chapter, to standard LCA indicators, a number of advantages and disadvantages can be identified: The main advantage of LCA-based AESI over standard LCA is its ability to evaluate the environmental sustainability of an anthropogenic systems in an absolute sense, in other words to, in principle, reveal not just whether system X has a lower environmental interference than a reference system, but also whether system X can be considered environmentally sustainable. In addition, the expression of environmental interferences as occupation of carrying capacity, either in the form of person years or an area, may have communication benefits compared to the somewhat abstract metrics of standard LCA, such as PAF (integrated over space and time) for the impact category ecotoxicity or ozone formation potential for the impact category photo-chemical ozone formation.

A number of disadvantages of LCA-based AESI versus standard LCA can also be identified: carrying capacity values, and thus indicator scores, largely depend on the inherently normative choice of the concerns goal, threshold basis and threshold value. Also, the concept of carrying capacity, as used in this thesis is inherently incompatible with marginal LCA indicators (see **article II** for an explanation). This is unfortunate seeing as how marginal LCA indicators are quite common. For example, ILCD recommended marginal indicators for 4 of the 10 midpoint indicators that link to the Area of Protection Natural environment. The incompatibility may, however, be circumvented by calculating stressor-generic factors that convert between marginal and linear CFs (if linear CFs are available) for each impact category. The carrying capacity based normalisation references can then be multiplied by these factors to make the references applicable to indicators scores based on marginal CFs. Note that for some impact categories stressor-specific conversion factor can vary a lot. In these cases the application of a stressor-generic conversion factor can entail a substantial increase in the uncertainty of normalised indicators scores. Another disadvantage of LCA-based AESI versus standard LCA is that carrying capacity does not apply to the areas of protection Human health and Natural resources. Other sustainability references may however complement the use of carrying capacity to potentially complete the coverage of LCA impact categories by AESI (see Chapter 5).

Since there are both inherent advantages and disadvantages of using AESI-LCA over using RESI-LCA, it can be concluded that the two approaches may complement each other in the study of environmental interferences of anthropogenic systems.

3 CARRYING CAPACITY ENTITLEMENT

In this chapter the second main research question is answered: “How can the carrying capacity entitlement of individual anthropogenic systems be calculated, how applicable are different valuation principles to calculating entitlements and how sensitive is this calculation to choice of valuation principle?”

3.1 CALCULATION FRAMEWORK

Natural science can conclude that humanity as a whole is environmentally unsustainable when it comes to emissions of greenhouse gases (IPCC 2013; **article II**), at least when basing the sustainability criteria on not exceeding a threshold of 2°C above pre-industrial temperatures. Likewise it may be concluded that current loads of nutrients to the Baltic Sea is unsustainable with regards to the negative effects of eutrophication (**article V**). The environmental interferences on the global climate and the Baltic Sea are not caused by a single anthropogenic system governed by a single absolute authority, but by a myriad a systems governed by decisions of individual consumers, and by private and public organisations of various sizes. How can the responsibility of each system to “do its share” in maintaining a sustainable whole be quantified? In other words, how can the entitlement to carrying capacity be granted to different anthropogenic systems that intervene with the same ecosystem(s)? Below, it is demonstrated that carrying capacity entitlement depends on the perceived value of a studied system relative to “competing systems” (existing or potential) that rely on occupying parts of the same carrying capacity for their functioning.

3.1.1 IDENTIFY COMPETING SYSTEMS

The calculation of entitlement depends, in part, on the number of systems competing for the same carrying capacity. Consider, for example, a hypothetical company emitting nutrients (nitrogen and phosphorous) to a local lake. Clearly, the share of that lake’s carrying capacity towards nutrients that can be entitled to the company must depend on the number of other anthropogenic systems relying on emitting nutrients into the same lake for their functioning. The identification of competing systems for the calculation of carrying capacity entitlement requires a spatial assessment. Below an ideal and simplified approach to such an assessment is outlined, based on **article IV**, and followed by the outlining of a pseudo-spatial identification of competing systems for spatially generic LCA indicators.

Ideally competing systems would be identified by combining a source-receptor fate model with a spatially differentiated emission inventory covering all anthropogenic systems of society in a chosen reference year: The fate model would first identify the spatial units affected by emissions of the studied system. The fate model would then identify within the comprehensive emission inventory all the anthropogenic systems that affect the spatial units previously identified. These systems would be labelled competing systems because they rely on occupying parts of the same carrying capacity as the studied system for their functioning. Note that the group of competing systems is potentially unique for each affected spatial unit (of which there may be thousands). This is in many cases impractical to operate with and therefore three simplifications may be introduced: 1) a cut-off criterion can be established whereby only spatial units receiving above a specified share of emissions from the studied system (e.g. 0.1%) are considered. The territory of these spatial units are termed $T_{affected}$ and its area is termed $A_{affected}$, 2) all emissions that occur within $T_{affected}$ can, in the entitlement part of the AESI, be assumed to occur in the spatial unit where the emission from the studied system occurs and thus can be assumed to have the same fate, 3) it can be assumed that no emissions within $T_{affected}$ leave $T_{affected}$ and that no emissions from outside enters. These three simplifications are visually presented in Figure 3.1.

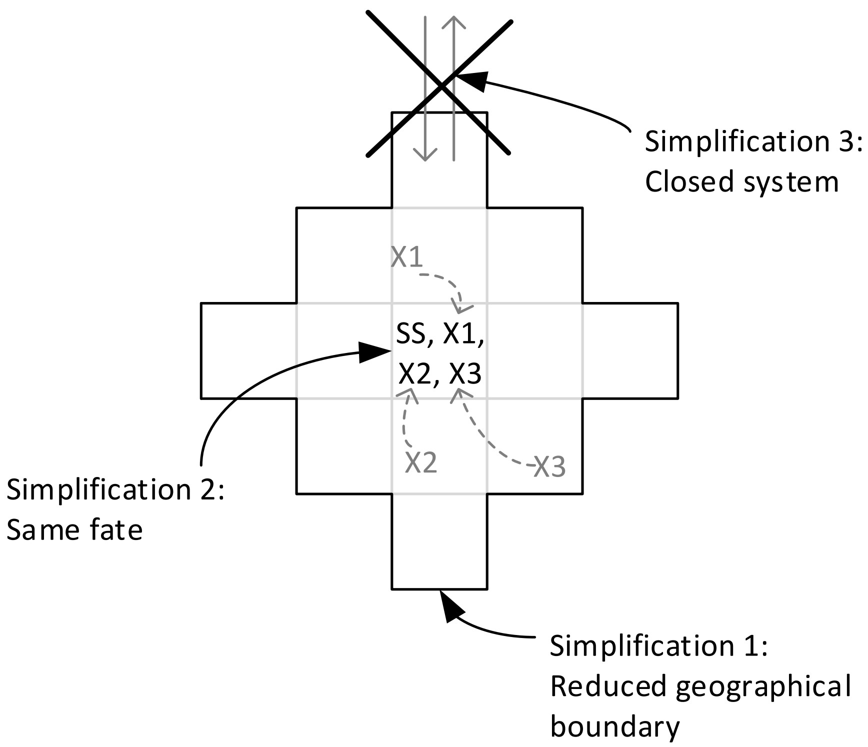

Figure 3.1: Illustration of three simplifications for identifying competing systems (X1-X3) of a studied system (SS) located in the middle grid cell and affecting 13 grid cells above an arbitrary emission distribution threshold. These 13 grid cells make up $T_{affected}$ and have the area $A_{affected}$. The dotted arrows indicate a change in location of X1, X2 and X3.

The consequence of the simplifications is that only one carrying capacity entitlement needs to be calculated for each emission location of a studied anthropogenic system and that the group of competing systems of all anthropogenic systems within $T_{affected}$ is the same. The simplifications can be defended in situations where potential competing systems are rather homogenously distributed in space and have emissions of similar magnitude. When this is not the case it may be more appropriate to follow the ideal approach outlined above to identifying competing systems. Note that when the studied anthropogenic system is of a life cycle nature (see Figure 1.2) the large number of emission locations could make the task of identifying competing systems very time demanding, even if the simplified approach is followed.

The existence of a spatially derived impact assessment model was assumed in the proposed ideal and simplified approaches to identifying competing systems. For some impact categories only spatially generic models are available, even when environmental interferences are local. This is, for example, the case for the impact category freshwater ecotoxicity, for which the scientific consensus model for impact assessment is currently the spatially generic[7] USEtox model (Hauschild et al., 2013; Rosenbaum et al., 2008). For spatially generic impact assessment models, competing systems may be identified from the typical travel distance of emissions that have the largest contribution to carrying capacity occupation. For example, these emissions with largest contributions were identified as metals (zinc and copper) emitted to air and freshwater in a case study of **article I**, which aimed to cover all chemical emissions within Europe. These zinc and copper emissions are known to have relatively short travel distances. Therefore competing systems may in such cases be pragmatically defined as all anthropogenic emitters of relevant compounds that are located within a typical travel distance of the studied system (a similar approach was used in **article I** to estimating the locations of carrying capacity occupied by a given emission).

3.1.2 Calculate the perceived relative value of a studied system

Having identified the competing systems, the perceived value of the studied system relative to that of competing systems must be considered. This relative value can be expressed as a value factor (VF) that numerically can be between 0 and 1. Different approaches from the fields of economy, sociology (including political science), psychology and philosophy can be taken to calculating VF. For example, Starkey (2008) evaluates whether different libertarian schools find the con-

[7] In USEtox a continental scale, reflecting an archetypical continent, is nested within a global scale. Therefore, while the model does distinguish impacts that happen within a continent from global impacts, it is not truly spatially resolved.

cept of equal per capita entitlement to greenhouse gas emissions just. Table 3.1 presents an illustrative list of valuation principles, the anthropogenic systems of study they apply to and how the VF, in principle, is calculated in each case. The list may not be exhaustive and its valuation principles are not necessarily mutually exclusive. Note that the task here is not to judge what is “fair” and “unfair” about each valuation principles, but rather to outline the principles so that their applicability in actual environmental assessments can be evaluated and the sensitivity of calculated carrying capacity entitlement to choice of principle can be analysed.

Table 3.1: List of valuation principles, their underlying perspectives, the anthropogenic systems they are compatible with (see Table 1.2) and value factor calculation principles.

Valuation principle	**Perspective**	**Anthropogenic systems of study**	**Value factor calculation**
Area (Borucke et al., 2013)	The value of a system is proportional to the physical area it takes up.	Territorial: nations.	Proportional to its share of total area (terrestrial or aquatic, depending on impact category).
Gross output (Krabbe et al., 2015)	The value of a system is proportional to its production volumes.	Territorial: organisations.	Proportional to its share of the total produced volume of a chosen reference product.*
Relevance for meeting human needs	The value of a system is proportional to the extent to which it contributes to meeting (essential) human needs, such as food and shelter.	Life cycle: single functions.	Proportional to its share of the total meeting of (essential) human needs, mediated by products and services.
Population (Starkey, 2008)	All humans are equally valuable.	Life cycle: person.	Total carrying capacity divided by population
Contribution to gross domestic product (GDP)	The value of a system is proportional to its contribution to GDP.	All.	Proportional to its share of GDP measured as value-added, income or final expenditure, depending on the anthropogenic system of study (Callen, 2012).
Grandfathering (Starkey, 2008)	Past environmental interferences of a system are deemed legitimate and all systems should thus play a proportionate part in achieving environmental sustainability.	All.	Proportional to its share of total environmental interferences in a chosen reference year.

*Note that reference products are industry-specific. Therefore another valuation principle must be used to calculate the carrying capacity entitlement of different industries. For this calculation Krabbe et al. (2015) refers to the 2DS GHG reduction scenario of the International Energy Agency which, presumably, is based on a combination of different valuation principles.

In some cases it can be desirable to adjust the VFs calculated from the valuation principles of table 3.1 (and others). Such an adjustment could reflect anticipated future changes in indicator scores due to structural changes in society. For example, changes in energy mix may reduce indicator scores of energy intensive anthropogenic systems and this could be a reason to lower the initial VFs of such systems. Adjustments could also reflect the creation of markets for trading carrying capacity entitlements, much like past and existing cap-and-trade-systems for pollutants such as CO_2. A third reason for adjustments of initially calculated VFs is that they in some cases can be considered unfair. For example, Starkey (2008) argues that the population valuation principle[8] may need to be complemented by adjustments based on the burden of historical emissions (or resource uses) and/or based on spatial variations in conditions that affect fossil fuel consumption, such as climate (affecting the need for heating and cooling) and transportation needs (generally higher for people living in scarcely populated areas). These adjustments are not dealt with in the remainder of this thesis, but they clearly deserve future academic attention in the context of AESI.

Continuing the hypothetical study from above, it can be demonstrated that the share of the lake's carrying capacity entitled to the studied company not only depends on the number of competing systems, but also on the valuation principle applied (see Table 3.1). Consider the following additional information: 1) There is just one competing system. 2) The studied system is a slaughterhouse, meaning that it produces a low-cost product associated with high levels of nutrient emissions. 3) The competing system is a company using refined metals to produce jewelleries, which is a high cost product associated with low levels of nutrient emissions. 4) In a reference year the studied system (the slaughterhouse) contributed to 90% of the total nutrient emissions to the lake, but only to 10% of the combined contribution to GDP of the two companies. The choice of valuation principle thus has a potential high influence of the carrying capacity entitlement: 90% of the value (VF = 0.9) would be ascribed to the studied system (the slaughterhouse) if the "grandfathering" valuation principle is chosen, meaning that 90% of the lake's carrying capacity would be entitled to the studied system. On the contrary, if the "contribution to GDP" valuation principle is chosen, the VF would be 0.1 and therefor just 10% of the lake's carrying capacity would be entitled to the studied system.

3.1.3 Integrate value factor in AESI scores

The integration of VF depends on whether carrying capacity references is integrated in normalisation references (see Chapter 2.4.1) or characterisation factors (see Chapter 2.4.3). When carrying capacity occupation is expressed as normalised indicator scores, the ratio of occupation to entitlement for a given impact category is calculated as follows (units in square brackets):

$$\frac{Carrying\ capacity\ occupation}{Carrying\ capacity\ entitlement} = \frac{IS_{normalised}}{P \cdot t_{duration} \cdot VF} \quad , \quad \frac{[person \cdot year]}{[person] \cdot [year] \cdot [-]} \tag{3.1}$$

[8] Starkey (2008) terms the population valuation principle "equal per capita allocation"

Here IS is the indicator score, P is the population of the relevant geographical territory and $t_{duration}$ is the duration of the relevant emission(s) or resource use(s). The integration of VF is different when carrying capacity references are instead integrated in characterisation factors and carrying capacity occupation thereby is expressed as area equivalents (m^2·year). If an ideal approach to identifying competing systems (see Chapter 3.1.1) is followed the ratio of occupation to entitlement is calculated as follows for a given impact category (units in square brackets):

$$\frac{Carrying\ capacity\ occupation_{i,j}}{Carrying\ capacity\ entitlement_{i,j}} = \frac{IS_{i,j}}{A_j \cdot t_{duration\_i} \cdot VF_{i,j}} \quad , \quad \frac{[m^2 \cdot year]}{[m^2] \cdot [year] \cdot [-]} \tag{3.2}$$

Here A is the area of a spatial unit, i is the spatial unit of emission or resource use, k is the environmental compartment (air, soil or water) of an emission and j is the receiving spatial unit (see also Chapter 2.2.2, where similar notation is used for the generic CF-equation). As mentioned above, the ideal approach is in many cases impractical because it involves the calculation of VF for every combination of emission location (i) and receiving cell (j), which reflects that a unique group of competing systems potentially exist for each combination. This also means that the ratio of occupation to entitlement becomes specific to each combination of i and j and that a choice must be made in aggregating these ratios to an overall ratio for each impact category. If a simplified approach to calculating VF (see above) is followed the ratio of occupation to entitlement simplifies to (units in square brackets):

$$\frac{Carrying\ capacity\ occupation_{i}}{Carrying\ capacity\ entitlement_{i}} = \frac{IS_{i}}{A_{affected\_i} \cdot t_{duration\_i} \cdot VF_{i}} \quad , \quad \frac{[m^2 \cdot year]}{[m^2] \cdot [year] \cdot [-]} \tag{3.3}$$

Here $A_{affected}$ is the area of $T_{affected}$ (see Chapter 3.1.1). Note that the ratio of occupation to entitlement for a given impact category is specific for each emission location (i). As in the previous case, a choice must therefore be made on how to aggregate these ratios to an overall ratio. Note also that for some impact categories the fate of different substances emitted varies substantially. This is for example the case for ecotoxicity, where metal emissions to air tend to deposit quite close to an emission source, while volatile organic compounds typically travel thousands of kilometres (see **Article I**). In these cases it may be appropriate to add a substance index (x) to the equations above, because the number of competing systems, and thus VF, depend on substance fate. Alternatively, it may be chosen to pragmatically let the parameters of the equations reflect only the substance(s) with the highest contribution to the total IS of an impact category.

3.2 Applicability of valuation principles

A valuation principle can be seen as applicable when it can be unambiguously used to calculate VF and when the data required for this calculation is available. Below, the applicability of the valuation principles is evaluated. The applicability for territorial studies (see Figure 1.2) is analysed at the corporate scale, which is the focus of many territorial studies. The applicability for life cycle studies is analysed at the scale of a single function (i.e. a product system), which is often the focus in LCA.

3.2.1 Territorial studies

The four valuation principles of Table 3.1 compatible with territorial studies are Area, Gross output, Contribution to GDP and Grandfathering. Of these, Area is only relevant at the national level and above. The Gross output valuation principle is somewhat ambiguous since many different products can be references, e.g. digital memory and computer processing capacity in the case of the electronics industry. In addition, this valuation principle relies on the initial use of another principle (e.g. contribution to GDP or grandfathering) to establishing carrying capacity entitlements for each industrial sector. The use of the Grandfathering principle to calculating VFs at the corporate level is less ambiguous, but the choice of reference year can influence calculated VFs. The Contribution to GDP principle is quite unambiguous. Contribution to GDP at the corporate scale can be calculated using the production approach (Callen, 2012) and the reference year should be the most recent for which data is available (e.g. the last financial year), because these data should approximate the current situation.

Regarding data availability for the use of the three valuation principles corporations most likely have data on their production of potential reference products, their contribution to GDP and their past emissions (of some pollutants) and resource uses. The same type of data for competing systems may be obtained from a combination of financial and corporate responsibility reporting as well as from national statistics collected by administrative bodies. This task is easiest if the boundaries of $T_{affected}$, in which the competing systems are located, follow administrative borders.

3.2.2 Life cycle studies

The four valuation principles of Table 3.1 that are compatible with life cycle studies are Relevance for meeting human needs, Population, Contribution to

GDP and Grandfathering. These principles are compatible to systems at specific scales, from single functions to nations and above.

The Relevance for meeting human needs principle is specific for studies of single functions, i.e. delivered by product systems. This principle is ambiguous due to the (perhaps purposely) ambiguous use of the term “need” by the Brundtland definition of sustainable development. Data availability depends on the interpretation of this principle to calculate VF. All in all, the principle therefore has low applicability unless “needs” becomes more precisely defined.

The population principle is unambiguous (all individuals equally valuable and thus entitled to occupy the same amount of carrying capacity via their consumption) and population data, needed to calculate VF, are generally available in statistics collected by administrative bodies.

The contribution to GDP principle is rather unambiguous, but the contribution can be calculated in different ways (Callen, 2012) and the most appropriate way depends on the scale (from functions to national consumption and beyond). For studies of product systems the expenditure approach (Callen, 2012) is deemed suitable. Here the cost to the user of a product system is a measure of the product system’s contribution to GDP. This is often straightforward to estimate, but the contribution to GDP from competing systems (i.e. the costs to their users), required for the calculation of VF, can be more difficult to estimate, especially when the borders of $T_{affected}$, in which the competing systems are located, do not follow administrative borders and considering that there is a $T_{affected}$ for each emissions location in the life cycle.

The Grandfathering valuation principle is, as noted above, rather unambiguous, except for the choice of reference year. It requires past data on emissions and resource uses. For studies of a product system, this data may be obtained from a life cycle inventory model based on unit processes whose temporal scope matches the reference year or from past corporate responsibility reports from the producer. Nationwide inventories of resource uses and emissions can be used to estimate the combined resource uses and emissions of competing systems in the chosen reference year. As for the contribution to GDP principle, the feasibility of this estimation depends on the extent to which the boundaries of $T_{affected}$ follow administrative bodies. Notwithstanding data availability, the grandfathering principle may have low applicability on product systems that are of a highly dynamic nature, i.e. that deliver functions that become obsolete or that replace obsolete functions by new functions. Consider the many relevant functions currently supplied by information and communications technologies that would have been unimaginable just two decades ago. In these cases the idea of a legitimate heritage (see Table 3.1) does not apply, because many functions have only exist-ed a very short time and are likely to be replaced by other functions in the near future.

3.3 Sensitivity of calculated entitlement

The case studies of **article V** and **article IV** (termed case 1 and 2 below) both analysed the influence of choosing different valuation principles (and combinations of principles) on study outcomes. Below, the case studies are presented to explore the topic beyond the hypothetical slaughterhouse example above.

3.3.1 Case 1

The environmental sustainability of all phosphorous emissions to the Baltic Sea was evaluated. The studied anthropogenic system was thus related to total production within a territory, spanned by a group of nations. The AESI was based on the planetary boundary for the phosphorous cycle, intended to avoid a major oceanic anoxic event (Rockström et al., 2009; Steffen et al., 2015). The emission inventory was based on data from the HELCOM monitoring system (HELCOM, 2011). Since the planetary boundary is global the competing systems were in this case all systems emitting phosphorous, either directly or indirectly, to an ocean.

Carrying capacity entitlements were calculated for two valuation principles of Table 3.1: The first was Gross output, according to which 1.6% of global carrying capacity should be entitled to anthropogenic systems within the Baltic Sea catchment (VF = 0.016), because 1.6% of global cropland was situated in the catchment (cropland area was taken as a proxy for gross output) in the reference year. The second valuation principle was Contribution to GDP, according to which 2.9% of global carrying capacity should be entitled to anthropogenic systems within the Baltic Sea catchment (VF = 0.029), because it contributed 2.9% to global GDP in the reference year.

When dividing the indicator scores by the two alternative carrying capacity entitlements both results were found to be below 1, meaning that the sum of anthropogenic systems within the Baltic Sea catchment were considered environmentally sustainable, with respect to the planetary boundary for the global phosphorous cycle, for both valuation principles.[9]

[9] As argued by Carpenter and Bennett (2011) this does not mean that anthropogenic systems within the Baltic Sea catchment collectively are environmentally sustainable for all types of carrying capacities. For example, the carrying capacity of freshwater ecosystems towards phosphorous is likely to be lower than the marine oriented planetary boundary for the phosphorous cycle. All other things being equal, this means lower carrying capacity entitlements.

3.3.2 CASE 2

The environmental sustainability of personal residential electricity consumption scenarios of 45 locations across contiguous United States[10] was evaluated. The anthropogenic systems studied thus belonged to the life cycle of personal consumption. The applied AESI was for the impact category terrestrial acidification, outlined in Chapter 2.4.3. Emission inventories were based on state specific average residential electricity consumption and power plant specific emission intensities.[11] Due to the largely overlapping deposition patterns of acidifying compounds for the 45 power plants and the convenience of operating with the same competing systems for all emissions locations, competing systems were approximated as all systems emitting acidifying compounds within contiguous United States.

Carrying capacity entitlements were calculated for two combinations of the valuations principles of Table 3.1: In the first combination contribution to GDP was used to calculate that 2.0% of carrying capacity within contiguous United States should be entitled to residential electricity consumption (VF = 0.02) because average US household on average spent 2.0% of its pre-tax income on residential electricity. This was combined with the Population principle by calculating the personal entitlement to residential electricity by dividing the total entitlement by the population of contiguous United States. In the second combination of valuation principles Grandfathering was used to calculate that US residential electricity consumption should be entitled to maintain its past share of total environmental interferences, which was estimated to be 9% in 2010 (VF = 0.09), which translates to a entitlement to 9% of carrying capacity within contiguous United States. Again, this was combined with the Population principle by dividing the total entitlement by the population of contiguous United States. When dividing indicator scores for the 45 scenarios by either of the two alternative carrying capacity entitlements the outcomes were above 1 for all scenarios in both cases. This translates into none of the scenarios being considered environmentally sustainable when applying either of the two combinations of valuation principles.

[10] The contiguous United States consists of the 48 adjoining U.S. states plus Washington, D.C. (federal district).

[11] Note that although the case study was life cycle oriented, environmental interferences upstream from the power plants, e.g. emissions of NO_X from the extraction and transportation of coal, were not considered.

3.4 CONCLUSION AND OUTLOOK

In this chapter the second main research question was answered: "How can the carrying capacity entitlement of individual anthropogenic systems be calculated, how applicable are different valuation principles to calculating entitlements and how sensitive is this calculation to choice of valuation principle?"

It was demonstrated that calculation of carrying capacity entitlement relies in part on the number of anthropogenic systems competing for the same carrying capacity as that partially occupied by the studied system and in part on the perceived value of the studied system relative to that of competing systems. The principles of an ideal and a simplified spatial assessment to identifying competing systems were presented, as were principles of an assessment suitable for a pseudo-spatial identification of competing systems for spatially generic LCA indicators. Furthermore, a variety of valuation principles and corresponding principles for calculating value factors (VF) were outlined and it was demonstrated how VF can be integrated in AESI scores to evaluate whether a studied anthropogenic system can be considered environmentally sustainable.

The applicability of valuation principles was analysed and it was concluded that the most critical element of entitlement calculations in many cases is the data on competing systems required to calculate VF. To ease the calculations of carrying capacity entitlement, it may therefore be justified to approximate the spatial boundary of competing systems by those of administrative units (as was done in Case 2). Thereby the information required to quantify the total value of all systems (e.g. in terms of contribution to GDP or gross output) may be found in statistical databases of administrative bodies such as municipalities or nations.

In the case studies presented in Chapter 3.3 the sensitivities of carrying capacity entitlement to choice of valuation principle were around the same: In each case study there was a factor 2-5 between the lowest and highest carrying capacity entitlement. Due to the potential importance of valuation principles, and since no single principle is objectively correct, it is important that decision-support from the use of AESI is transparent in terms of how entitlements are calculated.

A life cycle perspective is consistently taken in this thesis. Since the anthropogenic systems of study in LCA are typically product systems, the calculation of entitlement for such systems needs elaboration. Amongst the valuation principles in Table 3.1 that are compatible with the product system level, Relevance for meeting human needs was, for reasons given above, evaluated to have low applicability. The Grandfathering valuation principle has highest applicability on products systems that deliver functions that are stable in time, such as nutrition. The Contribution to GDP valuation principle is probably the most applicable at the product system scale. The principle is, however, problematic because it favours expensive products that the majority of mankind cannot afford (the higher the price, the higher the contribution to GDP and the higher the carrying capacity entitlement). This mechanism, essentially making the consumption of luxury items appear sustainable, can be seen as unacceptable from a social sustainability perspective, which includes the concept of social equity.

Instead the perspective may be lifted from individual product systems and the functions they provide to the sum of product systems and associated functions that make up the material components of a lifestyle. At this scale the Population valuation principle (possibly adjusted, see Chapter 3.1.2) could satisfy the social equity component (Starkey, 2008). It would thus be up to the individual to decide how to assemble the material component of a lifestyle, as long as the total carrying capacity occupation of these components does not exceed the personal carrying capacity entitlement. This freedom of (informed) choice within boundaries can be compared to the freedom, granted to consumers, of assembling a healthy diet based on information on shares (%) of recommended daily intakes of carbonhydrates, protein, fat, etc., taken up by food and beverages. Initial research on this topic has been done by CT (2012), who tested how consumers responded to the concept of personal carbon allowances when presented on the same media (printed on packaging) and in the same style as the familiar nutritional information. The normalisation references developed in **article II** are suitable for supporting such lifestyle related decisions, since normalised indicator scores are expressed in person years (or person equivalents) of carrying capacity occupation. The use of the global normalisation references would correspond to assuming that competing systems are all other individuals on Earth and that $T_{affected}$ is the global area of the ecosystem(s) in question. This assumption may be justified for the evaluation of many lifestyles, which tend to involve the consumption of products having life cycles that are composed of processes scattered around several continents. Thereby consumers in a global economy occupy parts of the same carrying capacity as all, or most, other consumers on the planet.[12]

[12] The perspective of the global human population being connected and sharing carrying capacity via global markets can, rightfully, be accused of being biased towards a free market ideal, which is a strong driver of the accelerated environmental degradation, that have been observed over the past centuries. Currently, however, the life cycles of the product systems that are typically evaluated in LCAs are, for various reasons, decidedly global. A more local approach to

calculating carrying capacity entitlements should be taken when evaluating products systems with local life cycles.

4 AESI IN STAKEHOLDER COMMUNICATION

In this chapter the third main research question is answered: "What characterises companies' use of AESI in corporate responsibility (CR) reports and how may answers to research question 1 and 2 contribute to an increase in companies' use of AESI?"

4.1 CORPORATE RESPONSIBILITY REPORTING

The practice of CR reporting started to gain momentum in the 1990s and since then an increasing number of reports have been published every year.[13] This shows that for large companies the practice of publishing CR reports has gradually become an informal requirement and no longer by itself signals that companies doing this are "environmental frontrunner". In other words, the publication of CR reports by companies may over time have become expected by critical stakeholders and thus a precondition for a companies' social licence (Gunningham et al., 2004). The share of small and medium sized companies that publish CR reports appears much smaller than that of large companies (CR, 2014a), probably because of the sizeable amount of resources required to create a CR report. CR reports are composed of a combination of qualitative and quantitative information. Typical qualitative information a visions, strategies and descriptions of technical or organisational processes within companies. Quantitative information related to environmental concerns is to a large extent in the form of environmental indicators. In CR reports indicators are used for different purposes, such as 1) to demonstrate that companies are aware of their environmental interferences (e.g. are able to quantify them), 2) to demonstrate improvements in environmental performance of new products compared to older ones, 3) to demonstrate superior environmental performance compared to competitors or market references, 4) to define performance targets that may be aligned with a sustainability strategy or vision.

4.2 PRESSURES ON COMPANIES TO ADOPT AESI

Recently, a number of initiatives from non-governmental organization (NGOs), non-profit organizations, think tanks, research organizations, consultancies and industry itself have encouraged companies to adopt AESI: McElroy and van Engelen (2012) call for companies to perform "context based sustainability reporting", where context refers to carrying capacity. Context based sustainability reporting is also encouraged by the latest G4 guideline of the Global Reporting

[13] The number of CR reports included in the comprehensive CorporateRegister database from the year 2000 is 459, while the number from the year 2013 is 4586 (CR, 2014a).

Initiative.[14] The World Business Council for Sustainable Development's Vision 2050 and Action 2020 encourage companies to commit to the challenge of staying within carrying capacities, based on the ecological footprint and planetary boundaries concepts (WBCSD, 2009, 2014). The One Planet Thinking model was developed to translate planetary boundaries to a business context (Ecofys, 2015). Other initiatives focusing exclusively on climate change have urged companies to reduce their greenhouse gas emissions in line with global reduction needs to meet various climate change targets that are based on avoiding the crossing of climatic tipping points (CDP, 2014; ClimateCounts, 2013; GreenBiz, 2014; Krabbe et al., 2015; Randers, 2012; WWF, 2013).

4.3 AESI IN CORPORATE RESPONSIBILITY REPORTS

In light of these recent encouragements for companies to adopt AESI and since companies are important potential users of AESI, the past and present use of AESI in CR reports was characterised in **article III.** The characterisation was based on a systematic screening of references to "ecological limits", which is an umbrella-term used in **article III** to cover both carrying capacity and threshold (and their synonyms) as these terms are defined in Chapter 2.1 of this thesis. The screening was based on a database of CR reports, CorporateRegister, that, as of November 2014, contained approximately 40.000 reports by 12.000 companies (CR, 2014a). The database can be assumed to cover essentially all CR reports written in English that has been published over the last two decades (CR, 2014b). Figure 4.1 shows the results of the screening as the numbers of references to ecological limits each year (4.1a) and that number divided by the number of CR reports published each year (4.1b).

[14] "This involves discussing the performance of the organization in the context of the limits and demands placed on environmental or social resources at the sector, local, regional, or global level. For example, this can mean that in addition to reporting on trends in eco-efficiency, an organization may also present its absolute pollution loading in relation to the capacity of the regional ecosystem to absorb the pollutant." (GRI, 2013). Note that this initiative has been criticized for not providing concrete guidance on this matter (Baue, 2013).

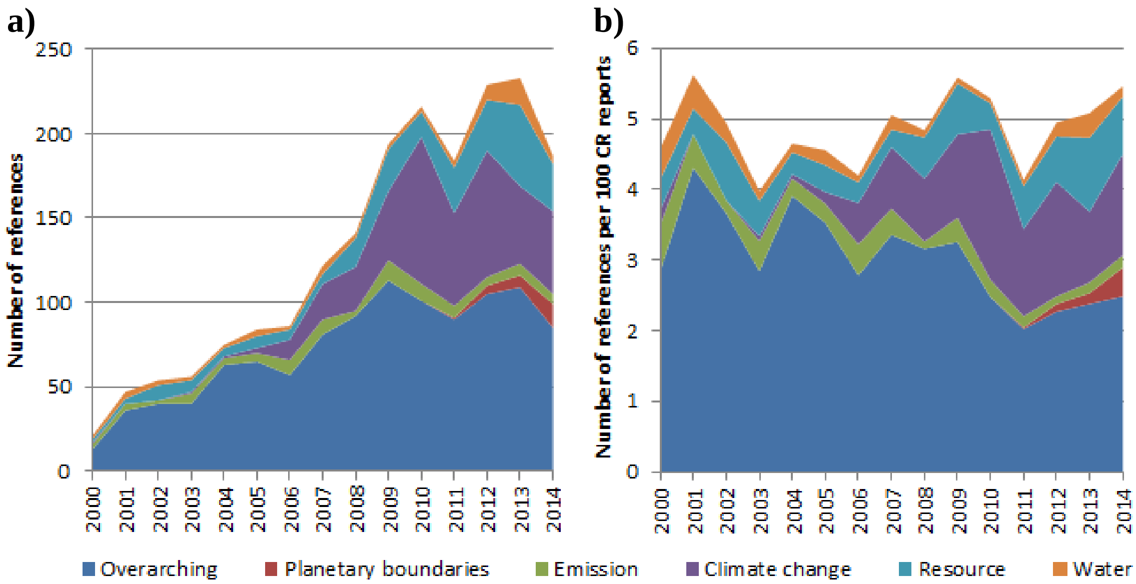

Figure 4.1: The absolute number of references to ecological limits in CR reports of the CorporateRegister database (a) and the number of references per 100 CR reports (b) in the period 2000 - 24. November 2014 grouped into six themes. Based on article III.

According to Figure 4.1a, the number of references to ecological limits across all CR reports increased by more than a factor 10 (from 21 to 233 references) from 2000 to 2013 (the coverage of reports from 2014 is incomplete). Figure 4.1b shows that, due to a similar increase in the number of published CR reports, the number of references to ecological limits per 100 CR reports was relatively stable around 5 throughout the entire period. This value leads to the estimation that the share of companies in the database referring to ecological limits was around 5% in any year of the 2000-2014 period, as elaborated in **article III.**

By classifying the context of each of the identified references to ecological limits, it was found that the vast majority (above 95%) of references were not associated with AESI. These references were instead made to 1) define the concept of (environmental) sustainability, 2) argue for the increasing importance of companies' products in a future with increasing scarcity of, e.g., energy and water resources, 3) report compliance with environmental legislation designed to avoid exceeding ecological limits of local ecosystems. Common for these types of references to ecological limits not associated with AESI is that they demonstrate awareness of ecological limits, but that this awareness is not accompanied by environmental sustainability performance targets based on ecological limits.

From the total pool of 12.000 companies, the context analysis identified just 23 that used AESI to define performance targets with deadlines for environmental sustainability. Table 4.1 presents these 23 companies and the ecological limit terms they referred to in CR reports. The following subsections present four trends that must be considered when trying to increase companies' use of AESI.

Table 4.1: Companies using AESI to define environmental sustainability performance targets with deadlines. Based on article III.

Company name	Sector	Nation	Publication years	Natural system	Ecological limit term
Ricoh Company Ltd	Technology Hardware & Equipment	Japan	2005, 2006, 2007, 2008, 2009	None specified	“tolerable impact”
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2007, 2008, 2009, 2010, 2011, 2012, 2013, 2014	Climate system	“2 degrees”
Spier Leisure Holdings	Travel & Leisure	South Africa	2008	Climate system	“tipping point”
Seiko Epson Corporation	Technology Hardware & Equipment	Japan	2008, 2009, 2010, 2011, 2012, 2014	Climate system	“absorption capacity”/ ”carrying capacity”
Ford Motor Company	Automobiles & Parts	USA	2008, 2009, 2011	Climate system	“450ppm”
Hitachi Ltd	Electronic & Electrical Equipment	Japan	2008, 2010	Climate system	“450 ppm”/”two degrees”
Acciona SA	Construction & Materials	Spain	2010	Climate system	“450 ppm”
Electrolux AB	Household Goods	Sweden	2010	Climate system	“2 degrees”
Hitachi Koki Co Ltd	Electronic & Electrical Equipment	Japan	2010, 2011, 2012	Climate system	“450 ppm”/ ”two degrees”
Toshiba Corporation Semiconductor Company	General Industrials	Japan	2010	None specified	“sustainable limit of the environment”
Unilever plc / NV	Food Producers	UK	2010	Climate system	“two degrees”
Iberdrola SA	Electricity	Spain	2011	Climate system	“2 degrees”
British Airways plc	Travel & Leisure	UK	2011, 2012, 2013	Climate system	“2 degrees”

Alcatel-Lucent	Technology Hardware & Equipment	France	2012	Climate system	"resource constraints"
Bridgestone Corporation	Automobiles & Parts	Japan	2012, 2013	Climate system	“ecological capacity”
PTT Public Company Limited	Oil & Gas Producers	Thailand	2012, 2013	Climate system	“2 degrees”
Skretting AS	Food Producers	Norway	2013	Fish stocks	“ecological limit”
Novelis Inc	Industrial Metals	USA	2013, 2014	Climate system	“safe limit”
Autodesk Inc	Software & Computer Services	USA	2014	Climate system	“Planetary limit”
Cisco Systems Inc	Technology Hardware & Equipment	USA	2014	Climate system	“2 degrees”
Colgate-Palmolive Company	Personal Goods	USA	2014	Climate system	“2 degrees”
Eneco Holding NV	Gas, Water & Multiutilities	The Netherlands	2014	Earth system	“planetary boundaries” of Rockström et al. (2009)
Implats	Mining	South Africa	2014	Climate system	“limits of the planet”

4.3.1 Trend 1: Few companies have been using AESI

The number of companies that have been using AESI to define performance targets with deadlines is very limited (less than 0.25% of the companies covered by the CorporateRegister database in the period 2000-2014), although an increase over the past 10 years can be observed. The low use may be partly caused by a lack of appropriate indicators (see below), but a lack of incentive may also play a large factor: Companies may perceive a long-term commitment to ecological limits based targets a risk. This is because companies are used to regularly adjusting targets and strategies in response to unforeseen changes in e.g. raw material prices, demands of products or rapid technological developments. Such unforeseen changes are, scientifically speaking, not valid reasons for adjusting targets and strategies motivated by ecological limits, although unforeseen changes can make it harder (or easier) to meet the targets and strategies. Abandoning or easing reduction targets (originally) based on an ecological limit could therefore be interpreted by critical stakeholders as a clear sign of abandoning the ambition of becoming a sustainable company. This creates a lock-in effect that is presumably generally undesirable to companies. As long as a committing to environmental sustainability performance targets is not part of the social license few companies will therefore do it.

4.3.2 Trend 2: AESI related to climate change are dominating

Of the 23 companies that have been using AESI to define environmental sustainability performance targets only 4 companies used AESI related to environmental interferences beyond climate change (see Table 4.1). Several factors may explain the predominance of climate change in companies' use of AESI in stakeholder communication. Firstly, climate change figures prominently on the political agenda worldwide. One indication of this is that the 2°C target, proposed by IPCC (Intergovernmental Panel on Climate Change), has been adopted in many policy documents. By contrast, critical stakeholders may perceive some of the other environmental problems as 'solved' or sufficiently controlled by regulation (at least in developed countries), which gives companies little reason to aim for emissions below legal thresholds. Secondly, the universality of the 2°C target and relatively high scientific certainty of associated global GHG emission reduction requirements means that these can be translated into company-specific emission reduction requirements, irrespective of the geographical setting of companies (i.e. a site-generic impact pathway model is sufficient). Thirdly, monitoring the manageable number of existing GHGs is relatively simple, and CO_2 emissions can be predicted relatively precisely based on consumption of fossil fuels. In other words, the relative popularity of AESI related to climate change is likely caused by a combination of pressure from stakeholders to address climate change and the convenience of using a universal AESI that has been made available from the work of scientific and political institutions such as IPCC and UNFCCC (United Nations Framework Convention on Climate Change).

4.3.3 Trend 3: AESI unrelated to climate change are incomplete

The few AESI not related to climate change used by the companies in Table 4.1 can be characterised as overall incomplete, when understanding a complete AESI as covering all 12 concerns of Table 2.1. For example, Toshiba and Ricoh did not specify any natural system or goal (concern 1 and 2) related to the ecological limit concepts that they in CR reports referred to ("sustainable limit of the environment" for Toshiba and "tolerable impact" for Ricoh).[15] Also, the links (concern 7) between an ecological limit in the form of a threshold (e.g. "sustainable limit of the environment") and an environmental sustainability performance target (i.e. an entitled carrying capacity expressed in LCA midpoint metrics) were not transparently presented by Toshiba or Ricoh. In fact, these companies referred to IPCC and UNFCCC for what they presented as reductions in total, unspecified, environmental interferences required for environmentally sustainability. In other words they used the recommendations of IPCC and UNFCCC for reduction of greenhouse gases directly on reduction needs for other types of environmental interferences.

This indicates that existing AESI (see Table 2.1 for examples) unrelated to climate change have either not been known, available or usable to companies. Existing AESI are generally not able to evaluate the absolute environmental sustainability of interferences expressed in LCA metrics (see Chapter 2). Existing AESI are therefore inconvenient for companies that are used to expressing their environmental interferences in LCA metrics. This may explain why Toshiba and Ricoh, who has been reporting environmental interferences in LCA metrics, did not use, e.g., the ecological footprint AESI to construct environmental sustainability targets.

Another reason for the low (or no) use of AESI unrelated to climate change may be that they can be technically demanding to use by companies due to the spatially variable nature of many environmental issues. For example, a

[15] Both companies used phrases like "reducing total impacts on the environment below tolerable impacts". Since the companies quantify environmental interferences using LCA, it is here assumed that the ecological limit concepts referred to in both cases apply to all natural systems covered by LCA.

multinational company with hundreds of production sites, scattered on different continents, may find it too resource demanding to compile data for the use of the spatially resolved (and time-resolved) water footprint indicator (Hoekstra et al., 2012) and to communicate indicator scores in CR reports. A noticeable exception from the trend of no, or poor, use of AESI unrelated to climate change is the case of the Dutch utility company Eneco, who has been basing their environmental sustainability targets on the One Planet Thinking model (Ecofys, 2015). This model aims to translate planetary boundaries to performance targets at the company level for LCA metrics, while taking into account spatial variations. The One Planet Thinking model shares similarities with the presented proposals for integrating carrying capacity in LCA in Chapter 2 and may prove usable to companies who already report environmental interferences in LCA metrics.

4.3.4 Trend 4: Entitled carrying capacities based on grandfathering principle

None of the companies in Table 4.1 explicitly argued why they considered themselves entitled to their proposed share of total carrying capacity. From the wording in the reports it could be inferred that all companies applied the grandfathering valuation entitlement principle (see Table 3.1) to calculate their entitlements. The companies defined a future target as a reduction of the environmental interferences in a past reference year of the same magnitude as overall reduction requirements proposed by e.g. IPCC or UNFCCC for GHGs to not exceed total carrying capacities.

A practical reason for the predominant use of the grandfathering valuation principle may be that the total reductions in environmental interferences required is known to companies (at least for GHG emissions) from the recommendations of e.g. IPCC or UNFCCC. By comparison the two other valuation principles that are compatible with the study of companies (see Table 3.1) require information on total[16] production volumes of key reference products (Gross output principle) or total generated revenue or profit (Contribution to GDP principle).

Additionally, a strategic reason for the predominant use of the grandfathering valuation principle may be that it has the inherent perspective that the past actions of a company are legitimate and that the future should "look like" the past. In other words, large polluters and resource consumers are allowed to remain large, relative to other systems, and the grandfathering principle is thus not

[16] Total, in this context, refers to the sum of the relevant quantity by the studied company and competing systems.

a threat to the market position of large companies. By contrast, the future according to the Contribution to GDP valuation principle may look very different than the past.

4.4 Increasing companies' use of AESI

The existence and causes of the four trends point to two complementary strategies for increasing the use of AESI by companies.

4.4.1 Strategy 1: Provide usable AESI

The first strategy is to make AESI for other issues than climate change available and usable to companies (and consultants hired by companies) to meet the apparent need of such AESI identified by the trends above. A critical factor for usability is for AESI to be compatible with the metrics companies use for reporting environmental interferences. These metrics are in many cases part of the LCA framework. There is thus a rational for the research community to continue the development of AESI based on modified LCA indicators, proposed in Chapter 2, and for making these accessible and usable to companies.

An important factor for the numerical values of companies' environmental sustainability targets, derived from AESI, is the extent to which companies themselves (or hired consultants) can make choices for the 12 AESI concerns identified in Table 2.1. Some concerns are entirely related to scientific understanding, such as control variable (concern 3) and modelling of safe limit (concern 7). For such concerns non-expert users should not be able to make choices. Companies (or hired consultants) should, on the other hand, be free to make choices for concerns that are partly related to value judgment, such as threshold value (concern 5) and carrying capacity entitlement (concern 12) in the AESI made available to them. Specifically, it is important that companies are asked to make an explicit choice on carrying capacity entitlement. Although most large companies will probably choose entitlement based on the grandfathering valuation principle, this choice should be transparent in a study and not merely a default invisible assumption. Transparency is important because the grandfathering principle in some cases can be considered "unfair."[17] The practical implementation of choices related to value judgement in LCA indicators and software may be inspired by

[17] For example, companies who have made no efforts in reducing environmental interferences before the chosen reference year will be favored with a high carrying capacity entitlement compared to companies that have achieved great reductions in and before the reference year. See **article III** for an elaboration on "unfairness" of different valuation principles.

the existing implementation of the normative choice of weighting factors for aggregating indicator scores into a single score in LCA indicators and software (Goedkoop et al., 2009; Thinkstep, 2015).

4.4.2 Strategy 2: Make AESI part of social license

The second strategy is for stakeholders to make the use of AESI part of the social license of companies. Obtaining or maintaining a social license is a key driver for companies' sustainability activities, as indicated by the growth in the publication of CR reports. The initiatives presented in Chapter 4.2 that encourage companies to adopt AESI can play an important role in increasing the awareness of AESI amongst critical stakeholders and thus contribute to making AESI part of the social license. Preferably the social licence should also come to include transparency about AESI choices related to value judgement, just as critical stakeholders currently expect LCA studies to be transparent about, e.g., system boundaries and weighting factors (if applied).

4.5 Conclusion and outlook

In this chapter the third main research question was answered: "What characterises companies' use of AESI in CR reports and how may answers to research question 1 and 2 contribute to an increase in companies' use of AESI?"

It was found that only 23 out of 12,000 large companies have been using AESI to define targets with deadlines for environmental sustainability at company level in the period 2000 to 2014. These 23 companies either used only AESI for climate change or used global reduction requirements for GHGs to calculate company-level environmental sustainability performance for other environmental issues than climate change. The 23 companies were all found to implicitly apply the grandfathering valuation principle in calculating the carrying capacity entitled to them. Based on these trends two strategies were outlined to increase the company use of AESI: 1) to make available to companies AESI that are compatible with LCA indicators of environmental interferences, and 2) to incentivise companies to use AESI by making it part of their social licence. Increasing companies' use of AESI is essential for the transition of societies to environmental sustainability given companies' direct influence on environmental stressors and substantial political power.

5 RECOMMENDATIONS

Chapters 2, 3 and 4 answered the three research questions presented in Chapter 1:

4. How can the carrying capacity concept be operationalized as sustainability reference value in environmental indicators in general and in LCA indicators specifically?
5. How can the carrying capacity entitlement of individual anthropogenic systems be calculated, how applicable are different valuation principles to calculating entitlements and how sensitive is this calculation to choice of valuation principle?
6. What characterises companies' use of AESI in CR reports and how may answers to research question 1 and 2 contribute to an increase in companies' use of AESI?

I believe that these answers can be used academically to advance the research in AESI and practically to increase the use of AESI in decision-making complementary to the use of RESI (relative environmental sustainability indicators). Below the potential uses of this thesis are elaborated.

5.1 A RESEARCH AGENDA FOR AESI IN A LIFE CYCLE PERSPECTIVE

The work of this thesis contributes to the development and understanding of AESI, but for every answer it provides, new questions emerge that motivates new research topics.

5.1.1 CARRYING CAPACITY BASED CFS

The mathematical equation developed in **article IV** could be used to construct AESI for impact categories beyond terrestrial acidification, to which the equation was applied in **article IV** and presented in Chapter 2. This could potentially lead to the development of a spatially derived LCIA methodology composed of a complete set of CFs for all impact categories. Such a methodology could include two sets of CFs, reflecting, respectively, impact category specific thresholds at midpoint and a single overarching threshold at endpoint related to potentially disappeared fraction of species (PDF).

5.1.2 CARRYING CAPACITIES BASED ON COMMON ENDPOINT THRESHOLD

The proposed principles for calculating carrying capacities based on a common threshold at endpoint, presented in Chapter 2.4.2, could be used to calculate carrying capacity based normalisation references at endpoint for all impact categories pertaining to the Natural environment Area of Protection. This would enable carrying capacity based normalisation at endpoint for linear LCA indicators. As mentioned in Chapter 2.5, the inherent incompatibility of carrying capacity based normalisation references with marginal LCA indicators may be circumvented by calculating stressor-generic factors that convert between marginal CFs and linear CFs (when existing) and applying these factors to the normalisation references.

5.1.3 Identification of important choices

The design of AESI requires the making of a choice for 12 concerns (see Table 2.1). Indicator scores are expected to exhibit varying sensitivities to changes in choices for the different concerns. It is important to identify the concerns that indicator scores are most sensitive to in the attempt of effectively reducing the overall uncertainty of indicator scores. These may be identified by, for each impact category, calculating indicators scores for one or more anthropogenic reference systems for all meaningful combinations[18] of choices for concerns related to the absolute environmental sustainability aspect of an environmental indicator (i.e. concerns 2-7, 11 and 12). This proposal to systematic sensitivity analysis was taken in **article V**, which analysed the influence of choices for the 12 concerns on uncertainties in indicator scores for 5 AESI that were adapted to studying environmental interferences from anthropogenic systems within the Baltic Sea catchment. This analysis showed that indicator scores for the systematic combination of choices for the 12 concerns potentially range 3 orders of magnitude. In other words, calculated occupations of carrying capacity for anthropogenic systems may be either over- or underestimated by more than a factor 1000, due to the choices made for the 12 concerns. There is thus a large potential of reducing indicator uncertainties by reducing the number of potential choices by different means (see below).

5.1.4 Consensus on AESI choices of scientific understanding

Choices of control variable, basis for threshold and modelling of carrying capacity (concerns 3, 4 and 7) are all related to scientific understanding of society-nature interactions. Choices like these could be made consistently for all impact categories and for both the integration of carrying capacity in CFs and in normalisation references. A scientific consensus process within the LCA community would be feasible for achieving this consistency in choices and could. Such a

[18] Not all combinations of choices are meaningful, since choices for some concerns (e.g. control variable) are restricted by choices for other concerns (e.g. goal).

process could, for example, mirror the consensus process that lead to great reductions of uncertainties in LCA indicators for human- and ecotoxicity (Rosenbaum et al., 2008). The systematic sensitivity analysis proposed above would be an important input for prioritisation in such a consensus process.

5.1.5 Managing AESI choices related to value judgement

As argued in Chapter 4.4.1, AESI should be designed with some flexibility for users to make their own value-based choices for concerns such as threshold value, aggregation of indicator scores and carrying capacity entitlement (concerns 5, 11 and 12). It may be feasible to design a few archetypical choices for each concern with the aim of excluding choices that are outside societal norms (see **article V**) and to reduce choices to a manageable number. This task could be inspired by the design and implementation in various impact assessment methodologies of the three archetypical "perspectives."[19] Archetypical choices for each concern may be defined in a multi-stakeholder process, which aims to include the views of all stakeholders.

5.1.6 LCIs in relation to AESI

Little attention is given to LCIs (life cycle inventories) in this thesis. However the use of AESI, compared to RESI, requires that at least three aspects of LCIs are considered.

Firstly, the spatially variable nature of many types of carrying capacities and the development of spatially derived CFs (Chapter 2.4.3), require that LCIs are spatially derived for utilizing the full potential of AESI. Currently, fine spatial information is typically only known for the foreground system (e.g. the production site(s) of a company commissioning an LCA study). In contrast, the locations of life cycle processes upstream and downstream of the foreground system are commonly uncertain. This is especially problematic when these up- and downstream processes account for large shares of the total stressors of a product life cycle, because it creates uncertainty on the occupations of carrying capacity quantified by AESI. The existence of global markets for many materials and semi-finished products means that the improvement of spatial information of LCIs will not be an easy task. The task may, however, become manageable due to the emergence of initiatives, such as WikiLCA (2015), designed for actors in a value chain to share environmental information and verify each other's information transparently via an online platform.

[19] These three archetypes is based on the work of Hofstetter (1998). See ReCiPe for an example of implementation in LCA methodology (Goedkoop et al., 2009).

Secondly, AESI is aligned with attributional LCA, and not consequential LCA. This is because (in short) AESI are designed to quantify the carrying capacity occupation that can be attributed to an anthropogenic system of study. In contrast, AESI are not designed to quantify the marginal change in the total occupation of carrying capacity (by all anthropogenic systems) caused by the addition of the studied anthropogenic system to the economy.[20] It is therefore important that attributional LCIs are used as input to AESI. A practical implication of this rule is that electricity production should be modelled based on grid mix and not based on marginal electricity production technologies (Ekvall et al., 2005).

Thirdly, the concept of environmental sustainability is inherently concerned about the future. Environmental unsustainability is harmful for future generations and anthropogenic systems that are evaluated as environmentally unsustainable by AESI should therefore preferably be transformed into systems that would be evaluated as environmentally sustainable at some point in the future. Prospective inventories are needed to make the latter type of evaluation. Prospective inventories take into account that some of the processes that LCIs are composed of are dynamic, due to e.g.: 1) changes to the studied system, such as core processes in the foreground system becoming more efficient in the future, 2) changes in the background system that happen independently of the studied system (such as changes in grid mix for electricity generation) (Miller and Keoleian, 2015). Prospective inventories are difficult to construct due to uncertainties in modelling the future. Yet, they are essential if AESI are to support societal transitions to environmental sustainability.

[20] The main reason for its attributional scope is that AESI must be based on linear, and not marginal, characterisation models (see Chapter 2.3.1 and **article II**). Linear characterisation models used with attributional LCIs ensure that results are scalable. In other words, that the indicator scores of 1,000,000 product systems are 1,000,000 times the indicator score of a single product system. In contrast, indicator scores from marginal characterisation models, used with consequential LCIs are, in principle, not scalable, because 1) the slopes of the current condition derivatives of response curves for CF components (e.g. concentration/response curves for EFs) could change between scales of anthropogenic systems, and 2) the concept of marginal technologies is only meaningful to apply to studies of anthropogenic systems at the micro-scale, and not to studies at the macro-scale. The macro-scale needs inherent consideration in AESI, due to their focus on quantifying the shares of total carrying capacity occupation accounted for by different anthropogenic systems.

5.1.7 Social sustainability references

The scope of this thesis is environmental sustainability in absolute sustainability indicators. Research is needed on how to construct social sustainability references to achieve a broader coverage of the sustainability concept in absolute evaluations. Here, the social component of sustainability is understood as covering anything that has an influence on the abilities of humans to meeting their needs and that is not mediated by environmental degradation (caused by exceeding thresholds). The social component thus includes: factors influencing human health, scarcity of mineral and fossil resources, wages, working conditions, personal security, etc. (Jørgensen et al., 2013). Some components of social sustainability have a strong material dimension, which may form the basis for absolute sustainability references. For indicators of human health social sustainability references might be based on science-based maximum exposures to individual stressors (i.e. by food intake or inhalation) or on policy goals for maximum acceptable DALYs relative to population, as proposed by Moldan et al. (2012). social sustainability references for indicators of mineral and fossil resource scarcity could, in turn, be based on considerations of required time for substitution of non-renewable resources (Goodland, 1995). Since human needs are somewhat more central in social sustainability components than in environmental sustainability components, the former are inevitably more normative than the latter due to the ambiguity of the concept of needs. Specifically references of absolute sustainability may be difficult to derive for social sustainability indicators that are focused on measuring effects on non-material human needs (e.g. working conditions and personal security). This is because the fulfilment of such needs are often more difficult to link to the processes of the life cycle of an anthropogenic system, let alone quantify.

5.1.8 Aggregation of indicator scores across impact categories

A core characteristic of LCA is that it covers a comprehensive set of impact categories and it is common practice to aggregate indicator scores across impact categories to arrive at a single score. This allows one-dimensional comparisons of the environmental interferences of product systems and thus answers questions such as “what product has the overall lowest environmental interference?” In AESI-LCA aggregation is technically possible since indicator scores are expressed in the same metric for all impact categories (person·year or ha·year, see Chapter 2.4). However, aggregation also means losing the absolute aspect of AESI: It is not meaningful to compare aggregated indicator scores to aggregated carrying capacity entitlements, because this would entail the assumption that exceeded carrying capacity entitlement for one impact category can be compensated by unused carrying capacity entitlement for another impact category, which is of course wrong. Yet, for identifying the anthropogenic system, amongst alternatives, that has the lowest overall occupation of carrying capacity, aggregation of indicator scores can be sensible. This can be done by simply adding scores for all impact categories. However, a weighting step may need to be developed as the consequences of exceeding carrying capacities can vary in severity between impacts categories. Some factors influencing the severity of exceedance are the social and/or economic consequences, the spatial extent and the time required for reversion of damage (see **article II**).

5.2 INCREASING THE USE OF AESI IN DECISION-MAKING

No actual decision-making supported by AESI was studied in this thesis. To increase the use of AESI in decision-making it is important to create an interaction between researchers and potential users and stakeholders of AESI. These include authors of LCA studies, decision makers commissioning these studies within industry, consultancy and regulatory bodies, standardization and guidance organizations, such as ISO and GRI (Global Reporting Initiative (2013)), NGOs, and consumer organizations.

5.2.1 INTERACTIONS BETWEEN RESEARCHERS AND POTENTIAL USERS

Interactions could be formalized through (in chronological order) 1) research projects involving the application of AESI to real case studies, 2) consensus processes on best recommended practice in indicator design and use, 3) standardization processes, 4) training events and workshops on the use of AESI to increase awareness and encourage user feedback on potential flaws and opportunities for improvements. Inspiration for point 2 and 3 may be sought in some of the consensus and standardization processes of LCA that have happened through ISO (2006a, 2006b), the UNEP SETAC Life Cycle Initiative (2015) and the Joint Research Centre of the European Commission (EC, 2010a).

5.2.2 INTEGRATING AESI IN LCA SOFTWARE

The LCA supported AESI proposed in this study are not as mature as existing LCA indicators, e.g. those identified as belonging to “best existing practice” by ILCD (Hauschild et al., 2013). Yet, the proposed AESI may be integrated as “pilot indicators” in LCA software such as SimaPro, GaBi and openLCA (GreenDelta, 2015; PRé, 2015; Thinkstep, 2015). This integration would both serve to increase the awareness of AESI amongst LCA users and also enable AESI researchers to receive user feedback on the pilot indicators.

At this point, the normalisation references developed in **article II** are ready for implementation in either software and are, additionally, available to users in a spreadsheet format for normalisation of characterised indicator scores exported from LCA software. The CFs that was developed for the terrestrial acidification AESI are spatially resolved by means of GIS coordinates. Currently no LCA software creates links between spatially explicit stressors and CFs with GIS co-ordinates. Links between spatially explicit stressors at the scale of nations and nation-specific CFs[21] are expected to be implemented in LCA software shortly, following the recent developments of spatially explicit CFs for many impact categories, e.g. within the Impact World+ (2015) methodology and the LC-Impact methodology (Huijbregts et al., 2015). A strategy for making the developed CFs for the terrestrial acidification AESI (and for others to come) available in LCA software is therefore to convert these from GIS referenced to nation averages. User guides should be developed for non-academic users in any software implementation of AESI.

The entitlement aspect is less mature for software integration. Ideally the user should only have to choose a valuation principle and define the duration of environmental interventions of each emission location of the foreground system. The software would then calculate carrying capacity entitlement and compare this to the corresponding indicator score (for each emission or resource use location if the assessment is spatially differentiated). This would require the software to be equipped with a fate model and to be linked to a complete spatially derived emission and resource use inventory that contains information needed to calculate entitlement, such as contribution to GDP, for each of the anthropogenic systems of the inventory.

[21] The reason for basing the resolution on nations rather finer spatial units (such as grid cells) is that most spatial information of typical in LCIs are rarely finer than the scale of nations.

6 MAJOR FINDINGS

1. Carrying capacity can be integrated as quantitative reference for environmental sustainability in LCA indicators in normalisation references or characterisation factors.
2. There is generally no objectively correct threshold value for which to calculate carrying capacities.
3. There is a risk of erroneously concluding that aggregated environmental interferences are environmentally sustainable when using a spatially generic impact pathway model,
4. AESI scores are potentially sensitive to many choices made in the design of AESI. The combined effect of variations in choices causes AESI scores to potentially vary at least 3 orders of magnitude.
5. The perceived carrying capacity entitlement of a studied anthropogenic system is a function of the valuation of the system relative to that of competing systems that rely on occupation of part of the same carrying capacity for their functioning.
6. The carrying capacity entitled to a studied system greatly influences whether that system can be considered environmentally sustainable.
7. Currently, the company practice of defining quantitative environmental sustainability targets based on AESI in stakeholder communication is very limited.
8. Companies that follow this practice almost exclusively focus on climate change and implicitly use the grandfathering valuation principle to calculate greenhouse gas reduction needs at the company level.

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Chemical footprint method for improved communication of freshwater ecotoxicity impacts in the context of ecological limits

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A chemical footprint method for improved communication of freshwater ecotoxicity impacts in the context of ecological limits

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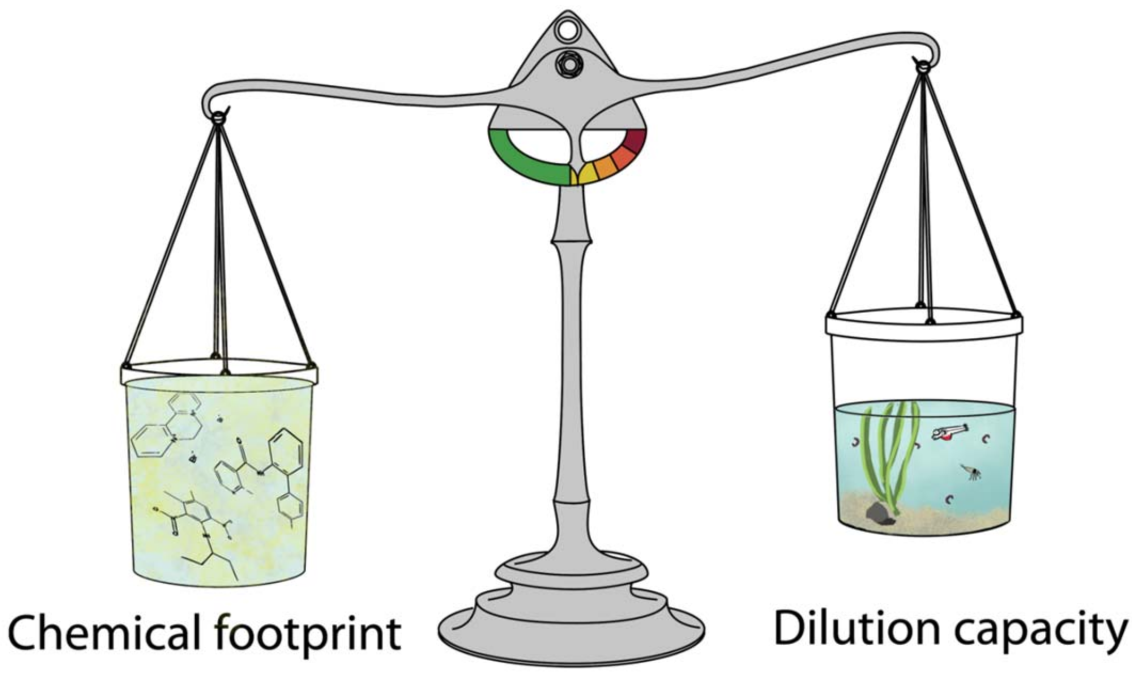

Abstract

The ecological footprint method has been successful in communicating environmental impacts of anthropogenic activities in the context of ecological limits. We introduce a chemical footprint method that expresses ecotoxicity impacts from anthropogenic chemical emissions as the dilution needed to avoid freshwater ecosystem damage. The indicator is based on USEtox characterization factors with a modified toxicity reference point. Chemical footprint results can be compared to the actual dilution capacity within the geographic vicinity receiving the emissions to estimate whether its ecological limit has been exceeded and hence whether emissions can be expected to be environmentally sustainable. The footprint method was illustrated using two case studies. The first was all inventoried emissions from European countries and selected metropolitan areas in 2004, which showed that the dilution capacity was exceeded for most European countries and all landlocked metropolitan areas. The second case study indicated that peak application of pesticides alone was likely to exceed Denmark's freshwater dilution capacity in 1999-2011. The uncertainty assessment showed that better spatially differentiated fate factors would be useful and that footprint indicator results are likely to be generally underestimated due to incomplete inventories and the exclusion of impacts from transformation products.

Introduction

Concerns over increased anthropogenic impacts on the environment have led to efforts to quantify local, regional and global ecological limits that should be respected in order to maintain or enhance ecosystem services.[1,2] Such limits (e.g., atmospheric CO_2 concentration and nitrogen depositions) are derived from sustainable levels of environmental impact and thus the limits represent absolute targets for which societies should strive. The most popular operationalization of ecological limits is the ecological footprint (EF) introduced by Matthias Wackernagel in 1994.[3,4] It translates and normalizes renewable material resource uses (timber, crops, fish, etc.) and CO_2 emissions into the occupation of productive area, followed by comparison with the availability of productive area (see S1 for background). EF is appealing because it easily conveys the concept of (un)sustainability at different scales, e.g. the consumption of an individual, a city, a country or humanity as a whole using a tangible metric of land area occupied. Its success as a communication tool can be seen by its widespread use and that it has given birth to other footprint methods such as the water footprint (WF), which is used to assess the freshwater impacts from water consumption and discharges from systems.[5] The critical dilution volume (CDV) is another indicator related to WF developed to assess detergents and adopted by the EU Eco-label scheme.[6,7] WF and CDV are both restricted to assessing impacts immediately after the release of pollutants emitted directly into freshwater and both use a precautionary risk assessment approach combined with politically defined water quality standards. Hence these indicators are not capable of expressing impacts from a system's chemical emissions to multiple recipients.

In an effort to improve the representation of chemical emissions in footprint methods Sala & Malgorzata[8] discussed requirements, possible approaches, and challenges to the development of a chemical footprint method. Challenges relate especially to the large number of anthropogenic and naturally occurring chemicals for which humans have altered their biogeochemical cycling, the variety of ways the chemicals can impact ecosystems and human health, and the limited understanding of the interaction between chemicals when exerting their toxicity.

Building on these earlier EF efforts, this study aims to develop an operational chemical footprint (ChF) method, which addresses the total toxic pressure on surface freshwater ecosystems from all emissions of an assessed system and supports the comparison to an ecological limit defined based on an estimated ecosystem effects level at a relevant spatial scale. Our main motivation is to build on the effectiveness of the EF concept at improving non-experts' understanding of environmental impacts from chemical emissions in relation to ecological limits. We therefore seek a balance between scientific rigor on one hand and relevance to, and accessibility by the public on the other. We deviate from CDV[7] and WF[5] by aiming for average rather than precautionary estimates, while acknowledging and quantifying uncertainties where possible. Below we describe the ChF approach, after which we apply the ChF to two case studies to evaluate its usefulness and relevance.

Methods

Framework and definition

The ChF method aims to characterize the impact of a chemical emission inventory using a metric that can be related to ecological limits for chemical pollution. The inventory should ideally contain all emissions caused by the system that it represents. Each inventory element should contain information on 1) the chemical ID (e.g. CAS no.), 2) the emitted quantity in kg, and 3) the compartment receiving the emission (soil, aquatic or air, if possible broken down into sub compartments).

In line with the logic of the EF, we defined the ChF as *the occupation of a (theoretical) freshwater volume needed to dilute a chemical emission to the point where it causes no damage to the ecosystem(s)* (see detailed rationale in S2). To operationalize the restriction of 'no damage to the ecosystem' we followed the approach of the EU Water Framework Directive, in which HC5(NOEC) is used to define environmental quality standards or "safe concentrations", assuming that functions and services of an ecosystem are protected when its structure is (sufficiently) protected.[9] HC5(NOEC) is therefore sometimes referred to as the predicted no effect concentration (PNEC).[9] HC5(NOEC) is the concentration at which maximum 5% of species are affected above their NOEC (no observable effect concentration), which is the highest concentration tested where no statistically significant chronic effects are observed.[10] Values of HC5(NOEC)s are derived from species sensitivity distributions (SSD), which are probabilistic models of the variation in sensitivity of species to exposure towards a particular stressor.[11] HC5(NOEC) is commonly derived through a precautionary approach, where safety factors are applied to ecotoxicity data, as was done in the EU Water Framework Directive[12] and also in the CDV and WF indicators[7,5]. Our approach instead derived average HC5(NOEC)s without use of safety factors, which avoids a precautionary bias that is important in absolute assessments, i.e. comparing impacts to ecological limits. We moreover deviated from the directive by assessing the effect of the mixture of chemicals in the emissions inventory rather than that of each individual substance, as simultaneous exposure to multiple chemicals in freshwater systems are the rule rather than the exception.[13] We assumed concentration addition, meaning that the ChF of individual chemicals in a mixture are simply added to calculate the ChF of the mixture. Evidence suggests that this assumption on average leads to slight overestimations of the actual toxicity in frequently occurring mixtures of substances with dissimilar modes of action.[13] In other cases, substances act synergistically and concentration addition thus underestimates the mixture toxicity.[14] The concentration addition assumption is therefore consistent with our average estimate approach.

Impact assessment modelling

To express an emission as a ChF we chose the USEtox impact assessment model, which was developed as a scientific consensus model, to represent the best application practice for characterization of toxic impacts of chemicals in Life Cycle Assessment (LCA).[15] USEtox was chosen since 1) its indicator for ecotoxicological impacts can be converted to the occupation of a compartmental volume, 2) it was developed for comparative assessment of chemicals and therefore is devoid of conservative assumptions, 3) it is a widely accepted and applied consensus model, and 3) its database (v. 1.01) covers ~2500 chemicals with calculated characterization factors for freshwater ecotoxicity.[16] The model's wide coverage of chemicals is important in the context of chemical footprinting where large numbers of chemical emissions are likely to jointly contribute to the exceedance of a given ecological limit. For substances not in the database, the USEtox model can be used to calculate characterization factors. Only impacts to surface freshwater ecosystems are modeled here because of the depth of knowledge of this system and its sensitivity to chemical pollution.

USEtox calculates the aggregated ecotoxicological impacts in freshwater, I_{USEtox} ([PAF]m^3*day), as the product of the emission quantity, E (kg), of chemical i into emission compartment j and its associated characterization factor, CF_{USEtox} ([PAF]m^3*day/kg), summed over all chemicals and emission compartments in the emissions inventory[21]:

$$I_{USEtox} = \sum_{ij} CF_{USEtox\_ij} \cdot E_{ij} \tag{1}$$

I_{USEtox} indicates the change in the potentially affected fraction of species, PAF (dimensionless), caused by the emission of chemicals to the environment, defined in terms of a volume of freshwater over a time interval. CF_{USEtox} is calculated as the product of three factors:

$$CF_{USEtox\_ij} = FF_{USEtox\_ij} \cdot XF_{USEtox\_i} \cdot EF_{USEtox\_i} \tag{2}$$

FF_{USEtox} is the fate factor representing the change in steady-state mass in compartment j, expressed in kg/(kg/day), XF_{USEtox} is the dimensionless exposure factor representing the fraction of the chemical in the water phase which is truly dissolved (sometimes expressed as BF or bioavailability factor[17]) and EF_{USEtox} is the effect factor representing the change in potential effect on species from a mass increase of that chemical in the freshwater compartment, expressed in [PAF]m^3*kg^{-1}. USEtox was developed as a relative model to compare and analyze incremental impacts of chemicals or chemical emissions from product systems. The ChF indicator, however, should be absolute, since impacts are to be compared to ecological limits.

Adaptation of effect factor

EF_{USEtox} needed to be adapted to express absolute impacts, while FF_{USEtox} and XF_{USEtox} did not require changing. The toxicity reference point of EF_{USEtox} is HC50(EC50), which is the concentration at which 50% of the species are exposed to a chemical above their chronic EC50, which is the concentration where 50% of a population displays an effect[16]. USEtox applies HC50(EC50) to estimate the marginal change in PAF caused by the assessed chemical(s):

$$EF_{USEtox\_i} = \frac{0.5}{HC50(EC50)_i} \tag{3}$$

For the ChF indicator we used HC5(NOEC) as the toxicity reference point and the ChF effect factor, EF_{ChF}, was therefore defined as follows[18]:

$$EF_{ChF\_i} = \frac{1}{HC5(NOEC)_i} \tag{4}$$

HC50(EC50) was converted to HC5(NOEC) in two steps. First we denominated HC50(EC50)/HC50(NOEC) as ɣ and thus:

$$HC50(NOEC)_i = \frac{HC50(EC50)_i}{ɣ_i} \tag{5}$$

Due to lack of toxicity measurements ɣ is difficult to obtain for many compounds. We calculated a default value of 9.8 as the geometric mean of ɣ for 11 chemicals exerting a wide range of toxic modes of actions[19] (see S3). In the second step of the conversion we used the relationship between HC50(NOEC) and HC5(NOEC) presented by Pennington et al.[20], who assumed a log-logistic model for the SSD (see S4):

$$HC5(NOEC)_i = \frac{HC50(NOEC)_i}{10^{2.94 \cdot \beta_i}} \tag{6}$$

β is a function of the standard deviation of HC5(NOEC) and describes the steepness of the SSD curve. We assigned β a default value of 0.4 that is representative of several toxic modes of action, judging from the average β value of 0.41 calculated by van de Meent & Huijbregts[21] from a sample of 261 chemicals. Combining both steps in the conversion via Equations 4, 5, and 6 yields EF_{ChF} in terms of HC50(EC50) (see S4):

$$EF_{ChF\_i} = \frac{ɣ \cdot 10^{2.94 \cdot \beta}}{HC50(EC50)_i} \tag{7}$$

ChF as a function of I_{USEtox}

We then defined the characterization factor for the ChF, CF_{ChF}, as the product of the USEtox fate factor, FF_{USEtox}, exposure factor, XF_{USEtox}, and the ChF effect factor, EF_{Chf}. Combining Equations 2, 3 and 7 gives (see S4):

$$CF_{ChF\_ij} = 2 \cdot ɣ \cdot 10^{2.94 \cdot \beta} \cdot CF_{USEtox\_ij} \tag{8}$$

When applying the default values for ɣ (9.8) and β (0.4), Equation 8 simplifies to (see S4):

$$CF_{ChF\_ij} = 290 \cdot CF_{USEtox\_ij} \tag{9}$$

To calculate the ChF (km^3*year) of an emission inventory, Equation 9 was combined with Equation 1 (see S4):

$$ChF = 8 \cdot 10^{-10} \cdot I_{USEtox} \tag{10}$$

A ChF result can be interpreted as the product of a water volume and the time it is occupied, required to dilute emissions to a safe level. This resembles the unit of the EF method of hectare*years, which represents the product of a land area and the time it is occupied, required to supply a given quantity of biomaterial.[4] Both methods thus assume interchangeability between time and space in the impact indicators. In cases where emissions are occurring over a known time, results can simply be divided by the time to be expressed in volumes. This is the case in ChF assessments of emissions occurring within a territory in e.g. 1 year. In other cases, such as product life cycles or aggregated consumptions of an economy, a common object of EF assessments, the time during which emissions occur is unknown and it is therefore more convenient to operate in ChF units of km^3*year.

Ideally the fate model of the ChF indicator should account for site-specific conditions which affect the fate of emissions and thus the ChF. The USEtox model does allow for some modifications of landscape parameters, but since it has a nested model structure it is not possible to calculate actual site specific CFs. We therefore recommend using default CFs from USEtox reflecting average environmental characteristics or the recently published USEtox CFs parameterized for 17 sub-continental zones[22], but to be aware of the potential for bias between the modelled situation and a specific local or regional situation as discussed later.

Comparison with dilution capacity

To relate ChF of territorial emissions to an ecological limit for chemical pollution, it can be compared to the territory's dilution capacity (DC), which is defined as the volume of surface freshwater available for diluting the emissions under consideration. When evaluating the ChF of emissions from aggregated human activities, comparing the ChF to the DC of a large territory, such as at a country-scale for which emissions data may be available, is straightforward, but will likely overestimate the DC relevant to emissions which are typically concentrated geographically within the territory. Thus, we recommend using a DC relevant to the geographic area that receives most of the emissions. In many cases, this area corresponds to a metropolitan area (large connected urban area) plus the area immediately surrounding it, because of the geographic concentration of emissions in and around urban areas.[23-26]

The geographical boundary of the DC of emissions from a metropolitan area depends on the travel distances of the emissions from sources to the receiving freshwater bodies. The most representative DC for all emission types can be identified by an analysis of the compartment(s) receiving emissions contributing the most to the ChF. If the ChF is dominated by emissions to soil, then the boundary of the metropolitan area is appropriate as DC boundary. When ChF is dominated by emissions to water, the boundary is expanded to include the entire volume of water bodies partly within the metropolitan boundary. Lastly, if the ChF is dominated by emissions to air, then the DC boundary can commonly be defined by a 50km radius from the air emission sources, which is the distance at which a range of gas- and particle-sorbed semi-volatile chemicals reach one tenth of their concentration at source.[27] This distance is suitable since it includes the distance for deposition of most particle-sorbed chemicals such as metals, which usually dominate impacts from air emissions in economy wide inventories.[28] We recommend using GIS software to define the DC boundary. The total volume of freshwater bodies within the boundary can then be estimated by importing the Global Lakes and Wetlands Database, levels 1 and 2, which contains data on all global freshwater bodies, excluding bogs, in shp format.[29] When the volumes of lakes and reservoirs are not given, we recommend using an ordinary least square empirical power regression to predict volume from surface area for lakes and reservoirs.[30] A river volume can be calculated by multiplying the area by the mean depth of the river. Continental mean river depths can be estimated by dividing their estimated total volume[31] with their total area extracted from the Global Lakes and Wetlands Database.[29] Alternative surface freshwater volumes may be obtained from grid cell based hydrological data.[32,33]

For assessment of an economy's consumption or a product system, the comparison of the ChF with DC is not feasible, since the exact locations of emissions and the time during which these occur are often not known with sufficient certainty. In these cases the ChF may be normalized to a reference DC of a relevant territory. Contrary to territory based assessments, such types of results do not reveal whether ecological limits of freshwater bodies receiving the emissions are exceeded. Instead results show the share of the sustainable level of impact of a reference environment that the studied system would occupy.

Case studies

The ChF method was applied to two case studies involving emissions that were heterogeneously distributed either spatially (Case 1) or temporally (Case 2).

The first case study estimated the ChF of the total inventoried chemical emissions of Europe in 2004 and thus evaluated the extent to which European freshwaters could dilute the emissions taking place within the region to a concentration at or below HC5(NOEC). We used the chemical emissions inventory of Laurent and co-workers[34] which covers approximately 300 substances, many of which are emitted into more than one compartment. The inventory was developed as a normalization reference in LCA and is mainly based on emission data compiled by EMEP/CEIP (air emissions from stacks and vehicles), OSPAR and HELCOM (riverine emissions of metals to the ocean) and Eurostat (pesticide and sludge application to farmland). Emissions from sewage treatment plants were back-calculated from riverine emissions to the ocean and pesticide emissions were modelled based on their field application.[34]

Here the inventory was disaggregated to country level[35] for Norway, Switzerland and all EU27-members, except Cyprus. I_{USEtox} was then calculated using equation 1 followed by the calculation of ChF using equation 10 and basing I_{USEtox} on the generic USEtox CFs. The ChF of each capital metropolitan area was then estimated by assuming that impacts from the metropolitan non-agricultural emissions are proportional to the fraction of a country's population living in the metropolitan area and that impacts from agricultural emissions (pesticides and pollutants in sewage sludge) do not occur within metropolitan areas. We furthermore assumed that emissions to water from coastal cities enter the sea directly. Thus:

$$ChF_{metro,i} = ChF_{country,i} \cdot \frac{P_{metro,i}}{P_{country,i}} \cdot \left(f_{non-agr,air,i} + a_i \cdot f_{non-agr,fw,i}\right) \qquad (11)$$

Where i is the country of the metropolitan area, ChF_{metro} and $ChF_{country}$ are the ChFs of the metropolitan area and country, P_{metro} and $P_{country}$ are the populations of the metropolitan area and country[36,37], $f_{non-agr,air}$ and $f_{non-agr,fw}$ are the fractions of $ChF_{country}$ accounted for by non-agricultural emissions to air and freshwater, and a is a bivariate parameter assigned the value 0 if the metropolitan area is coastal and 1 if it is landlocked. The GIS data of the capital metropolitan area of each country was extracted from Eurostat.[38,39] Google Earth Pro v.7.1 and ArcGIS v.10.1 were used to extract the data for all freshwater bodies falling within the defined metropolitan and national boundaries.

The second case study assessed the field application of pesticides in the agricultural sector in Denmark from 1999 to 2011. Unlike case study 1, active ingredients from pesticides are assumed to be uniformly distributed spatially in the freshwater compartment, since agricultural fields are distributed relatively evenly throughout Denmark. The active ingredients are, on the other hand, not uniformly distributed temporally since pesticides in Danish agriculture are mainly applied in discrete events in late spring and early autumn.[40] In this case, it is important to analyze whether ecological limits are exceeded at any points in time, since even an exceedance of HC5(NOEC) for a short period of time may be enough to cause adverse effects to ecosystems. To accommodate the short time period of pesticide application, we calculated the ChF for each month, assuming that emissions were continuous within each month. This allowed identifying the month with the highest ChF, much like the metropolitan focus in Case 1 allowed identifying the locations with the highest ChF. Emissions of active ingredients were modelled by 1) Collecting data on quantities of active ingredients applied annually and their distribution to different crop types (112 active ingredients were applied to a combination of 10 crop types, see S6.1[41]), 2) Calculating emissions for each month by assuming that farmers follow the standard practice for application time, see table S7.[40], 3) Estimating the fraction of applied active ingredients emitted to surface water and air after field application using the emission model PestLCI 2.0 on active ingredient level, see S6.2[42], I_{USEtox} was then calculated using equation 1, again based on generic CFs, followed by the calculation of ChF using equation 10. CFs for some active ingredients were not covered by the USEtox database and were therefore calculated by the USEtox model, see S6.3. The DC of Denmark was calculated following the method described for Case 1 with a few modifications, see S6.4.

Results

Case study 1: ChF of all inventoried emissions in Europe in 2004

The ChF of all countries was dominated by direct emissions to freshwater and therefore the boundary of DC included any freshwater body partly within the boundary of the metropolitan area. Figure 1 presents the ratios between ChF and DC for 28 metropolitan areas in Europe, which could be calculated since all emissions occurred within 1 year (see S5 for a contribution analysis for each country and metropolitan area).

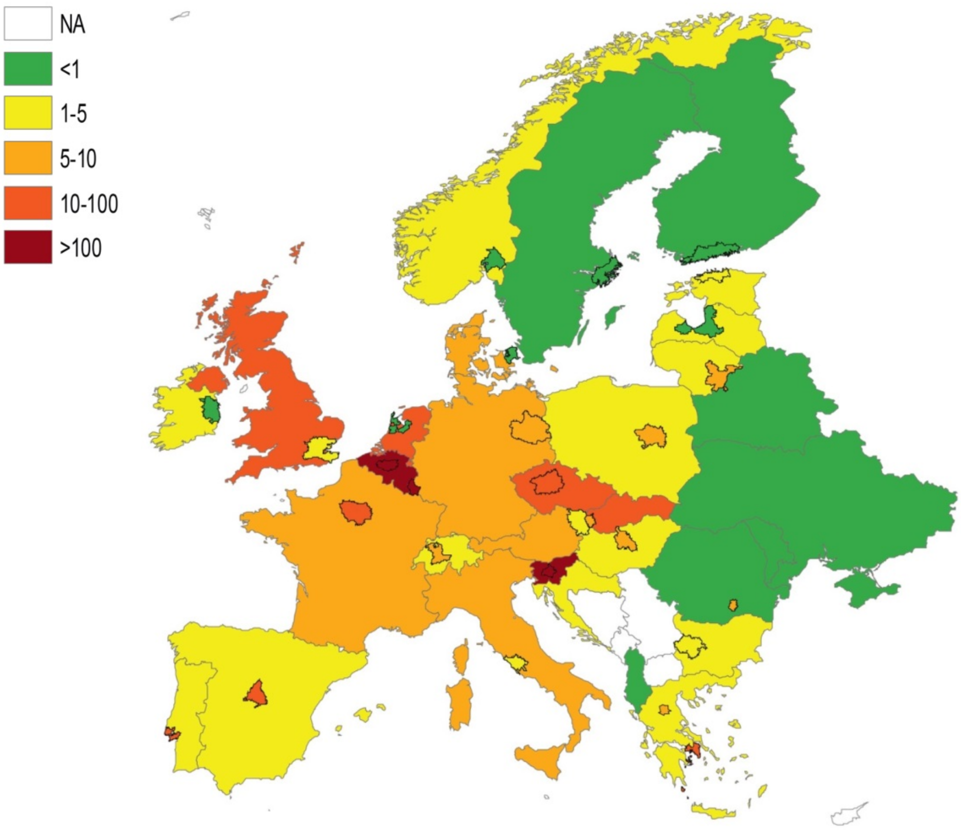

Figure 1: ChF/DC ratio by country and capital metropolitan areas (black borders). Values above 1 area assigned to surface freshwaters expected to have HC5(NOEC) exceeded on average.

The ChF of most countries exceeded their DC in 2004 and thus surface freshwaters of these countries are expected to have had, on average, their HC5(NOEC) exceeded for the ~300 chemicals considered. The most common country ratio was 1-5. Countries with ChF/DC ratios below 0.5 were Sweden, Finland and Ukraine. For Sweden and Finland this low score was driven by a relatively high DC per capita, while for Ukraine it was driven by a relatively low ChF per capita. In contrast, Slovenia, Belgium and Luxemburg had ratios above 100, driven by a very low DC per capita. Direct discharges to freshwater from WWTPs were on average responsible for 78% of each country's ChF, while emissions to air and from agricultural activities reaching the freshwater compartment were of minor importance.

Since WWTP discharges were assumed to be spatially proportional to population density, the ChF/DC ratio was typically 4X, and up to 28X, higher for metropolitan areas than the average ratio of the countries they were situated in, which is consistent with the findings of the urban metabolism literature.[23] For instance, the difference between Madrid (ratio 41) and Spain (ratio 4) is driven by a 12X lower

DC per capita in Madrid compared to the Spanish average, due to high population density in the city and little water available for dilution. Coastal metropolitan areas had lower ratios (e.g. 0.3 for Copenhagen) than the average of their countries (7.2 for Denmark) due to the assumption that coastal metropolitan areas discharged WWTP emissions directly into the sea (a=0). However, some coastal metropolitan areas had a higher ratio than the country average, e.g. Lisbon had a ratio of 53 compared to 1.9 for Portugal, since the very low DC in Lisbon of 3 m^3/capita was insufficient to dilute atmospheric emissions requiring the equivalent of 246 m^3/capita. Some inland metropolitan areas also had lower ratios than the country average but this was due to a high DC per capita for metropolitan areas located near lakes, e.g. ratio 4.2 for Vienna located nearby Lake Neusiedl versus 9.2 for Austria.

We evaluated the results by comparing them to those of Klepper and Van de Meent[43] who estimated that 33-63% of aquatic species were affected above their NOEC in major and regional surface waters in Netherlands in the year 1992. This estimate was based on measured water concentrations of 17 pesticides and 4 metals and thus underestimates the actual chemical emissions. Our study estimated that 18X the DC of the Netherlands was needed to dilute emissions to HC5(NOEC) in 2004. This translates into 54% of species affected above their NOEC (from equation 3 in Pennington et al.[20], which converts a pollutant concentration into PAF). This is well within the range given by Klepper and Van de Meent[43] (33-63%).

Uncertainties in these results include emission inventories of countries, particularly those with poor emissions reporting.[28] The emission inventories for WWTPs were particularly uncertain since they were back-calculated from riverine emissions to marine environments of Northern Europe and since differences in waste water treatment efficiencies within Europe were not taken into account.[44] This is a significant uncertainty since, as noted above, these discharges by far dominated the ChF on a country basis. In addition, the inventory of approximately 300 chemicals, though being state of the art, covers a relatively small fraction of the number of chemicals on the market, which was estimated at more than 100,000 for Europe in 1971-1981[45], and also covers less than the ≈4800 chemicals that OECD member countries in 2004 produced in volumes exceeding 1000 tons/year.[46] This leads to an uncertain underestimation of ChF estimates. The assumption of no agricultural emissions within the metropolitan areas also leads to underestimated results, but as agricultural emissions on average only accounted for 13% of a country's ChF and metropolitan areas, on average, covered 3% of the area of a country, the underestimation is negligible. A source of overestimating the ChF was the double counting of some emissions to air since they were registered both at the point of emission (e.g. a stack) and when they were discharged into the sea via rivers and thus classified as WWTP emissions by Laurent et al.[28] A final and important source of uncertainty and error was the ”disconnect” between using default compartmental volumes, including that of freshwater, to calculate the USEtox CFs and actual DCs available for each jurisdiction. As well, default values were used for all other environmental parameters. Recommendations to address this source of error are discussed below.

Case study 2: ChF of field emissions from pesticide use in Denmark in 1999-2011
The monthly ChF and ChF/DC for pesticides applied in Denmark was consistently highest in May followed by April and October, which is due to the intensive use of pesticides in these months (Fig. 2, Table S7). As expected, the annual average ChF (t=365 days) for pesticide use was 3-7X lower than that of May (t=30 days).

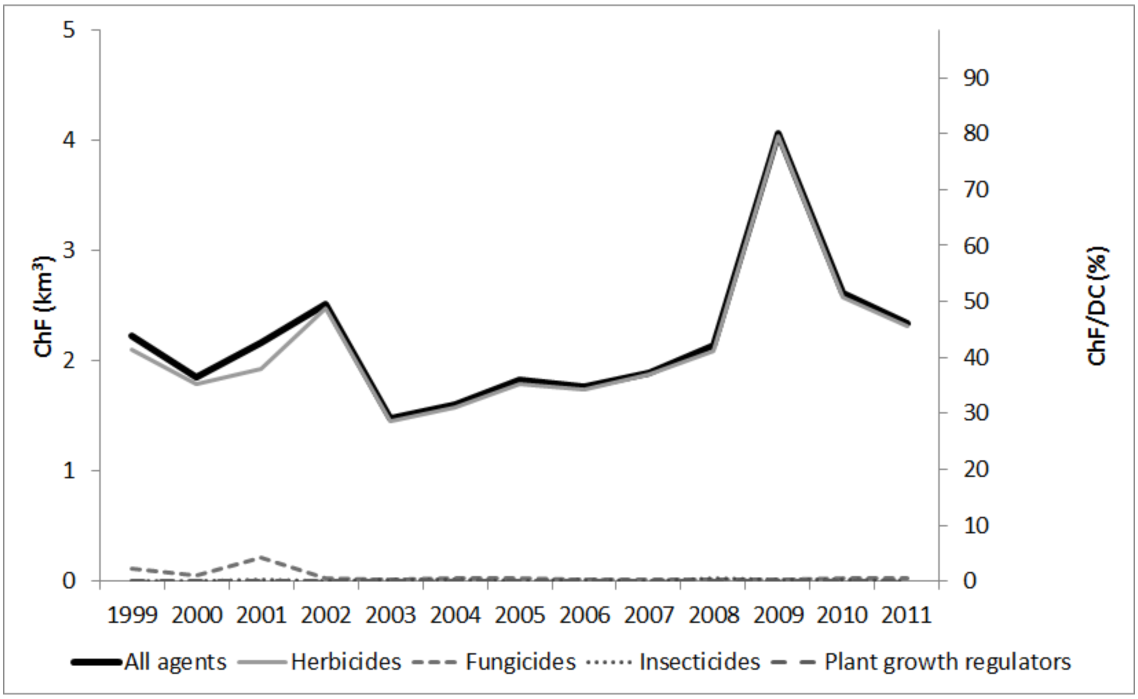

Figure 2: ChF and ChF/DC for the application of pesticides in Denmark in 1999-2011 for the month of May.

For the month of May the ChF and ChF/DC were ~1.5-2.5km^3 and ~30-50%, respectively, with the exception of a sharp peak of 4 km^3 and 80% in May 2009 due to a peak in the application of diquat dibromide used in certain herbicides. Although this compound only accounted for 5-10% of the active ingredient quantities applied, it contributed to 89-99% of the ChF due to its relatively high solubility and thus ability to reach the depth of the drainage tubes below the field and its relatively high toxicity. The assumption of concentration addition proved reasonable in this case according to Kudsk and colleagues, who found that additivity accounted for aquatic toxicity responses in 78% of 101 combinations of binary mixtures of pesticide applied in Denmark and five types of species.[47]

Modelled impacts were compared to those of Møhlenberg et al.[48] who tested for acute effects on algae and invertebrates in Danish freshwaters from realistic durations of peak concentrations of 14 pesticides. Of these, the herbicide pendimethalin was found to likely cause effects on algae, and pyrethroid insecticides were suspected of adversely effecting invertebrates depending on the particle-bound fraction. Diquat dibromide was not included in the study. Our results did not predict effects from any single compound (i.e. all single substance ChF/DC ratios are below 1), or from the combined emissions of all compounds. We calculated a high contribution to ChF from pendimethalin in April (data not shown), but negligible contributions from insecticides, including pyrethroids, in any month. Our results could therefore only partly reproduce measurements.

Uncertainties in this analysis include parameter and model structure uncertainty in PestLCI[42], which influences the predicted emission from field application. More importantly however, the assumption of continuous emissions leading to a steady-state impact within each month does not adequately deal with discrete field applications of pesticides. Pesticide applications can lead to peak concentrations in water bodies close to emission sources within hours[48]. Under these circumstances ChF results are hence underestimated. A dynamic model is required to calculate the dilution needed to avoid acute aquatic toxicity.

Discussion

The proposed ChF indicator was developed to be easily accessible and communicated to non-experts: It is relatively easily calculated, needing emission quantities (E), and a single constant (290) to convert USEtox indicator results to a ChF, which can be compared with a relevant DC. We also aimed for scientific validity and relevance. Below we evaluate the uncertainty accompanying ChF assessments beyond the scope of the two case studies.

Uncertainty assessment

We distinguished between parameter uncertainty and model structure uncertainty, following Van Zelm and Huijbregts[49] and U.S. EPA.[50] The assessment is elaborated in S7.

Parameter uncertainty

Incomplete inventories for, especially, organic chemical emissions are important source of parameter uncertainty for many types of ChF assessments, since there are no systems in place to monitor emissions of the vast number of synthesized chemicals on the market and their degradation products. The resulting ChF underestimation could lead to erroneously concluding that DC has not been exceeded for e.g. some countries and metropolitan areas in Case 1.

Parameter uncertainty also exists for DC, since the volumes of most global lakes are not included in the Global Lakes and Wetlands Database.[29] The applied power regressions predicting volume from surface area has high r^2 (0.89), but is based on only 36 lakes.[30] The uncertainty in volume prediction of individual lakes is up to one order of magnitude, but the uncertainty in DC of a territory goes down with an increasing number of lakes.

Henderson et al.[51] systematically analyzed the sensitivity of USEtox CFs to uncertain input parameters, and found that parameters that had both high uncertainty and large influence on CFs were ecotoxicity measurements (EC50s) and the chemical degradation half-lives that influence the fate factor. In case study 1 the ChF results are driven by metals and uncertainties in metal EC50s are therefore of special concern.

Model structure uncertainty

The use of default model coefficients β and ɣ, to translate EF_{USEtox} to EF_{ChF}, creates model structure uncertainty but likely not bias. An alternative approach to calculating EF_{ChF} is to use available NOEC data and supplement with ECOSAR (Ecological Structure Activity Relationships) model estimates to arrive at HC5(NOEC). ECOSAR estimates are, however, associated with parameter uncertainty of several orders of magnitude[52] and an overall lower uncertainty is therefore likely achieved by the conversion of EF_{USEtox} to EF_{ChF}.

The choice of HC5(NOEC) as a "safe" reference point obviously does not account for the role of key species in trophic chains, including microbes, and for the ecosystem as a whole and is therefore subject for improvement.[8,53,54] This problem is not unique to the ChF.

An important source of model uncertainty is the use of USEtox that is parameterized for average environmental parameters rather than site-specific applications used here. Kounina et al. found that CFs are generally sensitive to substance residence time, which varies orders of magnitude within the European continent.[22] Thus in site-specific assessments, as the ChF of metropolitan areas in case 1, this uncertainty is especially important to be aware of. Other environmental parameters with default values in USEtox are compartment areas and volumes, precipitation rate and soil erosion which also vary from territory to territory.[51] The use of site-explicit or specific fate models to replace USEtox fate factors could reduce this uncertainty however this solution requires a lot of data (see Pistocchi et al.[55] for a site explicit 1×1° grid based global pollutant fate model and Gandhi et al.[56] for a site specific fate model). A less data- intensive approach, that may greatly reduce uncertainty, is to apply archetypical CFs, based on freshwater residence time and chemistry (where the latter is important for assessing metals).[22,57]

Assuming additivity in the ecotoxicity calculations introduces uncertainty, in addition to uncertainty in ecotoxicity estimates themselves. Again, this is not a problem unique to the ChF. The msPAF approach of De Zwart and Posthuma[14] could reduce this uncertainty, but it requires knowledge of toxic modes of action of each compound in the inventory.

Model structure uncertainty also arises in assuming steady-state conditions. The assumption is reasonable for continuous emissions caused by e.g. WWTP and fossil fueled based power plants which typically contribute significantly to the ChF for economy wide inventories (see Case 1). As noted above, this assumption is less reasonable for the pesticide emissions considered in case study 2, where we expect ChF to be underestimated. Additional model structure uncertainties identified were neglecting metal speciation and chemical transformation products, and the choice of characterization model. These apply to LCIA modeling in general and are discussed in S7.

Overall, the uncertainty analysis suggests that the ChF tends to underestimate the actual impact due to incomplete emissions inventories and disregard of transformation products. Overall uncertainty is least for large inventories, since over- or underestimations of the impacts from individual substances will tend to balance each other.

Relevance and outlook

The ChF indicator presented here can be seen as a “first generation” attempt at developing an operational method that expresses chemical impacts in a metric that may be compared to an ecological limit. The main strength of our approach is that it combines well understood aspects of risk assessment to consider ecotoxicological effects with a widely accepted LCA impact assessment model having pre-calculated CFs for ≈2500 substances.

The ChF falls within the application domain of the EF method in that it compares the pressure that humans exert on the environment to its assimilative capacity. Like the EF, it is mainly intended to be used for communication to raise public awareness on the issue of chemical pollution and to place the issue on the political agenda together with other environmental issues. As a communication tool its novelty lies in its ability to compare ecotoxicological impacts to ecological limits.

We demonstrated the indicator on two territory-based case studies as this allowed comparison to the DC of these territories and also showed that indicator results were mostly in agreement with the observed state of ecosystems of the two case studies. From a sustainability perspective, consumption based assessments are of higher relevance than territory-based assessments, since they can reveal the embodied impacts of consumption and implications for the potential exceedance of DCs by emissions taking place in a multitude of locations as a consequence of global trade.[58] Such assessments rely heavily on high quality inventory data. In cases where this requirement is not fulfilled the ChF of consumption may, inspired by EF assessments, alternatively be compared to the DC of the national territory to evaluate the economy's potential ecological self-sufficiency or to the average global per capita DC to evaluate whether or not an economy is sustainable from an egalitarian perspective (humanity sharing global DC equally). When applying these alternative references to ChF results the focus moves from assessing actual environmental conditions to assessing whether systems are sustainable with reference to normative sustainability criteria.

Our indicator focuses on comparing impacts to ecological limits 1) at the local scale, 2) in one compartment (freshwater) and 3) using an aggregated ecotoxicological metric (PAF). With regards to scale, impacts from persistent pollutants, such as mercury, are not restricted to the local scale as they tend to be found in ecosystems far away from human activities[59]. For such compounds it may be relevant to define global ecological limits, in other words planetary boundaries for chemical pollution as has been proposed by Rockström et al.[60] and further discussed by Persson et al.[11] and Diamond et al. Thus, the ChF and planetary boundaries for chemical pollution can be complementary, noting the challenges associated with estimating the latter.

In addition to the freshwater compartment, a ChF should also be defined for the terrestrial and marine compartments as significant chemical impacts commonly occur in these.[59] Chemicals furthermore exhibit different toxic modes of action resulting in a variety of end points, each of which may be associated with thresholds or the lowering of ecosystem resilience. A limit for each relevant endpoint, or a differentiated treatment within the proposed ChF method, may therefore be more appropriate than the current representation by the aggregated PAF metric. In addition, as argued by Rockström et al.[60], impacts from chemical pollution may influence the ecological limits of other types of impacts and vice versa. For instance a lake that is both affected by emissions of nutrients and toxic chemicals may have reduced ecological limits for both types of stressors compared to one alone. This calls for careful considerations of context when conducting absolute environmental assessments.

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Supporting Information
The supporting information is available free of charge and contains: background information on the EF concepts, methodological details, detailed case study results and discussion of additional model structure uncertainties.

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Supporting Information for:

A chemical footprint method for improved communication of freshwater ecotoxicity impacts in the context of ecological limits

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S1 Ecological footprint background

The concept of ecological footprints was introduced as an estimation of humans' appropriation of biologically productive areas by Matthias Wackernagel in 1994[1,2]. The ecological footprint translates different renewable material resource uses (such as timber, crops and fish) into a unit termed 'global hectares' (gha), which reflects the area of productive land required to generate these material resources in a given amount of time (1 year per default). Also CO_2-emissions from fossil fuels are included in the footprint as the amount of productive area required to sequester the emitted carbon dioxide and thus prevent its accumulation in the atmosphere[1].

Expressing environmental impacts as a physical area on a finite planet intuitively leads to considerations of environmental limits. If the ecological footprint of a country's production of renewable material resources exceeds the availability of biologically productive land in that country (the biocapacity, gha*year), the result is a depletion of that country's biological capital, for instance more timber is being harvested each year than what is being generated by photosynthesis. When the ecological footprint of a country or region systematically exceeds its biocapacity year after year, the biocapacity may decrease, for instance due to soil erosion or collapsed fishing stocks (this is not true for the CO_2 part of the ecological footprint, since an increasing atmospheric CO_2 concentration is believed to increase, rather than decrease, net primary production.[3]

According to the latest assessments of the ecological footprint network the ecological footprint of the entire planet already exceeded the total availability of biologically productive lands (biocapacity) around 1970 and in 2008 the global ecological footprint was 50% larger than the global biocapacity, largely due to the contribution from CO_2 emissions.[2] The ecological footprint method is based on estimating nature's maximum short term capacity of delivering environmental provisioning services. This capacity can be increased by the use of agriculture technologies like irrigation and pest control resulting in a higher yield. Long term negative effects from the use of these technologies are not taken into account. Therefore it has been argued that aiming for a footprint just below the biocapacity is not sufficient to guarantee the environmental sustainability of a given country, region or the globe. Following this reasoning, a fraction of biocapacity should be reserved to safeguard biodiversity, but estimates of this fraction vary greatly from 12% to 75%[4]. A universal number is not meaningful since the effect on biodiver-

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[1] An alternative interpretation of the CO_2-footprint is that it is the amount of productive lands that would have been required if the energy obtained from the fossil fuels had instead been obtained from renewable energy resources (such as timber)

sity is largely depending on the geographical distribution of the footprint (influencing the connectiveness of ecosystems) and the type of footprint (cropland usually has a greater impact on biodiversity than forest land). In addition, local environmental conditions affect the amount of land that needs to be left undisturbed in order to sufficiently protect biodiversity.

It should be noted that the term 'footprint' is commonly used to describe a variety of indicators within environmental assessment, some of which are not related to environmental limits.[5] In the development of our indicator we follow the definition of Hoekstra[6], in which a footprint is a quantitative measure of the appropriation of (finite) natural resources. This definition allows for expressing footprints in units other than global hectares, which is not an appropriate unit for chemical impacts on the freshwater system.

S2 Detailed rational behind the definition of the chemical footprint

In the ecological footprint framework, the footprint of an emission is generally interpreted as the physical area needed to absorb the emission (carbon dioxide is given as an example). However for "non-degradable pollutants" (e.g. plastics) the footprint is interpreted as the physical area needed to make up for the loss of biocapacity resulting from the emission.[7,8] These definitions are not fully operational for chemical pollution. First of all, there is no universal classification into degradable and non-degradable chemical emissions. Some show persistences similar to non-degradable plastics, e.g. POPs and metals, while others have environmental halflives of less than a day. Most are somewhere in between. Secondly, even chemical emissions with short halflives may cause a loss of biocapacity if they are emitted as strong pulse-emissions from a point source with the potential to induce acute eco-toxicological effects on organisms close to the emission point.

As eco-toxicological effects are related to a loss in bio capacity (biomass production in freshwater) and offhand choice would be the alternative footprint interpretation (the physical area needed to make up for the loss of biocapacity resulting from the emission). It is however not straight forward to quantify this relationship between eco-toxicological effects and loss of biocapacity, since the PAF can cover many different forms of effects on biological organisms (e.g. immobilization, growth inhibition and mortality), all of which relate to a loss of biocapacity, but in different ways. Some chemical emissions may even through release of essential micro or macronutrients result in an increased biocapacity. This is further complicated by the fact that biocapacity in the context of ecological footprint only covers the part of net primary productivity that is useful to humans. The difficulties of establishing a link between emissions and loss of net primary productivity is reflected by the fact that it has so far not been operationalized in any LCIA method, although some have suggested it (e.g. LIME[9]). This issue is also noted by Sala & Goralczyk[5] in their proposed framework for a chemical footprint.

Instead, we have chosen to define a chemical footprint as *the occupation of a (theoretical) freshwater volume needed to dilute a chemical emission to the point where it causes no damage to the ecosystem(s)*. This is inspired by the general interpretation of a chemical footprint mentioned earlier, where the word *absorb* has been replaced by the word *dilute*, and where a restriction related to damage on organisms has been introduced. The occupation of a water volume or area is theoretical in many cases because chemical emissions are not instantaneously diluted into the environmental compartments and often not sufficiently diluted to avoid environmental impacts nearby their point sources, even though this is assumed when basing the calculations of the ChF on the USEtox LCIA model. Nevertheless the approach shares the strength of the indicators included in the ecological footprint framework, that the impact can be compared to the total available area or volume (the biocapacity), which can serve as a proxy for a (local, regional or planetary) boundary for chemical pollution.

S3 Conversion from HC50(EC50) to HC50(NOEC)

The 11 substances selected for analysis, based on Larsen & Hauschild[10] are shown in Table 1. The toxicity data available for these chemicals were obtained from the US EPA ECOTOX (2013) database. The advanced database query function was used for the 11 chemicals, while restricting the test species to the ones on the recommended species list.[10] NOEC, EC%, LC% and IC% endpoints were queried. EC10 was approximated as NOEC following EC.[11] All units were converted into µg/L. If no mean effect concentration was given it was calculated based on the reported range. If only a maximum (<) or minimum (>) value was reported the data point was as a general rule deleted. The only exception was when such data was the only data available at the trophic level for that endpoint. In this case the data was used provided that its influence on HC50 was an increase for minimum values (>) or a decrease for maximum values (<). In a few cases data points were deleted if they were clearly erroneous, for instance if a single NOEC value was larger than all the values in a group of corresponding EC50 values for the same species. The calculation of HC50 based on EC50 and NOEC respectively followed that of Larsen and Hauschild[10]:

1. The geometric average is calculated for each individual species.
2. Based on this the geometric average of each of the three trophic level included (algae, crustaceans, fish) is calculated.
3. Based on this the aggregated geometric average is calculated, which corresponds to HC50.
4. EC50 data is almost exclusively based on tests with acute exposure conditions. Since HC50 must be based on chronic exposure conditions HC50(EC50) was divided by a conversion factor of 2 following Larsen and Hauschild (2007a).

After HC50 was calculated for EC50 and NOEC respectively the fraction HC50(EC50)/HC50(NOEC) was calculated for each chemical, see Table S1.

Table S1: Substances selected for HC50(EC50)/HC50(NOEC) conversion, data availability and results.

Substance	CAS	End-point	Total number of data	Total number of species	Number of Algae species	Number of crustaceans species	Number of fish species	HC50(EC50)/HC50(NOEC)
2.3.4.6-Tetrachlorophenol	58-90-2	EC50	33	11	2	2	7	13
		NOEC	2	1	0	1	0	
4-Methyl-2-pentanone	108-10-1	EC50	13	5	1	1	3	8.1
		NOEC	10	2	0	1	1	
2.4-Dichlorophenol	120-83-2	EC50	67	13	3	1	9	11
		NOEC	8	2	0	1	1	
2-Chloroaniline	95-51-2	EC50	20	5	1	1	3	6.9
		NOEC	2	1	0	1	0	
4-Nitrophenol	100-02-7	EC50	70	11	3	1	7	2.6
		NOEC	42	4	2	1	1	
Dicamba	1918-00-9	EC50	22	9	3	2	4	0
		NOEC	0	0	0	0	0	
Metribuzin	21087-64-9	EC50	31	11	6	2	3	9.6
		NOEC	5	3	1	2	0	
Terbutylazine	5915-41-3	EC50	20	8	4	1	3	92
		NOEC	3	1	1	0	0	
Pendimethalin	40487-42-1	EC50	33	9	5	1	3	95
		NOEC	28	4	1	1	2	
Azoxystrobin	131860-33-8	EC50	9	6	3	1	2	2.2
		NOEC	3	2	0	1	1	
Dimethoate	60-51-5	EC50	98	10	3	1	6	2.1
		NOEC	52	5	1	1	3	
Geometric average								9.8

From Table 1 it can be seen that EC50 data is generally more abundant than NOEC data: The number of species covered by toxicity data for each of the chemicals ranged from 5 to 13 for EC50, compared to 0 to 5 for NOEC. Also data for all three trophic levels were included in the calculation of HC50(EC50) for all chemicals, while this was only the case for 4 of the 11 chemicals in the calculation of HC50(NOEC). When comparing to the data requirements for HC50(EC50) and HC50(NOEC), it can be seen that HC50(EC50) for all chemicals fulfill the requirements of being based on EC50 values for at least 3 species covering all 3 trophic levels.[12,13]On the contrary HC50(NOEC) does not fulfill the requirements of covering at least 10 species from 8 different taxonomic groups for any of the 11 chemicals.[11,12] This illustrates the reason why HC50(EC50) is applied as a working point in most recent LCIA methods, since its low data demand combined with a relative abundance of data ensures a more certain parameter than HC50(NOEC). However due to the definition of the chemical footprint, HC50(NOEC) is needed and we therefore have no choice but to use the values calculated to estimate HC50(EC50)/HC50(NOEC). From Table 1 it can be seen that ɣ varies from 2.1 to 95. The geometric average of the ɣ values for the 11 chemicals was found to be 9.8. The geometric average was chosen as the relevant statistical measure rather than the arithmetic average since the variations in biological material often display a lognormal distribution.[10] No bias was found between ɣ and data availability (such as relatively low data availability leading to either very high or very low values of HC50(EC50)/HC50(NOEC)). We therefore recommend that 9.8 is applied as a generic conversion factor from HC50(EC50) to HC50(NOEC). In any application its uncertainty should be kept in mind when interpreting the results.

S4 Step by step derivation of formulas

$$I_{USEtox} = \sum_{ij} CF_{USEtox\_ij} \cdot E_{ij} \tag{1}$$

$$CF_{USEtox\_ij} = FF_{USEtox\_ij} \cdot XF_{USEtox\_ij} \cdot EF_{USEtox} \tag{2}$$

$$EF_{USEtox\_ij} = \frac{0.5}{HC50(EC50)} \tag{3}$$

$$EF_{ChF\_ij} = \frac{1}{HC5(NOEC)} \tag{4}$$

In the first step in the conversion from HC50(EC50) to HC5(NOEC) we denominated HC50(EC50)/HC50(NOEC) as ɣ and thus:

$$HC50(NOEC) = \frac{HC50(EC50)}{ɣ} \tag{5}$$

In the second step of the conversion we used the relationship between HC50(NOEC) and HC5(NOEC) presented by Pennington et al.[14], who assumed a log-logistic model for the SSD:

$$\frac{HC50(NOEC)}{HC5(NOEC)} = 10^{2.94 \cdot \beta} \leftrightarrow HC5(NOEC) = \frac{HC50(NOEC)}{10^{2.94 \cdot \beta}} \tag{6}$$

Combining both steps in the conversion via Equation 4, 5, and 6 gave the expression of EF_{ChF} as a function of HC50(EC50):

$$EF_{ChF\_ij} = \frac{1}{HC5(NOEC)} = \frac{10^{2.94 \cdot \beta}}{HC50(NOEC)} = \frac{ɣ \cdot 10^{2.94 \cdot \beta}}{HC50(EC50)} \tag{7}$$

We then defined the characterization factor for the ChF, CF_{ChF}, as the product of the USEtox fate factor, FF_{USEtox}, exposure factor, XF_{USEtox}, and the ChF effect factor, EF_{Chf}. Combining Equation 2, 3 and 7 this gives:

$$CF_{ChF\_ij} = FF_{USEtox\_ij} \cdot XF_{USEtox_{ij}} \cdot EF_{ChF\_ij} = FF_{USEtox\_ij} \cdot XF_{USEtox\_ij} \cdot \frac{\gamma \cdot 10^{2.94 \cdot \beta}}{HC50(EC50)} = FF_{USEtox\_ij} \cdot XF_{USEtox\_ij} \cdot 2 \cdot \gamma \cdot 10^{2.94 \cdot \beta} \cdot EF_{USEtox\_ij} = 2 \cdot \gamma \cdot 10^{2.94 \cdot \beta} \cdot CF_{USEtox\_ij} \tag{8}$$

When applying the default values for ɣ (9.8) and β (0.4), Equation 8 simplifies into:

$$CF_{ChF\_ij} = 2 \cdot \gamma \cdot 10^{2.94 \cdot \beta} \cdot CF_{USEtox\_ij} = 2 \cdot 9.8 \cdot 10^{2.94 \cdot 0.4} \cdot CF_{USEtox\_ij} = 290 \cdot CF_{USEtox\_ij} \tag{9}$$

To calculate the ChF (km^3*year) of an emission inventory, Equation 9 was combined with Equation 1 and the units of space and time converted:

$$ChF = \frac{1}{365 \cdot 10^9} \cdot \sum_{ij} 290 \cdot CF_{USEtox\ ij} \cdot E_{ij} = 8 \cdot 10^{-10} \cdot I_{USEtox} \tag{10}$$

S5 Case I

Tabel S2 shows the contribution analysis, the per capita ChF, per capita DC and ChF/DC for each of the countries in the inventory. Tabel S3 shows the inputs needed in calculating the ChF for metropolitan areas (see equation 11 in the MS) and the resulting ChF, DC and ChF/DC.

Tabel S2: Contribution analysis and results for countries in inventory

	Non-agriculture			Agriculture			**ChF/10^6 cap**	**DC/10^6 cap**	**ChF/DC**
Country	**Air**	**WWTP**	**Total**	**Pest-icides**	**Sludge**	**Total**			
Albania	0.06	0.59	**0.65**	0.32	0.03	**0.35**	0.57	0.64	0.89
Austria	0.08	0.89	**0.98**	0.01	0.02	**0.02**	5.77	0.63	9.17
Belarus	0.50	0.40	**0.90**	0.08	0.02	**0.10**	0.83	1.35	0.62
Belgium	0.05	0.92	**0.97**	0.03	0.01	**0.03**	8.16	0.01	703.32
Bulgaria	0.11	0.83	**0.93**	0.00	0.07	**0.07**	0.54	0.39	1.39
Croatia	0.07	0.77	**0.85**	0.12	0.04	**0.15**	1.48	0.34	4.36
Czech Re-public	0.17	0.77	**0.93**	0.03	0.03	**0.07**	1.97	0.19	10.58
Denmark	0.02	0.91	**0.93**	0.02	0.05	**0.07**	6.73	0.94	7.17
Estonia	0.13	0.87	**1.00**	0.00	0.00	**0.00**	6.90	5.10	1.35
Finland	0.03	0.97	**0.99**	0.00	0.00	**0.01**	27.81	84.43	0.33
France	0.07	0.77	**0.85**	0.07	0.09	**0.15**	2.01	0.31	6.47
Germany	0.28	0.64	**0.92**	0.01	0.06	**0.08**	2.86	0.37	7.79
Greece	0.08	0.82	**0.90**	0.08	0.02	**0.10**	3.29	1.30	2.53
Hungary	0.10	0.77	**0.87**	0.07	0.05	**0.13**	1.87	1.82	1.03
Ireland	0.02	0.95	**0.97**	0.00	0.03	**0.03**	10.62	3.34	3.18
Italy	0.06	0.90	**0.95**	0.03	0.02	**0.05**	4.73	0.51	9.33
Latvia	0.02	0.97	**0.99**	0.00	0.01	**0.01**	11.48	3.66	3.13
Lithuania	0.05	0.93	**0.98**	0.00	0.02	**0.02**	6.55	1.48	4.42
Luxembourg	0.08	0.89	**0.97**	0.00	0.03	**0.03**	11.80	0.01	1037.38
Moldova	0.06	0.13	**0.19**	0.81	0.01	**0.81**	0.81	0.94	0.86
Netherlands	0.02	0.97	**0.99**	0.01	0.00	**0.01**	9.43	0.54	17.52
Norway	0.00	0.99	**0.99**	0.00	0.01	**0.01**	27.17	18.32	1.48
Poland	0.03	0.93	**0.97**	0.02	0.01	**0.03**	2.90	0.65	4.43
Portugal	0.10	0.56	**0.66**	0.07	0.27	**0.34**	1.74	0.92	1.90
Romania	0.07	0.64	**0.71**	0.29	0.00	**0.29**	0.77	1.19	0.64
Slovakia	0.23	0.70	**0.92**	0.04	0.04	**0.08**	1.59	0.13	12.59
Slovenia	0.09	0.89	**0.98**	0.02	0.00	**0.02**	2.61	0.01	257.45
Spain	0.11	0.72	**0.84**	0.06	0.10	**0.16**	3.87	0.94	4.13
Sweden	0.02	0.97	**1.00**	0.00	0.00	**0.00**	17.48	71.03	0.25
Switzerland	0.08	0.90	**0.98**	0.01	0.01	**0.02**	7.81	3.28	2.38
Ukraine	0.19	0.11	**0.30**	0.69	0.01	**0.70**	1.82	3.75	0.49
United Kingdom	0.03	0.87	**0.90**	0.01	0.09	**0.10**	5.16	0.38	13.59

Table S3: Inputs and results for metropolitan areas

Country	Metro	P_metro/ P_country (%)	a	ChF metro/ 10^6 cap	DC metro/ 10^6 cap	ChF/DC metro
Austria	Vienna	30	1	5.63	1.35	4.16
Belgium	Brussel	26	1	7.91	0.00	2101.02
Bulgaria	Sofia	21	1	0.51	0.24	2.14
Croatia	Zagreb	26	1	1.29	0.13	9.82
Czech Re-public	Prague	23	1	1.84	0.16	11.43
Denmark	Copenhagen	34	0	0.11	0.38	0.30
Estonia	Tallinn	38	0	0.86	0.23	3.81
Finland	Helsinki	27	0	0.70	1.96	0.36
France	Paris	18	1	1.70	0.05	35.82
Germany	Berlin	6	1	2.64	0.39	6.76
Greece	Athens	36	0	0.27	0.01	41.06
Hungary	Budapest	28	1	1.63	0.23	7.01
Ireland	Dublin	39	0	0.19	0.34	0.57
Italy	Rome	6	0	0.27	0.20	1.34
Latvia	Riga	48	0	0.21	1.59	0.13
Lithuania	Vilnius	25	1	6.51	0.92	7.04
Luxembourg	Luxembourg	100	1	11.49	0.01	1020.39
Netherlands	Amsterdam	14	0	0.19	2.48	0.08
Norway	Oslo	22	0	0.11	8.66	0.01
Poland	Warsaw	8	1	2.81	0.32	8.85
Portugal	Lisbon	26	0	0.18	0.00	53.05
Romania	Bucharest	10	1	0.55	0.06	9.41
Slovakia	Bratislava	11	1	1.47	0.21	7.02
Slovenia	Ljubljana	25	1	2.55	0.02	126.10
Spain	Madrid	13	1	3.23	0.08	40.67
Sweden	Stockholm	21	0	0.36	20.64	0.02
Switzerland	Bern	13	1	7.66	1.30	5.89
United Kingdom	London	21	0	0.17	0.04	4.23

*DC_metro arbitrarily assigned to 0.01, since no surface freshwater is present within the boundary of the metropolitan areas according to WWF.15

S6 Case II

S6.1 Calculation of applied quantities

Each year the number of theoretical hectares (A, in ha) treated with an individual AI and its recommended dose (D, in tons/ha) for each combination of AI (i) and crop type (j) is published.[16] Assuming that on average the recommendations are followed by the farmers, the quantity (Q, in tonnes) can thus be calculated:

$$Q_{i,j} = A_{i,j} \cdot D_{i,j}$$

As a quality check the total application across crop types was calculated for each AI and compared with the corresponding quantities reported as sold. For a few AIs the numbers did not match, which is judged to be due to a combination of entering errors (e.g. missing A for a specific combination of i and j) and errors in the expert judgment behind the derivation of A. This resulted in an initial total deviation between the sum of Q for all i and j and the sum of the reported quantities sold of up to 3% for each individual year (calculated quantities are consistently lower than reported). To eliminate this deviation the missing quantity was allocated to the crop type (j) with the highest A among the crop types with missing D, in cases where one or more crop types had missing recommend doses, and to thecrop type of with the highest tentative Q, in the cases where no Ds were missing.

S6.2 Inventory modeling with PestLCI 2.1

The PestLCI model requires a large number of application parameters to estimate the fractions of AI being emitted from the field to air via wind drift, leaf volatilization and topsoil volatilization processes and to freshwater via top soil runoff and drainage systems processes. Table S4 lists all these parameters and the values applied in the model.[17]

Table S4: Specification of input parameters in PestLCI 2.0

Parameter	Value	Comments
Pesticide	Variable	
Crop type selection	Variable	See Table S5 and S6
Soil selection	PetsLCI i: Tune	Based on soil sample from western Zealand (DK)
Climate selection	02 - Temperate maritime I: Tranebjerg (DK)	The only Danish weather model available in PestLCI 2.0[17]
Month selection	Variable	See Table S6
Application method	IMAG conv boom	Crop specification dependent on crop type selection
Application rate	1 (default)	Value irrelevant since only output fractions are of interest
Field width	200m	On average Danish fields are 4 hectare (4000m^2).[18] Assumed quadratic.
Field length	200m	On average Danish fields are 4 hectare (4000m^2).[18] Assumed quadratic.
Slope	1%	Birkved & Hauschild[19]
Depth of drainage system	0.6m	Birkved & Hauschild[19]
Fraction drained	0.55	Birkved & Hauschild[19]
Annual irrigation	0mm	Birkved & Hauschild[19]
Tillage type	Conventional tillage	Assumption
Solid material density	2.65kg/L	PestLCI 2.0 default value
Soil solid matter fraction	0.5	PestLCI 2.0 default value
Soil water fraction	0.25	PestLCI 2.0 default value[17]
E(a)/E(p)	0.8	PestLCI 2.0 default value[17]
Rainfall time (h)	3.5	PestLCI 2.0 default value[17]
Interception fraction	0	PestLCI 2.0 default value[17]
Reference soil moisture content for soil biodegr.	0.5	PestLCI 2.0 default value[17]
Response factor soil biodegra. rate on soil moist. Cont.	0.7	PestLCI 2.0 default value[17]
Q-value	2.1	PestLCI 2.0 default value[17]
Air pressure	1atm	PestLCI 2.0 default value[17]
Air boundary level	0.00475m	PestLCI 2.0 default value[17]
Nozzle distance	0.2m	PestLCI 2.0 default value[17]

In the following the handling of the three variable parameters will be explained one by one.

Pesticide properties

The model was run with each of the AI included in the study. Of the 115 AI reported 3 were excluded as biological or metallic agents that can currently not be handled by fate models. Out of these 37 were not a priori included in the PestLCI 2.0 substance database. In order to integrate them a number of chemical and physical properties were needed as input.[17] Table S5 shows input data sources.[20-22]

Table S5: Primary and secondary data sources for PestLCI 2.0 input data

Parameter	**Unit**	**Primary Source**	**Secondary source**
Molar weight	$g.mol^{-1}$	PPDB	EPI Suite
Molar volume	$cm^3.mol^{-1}$	Chemsketch	-
Solubility	$g.l^{-1}$	PPDB	EPI Suite: WSKOW method
Vapour pressure	Pa	PPDB	EPI Suite: modified grain method
pK_a	-	PPDB	Other literature sources (regulatory reports, etc.)
Log K_{ow}	-	PPDB	EPI Suite: experimental > KOWWIN estimate
K_{oc}	$l.kg^{-1}$	PPDB	EPI Suite: KOCWIN MCI method
Soil $t_{½}$	days	PPDB	EPI Suite: half-life in soil, level III fugacity model
Reference temperature for soil t½	°C	PPDB	-
Atmospheric OH rate*	$cm^3.molecules^{-1}.s^{-1}$	EPI Suite	EPI Suite: AOP Program
No-sprayzone width	m	national regulations	-

In case of cis/trans isomeric compounds the value corresponding to the cis version of the compound was arbitrarily applied. Due to insufficient chemical and physical property data the AI mesosulfuron (CAS 400852-66-6) was approximated with the AI mesosulfuron-methyl (CAS 208465-21-8), as this was deemed a better solution than excluding the AI from the assessment.

Crop type and month selection

The translation of crop type categories in the Danish EPA statistics[19] to the PestLCI 2.0 crop type categories are shown in Table S6.

Table S6: Translation of crop type categories from MST to PestLCI 2.0

MST (Danish)	MST (English translation)	PestLCI	Notes
Korn, vintersæd	Cereal, Winter seed	Cereal	
Korn, vårsæd	Cereal, Spring seed	Cereal	
Raps, vinter	Rapeseed, winter	Oilseed rape	
Raps, vår	Rapeseed, spring	Oilseed rape	
Andre frø	Other seeds	Grass	production of seed for grass was found to constitute >90% of the area assigned to "other seed".[16,23]
Kartofler	Potatoes	Potatoes	
Roer	Beets	Sugar beets	
Ærter	Peas	Peas	
Majs	Maize	Maize	
Grøntsager	Vegetables	Carrots	Carrots were the vegetable taking up the largest agricultural area in Denmark in
Græs og kløver	Grass and clover	Grass	
Areal udenfor vækst/mellem afgrøderne	Area outside growth season/in between crops	bare soil - pre-emergence	

In PestLCI 2.0 the AI can be applied at various crop stages (usually between 3 and 4 for each crop), the selection of which will affect the fraction of AI emitted from the field. The fractions are also affected by the month of pesticide application through variations in temperatures and precipitation. In order to most correctly model realistic pesticide application practices an agricultural consultant was consulted.[25] His expert judgment led to the population of Table S7 stating for each pesticide type (herbicides, growth regulators, insecticides and fungicides) the month and crop stage of application to each crop type (listed in Table 2). The data received by Petersen[25] did not always match the data derived from The Danish Ministry for the Environment.[16] Petersen[25] reported “usually no treatment” for some combinations of crop and pesticide type, where The Danish Ministry for the Environment[16] indicated that treatment is taking place. For these cases the crop stage assumed was based on the typical crop stage of application on other crops of that particular pesticide type. In such cases the month and crop stage of application was assumed to follow the month of application of other pesticides of the same class applied to the same crop.

Table S7: Applied combination of crop stage and month of pesticide application for different crop and pesticide types

		Crop stage				
	Crop type (model crop)	**Prior to seeding***	**I**	**II**	**III**	**IV**
Herbicides	Cereals, winter seed (Cereals)		October			
	Cereals, spring seed (Cereals)		April			
	Rapeseed, winter (Oil seed rape)	August				-
	Rapeseed, spring (Oil seed rape)		April			-
	Andre frø (Grass)			April	-	-
	Potatoes (Potatoes)	May				
	Beets (Sugar beets)		April			-
	Peas (Peas)		April			-
	Maize (maize)		April			
	Vegetables (Carrots)		April			
	Grass and clover (Grass)			May	-	-
	Area outside growth season/in between crops (bare soil - pre-emergence)	September	-	-	-	-
Insecticides	Cereals, winter seed (Cereals)		September			
	Cereals, spring seed (Cereals)				June	
	Rapeseed, winter (Oil seed rape)				April	-
	Rapeseed, spring (Oil seed rape)				May	-
	Other seeds (Grass)		May	-	-	-
	Potatoes (Potatoes)				May	
	Beets (Sugar beets)	April				-
	Peas (Peas)				May	-
	Maize (maize)	Usually no treatment***				
	Vegetables (Carrots)				June	
	Grass and clover (Grass)	Usually no treatment***				
	Area outside growth season/in between crops (bare soil - pre-emergence)	Usually no treatment				
Fungicides	Cereals, winter seed (Cereals)				May	
	Cereals, spring seed (Cereals)				May	
	Rapeseed, winter (Oil seed rape)			September		-
	Rapeseed, spring (Oil seed rape)	Usually no treatment				

	Andre frø (Grass)		May	-	-	-
	Potatoes (Potatoes)				Juli	
	Beets (Sugar beets)				August	-
	Peas (Peas)				May	-
	Maize (maize)				Juli	
	Vegetables (Carrots)				Juli	
	Grass and clover (Grass)	Usually no treatment****				
	Area outside growth season/in between crops (bare soil - pre-emergence)	Usually no treatment				
Growth regulators	Cereals, winter seed (Cereals)				April	
	Cereals, spring seed (Cereals)				May	
	Rapeseed, winter (Oil seed rape)			September		-
	Rapeseed, spring (Oil seed rape)	Usually no treatment				
	Andre frø (Grass)		May	-	-	-
	Potatoes (Potatoes)	Usually no treatment				
	Beets (Sugar beets)	Usually no treatment				
	Peas (Peas)	Usually no treatment				
	Maize (maize)	Usually no treatment				
	Vegetables (Carrots)	Usually no treatment**				
	Grass and clover (Grass)	Usually no treatment				
	Area outside growth season/in between crops (bare soil - pre-emergence)	Usually no treatment				

*Prior to seeding is modeled as 'bare soil - pre-emergence'

** Maleic hydrazide is reported to be applied to 'Vegetables'. It is assumed applied in Crop stage III in June

*** alpha-cypermethrin, cypermethrin, lambda-cyhalothrin, dimethoat and esfenvalerat are all reported to be applied to 'Maize' and 'Grass and Clover'. They are assumed to be applied in crop stage I for 'Grass and Clover' in May and crop stage III for 'Maize' in July. Carbofuran is reported to be applied to 'Grass and Clover'. It is assumed applied in Crop stage I in May. Malathion is reported to be applied to 'Maize'. It is assumed to be applied in crop stage III for 'Maize' in July.

****Mandipropamid is reported to be applied to 'Grass and Clover'. It is assumed applied in Crop stage I in June.

S6.3 Calculating missing CFs in USEtox

Of the 112 AI in the assessment CFs were not available for 27 in the USEtox database. Much like PestLCI 2.0 USEtox requires a number of chemical and physical property input data in addition to eco-toxicological data to calculate CFs for the emissions to air (continental compartment) and freshwater, estimated by PestLCI 2.0

The chemical and physical data required in the fate modeling module are to a large extend similar to those required by PestLCI 2.0, and thus the same sources are used for obtaining the data, see table S8.[20-22]

Table S8: Primary and secondary data sources for USEtox input data

Parameter	Unit	Primary source	Secondary source
MW	$g.mol^{-1}$	PPDB	EPI Suite
K_{OW}	-	PPDB	EPI Suite: experimental > KOWWIN estimate from water solubility
Koc	$L.kg^{-1}$	PPDB	EPI Suite: KOCWIN MCI method
K_H25C	$Pa.m^3 \cdot mol^{-1}$	PPDB	EPI Suite: HenryWin Bond Estimate
Pvap25	Pa	PPDB	EPI Suite: Antoine estimate
Sol25	$mg.L^{-1}$	PPDB	EPI Suite: WskowWin estimate
$kdeg_A$	s^{-1}	EPI Suite: Based on Half-Life in air (t½), Level III Fugacity Model (k = ln(2)/t½)	
$kdeg_W$	s^{-1}	EPI Suite: Based on Half-Life in water (t½), Level III Fugacity Model (k = ln(2)/t½)	
$kdeg_{Sd}$	s^{-1}	EPI Suite: Based on Half-Life in sediment (t½), Level III Fugacity Model (k = ln(2)/t½)	
$kdeg_{Sl}$	s^{-1}	PPDB	EPI Suite: Based on Half-Life in soil (t½), Level III Fugacity Model (k = ln(2)/t½)
$av_{logEC50}$	$mg.L^{-1}$	ECOTOX data-base*	PPDB

* See description in text

For effect modeling the ECOTOX database[26], developed by the US EPA, was used as a primary source for EC_{50} values. The advanced database query function was used for each individual chemical, while restricting the test species to only including the recommended species listed in Larsen & Hauschild[10]. For some AI the search gave no results and the EC_{50} values were therefore supplemented with EC_{50}-values from the PPDB database[20]. Some of the EC_{50} data found in the ECO-TOX[26] and PPDB[20] databases are stated as risk assessment oriented intervals with no upper boundary (e.g. > 100mg/L) rather than absolute values (e.g. 100mg/L). For taxonomic groups where only intervals with no upper boundary are stated as EC_{50} data for a given chemical, the lower value in the interval has been included. For taxonomic groups where EC_{50} values exist for a given chemical, any existing EC_{50} data expressed as intervals with no upper boundary has been excluded from the calculation of the geometrical mean. For EC_{50} data represented as an interval the average value was applied. After gathering the EC50 data, the official USEtox procedure was used to calculate avlogEC50.[27]

1. Gather experimental or estimated EC50 data for the chemical of interest;
2. Specify for every EC50-value whether it is chronic or acute exposure;

3. Calculate the geometric mean chronic or acute EC50 (mg/l) for every individual species (this can e.g. be done with the function =GEOMEAN() in Excel).
4. In case of acute EC50-data, derive the chronic-equivalent EC50 per species by dividing by a factor of 2 (acute-to-chronic extrapolation factor)
5. Take the log of the geometric mean EC50s and calculate the average of the log-values. This average equals the logHC50 (log mg/l).
6. Implement this value in column 20 of the sheet "Substance data" of USEtox.xls.
7. Always be careful with the units!

The avlogEC50 values were calculated for the 27 AI not included in the USEtox substance database. In addition ecotoxicity data was improved for the 6 AIs included in the substance database that are currently flagged 'interim' due to poor ecotoxicity data: EC_{50} values for at least 3 taxonomic groups are required for the data set to be flagged 'recommended' (otherwise it is flagged 'interim').

S6.4 Calculating the dilution volume

After importing the Global Lakes and Wetlands Database[28] level 1 and 2 into ArcGIS v.10.1, the area of each freshwater body from the database was exported to excel and converted to a volume, using the ordinary least square empirical power regression of surface area and volume for unregulated lakes, developed by Hendriks et al.[29]: $V = 0.003942 \cdot A^{1.30}$, where V is the volume in km^3 and A is the surface area in km^2. This yielded a total lake volume of 4.51km^3. Some Danish lakes were not covered by the database, since their total surface area is reportedly 516km^2, which is 64km^2 less than the 580km^2 of total Danish lake area as reported by Søndergaard et al.[30] When assuming the average depth of the database lakes, calculated by using the formula above, to the missing lakes, their total volume was estimated at 0.33km^3. Data on the streams of Denmark was obtained from 486 measurement stations.[31] These stations are geographically well spread and their measurements are thus assumed to be representative for the Danish streams as a whole. Each station measures the width and cross-sectional area (in addition to a number of other parameters). By dividing the cross-sectional area with the width for each measurement, the average depth of that measurement was calculated. Considering that the total length of Danish streams is 64.000km[32] the average distance between each measurement station is 131km. The total volume of Danish streams is then estimated by treating them as a series of rectangles each having the length 131km, and the width and average depth as described above. The gives a total volume of Danish streams of 0.23km^3. The total estimate of Danish surface freshwater is thus 5.07km^3 (4.51km^3+0.33km^3+0.23km^3).

S6.5 Elaboration of results

The results are shown for the year 2011, May an annual average, in Table S9 and Figure S1-S7 shows the corresponding time series.

Table S9: Methodological stages and results for the chemical footprint of Danish pesticide application in May 2011 (annual average results in parenthesis). The numbers of active ingredients are written in parenthesis after the pesticide classes.

		Herbicides (64)	**Growth regulators (5)**	**Fungicides (29)**	**Insecticides (14)**	**All pesticides (112)**
Input to PestLCI 2.0	Application quantity (tons)	80 (3512)	39 (158)	304 (539)	1 (27)	424 (4236)
Outputs from PestLCI 2.0/inputs to USEtox	Emissions to surface freshwater (tons)	1.3 (1.6)	$2.4*10^{-4}$ ($2.7*10^{-4}$)	$7.8*10^{-2}$ ($8.0*10^{-}$)	$1.3*10^{-5}$ ($1.5*10^{-4}$)	1.4 (1.7)
	Emissions to air (tons)	1 (59)	0.10 (0.11)	1.8 (6.5)	$2.5*10^{-2}$ ($3.1*10^{-2}$)	3 (66)
USEtox results	USEtox impact from surface water emissions ([PAF]m^3*day)	$2.3*10^8$ ($2.6*10^8$)	$1.0*10^2$ ($1.1*10^2$)	$1.6*10^6$ ($1.7*10^6$)	$8.1*10^4$ ($1.4*10^6$)	$2.3*10^8$ ($2.6*10^8$)
	USEtox impact from air emissions ([PAF]m^3*day)	$9.8*10^6$ ($1.3*10^8$)	$3.1*10^4$ ($3.1*10^4$)	$1.0*10^6$ ($3.0*10^6$)	$6.1*10^5$ ($7.3*10^5$)	$1.2*10^7$ ($1.3*10^8$)
	USEtox impact total, I_{USEtox} ([PAF]m^3*day)	$2.4*10^8$ ($3.8*10^8$)	$3.1*10^4$ ($3.1*10^4$)	$2.7*10^6$ ($4.7*10^6$)	$7.0*10^5$ ($2.2*10^6$)	$2.4*10^8$ ($3.9*10^8$)
Chf Results	Chemical footprint (km^3)	2.3 (0.3)				
	DC (km^3)	5.1 (5.1)				
	Fraction of DC occupied (%)	46 (6)				

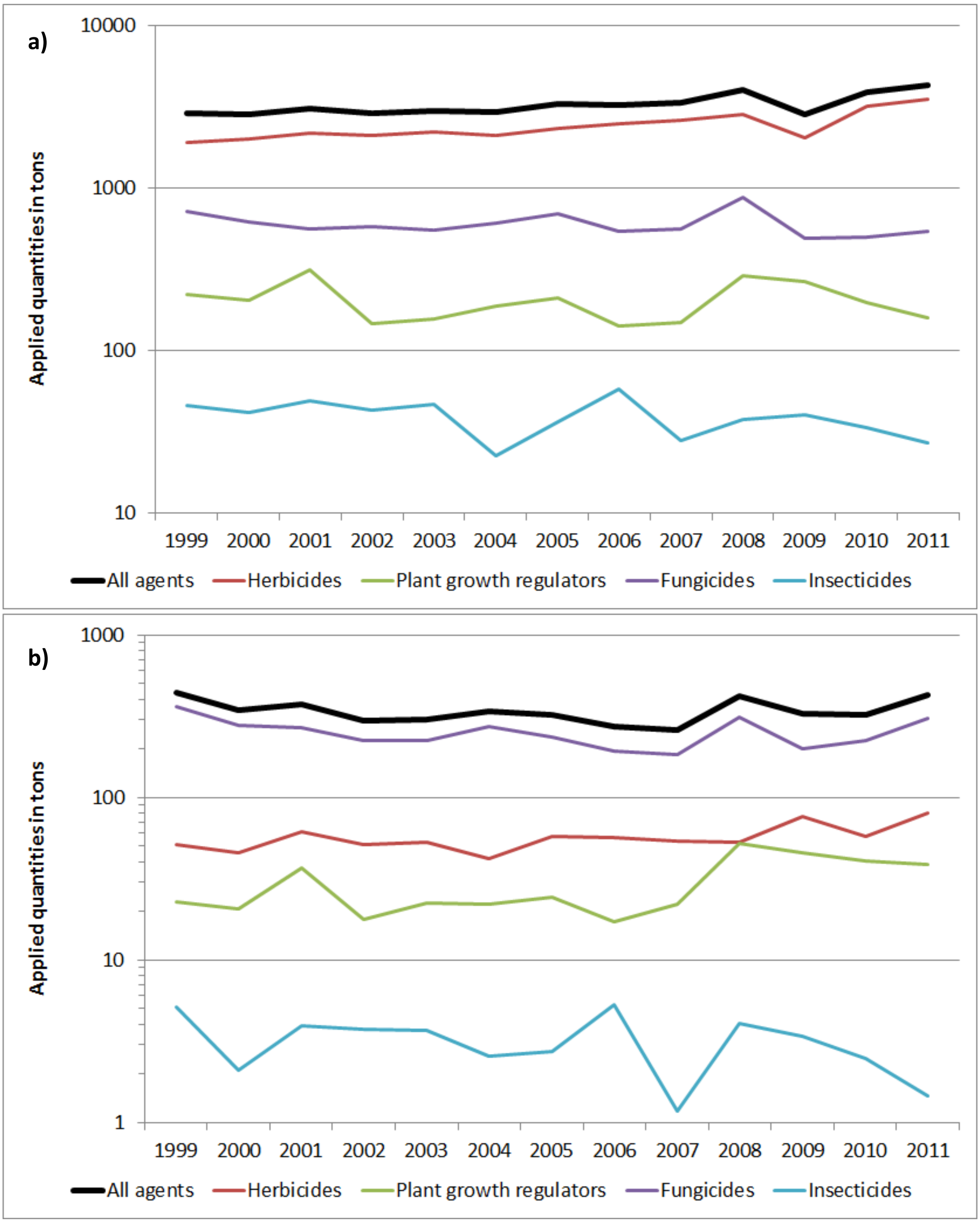

Figure S1: Trend in applied quantities (tons), aggregated into pesticide classes for a) the whole year and b) May only

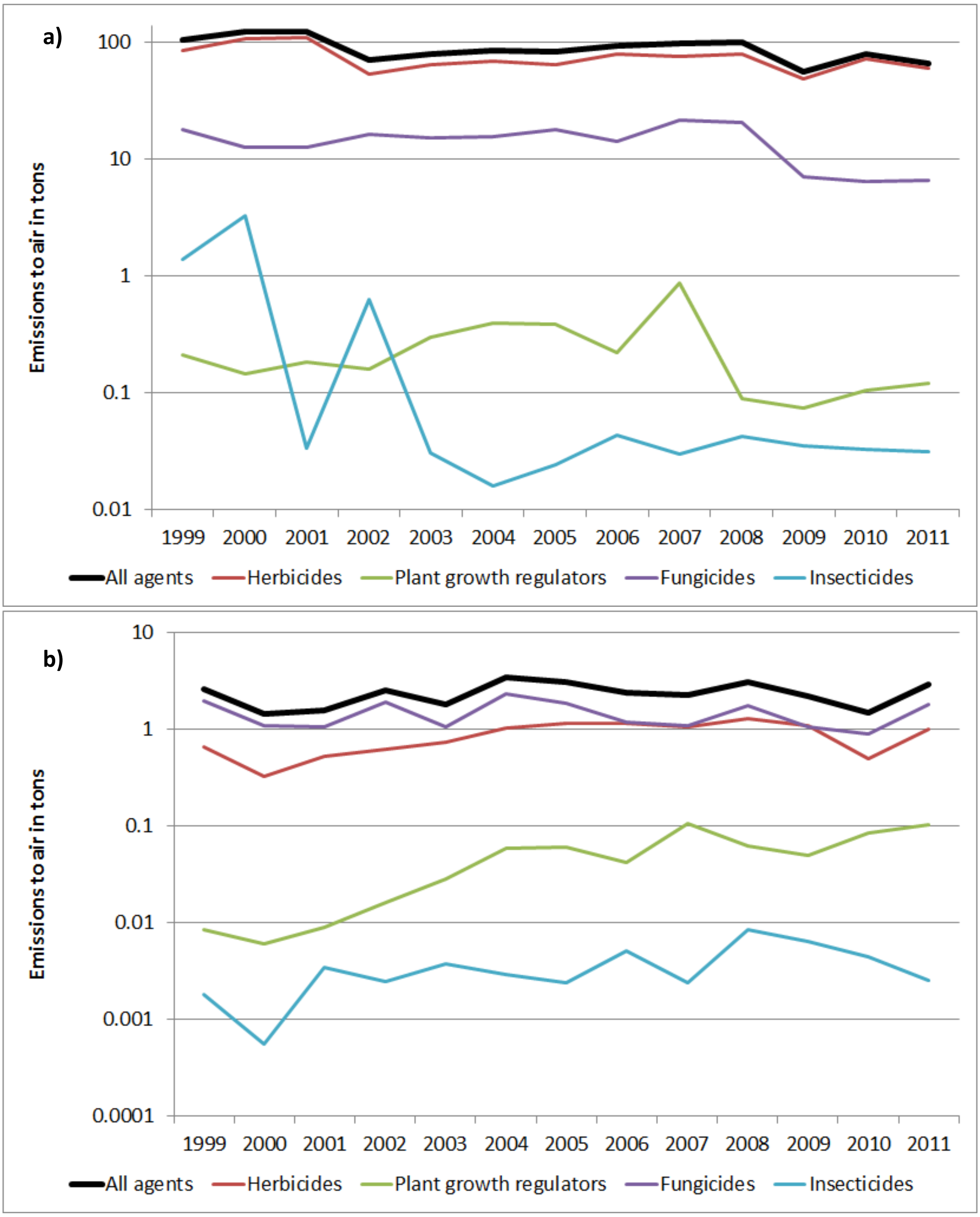

Figure S2: Trend in emissions to air (tons), aggregated into pesticide classes for a) the whole year and b) May only

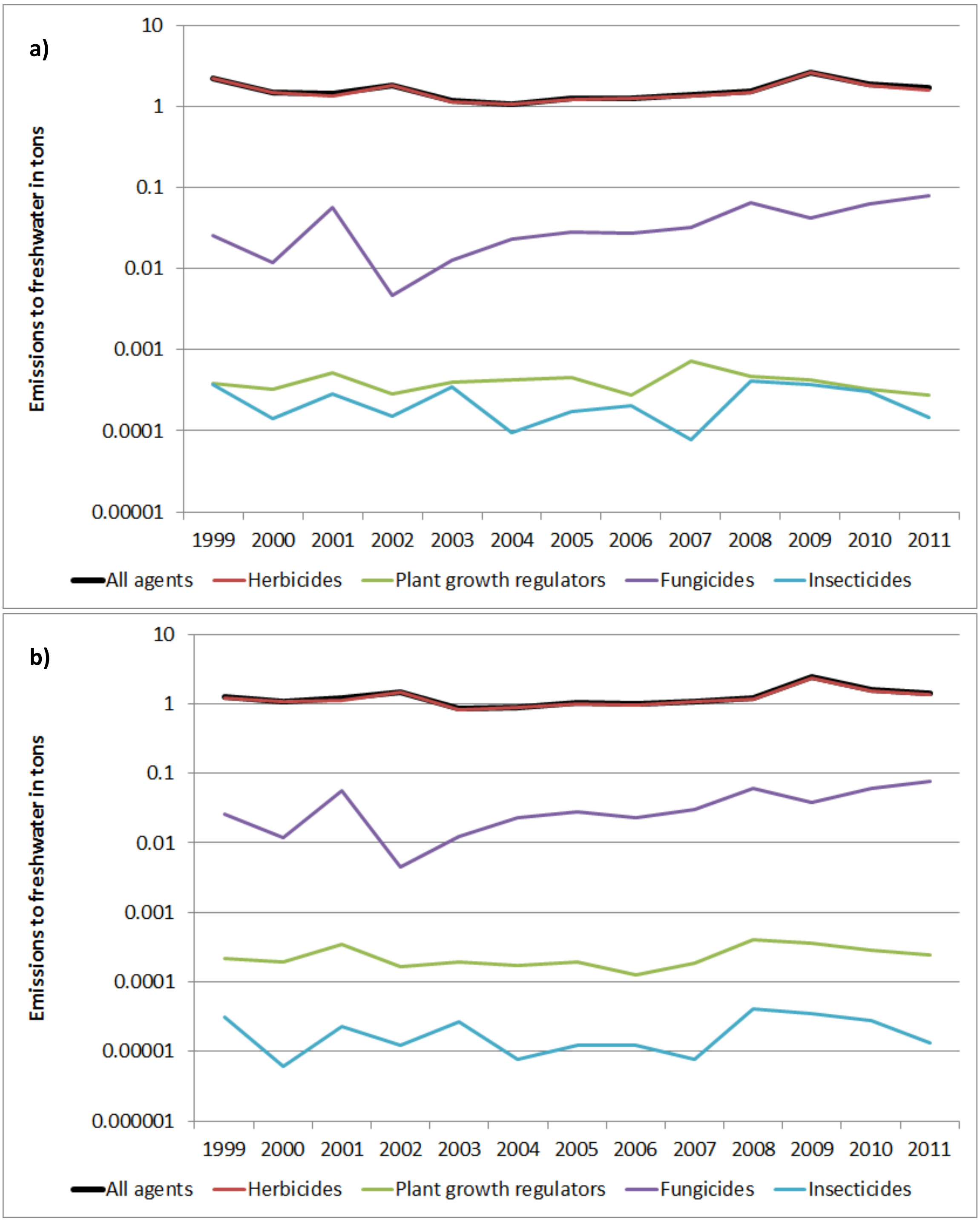

Figure S3: Trend in emissions to freshwater (tons), aggregated into pesticide classes a) the whole year and b) May only

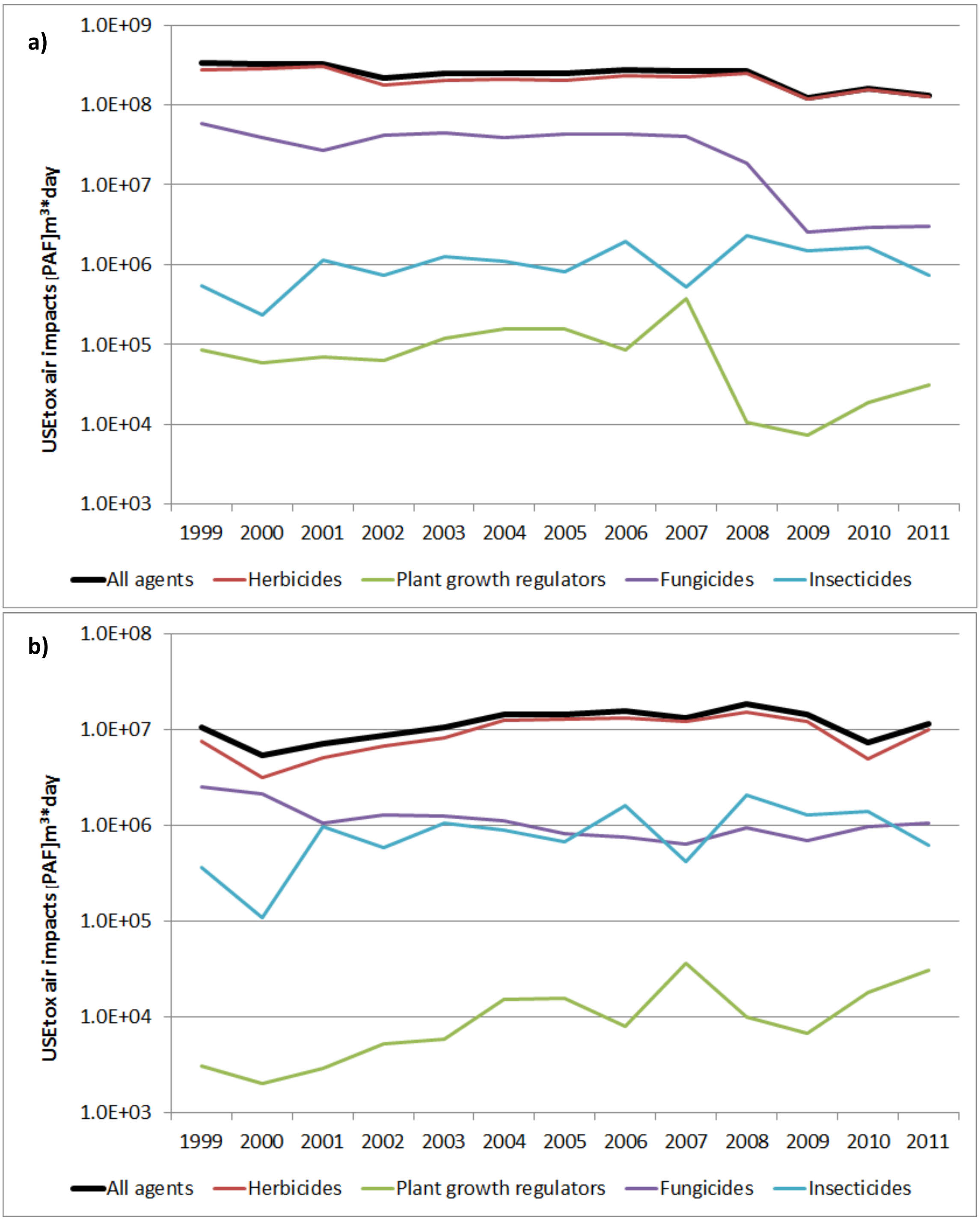

Figure S4: Trend in USEtox air impacts ([PAF]m3*day), aggregated into pesticide classes for a) the whole year and b) May only

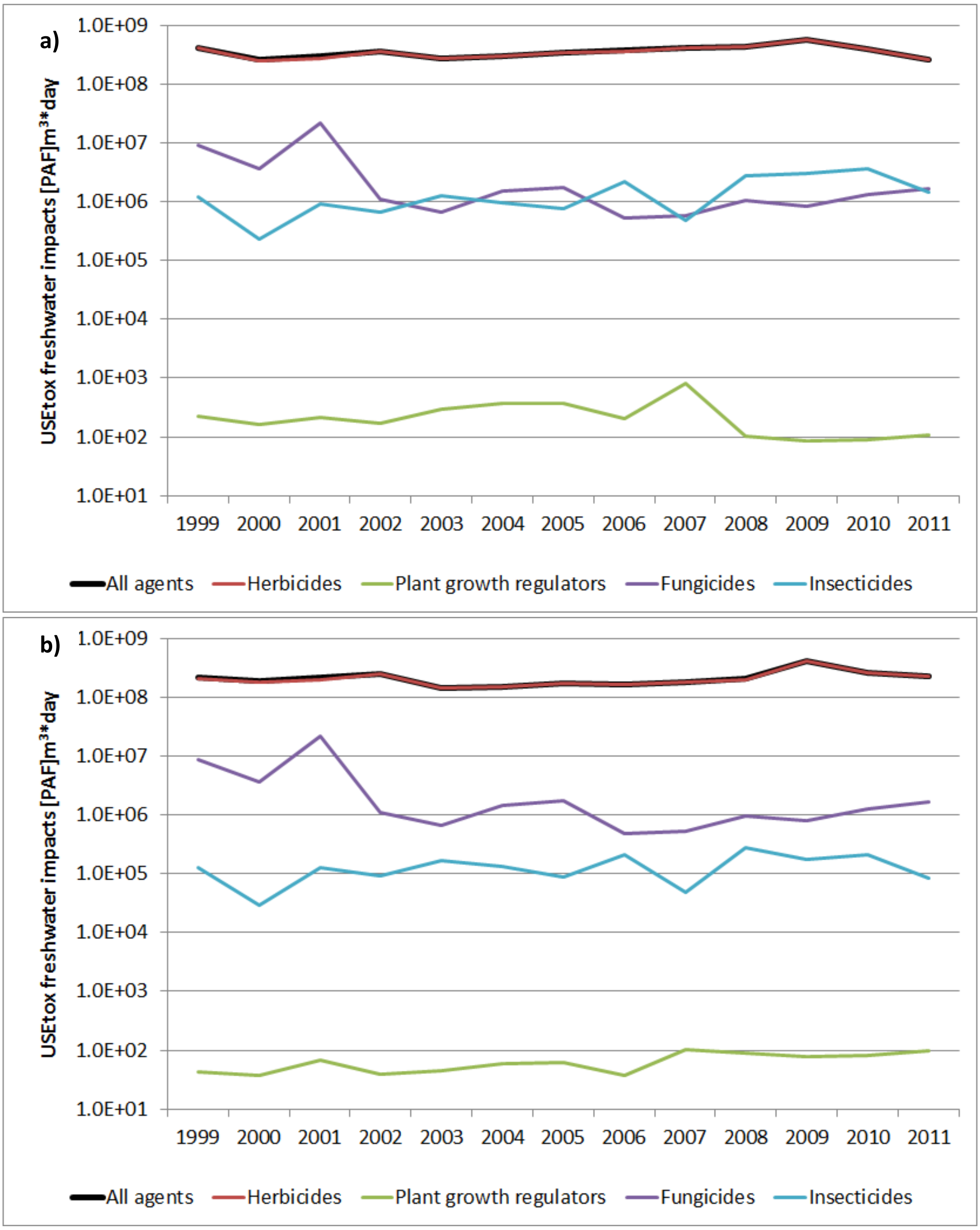

Figure S5: Trend in USEtox freshwater impacts ([PAF]m3*day), aggregated into pesticide classes for a) the whole year and b) May only

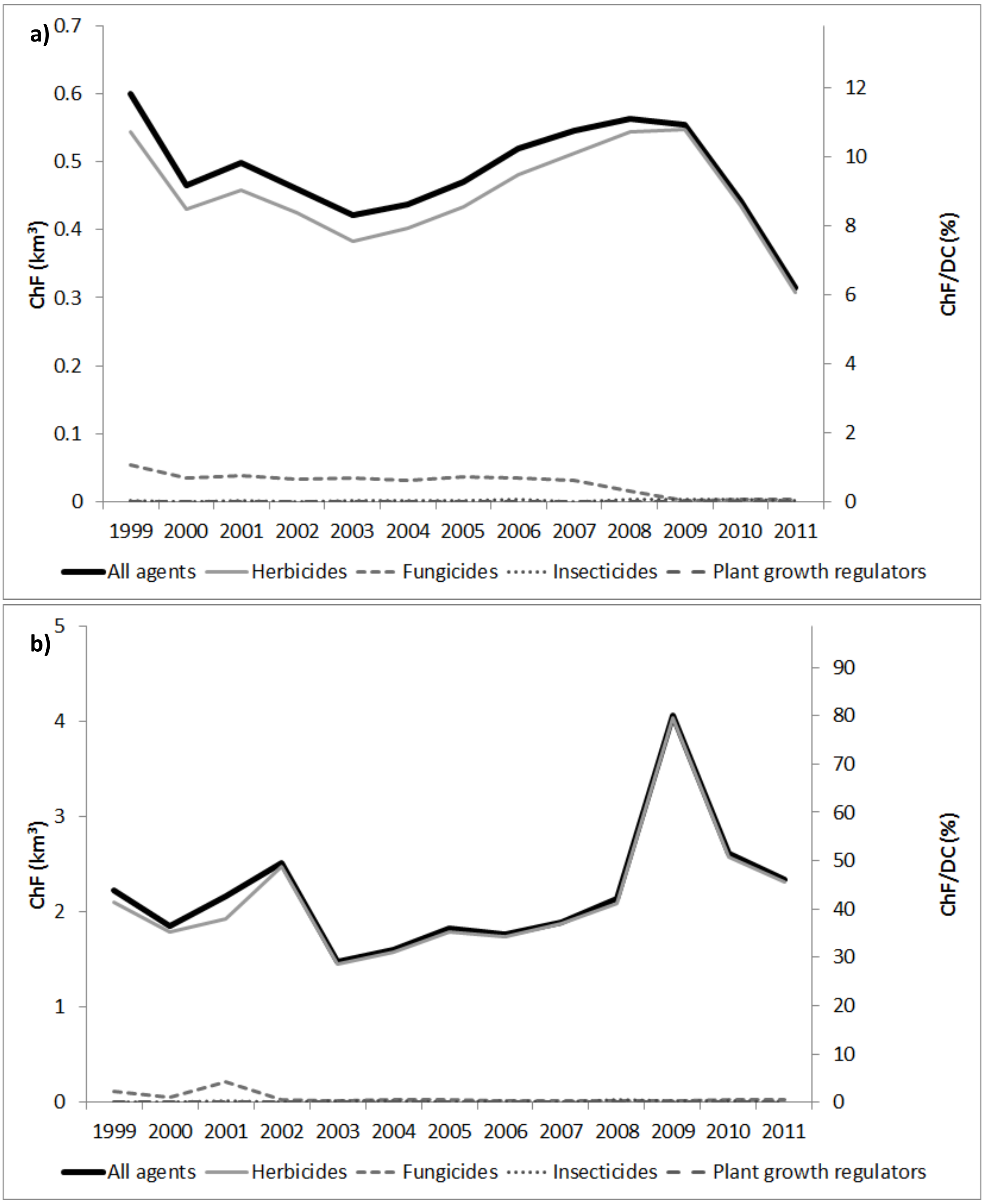

Figure S6: Trend in chemical footprint (km3) and ChF/DC fraction, aggregated into pesticide classes for a) the annual average and b) May average

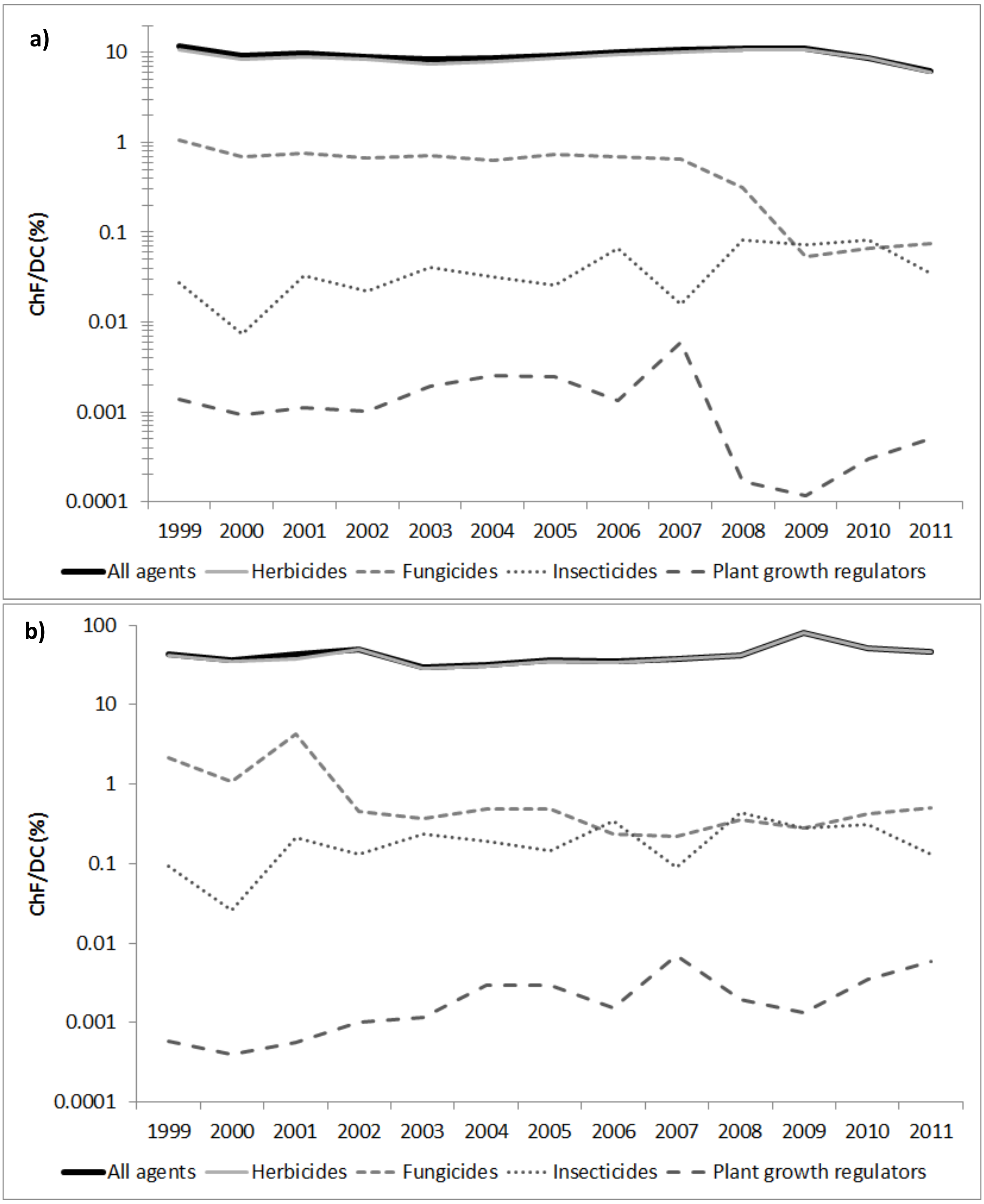

Figure S7: Trend in chemical footprint (km3) and ChF/DC fraction, aggregated into pesticide classes, logarithmic, for a) the annual average and b) May average

S7 Additional uncertainties

Table S10 presents an overview of uncertainties discussed in the manuscript in addition to three model structure uncertainties general in LCA (discussed below).

Table S10: Elements of uncertainty assessment and corresponding impact on ChF results

Type	Uncertainty	Magnitude and direction	Potential to reduce uncertainty
Parameter	Emissions inventory	Unknown underestimation	Improve substance coverage and data quality. Cross-check with comprehensive inventories with overlapping coverage
	Dilution Capacity (DC)	Up to one order of magnitude of DC for individual lakes	Base prediction of volume from surface area on more data points and include topography of landscape as explaining variable[33]
	USEtox parameters: EC50s, $t_{½}$ for water, soil and air	Several orders of magnitude depending on the substance	Increase measurement quantity and quality of chemical and ecotoxicological properties
Model structure	Assuming default β	25-800% of estimation.* Reduced in large inventories	Substance specific β can be calculated if the toxicity measures on which EF_{USEtox} is based become available. Alternatively a professional tool can be applied, such as AiiDA[34]
	Assuming default ɣ	20-1000% of estimation.** Reduced in large inventories	Including measurements of all substances that have been tested at both the EC50 and NOEC endpoint in the calculation of the default ɣ
	HC5(NOEC) as a safe reference point	Unknown	Use NOEC of key species or knowledge of food web collapse
	Generic fate model	Unknown	Application of local fate models to calculate site specific fate factor
	Assuming effect addition	Unknown, but likely to be relatively small[38]	Apply msPAF approach[35]
	Steady-state assumption	Unknown underestimation in some cases	Define t as length of emission interval or use a dynamic model to calculate peak mass increase
	Disregarding metal speciation	Up to 2 orders of magnitude.[39] Reduced in large inventories	Ongoing research is expected to improve our understanding of the consequences of metal speciation

	Disregarding transformation products	Underestimation of up to 5 orders of magnitude in extreme cases	Include known transformation products as sensitivity check of conclusion of study
	Choice of characterization model	1-2 orders of magnitude for individual substances. Reduced in large inventories	-

* Based on Pennington et al.[14], who reports that for most chemicals exhibiting a specific mode of action β is in the range of 0.2-0.7.

**Representing the range of ɣ for the 11 chemicals evaluated (se supporting information).

The disregard of metal speciation and its consequences for bioavailability by state of the art impact assessment models also represent model structure uncertainty. This is problematic since emissions from metals often dominate aggregated impact; in case 1 they were found to contribute on average 90% to the total ChF, mainly driven by copper and zinc emissions to freshwater. Gandhi et al.[36] suggested means to include these parameters in the characterization model and found that this caused USEtox characterization factors to change with up to two orders of magnitude, depending on the metal and freshwater archetype. However no consistent bias was found, meaning that the current CFs for individual metals may be either under- or overestimated.

Degradation is an important removal pathway in multimedia fate models associated with model structure uncertainty, since transformation products are usually neglected, even though they may turn out to have an impact comparable to their parent compounds.[37] This is especially relevant for large organic molecules (such as pesticides) likely to form transformation products before they completely degrade. A challenge for including transformation products in the ChF is that experimental measurement for their physical, chemical and toxicological properties are largely missing, which necessitates the reliance of uncertain ECOSAR estimates. Van Zelm et al.[37] demonstrated that the inclusion of transformation products resulted in a median impact increase that varied from negligible to more than 5 orders of magnitude.

No single characterization model can claim to have the "right" model structure for all types of applications. Rosenbaum et al.[38] observed that model structure differences between the most used characterization models for freshwater ecotoxicity resulted in 1-2 orders of magnitude variations for aquatic ecotoxicity impacts across the substances covered by USEtox. This means that the assessment of individual chemicals is highly uncertain, but that the uncertainty is smaller for aggregated assessments of large chemical inventories, since individual contributions being either under- or overestimated in combination have been found to more or less cancel each other out.

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II

Introducing Carrying Capacity Based Normalization in LCA: Framework and Development of References at Midpoint Level

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Introducing carrying capacity based normalisation in LCA: framework and development of references at midpoint level

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Abstract

Purpose: There is currently a weak or no link between the indicator scores quantified in life cycle assessment (LCA) and the carrying capacity of the affected ecosystems. Such a link must be established if LCA is to support assessments of environmental sustainability and it may be done by developing carrying capacity-based normalisation references. The purpose of this article is to present a framework for normalisation against carrying capacity-based references and to develop average normalisation references (NR) for Europe and the world for all those midpoint impact categories commonly included in LCA that link to the area of protection Natural environment.

Methods: Carrying capacity was in this context defined as *the maximum sustained environmental intervention a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert.* A literature review was carried out to identify scientifically sound thresholds for each impact category. Carrying capacities were then calculated from these thresholds and expressed in metrics identical to midpoint indicators giving priority to those recommended by ILCD. NR was expressed as the carrying capacity of a reference region divided by its population and thus describes the annual personal share of the carrying capacity.

Results and Discussion: The developed references can be applied to indicator results obtained using commonly applied characterisation models in LCIA. The European references are generally lower than the global references, mainly due to a relatively high population density in Europe. The references were compared to conventional normalisation references (NR') which represent the current level of intervention for Europe or the world. For both scales the current level of intervention for climate change, photochemical ozone formation and soil quality were found to exceed carrying capacities several times.

Conclusion: The developed carrying capacity-based normalisation references offer relevant supplementary reference information to the currently applied references based on society's background interventions by supporting an evaluation of the environmental sustainability of product systems on an absolute scale.

Recommendations: Challenges remain with respect to spatial variations to increase the relevance of the normalisation references for impact categories that function at the local or regional scale. For complete coverage of the midpoint impact categories, normalisation references based on sustainability conditions should be developed for those categories that link to the areas of protection Human health and Natural resources.

Keywords: Carrying capacity, normalisation, impact assessment, midpoint, sustainability conditions, threshold, severity, single score.

1. Introduction

Recent years have seen an increasing focus on environmental sustainability of products and technologies and a growing use of LCA and life cycle thinking in industry and the public sector. Still, the state of the environment is deteriorating globally by and large (MEA 2005; Steffen et al. 2004). This trend reflects that increases in eco-efficiency, achieved with the aid of LCA, are generally insufficient to offset the effects of an increasing global population that is achieving a higher material affluence. With many environmental impacts on the rise globally, the end goal of eco-efficiency improvements becomes increasingly important, namely that ecological impacts and resource intensities of product life cycles should be reduced to "…a level at least in line with the Earth's estimated carrying capacity" (WBCSD 2000). This end goal can be seen as a condition for environmental sustainability, originally defined as "…seek[ing] to improve human welfare by protecting the sources of raw materials used for human needs and ensuring that the sinks for human wastes are not exceeded, in order to prevent harm to humans" (Goodland 1995). Attempts to quantify carrying capacities have been made for decades most recently at the global scale through the introduction of the planetary boundaries concept (Rockström et al. 2009; Steffen et al. 2015).

Carrying capacity is currently considered in some LCA indicators, for instance in the form of critical loads for terrestrial acidification in Posch et al. (2008). In such indicators only interventions above carrying capacities are accounted for, meaning that resource uses and emissions that push a natural system closer to carrying capacity exceedance get a free ride. If LCA is to support a development towards environmental sustainability, understood as the non-exceedance of carrying capacities, measures of how much environmental intervention change the level of carrying capacity exceedance are not sufficient for decision support. In other words the path to environmental sustainability cannot be illuminated solely by indicators designed to measure environmental unsustainability. Existing LCA indicators must therefore be supplemented by measures that quantify the share of carrying capacity occupied by environmental interventions of a studied product system. Such measures can be established by using carrying capacity as environmental sustainability reference in LCA. A first step was taken by Hauschild and Wenzel (1998) who derived carrying capacity based distance-to-target weighting factors, albeit using varying definitions of carrying capacity across life cycle impact categories. Tuomisto et al. (2012) recently attempted to adapt initial planetary boundaries of Rockström et al. (2009) as weighting factors for 8 impact categories. Following the suggestion of Sala et al. (2013) in the context of life cycle sustainability assessment we here propose to use carrying capacity as consistent environmental sustainability reference in the normalisation step of LCA to facilitate the comparison of indicator scores to sustainable levels of interventions. According to ISO 14044, normalisation is "the calculation of the magnitude of the category indicator results relative to some reference information. The aim of the normalisation is to understand better the relative magnitude for each indicator result of the product system under study" (ISO 2006). In existing normalisation practice the reference information is commonly the sum of all characterized environmental interventions taking place in a specified year within a specified region, often scaled per capita (Laurent et al. 2011a). Normalisation thus allows for the translation of interventions in person equivalents (or person years) and facilitates some level of comparison across impact categories. However since common references are solely based on activities within the technosphere they cannot be used to compare and aggregate the severity of different types of interventions in the ecosphere. The subsequent weighting step is designed to capture the severity of characterized interventions, but as weighting is often based on personal perspectives on the prioritization of problems or policy goals, this expression of severity has a strong subjective element, which is also why ISO 14044 does not allow weighting in "LCA studies intended to be used in comparative assertions intended to be disclosed to the public" (ISO 2006). Without weighting the user of the LCA results is left with the normalized results. When understanding carrying capacity occupation as a measure of severity normalizing according to carrying capacity instead of total characterized interventions can improve the representation of the severity of different interventions.

The purpose of this article is to present a framework of carrying capacity-based normalisation references in LCA and to develop European and global carrying capacity-based normalisation references compatible with characterised indicator scores at midpoint for impact categories that link to the area of protection Natural environment. After presenting definition and framework, the concept of carrying capacity is made operational for Life Cycle Impact Assessment (LCIA), and European and global carrying capacity based normalisation references s for each midpoint indicator are developed. The new references are analysed by internal comparison and comparison to traditional normalisation references and their implications are discussed followed by an outlook.

2. Methods

2.1. Definition and operationalization

Carrying capacity generally refers to a certain quantity of X that some encompassing Y is able to carry (Sayre 2008). X and Y can refer to different entities depending on the discipline in which carrying capacity is applied.[2] In all applications carrying capacity aspires to idealism, stasis, and numerical expression (Sayre 2008). In ecology, for instance, carrying capacity describes the maximum equilibrium number of organisms of a species (X) that a given environment (Y) in theory can support indefinitely (Odum 1971). In the common definition of eco-efficiency (WBCSD 2000) X is impacts of unspecified environmental interventions and Y is the planet. In this form carrying capacity thus acts as the boundary between global environmental sustainability and unsustainability. Following this use of the term we define carrying capacity as *the maximum sustained environmental intervention a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert.* Here a natural system may refer to ecosystems or, more broadly, Earth´s interacting physical, chemical, and biological processes, which for instance make up the climate system. By considering both functioning and structure our carrying capacity definition aims for a balanced approach: Whereas the concept of ecosystem functioning may have an anthropocentric bias, in that it tends to focus on functions valuable to humans, the concept of ecosystem structure is eco-centric because no judgement is made on the relative inherent value of organisms.[3]

[2] Wildlife management, chemistry, medicine, economics, anthropology, engineering, and population biology are listed as examples by Sayre (2008).

[3] The concept of resilience may offer a bridge between anthropocentric and eco-centric approaches to environmental management since studies generally show that ecosystems with high genotype- and species diversity has a high resilience, meaning in general terms, that they are

We calculated carrying capacities from science based thresholds identified in the literature. Thresholds are numerical values of control variables, which in turn are numerical indicators of the structure and/or functioning of natural systems (Scheffer et al., 2001; Carpenter et al. 2001; Steffen et al. 2015). In the example of aquatic eutrophication a threshold can be expressed as a specific nutrient concentration (the control variable), which demarcates an oligotrophic (clear water) stable state from a eutrophic (turbid water) stable state, both characterized by distinct ecosystem structure and functioning. When thresholds are crossed, reverting the natural system to the original state can require a considerable amount of time with reduced interventions due to the initiation of feedback mechanisms stabilising the natural system in the new state after the threshold crossing. Here we characterize an interaction between humans and natural systems that does not lead to the exceeding of thresholds as environmentally sustainable.

Figure 1a shows the impact pathway for the example of how demand for food drives a chain of events that ultimately leads to increased risk of threshold exceedance for nutrients, which would entail significant impacts on structure and functioning of the affected aquatic ecosystem(s). Figure 1b shows the elements of an LCA that are used as indicators for and mechanistic translators between the points of the impact pathway in Figure 1a and shows conceptual cause/effect curves for the translation between points. Here we use "environmental interference" as a generic term for anthropogenic changes to any point in the impact pathway. Here we expressed carrying capacity at the point in the impact pathway where the concerned midpoint indicator expresses environmental interference. A translation from threshold to carrying capacity therefore involved different LCA elements depending on the point of the impact pathway, marked with a cross in Figure 1c, where the concerned midpoint indicator is expressed (see Section 3). For instance for indicators expressed at the pressure point the translation from threshold to carrying capacity involved a fate factor. For impact categories where LCIA models did not model the control variable for which the science based threshold was expressed, alternative approaches were taken in translating threshold to carrying capacity (see Section 3).

better at adapting to sudden changes in conditions than ecosystems with lower diversity (Scheffer et al., 2001; Carpenter et al. 2001). Thus the protection of ecosystem structure can be seen both as eco-centric and as being in the enlightened self-interest of man.

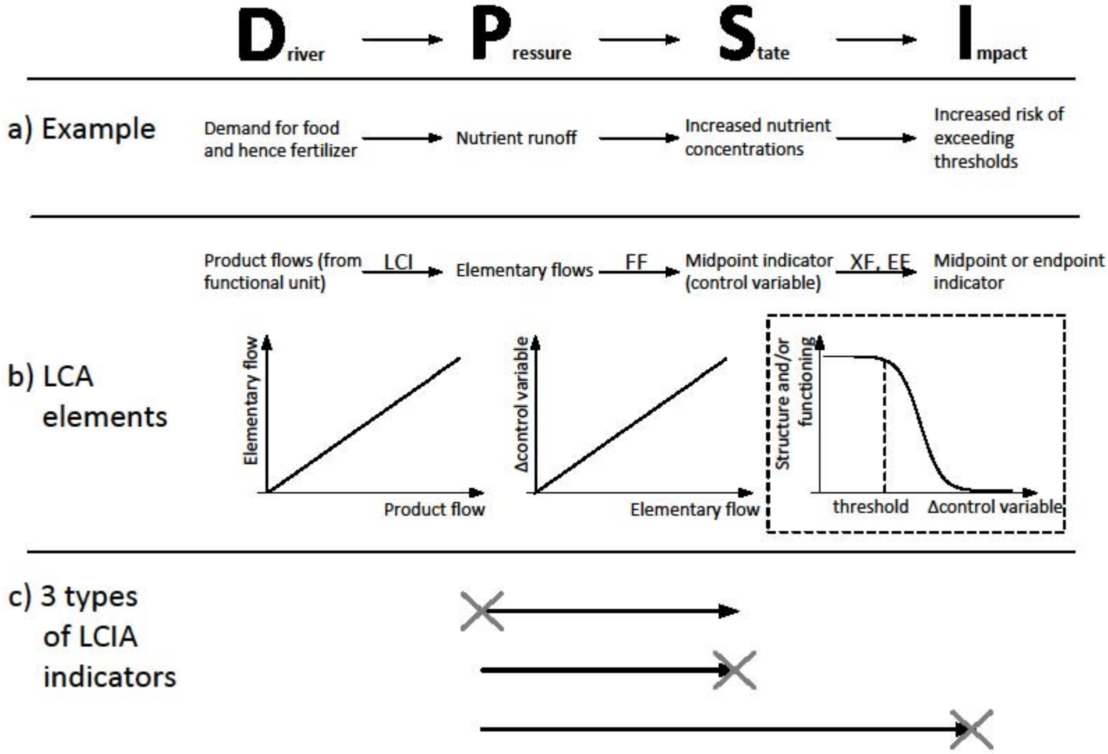

Figure 1: Elements of LCA placed in the DPSIR impact pathway framework (EEA, 1999) (response category not included). Figure 1a shows the example of an impact pathway leading to aquatic eutrophication. Figure 1b maps elements of LCA and their interactions. The punctured frame around the cause/effect curve between the state and impact points indicate that our adopted science based thresholds are external references to LCA for impact categories where thresholds are not considered by LCIA models. Figure 1c shows three types of midpoint indicators characterised by the point in the impact pathway where interferences are modelled (arrow) and expressed (cross).

Our carrying capacity definition is concerned with environmental sustainability and we therefore only derived carrying capacities for midpoint impact categories linking to the area of protection Natural environment. References based on sustainability conditions for impact categories linking to the areas of protection Human health and Natural resources may also be developed, but this falls outside the scope of this article. Carrying capacities were hence quantified for the following ten midpoint categories from the EU Commission's ILCD methodology (Hauschild et al. 2013): climate change, ozone depletion, photochemical ozone formation, terrestrial acidification, terrestrial eutrophication, freshwater eutrophication, marine eutrophication, ecotoxicity, land use and water depletion.[4] Several

[4] Ionizing radiation effects on the natural environment was excluded since the recommended LCIA model was classified as interim by Hauschild et al. (2013).

LCIA models exist for calculating indicator scores within each of these impact categories. When possible we followed the recommendations for best existing practice by Hauschild et al. (2013) when choosing the characterisation model and factors with which NR should be compatible. Exceptions were made when recommended models were of a marginal nature. Marginal characterization models base translations between points in the impact pathway on the derivative at the estimated current level of environmental interference. Because carrying capacities should ideally be calculated without considering background interference (see below) marginal characterization models were replaced by characterization models using a linear approach (i.e. using the same factors to translate between points in the impact pathway no matter the current level of interferences) when these were available. This procedure led to the replacement of ILCD recommended models for terrestrial acidification, terrestrial eutrophication, land use and water depletion by models using a linear approach.

2.2. *Derivation of normalisation references*

Normalisation references (NR) were calculated as the carrying capacity (CC, indicator score/year) for impact category i in region j, divided by the population in the region (P):

$$NR_{i,j} = \frac{CC_{i,j}}{P_j}$$

When dividing characterised LCIA results by NR they are converted into normalized results expressed in units of person equivalents (or person years). Here 1 person equivalent can be interpreted as a level of environmental intervention equivalent to the annual personal share of the carrying capacity for impact category i. This normalisation replaces the traditional normalisation, where indicator scores of a product system is compared to those of society's background interventions (Laurent et al. 2011a). If $NR'_{i,j}$ denotes the traditional normalisation reference, $\frac{NR'_{i,j}}{NR_{i,j}}$ can be interpreted as a distance-to-target indicator, where a value above 1 means that the current per capita interventions exceed the carrying capacity and are hence environmentally unsustainable (Seppälä and Hämäläinen 2001).

2.3. *Spatial and temporal concerns*

The choice of reference region for the normalisation inventory depends on the spatial extent of the impact category. Local and regional scale impact categories such as freshwater depletion and aquatic eutrophication should ideally be related to carrying capacities of relevant local and regional territories corresponding to

the spatial information of the LCI. On the contrary global scale impact categories such as climate change and ozone depletion should be related to a single global carrying capacity. As a first step we here developed European (the continent, not the union) and global average carrying capacities for each impact category. Issues related to spatial variation are further discussed in Section 4.
Carrying capacities are in practice dynamic due to: 1) Natural dynamics related to for instance the diurnal and seasonal cycles and stochastic weather events. 2) Anthropogenic interventions that can lead to temporary or permanent carrying capacity reductions if thresholds are exceeded. For instance if a reproductive threshold for a fish stock is exceeded, its carrying capacity expressed as a maximum sustainable yield (kg fish caught per year) will decrease temporarily. Likewise if the threshold of a natural system has been exceeded the original carrying capacity could in theory decrease if parts of the natural system, such as bacteria capable of metabolising pollutants, have been weakened or entirely eliminated due to the threshold exceedance. Here we did not consider the effects on carrying capacity caused by natural dynamics because it would involve complex dynamic modelling and because the short time scale of some natural dynamics, often hours to months, is incompatible with the limited time information of typical LCIs. For impact categories of a dynamic nature, such as photochemical ozone formation, we instead expressed thresholds at a form compatible with the time constraints of relevant LCIA models. We also did not consider dynamics in carrying capacity caused by human interventions because carrying capacities were calculated from ideal scenarios where interactions between natural and humans systems are at a steady state characterized by numerical values of control variables being below threshold values. In summary, calculated carrying capacities were treated as static in this work, which is in line with the general understanding of carrying capacity as a static concept (Sayre 2008).

In calculating NR we applied the populations of 2010 (6.916 billion globally and 740 million for continental Europe (UNDESA 2012)).We do however note that NR can be considered time dependent because the human population, the denominator of formula 1, is changing in most regions and increasing globally. Practitioners may therefore choose a projected population for the median year of the time horizon considered in a study. For instance an LCA of a system that will be operating from 2015 to 2035 would then use the projected population in 2025 as P.

2.4. *Choice of precaution*

In our carrying capacity quantifications we adhered to the consensus within LCA modelling to aim for best estimates. Therefore whenever an uncertainty range or confidence interval was given for an identified threshold and parameters used to translate this threshold to a carrying capacity, the medium or average value was chosen, corresponding to a medium level of precaution. A best estimate approach is suitable in LCA where the purpose is to compare indicator scores across assessed product systems and impact categories. A more precautionary approach to quantifying carrying capacities, as e.g. taken by Rockström et al. (2009) and Steffen et al. (2015), may be more appropriate in other decision support contexts, e.g. the design of emission standards in a specified jurisdiction.

3. Results

The following sections present the principles behind the derivations of global average carrying capacity based normalisation references for each impact category and the choice of characterisation model in cases where the recommendation of ILCD on best existing practice for characterisation modelling were not followed. See Table 1 for a summary, S1 for a detailed description including derivations of European references, which were calculated in much the same way as global references, and S2 for calculations in a spreadsheet.

3.1. *Climate Change*

There is evidence of several thresholds in the climate system expressed as average temperature increases above pre-industrial levels. These include disintegration of the Greenland ice sheet (1-1.5°C), widespread bleaching of coral reefs (>1°C), broad ecosystem impacts with limited adaptive capacity (1-2°C), complete melting of the Greenland ice sheet, (3°C) and shutdown of thermohaline circulation (3°C) (Haines-Young et al. 2006). In comparison the current temperature increase is around 0.8°C (IPCC 2013). The crossing of each of these thresholds can lead to irreversible changes in the functioning of the climate system with cascading effects on functioning and structure of various eco-systems. Here we propose one carrying capacity based on the 2°C target, which aims to limit global warming to 2 degrees above pre-industrial levels, and another more precautionary carrying capacity based on reducing current radiative forcing from greenhouse gases to 1 W/m^2 (corresponding to a steady state temperature increase of 1.06 degrees above pre-industrial levels, see S1) as proposed by Rockström et al. (2009). The 2°C threshold has highest acceptance as a policy target, while the 1 W/m^2 threshold is most in line with our definition of carrying capacity, since a temperature increase of 2°C will possibly lead to irreversible changes in functioning and structure of the climate system (Rockström et al. 2009). These thresholds were converted into carrying capacities, expressed at the pressure point of the impact pathway as GWP100 based kg CO_2-eq/year. This conversion was made using the GEOCARB model for CO_2 (Berner & Kothavala 2001) and the model of Shine et al. (2005) for other greenhouse gases, from which we calculated the sustained level of emissions that for each greenhouse gas alone would lead to a steady state concentration corresponding to each of the two proposed thresholds.[5] The carrying capacity was then calculated as the average of the GWP100-based indicators of all gasses, weighted according to their contribution to the total climate change indicator score in 2010, and this lead to a NR_{Global} of 985 kg CO_2-eq/pers/year for the 2°C threshold and 522 kg CO_2-eq/pers/year for the 1W/m^2 threshold (see S1 for details). The calculation of a weighted average was required due to the 100 year time scale of the GWP100 indicator and high variation of atmospheric life time of greenhouse gases. Had the time scale of the characterisation model instead been infinite, specific carrying capacities of the different gasses would be identical. The hidden variance of gas specific carrying capacities in the derived normalisation references is important to communicate to practitioners and decision makers. Specifically for CO_2 (having a very long atmospheric life time) the per capita carrying capacity is just 4-8 kg/year depending on the chosen threshold (see SI).[6]

3.2. Stratospheric ozone depletion

Rockström et al. (2009) proposed a planetary boundary of 5-10% decrease in column ozone levels for any particular latitude with respect to 1964–1980 values. The threshold was not based on a single well-established threshold in the climate system, but rather on the precautionary principle to acknowledge the complexity of the system of which knowledge is currently incomplete. Stratospheric ozone provides the regulatory function of filtering harmful ultraviolet radiation from the sun. Due to the long life time of many ozone depleting substances, ozone degradation in the stratosphere takes decades to recover. The threshold of 7.5% decrease in ozone levels (medium value) was converted to a carrying capacity expressed at the pressure point of the impact pathway in ozone depletion potential (ODP) based kg CFC-11-eq/year of Montzka & Fraser (1999). This conversion

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[5] The reason we could not use the FF of the GWP100 model to make the conversion is that the FF calculates a time integrated increase in radioactive forcing caused by an emission rather than the steady state increase in radioactive forcing or temperature required to convert the two thresholds (1 W/m^2 and 2°C) into carrying capacities according to our definition.

[6] Note that this carrying capacity is much lower than the 2050 goal of 2 tons per capita often mentioned in the climate change debate. The 2 tons per capita target was derived from the RCP2.6 reduction pathway designed to stay below the 2°C threshold by 2100 (IPCC, 2013; van Vuuren et al. 2011). In the year 2100 of the RCP2.6 reduction pathway CO_2 emissions are nearly zero, which is consistent with our low carrying capacity figures for CO_2.

was based on the model of Velders and Daniel (2013), which was used to calculate the sustained CFC-11-eq emissions that would lead to this decrease in ozone levels at steady state.[7] This resulted in a NR_{Global} of 0.078 kg CFC-11-eq/pers/year.

3.3. *Photochemical ozone formation*

We could not find a globally applicable threshold for this impact category and therefore based the carrying capacity on a time integrated ozone concentration threshold of 3 ppm*hour AOT40 for daylight hours during May-July which is applied in European regulation. AOT40 is an effect measure calculated as the accumulated ozone exposure during daylight hours above a threshold value of 40 ppb (EEA 1998). We here outline the derivation of the European carrying capacity and refer to SI for details and approximation at the global scale. The threshold, which was developed by WHO and adopted as a policy target by the European Environmental Agency (EEA 1998), was designed to prevent negative effects on growth and/or seed production for (semi-) natural sensitive perennial and annual species (Umweltbundesamt 2004).

We converted the time integrated threshold into an average concentration threshold of 44ppb ozone which applies to the 8 consecutive daily hours[8] with the highest ozone concentrations of May-July. This threshold was back calculated to a carrying capacity expressed at the pressure point of the impact pathway as kg NMVOC-eq/year applying the fate factor of the recommended indicator of Van Zelm et al. (2008) modified to calculate a change in maximum daily 8-h average ozone concentrations in Europe during May-July as a function of a change in emission. This resulted in a NR_{Europe} of 2.5 kg NMVOC-eq/pers/year.

3.4. *Terrestrial acidification*

Thresholds were here based on the critical load concept, for which acidification is defined as the highest deposition of acidifying compounds that will not cause chemical changes leading to long-term harmful effects on ecosystem structure

[7] We could not use the FF of CFC-11 of the ODP model because it is expressed relative to a reference substance (CFC-11) and not as an absolute steady-state ozone response to changes in emission.

[8] Although the number of daylight hours exceed 8 per day during May-July at all latitudes within Europe, we chose a time frame of 8 hours per day for the translation of the time integrated concentration threshold (3 ppm*hour AOT40) to a concentration threshold (44ppb) to be compatible with the time frame of the recommended indicator of Van Zelm et al. (2008). Had we chosen a longer time frame, e.g. 12 hours per day, the concentration threshold would have been only slightly lower (43ppb instead of 44ppb) and so would the resulting carrying capacity calculated.

and function (Umweltbundesamt 2004)[9]. Exceeding critical loads can lead to the reductions in crop and forest yields, which can take decades to recover (Hettelingh et al. 2007).. We calculated a world average critical load of 1170 mole H^+ eq/ha/year based on Bouwman et al. (2002), who developed a global map of critical loads based on acid buffering capacity of soils. From this critical load we subtracted global average natural depositions of 90 mole H^+ eq/ha/year. We converted the threshold (critical load) to a carrying capacity expressed at the state point of the impact pathway as mole H^+ eq deposition/year to be aligned with the OT indicator of Posch et al. (2008) based on average European conditions. This indicator was chosen instead of the indicator recommended by ILCD, Accumulated exceedance of Posch et al. (2008), because that indicator is of a marginal nature as it accounts for the share of emissions depositing on soils for which critical loads are modelled to be exceeded by background depositions. For this impact category the carrying capacity was to be expressed at the same point in the impact pathway as the threshold (the state point). Therefore the carrying capacity was simply calculated by multiplying the global average critical load with the global terrestrial area ($1.49*10^{10}$ ha)This resulted in a NR_{Global} of $2.3*10^3$ mole H^+ eq/pers/year.

3.5. Terrestrial eutrophication

Again thresholds were based on the critical load concept, which for terrestrial eutrophication is defined as the highest deposition of nitrogen as NH_x and/or NO_y below which harmful effects in ecosystem structure and function do not occur according to present knowledge (Umweltbundesamt 2004). Exceeding critical loads can reduce crop and forest yields and changes in species compositions (disappearance of species adapted to nutrient poor conditions), which may be practically irreversible (Bobbink et al. 2010). We calculated a world average critical load based on the global critical load map of Bouwman et al. (2002), which was constructed by extrapolations from a study covering critical loads of natural and semi-natural vegetation in Europe. From this estimate we subtracted estimated global average natural depositions which gave a global threshold of 1340 mole N eq/ha/year. As for terrestrial acidification we converted the threshold to a carrying capacity expressed at the state point of the impact pathway as mole N eq deposition/year based on the OT indicator of Posch et al. (2008) which is based on average European conditions. This indicator was chosen instead of the one recommended by ILCD for the reason given for terrestrial acidification above. Again the carrying capacity was calculated by multiplying the global average critical load with global terrestrial area. This resulted in a NR_{Global} of $2.7*10^3$ mole N eq/pers/year.

3.6. *Freshwater and marine eutrophication*

For freshwater and marine eutrophication a threshold demarcates oligotrophic (clear water) from eutrophic (turbid water) states (Carpenter et al. 2001). Thresholds may vary spatially, depending on e.g. temperature, salinity and depth. We chose 0.3 mg P_{tot}/L as a generic threshold for freshwater (usually P-limited) based on Struijs et al. (2011) who stated that concentrations above this value are considered a potential cause of encroachment of aquatic life due to nutrient enrichment. For marine environments (usually N-limited), we chose 1.75 mg N_{tot}/L as the medium of the concentration limit range proposed by de Vries et al. (2013) in their development of planetary boundaries for nitrogen emissions. The concentration threshold was converted to a carrying capacity expressed at the pressure point of the impact pathway as increase in P (freshwater) and N (marine) concentrations to be compatible with the midpoint indicators of Struijs et al. (2009) based on average European conditions. For the conversion we used FFs of P and N of Struijs et al. (2009), which links a marginal emissions increase (kg/year) to a steady state concentration increase (kg P or N per m^3). After a linear scaling to account for global water volumes and the subtractions of natural flows of N and P, NR_{Global} was calculated as 0.84 kg P eq/pers/yr for freshwater and 29 kg N eq/p/yr for marine waters.

3.7. *Freshwater ecotoxicity*

The carrying capacity calculation was based on the threshold HC5(NOEC), which has been adopted as a quality target in several regulatory frameworks, such as the EU Water Framework Directive (EC 2011). HC5(NOEC) is the concentration at which maximum 5% of species in an ecosystem are affected and it is derived from species sensitivity distributions, which are probabilistic models of the variation in sensitivity of all species in a model ecosystem to a particular stressor (Posthuma et al. 2002). The HC5(NOEC) threshold was converted to a carrying capacity expressed at the impact point of the impact pathway as [PAF]*m^3*day/year to be compatible with the spatially generic USEtox indicator (Rosenbaum et al. 2008). The conversion was carried out by modifying the effect factor of USEtox from being based on the HC50(EC50) effect level to being based on HC5(NOEC) following Bjørn et al. (2014). In accordance with USEtox full concentration addition was assumed, i.e. if two chemicals are each present at their HC5(NOEC) in the same freshwater volume then the carrying capacity of the compartment is assumed to be exceeded by 100%. The procedure resulted in a NR_{Global} of $1.9*10^4$ [PAF]*m^3*day/pers/year.

3.8. *Land Use*

To reflect the multitude of functions and services of land we calculated carrying capacities based on thresholds for two control variables representing different impact pathways. The first threshold concerns erosion regulation and the second threshold regional scale biodiversity.

The soil erosion carrying capacity was based on Verheijen et al. (2009), who provided a threshold interval for Europe of 0.3-1.4 ton/ha/year for 'tolerable soil erosion', defined as 'any actual soil erosion rate at which a deterioration or loss of one or more soil functions does not occur'. The threshold range was based on the estimated rate of natural soil formation caused by mineral weathering and dust deposition. We chose the middle value of 0.85 ton/ha/year and converted this to a carrying capacity expressed at the state point of the impact pathway as ton of eroded soil/(ha*year) to be compatible with global average CFs of the indicator for erosion resistance of Saad et al. (2013). The indicator of Saad et al. (2013) was chosen instead of the one recommended by ILCD based on soil organic matter (SOM) of Milà i Canals et al. (2007), because that indicator is of a marginal nature as it accounts for the change in SOM compared to an alternative land use scenario reference. As for terrestrial acidification and eutrophication the carrying capacity was expressed at the same point in the impact pathway as the threshold, the state point. Therefore the carrying capacity was simply calculated by multiplying the threshold with the global terrestrial area ($1.49*10^{10}$ ha).This gave a NR of 1.8 ton/pers/year.

The land use threshold for biodiversity was based on Noss et al. (2012), who meta-reviewed 13 studies that reported science-based local or regional conservation targets expressed as a share of natural lands that should be conserved, i.e. practically undisturbed by humans, to maintain sufficient levels of biodiversity in the region in question. Such conservation targets have the inbuilt perspective that loss of local biodiversity, due to e.g. intensive agriculture or infrastructure land use, is acceptable as long as regional biodiversity is maintained. The relationship between land use and regional biodiversity levels show threshold behaviour as ecosystems not directly affected by the land use (e.g. situated close to a clear-cut forest) are known to undergo state shifts due to the effects of neighbouring land use (Barnovsky et al. 2012; Noss et al. 2012). As a threshold we chose the median value, 31%, of the data series of Noss et al. (2012) for the share of terrestrial land that needs to be conserved as a threshold.[10] The threshold was converted to a carrying capacity expressed at the pressure point of the impact pathway as m^2*year/year land occupation to be directly compatible with any LCI. For reasons given above we did not align the carrying capacity with the ILCD recommended indicator and instead chose to align it directly to any LCI since the threshold is independent on types of land use (i.e. paved road counts as none-conserved land just as managed forest). The conversion of the threshold to carrying capacity was carried out simply by taking 31% of global terrestrial land. This gave a NR_{Global} of $1.5*10^4$ m^2*year/pers/year. In practice a set of CFs with the value 1 for all relevant elementary flows could be created in LCA software to form an indicator compatible with the NR.

Note that land transformations were not considered in the derivation of the two carrying capacities because indicators of land transformation are inherently marginal as they are based on an alternative land use scenario reference.

3.9. Water depletion

The carrying capacity was based on the so-called environmental flow requirements for good conditions (EFR_{good}), which is a threshold measure of the minimum water flow required to sustain rivers in a "good ecological state" (Smakhtin et al. 2004). This threshold was supplemented by another threshold for the minimum water flow required to sustain terrestrial ecosystems in the river catchment. In deriving a combined threshold for aquatic and terrestrial ecosystems we followed Gerten et al. (2013), who estimated the global accessible blue water resource (16.300km^3/year) and subtracted a global EFR_{good} quantification of 57% of blue water and another 30% of blue water to avoid physical water stress of terrestrial ecosystems. In the impact pathway of water depletion a change in pressure, expressed in m^3/year water consumed, causes a change in control variable, expressed in m^3/year water availability, of similar magnitude. EFR_{good} can therefore be interpreted as a pressure based carrying capacity and no conversion from threshold to carrying capacity was hence needed. As for the carrying capacity of land use related to regional biodiversity the carrying capacity is aligned directly to any LCI since the EFR_{good} estimates of Gerten et al. (2013) made no distinction between different types of blue water consumption such as lake or river water. We deviated from the ILCD recommended water scarcity indicator of Frischknecht et al. (2008), because this indicator is of a marginal nature as it

[10] This number is in good agreement with recent conclusions that around 34% of global terrestrial coverage should be conserved to achieve biodiversity protection goals given patterns and effects of current land conservation (Butchart et al. 2015)

models the scarcity created by background water consumption. This procedure gave a NR_{Global} of 306 m^3/pers/year. As for the land use impact category (regional biodiversity) a set of CFs with the value 1 for all relevant elementary flows could be created in LCA software to form an indicator compatible with the NR.

3.10. Comparison with traditional normalisation references and across spatial scale

Table 1 presents an overview of the developed carrying capacity-based normalisation references (NR) globally and for Europe and a comparison with traditional normalisation references based on characterized global background interventions (NR’). NR'_{global} was based on Laurent et al. (2013) who calculated global normalisation references for the ILCD methodology for the year 2010 (or 2000 for impact categories were more recent data was unavailable). NR'_{Europe} was based on Benini et al. (2014) and Sala et al. (2015) who calculated normalisation references for EU-27 for the ILCD methodology, also for the year 2010. When comparing NR'_{Europe} to NR_{Europe} it should be noted that NR_{Europe} has a wider geographical coverage as it is based on the European continent. For impact categories where our developed NR was not aligned with the ILCD methodology NR’ was calculated using the underlying inventories of Laurent et al. (2013) and Sala et al. (2015), with the exception of water depletion for which blue water consumption could not be extracted from the inventories of these two studies. More details can be found in SI1 and SI2.

Table 1: Developed global normalisation references based on carrying capacity, comparison across scales and with traditional normalisation references. Bold values indicate that NR'/NR fractions are above 1. Italics CF references mean compatibility with characterisation methods recommended by Hauschild et al. (2013).

Impact category	**NR_Global** (per person year)	$\frac{NR'_{Global}}{NR_{Global}}$	**NR_Europe** (per person year)	$\frac{NR'_{Europe}}{NR_{Europe}}$	$\frac{NR_{Global}}{NR_{Europe}}$	**CF compatibility**	**Threshold**
Climate change	985kg CO_2-eq	**8.2**	985kg CO_2-eq	**9.4**	1	*GWP100 (CO_2-eq) (Forster et al. 2007)*	Temperature increase of 2°
	522 kg CO_2-eq	**15**	522 kg CO_2-eq	**18**			Radioactive forcing increase of $1W*m^{-2}$
Ozone depletion	0.078kg CFC-11-eq	0.53	0.078kg CFC-11-eq	0.28	1	*ODP (Montzka and Fraser 1999)*	7.5% decrease in average ozone concentration
Photochemical ozone formation	3.8 kg NMVOC-eq	**15**	2.5 kg NMVOC-eq	**13**	1.6	*Tropospheric ozone concentration Increase (Van Zelm et al. 2008)*	Tropospheric ozone concentration of 3 ppm* hour AOT40
Terrestrial acidification	$2.3*10^3$ mole H+ eq	0.34	$1.4*10^3$ mole H+ eq	0.53	1.7	OT method of Posch et al. (2008)	Deposition of 1170 and 1100 mole H^+ eq*ha^{-1}*$year^{-1}$ globally and for the EU
Terrestrial eutrophication	$2.8*10^3$ mole N eq	0.13	$1.8*10^3$ mole N eq	0.30	1.5	OT method of Posch et al. (2008)	Deposition of 1340 and 1390 mole N eq*ha^{-1}*$year^{-1}$ globally and for the EU
Freshwater eutrophication	0.84kg P eq	0.74	0.46kg P eq	**3.22**	1.8	*P concentration increase (Struijs et al. 2009)*	P concentration of 0.3mg/L
Marine eutrophication	29 kg N eq	0.32	31kg N eq	0.55	0.95	*N concentration increase (Struijs et al. 2009)*	N concentration of 1.75 mg/L
Freshwater ecotoxicity	$1.9*10^4$ [PAF]*m^3*day	0.036	$1.0*10^4$ [PAF]*m^3*day	0.85	1.8	*CTU (Rosenbaum et al. 2008)*	HC5(NOEC)
Land use, soil erosion	1.8 tons eroded soil	**4.9**	1.2 tons	**9.3**	1.6	Saad et al. (2013), land occupation CFs only	Tolerable soil erosion of 0.85 tons*ha^{-1}*$year^{-1-}$)
Land use, biodiversity	$1.5*10^4$ m^2*year	0.42	$9.5*10^3$ m^2*year	0.79	1.6	LCI data, land occupation only	31% conserved land area

Water depletion	306 m^3	**1.3**	490 m^3	0.52	0.63	LCI data classified as blue water consumption	Conservation of 57% of river flows for aquatic ecosystems and 30% for terrestrial

NR'/NR-values above 1 mean that current levels of interventions exceed the carrying capacity and that normalized indicator scores will become higher when a traditional normalisation reference is replaced by a carrying capacity-based one. This is the case for climate change (both thresholds), photochemical ozone formation and land use (soil erosion) both at the global and European scale, for freshwater eutrophication at the European scale and for water depletion at the global scale. The NR'/NR ratios for the remaining impact categories are all below 1 and normalized indicator scores of these categories thus become smaller when replacing traditional normalisation references with carrying capacity based ones. When comparing across scale (column 6 in Table 1) it can be seen that for all impact categories except water depletion and marine eutrophication NR_{Europe} is smaller than NR_{Global}, which is mainly due to Europe's relatively high population density.

The interpretation of results for climate change, photochemical ozone formation, land use and water depletion is that humanity is globally unsustainable according to our carrying capacity definition. Global degrees of unsustainability are seemingly greatest for climate change (when carrying capacity is based on the 1 W/m^2 threshold) and photochemical ozone formation where in both cases characterized interventions need to decrease by a factor of 15, compared to those of the year 2010 and 2000 respectively, to reach sustainable levels characterized by no exceedance of reference thresholds on average.

For the remaining impact categories current interventions appear environmentally sustainable when averaging over the global situation because NR'_{global}/NR_{Global} is below 1. The relevance of this perspective is discussed in the next section.

4. Discussion and outlook

The new normalisation references are compatible with commonly used midpoint indicators and provide reference information of a different relevance than society's background interventions, giving better indications of the severity of interventions compared to sustainable levels. The references can be integrated in LCA software for the application in LCA studies. Practitioners should be aware of uncertainties of the references discussed below and that updated references in the future may replace the ones proposed here. Using the developed references in LCA serves mainly two purposes: 1) To provide absolute references that can inform criteria for environmental sustainability of systems. 2) To provide a scientific basis for aggregating indicator scores across impact categories in LCA.

4.1. *Criteria for environmental sustainability*

Regarding the first purpose the normalisation references offer a pedagogical expression of interventions in environmental sustainability person equivalents, which serves to communicate how large a share of the carrying capacity a given system or activity takes up. This can help shifting the perspective of environmental assessments from comparing eco-efficiencies of product systems to addressing eco-efficiency improvements required to achieve environmental sustainability at a societal scale (i.e. through the NR'/NR ratio). Criteria for environmental sustainability of societal subsystems are inherently subjective because they involve the allocation of carrying capacity to systems that meet different human needs (and wants). However it may be feasible to agree upon a moral rule that carrying capacities should be shared equally amongst people living within its geographical boundaries or an alternative rule that global carrying capacities should be shared equally within the global population.[11] Moral rules like these would not restrict personal freedom by enforcing a specific consumption pattern. Instead they would translate into equal personal carrying capacity budgets that could be used according to personal preferences, much like a salary. As a supplement to the perspective of personal carrying capacity consensus on the allocation of carrying capacity between products belonging to different sectors may be based on sector specific reduction scenarios of e.g. IPCC, IEA or national and municipal environmental strategies.

4.2. *Aggregation of normalized indicator scores*

Regarding the second purpose, the developed normalisation references allows for the aggregation of indicator scores expressed in carrying capacity occupation across impact categories to a single score. In this process an additional weighting step is needed as the exceeding of the considered carrying capacities are not necessarily equally severe for all categories of impact. Factors that influence the severity of exceeding a carrying capacity include the type of damage that is caused, the social and/or economic impact, the spatial extent, the time required for reversion of damage, whether a threshold is characterized by a hysteresis,[12] and effects

[11] The difference between these two rules is not trivial. Consider the potentially large differences between per capita domestic carrying capacities of Canada and Singapore for the many impact categories related to the availability of land and water as source or sink.

[12] A hysteresis is a phenomenon which causes the exceedance of a threshold to be difficult to revert because the natural system has entered a new stable state characterized by stabilizing feedback mechanisms. In practice this means that a reduction in environmental intervention of a similar magnitude as the increase in interventions that previously caused the threshold to be exceeded is not sufficient to bring the system back to its original state. Hysteresis has been observed for e.g. the response of shallow lakes to changes in phosphorous loadings (Scheffer 2001).

on other carrying capacities.[13] As an example it could be argued that carrying capacity normalised indicator scores for climate change should have a higher weight than corresponding scores for photochemical ozone formation, given for instance that effects of crossing climate system thresholds are both more pervading and difficult to reverse than the effects of crossing the tropospheric ozone threshold for vegetation used in this work.

4.3. Uncertainties and future work

The introduction of the carrying capacity based normalisation reference on one hand eliminates the inventory-related uncertainties that accompany the classical normalisation reference (NR'), and these uncertainties are large, especially for the toxicity-related impact categories (Laurent et al. 2011b). On the other hand additional uncertainty related to quantification of carrying capacity is introduced. A central question is whether control variables, and thus thresholds, should be located at midpoint or endpoint[14] in the impact pathway. In this work control variables, often expressed in a concentration metric, were located at midpoint. A control variables related to effects on species (e.g. potentially disappeared fraction of species, PDF) at endpoint could alternatively have be chosen consistently for all impact categories, along with a threshold value. Carrying capacity based normalisation references could then be calculated at either midpoint or endpoint from such an overarching threshold value. This approach is expected to lead to higher uncertainties than the approach taken here of calculating carrying capacities from thresholds at midpoint, because it would involve a translation through more processes in the impact pathway (i.e. from driver to impact in the DPSIR framework, see Figure 1). Also, a control variable at endpoint, such as PDF, is not necessarily a good indicator of ecosystem functioning (Mace et al. 2014), although it is a direct measure of ecosystem structure. Yet, a consistently chosen threshold value at endpoint would lead to the calculation of carrying capacities that reflect the same level of species protection across impact categories, which is appealing in the comparative setting of LCA. This approach should therefore be further explored.

[13] For instance increased run-off due to the exceedance of the climate change carrying capacity can lead to a higher loss of reactive nitrogen and phosphorous from fertilizer application, thereby increasing the risk of exceeding carrying capacities for freshwater and marine eutrophication. See Steffen et al. (2015) for elaboration on this topic.

[14] Midpoint is here understood as the point at which the impact pathway of different substances converge (Hauschild et al. 2013). Because this point of convergence varies the impact pathway location of the midpoint varies across impact categories. In comparison the endpoint is consistently located at the of the impact pathway and typically expressed in a metric related to the disappearance of species (Hauschild et al. 2013).

Another type of uncertainty relates to spatial variations. Our derived carrying capacities reflect average conditions of Europe and the world and have been developed to fit site generic characterisation factors. This is useful in LCA, where locations of environmental interventions are often not known with great accuracy. However the spatially generic approach hides variations emission fate and carrying capacity of receiving environments, which is problematic in cases where locations of environmental interventions are in fact known and spatially derived impact assessment models exist. Our spatially generic approach, combined with the fact that emission sources are rarely homogenously distributed in space, is the reason that our method predicts that carrying capacities have not been exceeded for the majority of impact categories (see Table 1 and Bjørn et al. (2014) for an elaboration of this issue for freshwater ecotoxicity). This prediction is invalidated by observations since exceedances of carrying capacities are quite frequent for many types of environmental interferences operating at the local to regional scale (MEA 2005; Steffen et al. 2015). A pragmatic way of accounting for this bias is to subtract the carrying capacity of remote areas, classified based on e.g. a population density threshold, from the calculation of spatially aggregated carrying capacities. Thereby land, water and air in scarcely populated areas would be considered unavailable as resources and for assimilating emissions, and the carrying capacity estimates would consequently be reduced. This was done by Gerten et al. (2013), who estimated the accessible blue water to be 40% of global blue water resources, meaning that roughly 60% of the theoretical global carrying capacity for water use (i.e. total flow minus environmental flow requirements) was considered unavailable This estimate of unavailable carrying capacity gives an impression of the extent at which our derived carrying capacities may be overestimated, but it needs to be assessed for each impact category since it is 0 for climate change and stratospheric ozone depletion and may be higher than 60% for other impact categories. Such a modification might change the ranking between the normalised indicator scores but it would not solve the problem of spatial variability in degrees of carrying capacity occupation of a given emission within the remaining non-remote areas where carrying capacity is judged available. Normalisation references could be developed at finer scales than what was demonstrated in this article to take into account spatial variation in carrying capacity and the spatial distribution of the processes making up an LCI. However at a high resolution (e.g. 0.5°·0.5°) such references would need to take into account transboundary emissions. Alternatively carrying capacity could be integrated in spatially differentiated characterisation models rather than in the normalisation step. In this way indicator scores could be expressed in hectare years, which could be compared to the availability of land, thus following the style of the ecological footprint indicator (Borucke et al. 2013).

Beyond the location of control variable in the impact pathway and the handling of spatial variations additional sources of uncertainties related to quantification of carrying capacity needs consideration: the selection of threshold on which to base the carrying capacity in some cases involves a choice between more alternatives. For instance we aimed to base carrying capacities on scientific consensuses on threshold reflecting the state of natural systems that should be protected to ensure their structure and functioning. Yet, a clear scientific consensus could not be identified in all cases. For example, the threshold for stratospheric ozone depletion (Section 3.2) was here based on the planetary boundary of Rockström et al. (2009), which is to a larger extent a precautionary first estimate than a scientific consensus, due to the imperfect understanding of the relationship between control variable and structure and functioning of natural systems. In other cases the relationship is better understood, but may not be characterized by a single sharp threshold, but rather by a sequence of thresholds or be close to linear (Dearing et al. 2014). In such cases value judgement on what can be considered a minimum environmentally sustainable level of structure and functioning is required for the calculation of carrying capacities. Other sources of uncertainties in the calculated carrying capacity based normalisation references are: 1) choice of structure and functioning to be protected (land is, for example, associated with a multitude of functions beyond erosion resistance and host of biodiversity (Saad et al. 2013)), 2) choice of control variable (for example, total concentration of nitrogen may not be the best control variable for indicating structure and/or functioning of marine ecosystems (HELCOM 2013)), 3) choice of impact pathway model to translate threshold to carrying capacity (the translation for photochemical ozone formation in this work, for example, involved different time frames and could be improved). Identifying all sources of uncertainties, analysing their magnitudes and consequently managing and reducing them is an important future task that could take point of departure in the proposal of Bjørn et al. (2015).

This article only provided normalisation references for midpoint impact categories that link to the area of protection Natural environment. To increase the usefulness of the references they should be supplemented with normalisation references based on sustainability conditions for the impact categories linking to the areas of protection Human health and Natural resources, thus covering all midpoint impact categories of LCA. For midpoint impact categories such as climate change and photochemical ozone formation that link to more than one area of protection the lowest normalisation reference amongst the complete set of references should then be used. Using sustainability conditions as references in impact assessment may also be explored in life cycle sustainability assessment.

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Supporting information
SI1 elaborates on the carrying capacity quantifications for each impact category and SI2 contains all calculations in spread sheet format.

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Supporting information S1 for:

Introducing carrying capacity based normalisation in LCA: framework and development of references at midpoint level

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1 Climate change

Threshold

As threshold we applied the 2°C target, which aims to limit an increase in the atmospheric global average temperature to 2 degrees above pre-industrial levels. In December 2010 parties to the UN Framework Convention on Climate Change (UNFCCC) agreed to commit to the 2°C target. The target is partly political since the climate system is not associated with a single predictable threshold, but rather characterized by a number of possible thresholds related to different parts of it. These include disintegration of Greenland ice sheet (1-1.5°C), widespread bleaching of coral reefs (>1°C), broad ecosystem impacts with limited adaptive capacity (1-2°C), complete melting of the Greenland ice sheet, (3°C) and shutdown of thermohaline circulation (3°C in 100 yr) (Haines-Young et al. 2006). Respecting the 2°C target is thus no guarantee of not crossing thresholds in the climate system (for elaborations on this issue, see Wijkman and Rockström (2012)). We nevertheless choose the 2°C target as one alternative on which to base the carrying capacity, since it is used as a reference for the IPCC scenarios on which a conversion from temperature target to carrying capacity relies.

As an alternative threshold we applied the more precautionary planetary boundary of net increase in radioactive forcing of 1 W/m^2, proposed by Rockström et al. (2009). Rockström and co-workers proposed a complementary planetary boundary for climate change of a 350 ppm CO_2 concentration, but we refrained from using that, since it does not consider the contribution from non-CO_2 greenhouse gasses.

Translation of threshold to carrying capacity and calculation of NR

For the two degree target we applied the Absolute Global Temperature change Potential for a sustained emission change ($AGTP_S$) of Shine et al. (2005) for non-CO_2 gasses. The model behind the $AGTP_S$ metric was used to calculate the steady state temperature increase following a sustained emission of 1kg/year of any non-CO_2 greenhouse gas:

$$\mathrm{AGTP}_S^x(t) = \frac{\alpha_x A_x}{C}\left\{\tau\left[1-\exp\left(-\frac{t}{\tau}\right)\right] - \frac{1}{(\tau^{-1}-\alpha_x^{-1})}\left[\exp\left(-\frac{t}{\alpha_x}\right)-\exp\left(-\frac{t}{\tau}\right)\right]\right\}$$

Here $AGTP_S$ is expressed in K/(kg/year), α_X is the lifetime (year) and A_X the radiative efficiency (W /m^2/kg) for gas X, C is the heat capacity of the climate system (4.2*10^8 J/K/m^2), τ is a time constant and the product of C and λ, which is the climate sensitivity parameter (1.06K/(W/m^2)). Parameter values were obtained from Shine (2005) and supplemented with updated values from the IPCC's 5th assessment report (IPCC 2013). $AGTP_S$ is a function of time (t) and we chose 1 million years as an approximation of steady state. From this the sustained emissions leading to a steady state temperature increase of 2 degrees was calculated by linear scaling. For CH_4 and N_2O this resulted in carrying capacities of $1.2*10^8$ kg/year and $4.1*10^{10}$ kg/year respectively. See SI2 for results for other greenhouse gasses.

For CO_2 steady state takes hundreds of thousands of years to occur after a change in emissions. The GTP_S model for CO_2 of Shine et al. (2005) is not valid for such a long time horizon. We therefore used the GEOCARB model (Berner and Kothavala 2001; UC 2014) to calculate temperature increases following a change in natural carbon emissions after 1.95 million years, the last time step the model extends to. At this point in time there was no change in the first 4 digits of the temperature (compared to the previous time step, 50.000 years before), which was therefore taken as a steady state temperature. See Figure S4 for input parameters based on Archer (2014).

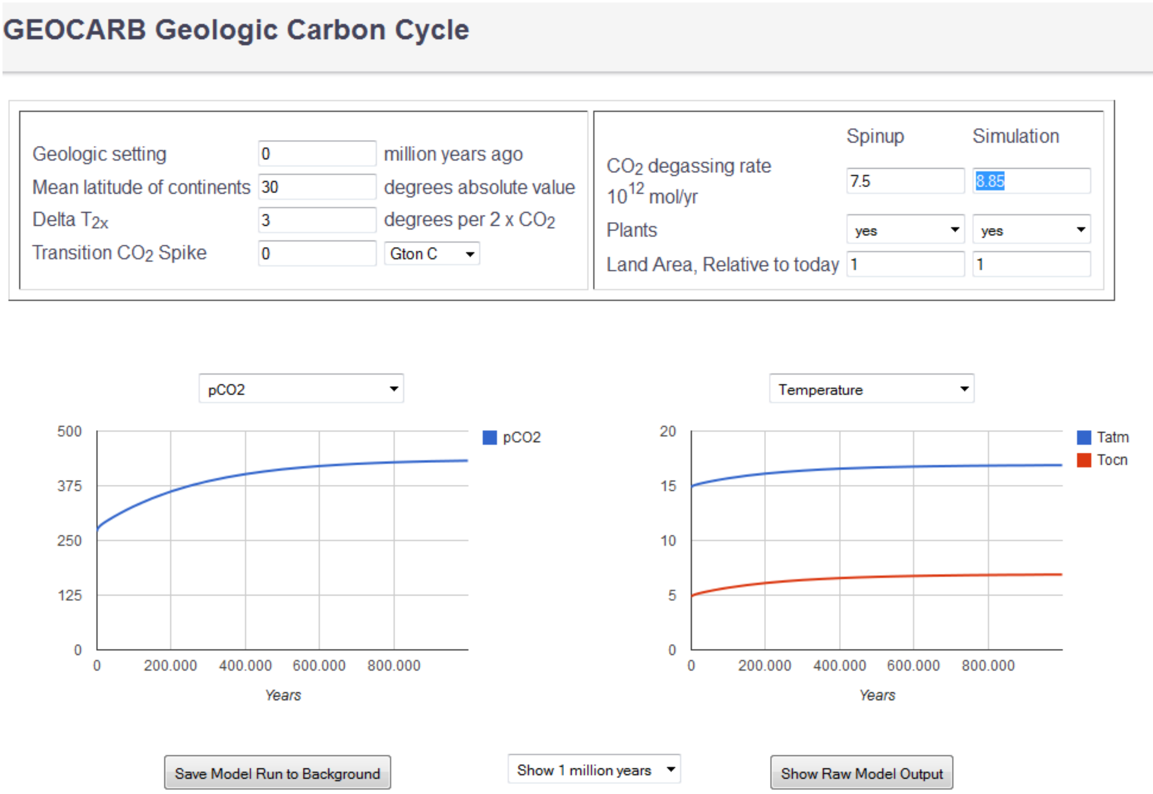

Figure S4: Input parameters for the modeling of atmospheric CO_2 concentration (left graph) and atmopshric and oceanic temperatures (Tatm, Tocn) following a change in natural carbon emissions. The Spinup stage indicates initial natural conditions (not shown in graph). The simulation stage models the response of the system due to a pulse emission (Transition CO_2 spike, here 0) or change in sustained emissions (CO_2 degassing rate, here 8.85).

By changing the emissions stepwise we found that a sustained emission of $8.83*10^{12}$ mole/yr CO_2 lead to a steady state atmospheric temperature increase of 2 degrees. When subtracting natural emissions ($7.5*10^{12}$ mol/yr CO_2) the carrying capacity for CO_2 was found to be $1.33*10^{12}$ mole/yr CO_2 or $5.9*10^{10}$ kg/year.

For the alternative carrying capacity based on the planetary boundary of 1 W/m^2 the $AGTP_S$ formula above was modified by division with λ, the climate sensitivity parameter, which indicates the change in equilibrium surface temperature per unit radiative forcing (1.06 $K/(W*m^{-2})$). The formula was thereby modified to calculate the steady state radioactive forcing increase following a sustained emission of 1kg/year. Again the sustained emissions leading to a steady state radioactive forcing increase of $1W/m^2$ was calculated by linear scaling. This resulted in carrying capacities of $6.3*10^{11}$ kg/year and $2.2*10^{10}$ kg/year for CH_4 and N_2O respectively. See SI2 for results for other greenhouse gasses. As for the 2°C target the GEOCARB model was applied to calculate the CO_2 emission leading to a steady state atmospheric temperature increase of 1.06K (=1W/m^2) and this figure was found to be 0.68 mole/yr CO_2 or 3.0*10^{10} kg/year.

The calculated carrying capacities vary between substances because of their different atmospheric life times: The longer the life time, the lower the carrying capacity, because CFs for gasses with long life times are artificially low, due to the disregard of impacts taking place after 100 years. As a consequence the carrying capacity based on the 2°C target for CH_4, having a life time of 12 years, was 3.3*10^{13} kg CO_2-eq, while it was just 5.9*10^{10} kg CO_2-eq for CO_2,who's fate is partly governed by the slow turnover rates of the geological carbon cycle (Archer 2014). To calculate a single carrying capacity for all considered greenhouse gasses[15] their average carrying capacity was weighted according to substance contribution in 2010 (Laurent et al. 2013):

$$CC_{climate\ change,i} = \sum_{x} \frac{CC_{climate\ change,i,x} \cdot IS_{i,x}}{IS_{total}}$$

Here IS is the impact score (kg CO_2-eq.) in 2010, based on Laurent et al. (2013) and i either the 2°C target or the 1 W/m^2 planetary boundary. This resulted in carrying capacities of 6.8*10^{12} kg CO_2-eq/year for the 2°C target and 3.6*10^{12} kg CO_2-eq/year for the 1 W/m^2 planetary boundary. After dividing by P (6.9 billion) NR_{Global} was found to be 985 kg CO_2-eq/pers/year for the 2°C target and 522 t CO_2-eq/pers/year for the planetary boundary. Both carrying capacities are compatible with CFs from any LCIA method based on the GWP100 approach. Due to the global nature of the impact category NR_{Europe} is equal to NR_{Global}. See S2 for calculations.

NR' (traditional normalisation reference)

NR'_{Global} was obtained from Laurent et al. (2013) for the year 2010 as 8.1 tons CO_2-eq/pers/year and NR'_{Europe} from Benini et al. (2014)for the year 2010, covering nations in EU-27, as 9.2 tons CO_2-eq/pers/year.

[15] All substances covered by the normalisation reference of Laurent et al. (2013), were included, except CO and PFCs, due to missing substance parameters (IPCC 2013). The combined contribution of CO and PFCs were just 3.3% to the total climate change impact score of 2010 and their exclusion is therefore thought to be acceptable.

2 Stratospheric ozone depletion

Threshold

The planetary boundary 7.5%[16] decrease in column ozone levels for any particular latitude with respect to 1964–1980 values was applied as threshold. Although a threshold has been observed for the occurrence of the Antarctic ozone hole, Rockström et al. (2009) reports that there is no clear threshold for global, extra-polar stratospheric ozone around which to construct a boundary. Even when column ozone levels were the lowest (282DU on average in 1993) no ecosystems in regions outside the Antarctic ozone hole were likely affected. The threshold of 7.5% decrease is therefore rather arbitrary, but since the effect of ozone-depletion on the climate system is complex (e.g. formation of local ozone holes and influence on cloud formation) a precautious threshold may in this case be justified.

Translation of threshold to carrying capacity and calculation of NR

The observed decrease in ozone from 1980 to the stable period of 1996-2009 was 3.5%. This corresponds to an increase in EESC (Equivalent Effective Stratospheric Chlorine) of 0.704 ppt (1.852ppt - 1.148 ppt) (Daniels and Velders 2011). By linear scaling, the increase in EESC that will result in a 7.5% ozone decrease (the threshold value) compared to 1980 is 1.51 ppt. The model of Daniels and Velders (2011) was then used to calculate the sustained CFC-11-eq emissions that would lead to this decrease in EESC at steady state and the result was 540000 ton CFC-11-eq/year[17]. This gave a NR_{Global} of 78g CFC-11-eq/pers/year, which is compatible with the ozone depletion potential (ODP) indicator of Montzka and Fraser (1999). As for global warming NR_{Europe} is the same as NR_{Global}. See S2 for calculations.

NR' (traditional normalisation reference)

NR'_{Global} was obtained from Laurent et al. (2013) for the year 2010 as 41 g CFC-11-eq/pers/year and NR'_{Europe} from Benini et al. (2014) for the year 2010, covering nations in EU-27, as 22 g CFC-11-eq/pers/year.

[16] The official planetary boundary is a 5% decrease, but 7.5% is in the middle of the "zone of uncertainty" in accordance with the average approach in LCA (see section 2.6 in the manuscript) (Rockström et al. 2009)

[17] The model was run by Velders (2014)

3 Photochemical ozone formation

Threshold

We applied the 3 ppm*hour AOT40[18] time integrated concentration limit for daylight hours during May-July adopted as a policy target by the European Environmental Agency (EEA 1998) and assumed that this target is also valid for global conditions (see below). The limit was based on preventing negative effects on growth for (semi-) natural sensitive perennial or annual species (Umweltbundesamt 2004). Experiments have shown that the most sensitive species will only show a significant effect after exposures with AOT40 above values of 3 ppm h. Recent findings by van Goethem (2013a) support the concentration limit: For endpoints related to reduction in biomass growth of annual and perennial natural grassland species HC5(EC10)[19] of 1.37-2.81 ppm h AOT40 was found.

Translation of threshold to carrying capacity and calculation of NR

The time integrated concentration threshold (T_{time}, 3 ppm*hour AOT40) was first converted into an average concentration threshold, T_{conc}, expressed in ppb and valid for the 8 consecutive hours with the highest daily average ozone concentration during the months of May, June and July in Europe:

$$T_{conc.} = 40ppb + \frac{T_{time}}{timefram} = 40ppb + \frac{3000ppb * hour}{8\ hours * 365/4} = 44ppb$$

This threshold could be converted to a carrying capacity using the fate factor of Van Zelm et al. (2008) for NMVOC (the substance equivalent for which indicator scores are expressed) which calculates a change in maximum daily 8-h average ozone concentrations, averaged over 1 year, as a function of a change in emission. However the fate factor needed to be corrected to consider specifically the period of May-July to which the threshold applies. This correction was done by considering annual variations in the diurnal cycle of ozone concentration obtained from measurement stations across Europe as reported by Katragkou et al. (2015) and shown in Figure S6.

[18] AOT40 refers the ozone concentrations accumulated over a threshold of 40 ppb, which is close to natural conditions (Umweltbundesamt 2004)

[19] This corresponds to the concentration where 5% of tested species showed an effect above EC10, the level at which 10% of individuals of species are effected.

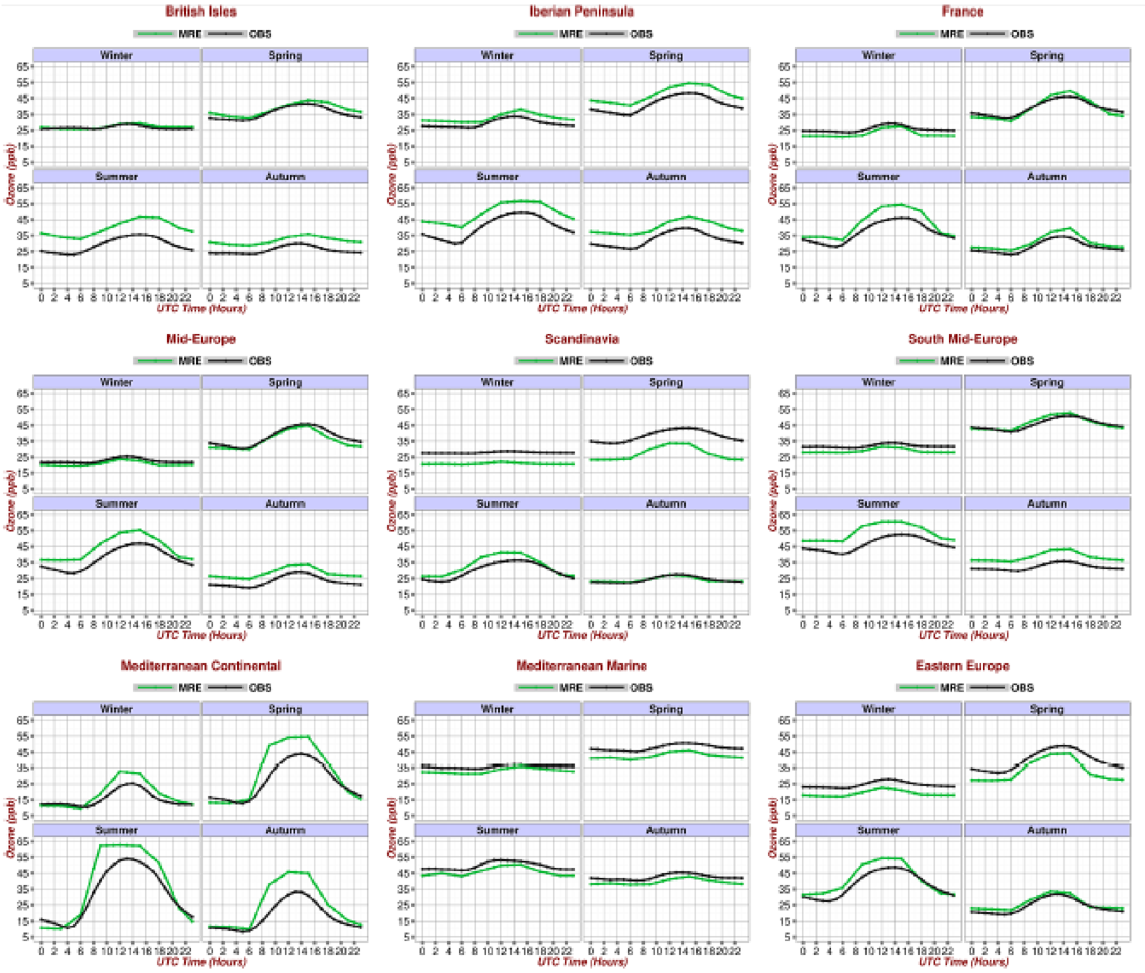

Figure S6: Mean 2003-2012 diurnal cycle of near surface ozone for the different European subregions based on MRE simulation (green line) and observations (black line) for winter (DJF), spring (MAM), summer (JJA) and autumn (SON). After Katragkou et al. (2015).

For each season and subregion the maximum daily 8-h average ozone concentrations was inferred from the observations of Figure S6. From this we divided the average ozone level of the threshold period (calculated as spring average*1/3 + summer average*2/3, since spring contains 1 month of the May-July threshold period and summer contains 2) by the average ozone level over the whole year. This gave a fraction that varied between 1.1 and 1.4 for the 9 European subregions. The average of these 9 fractions was found to be 1.2. We assumed that the emission pattern for substances with ozone formation potential was the same in all four seasons and hence that the observed differences in the maximum daily 8-h average ozone concentrations across the seasons could be explained solely from seasonal differences in the fate of the compounds. We therefore used 1.2 as a correction factor to convert the NMVOC fate factor of Van Zelm et al. (2008) from calculating average annual ozone increases to calculating average ozone increases over the months May-July. The fate factor was thus converted from the original

$5.80*10^{-14}$ (kg/m^3)/(kg NMVOC/day) to $6.96*10^{-14}$(kg/m^3)/(kg NMVOC/day). The modified fate factor was converted into a unit compatible with the 44 ppb threshold, giving a value of $9.72*10^{-8}$ ppb/(kg NMVOC/year) which was divided by the threshold to obtain a carrying capacity of $4.5*10^8$kg NMVOC-eq/year. This carrying capacity applies to emissions during May, June and July and emissions outside this time period thus make not contributing to occupying the carrying capacity. Since the points or durations in time of emissions are rarely known in an LCI we multiply the carrying capacity by 4 to make it applicable to an average emission assuming that the likelihood of an emission occurring in any month is the same. This gives a European carrying capacity of $1.8*10^9$kg NMVOC-eq/year. Note that a similar correction to take into account the share of emissions occurring outside the maximum daily 8-h average period was not needed since the lifetime of substances with ozone formation potential is such that substances emitted at night can be assumed to contribute to the increase of ozone during the maximum daily 8-h average period of the following day. To approximate a global average carrying capacity we applied a simple linear scaling based on land surface (since vegetation damage from photo oxidants occur over land) considering that the fraction of the Earth's land-covered area taken up by continental Europe is 7%. This resulted in a global carrying capacity of $2.7*10^{10}$ kg NMVOC-eq/year. This should be considered a tentative estimate given that a global average threshold and fate may be different from the European average threshold and fate given differences in solar irradiation. After dividing by 2010 European and global population respectively, NR_{Europe} and NR_{Global} were found to be 2.5 kg NMVOC-eq/pers/year and 3.8 kg NMVOC-eq/pers/year respectively. These references are compatible with the ILCD recommended midpoint indicator of Van Zelm et al. (2008). Note that this indicator models to the exposure level since human intake fractions of ozone are considered. However as indicator results are expressed at the pressure level in kg NMVOC-eq/pers/year our developed references, based on the fate factor of Van Zelm et al. (2008), are compatible with the indicator.

Spatial concerns

Due to spatially inhomogeneous emissions sources and the fact that the transportation distance of ozone and other formed photo oxidants is local to regional, the occupation of carrying capacity is likely to vary spatially. Also there might be variations in the sensitivity of ozone to (semi-) natural species around the globe, which means that the concentration target designed to European conditions might not be representative for thresholds globally.

NR' (traditional normalisation reference)

NR'$_{Global}$ was obtained from Laurent et al. (2013) for the year 2000 as 57 kg NMVOC-eq/pers/year and NR'$_{Europe}$ from Benini et al. (2014) for the year 2010, covering nations in EU-27, as 32 kg NMVOC-eq/pers/year.

4 Terrestrial acidification

Threshold

The critical load (CL) concept was chosen as threshold basis. Global and European average CLs for terrestrial acidification were estimated at 1170 and 1100 mole H^+ eq/ha/year, following Bouwman et al. (2002).[20] To obtain critical loads for manmade depositions a global average natural deposition of 90 mole H^+ eq/ha/year (Tegen et al. 1994; Bey et al. 2001; Roy 2014) was subtracted giving CL figures of 1080 and 1010 mole H^+ eq/ha/year globally and for Europe respectively. CL for terrestrial acidification is defined as "the highest deposition of acidifying compounds that will not cause chemical changes leading to long-term harmful effects on ecosystem structure and function Umweltbundesamt (2004)". CL is exceeded when the soils capacity to neutralize acidification is exceeded. This capacity is primarily defined by the mineralogical composition of soil materials as some classes of minerals weather faster than others. Base cations from weathering exchange with protons in the soil and offset the decrease in pH. In addition factors such as precipitation, vegetation and slope influences the critical load (Umweltbundesamt 2004). The model definition of CL in Umweltbundesamt (2004) leads to a non-linear critical load function for sulphur and nitrogen, since these compounds have different effects on the soil. Bouwman et al. (2002) used a simplified approach, where a single CL applying for both S and N was derived, based on a global soil classification into sensitivity classes. Each class was assigned a critical load. We chose the approach of Bouwman et al. (2002) as a basis for the NR, since the non-linear CL-function would be difficult to operationalize as a normalisation reference.

Translation of threshold to carrying capacity and calculation of NR

[20] The numerical data behind Figure 1a of Bouwman et al. (2002) is no longer available, so we estimated the average CLs for the two territories from an assessment of the area covered each interval via image analysis, correcting for the non-area conserving projection of the map. The medium value for each interval was used as weighting factor when calculating the average CLs. For intervals <25 and >200 meq m−2 yr−1 assigned weighting factors were 25 and 200 meq m−2 yr−1 respectively. Areas labelled 'no data' and 'agriculture' was omitted from the calculations of the average CLs.

To translate critical loads based threshold into global and European carrying capacities, expressed as mole H^+ eq deposition/year, threshold were multiplied with the land surface of Europe and the globe to obtain of carrying capacities of $1.0*10^{12}$ and $1.6*10^{13}$ mole H^+ eq/year for Europe and the global average. This gave NR_{Europe} of $1.4*10^3$ and NR_{Globe} of $2.3*10^3$ mole H^+ eq/pers/year, which are compatible with the OT indicator of Posch et al. (2008). This indicator models the fraction of emissions depositing on the terrestrial environment. The unit is mole H^+ eq/kg and thus states the moles of H^+ equivalent depositing on soil per kg of emission. See S2 for calculations.

Spatial concerns

Three issues needs to be mentioned: 1) the spatially generic CFs of the OT-method of Posch et al. (2008) are based on the European continent only. When applying these on a global scale, it is implicitly assumed that deposition fractions to the terrestrial environment are equal, 2) the global average CL of 1130 mole H^+ eq/ha/year hides regional and local variations from 125 to more than 3000 eq/ha/year (Bouwman et al., 2002), 3) Depositions are not uniform across all terrestrial environments, due to the spatial inhomogeneity of emissions sources. The first point is likely to only lead to minor uncertainties, while point 2 and 3 both entail that local or regional carrying capacities may have been exceeded even though the global average carrying capacity has not.

NR' (traditional normalisation reference)

No normalisation reference has been developed for the OT method of Posch et al. (2008), so we calculated references on our own based on inventories. NR'_{Global} for the year 2000 was calculated by applying CFs of the OT method (Posch et al. 2008) to the global inventory of Sleeswijk et al. (2008), on which the normalisation references of Laurent et al. (2013) was based, and dividing by the global population in 2000 (see S2). This resulted in an NR'_{Global} of $7.8*10^2$ mole H^+ eq/pers/year. NR'_{Europe} was calculated by applying CFs of the OT method (Posch et al. 2008) to the underlying inventory for the year 2010 of EU-27 of Sala et al. (2015) and dividing by the population of EU-27 of 2010 (see S2). This resulted in an NR'_{Europe} of $7.4*10^2$ mole H^+ eq/pers/year.

5 Terrestrial eutrophication

Threshold

As for terrestrial acidification the critical load (CL) concept was chosen as threshold basis. Global and European average CLs for terrestrial eutrophication were estimated at 1340 and 1390 mole N eq/ha/year, following Bouwman et al.

(2002)[21]. To obtain critical loads for manmade depositions a global average natural deposition of 70 mole N eq/ha/year was subtracted giving CL figures of 1270 and 1320 mole H^+ eq/ha/year globally and for Europe respectively. CL for terrestrial eutrophication is defined as "the highest deposition of nitrogen as NH_x and/or NO_y[22] below which harmful effects in ecosystem structure and function do not occur according to present knowledge" (Umweltbundesamt 2004). CL is exceeded when the combined removal from different pathways cannot prevent an increase of N in the soil. Removal pathways include harvest (in case of crops), denitrification, immobilization and leaching below the root zone (Umweltbundesamt 2004). Modeled CLs are generally supplemented by CLs derived from empirical observations. This approach was taken by Bouwman et al. (2002), who assigned global CLs for terrestrial eutrophication based on European values and ecosystem classifications.

Translation of threshold to carrying capacity and calculation of NR

To translate critical loads based threshold into global and European carrying capacities, expressed as mole H^+ eq deposition/year, threshold were multiplied with the land surface of Europe and the globe to obtain of carrying capacities of $1.3*10^{12}$ and $1.9*10^{13}$mole N eq/ year for Europe and the global average. This gave NR_{Europe} of $1.4*10^3$ and NR_{Globe} of $2.3*10^3$ mole H^+ eq/pers/year, which are compatible with the OT indicator of Posch et al. (2008). This indicator models the fraction of emissions depositing on the terrestrial environment. The unit is mole N eq/kg and thus states the moles of nitrogen equivalent depositing on soil per kg of emission. See S2 for calculations.

Spatial concerns

As for terrestrial acidification three issues needs to be mentioned: 1) the spatially generic CFs from the OT-method of Posch et al. (2008) are based on the European continent only. When applying these on a global scale, it is implicitly assumed that deposition fractions to the terrestrial environment are equal, 2) the global average CL of mole N 1340 eq/ha/year hides regional and local variations from 200 to 2850 eq/ha/year (Bouwman et al. 2002), 3) Depositions are not uniform across all terrestrial environments, due to the spatial inhomogeneity of emissions sources. The first point is likely to only lead to minor uncertainties, while point 2 and 3 both entail that local or regional carrying capacities may have been exceeded even though the global average carrying capacity has not.

[21] The numerical data behind Figure 1a of Bouwman et al. (2002) is no longer available, so we estimated the average CLs for the two territories from an assessment of the area falling into each interval via image analysis, correcting for the non-area conserving projection of the map.

[22] $NH_x = NH_3 + NH_4^+$; $NO_y = NO + NO_2 + NO_2^- + NO_3^-$

NR’ (traditional normalisation reference)

As for terrestrial eutrophication no normalisation reference has been developed for the OT method of Posch et al. (2008), so we calculating references on our own based on inventories. NR'_{Global} for the year 2000 was calculated by applying CFs of the OT method (Posch et al. 2008) to the global inventory of Sleeswijk et al. (2008), on which the normalisation references of Laurent et al. (2013) was based, and dividing by the global population in 2000 (see S2). This resulted in an NR'_{Global} of $3.5*10^2$ mole N eq/pers/year. NR'_{Europe} was calculated by applying CFs of the OT method (Posch et al. 2008) to the underlying inventory for the year 2010 of EU-27 of Sala et al. (2015) and dividing by the population of EU-27 of 2010 (see S2). This resulted in an NR'_{Europe} of $5.5*10^2$ mole N eq/pers/year.

6 Aquatic eutrophication

Threshold

A concentration threshold of 0.3 mg/L for P in freshwater was chosen, as this concentration is considered to give nutrient enrichment (algae bloom) (Struijs et al. 2011). For marine an average threshold of 1.75 mg/L N (ranging 1-2.5) was identified by de Vries et al. (2013) following an extensive empirical literature review of N-limited aquatic environments. Both thresholds are assumed to apply to European and global average conditions alike, as the references do not specify spatial applicability. The thresholds for both freshwater and marine waters are based on empirical data. Aquatic eutrophication is a well-known example of threshold-behaviour within resilience theory (Carpenter et al. 2001). The threshold represents the point where the system changes from reacting to pressure with negative to positive feedback, see Figure S5, which applies to eutrophication in shallow lakes. When P is emitted to a lake it may be taken up by primary producers (e.g. algae) or the lake sediment in the form of phosphorous-iron complexes. When the oxygen concentration in the lake is high, a relatively large share of the incoming P to the lake system is stored in the sediment, little sediment P is recycled into the water and the lake remains in its clear water/high oxygen regime. In this regime the sediment provides negative feedback in response to inputs of P to the lake system. As concentrations of P in the sediment increase so does the recycling rate of P from the sediment to the water (the sediment approaches P saturation). This lowers the resilience of the lake because the increase of algae following a sudden increase of P inputs to the lake system can reduce oxygen levels at the surface of the sediment sufficiently for the phosphorous-iron complexes to become unstable. The phosphorous-iron complexes are then chemically reduced and P is released into the water. This shift from the sediment providing negative to providing positive feedback brings the lake system into a new regime characterised by turbid water and low oxygen concentration. In the clear water regime it is possible that some species adapted to nutrient poor conditions will be affected at concentrations even below the threshold, but the aquatic ecosystem as a whole is in a clear state.

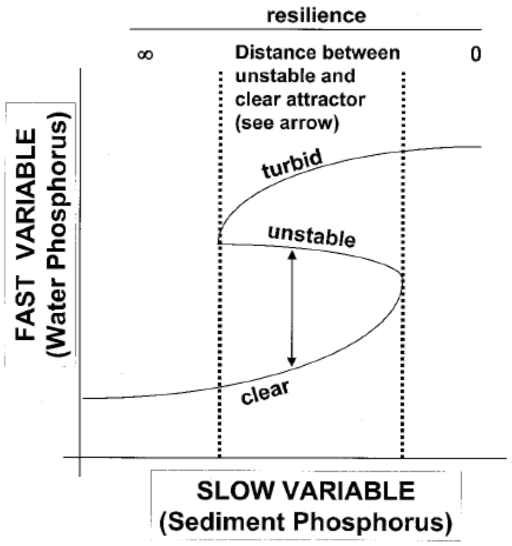

Figure S5: Threshold behaviour for eutrophication in shallow lakes. Vertical dashed lines show where the resilience of one of the stable states becomes zero. After Carpenter et al. (2001)

Not all aquatic systems show the same behaviour as the one in Figure S5. Some do not show thresholds behaviour and for the ones that do the values of the system parameter that lead to their crossing may vary depending on characteristics such as temperature and depth of water column. The applied carrying capacities for both freshwater and marine eutrophication are believed to represent average conditions, but it must be noted that the one for marine eutrophication was based on a combination of ecological and toxicological effects of inorganic N pollution, quality standards and political targets.

<u>Translation of threshold to carrying capacity and calculation of NR</u>

The concentration thresholds could be translated to carrying capacities by division with fate factors of Struijs et al. (2009), which models the average increase in steady state concentration of P and N (kg/m^3) in the European freshwater and marine compartments as a function of a marginal increase in emission (kg/year). These fate factors are $3.44*10^{-4}$ (kg P/km^3)/(kg P-eq/year) and $7.17*10^{-5}$ (kg N/km^3)/(kg N-eq/year) respectively. Intermediate European carrying capacities were thereby calculated as $8.72*10^{8}$ kg P eq/yr and $2.44*10^{10}$ kg N eq/yr, and from these we subtracted natural flows estimated to be 0.53Tg/year P[23] and 1.7Tg/year N[24] for continental Europe (Bennet et al. 2001; Green et al. 2003). This gave European carrying capacities of $3.41*10^8$ kg P eq/yr for freshwater and $2.27*10^{10}$ kg N eq/yr for marine waters, which resulted in NR_{Europe} of 0.49 kg P eq/pers/year for freshwater and 33 kg N eq/pers/year for marine waters. We applied simple linear scaling to calculate global carrying capacities based on the European fraction of global surface freshwater for freshwater eutrophication and the European fraction of global coastal waters for marine eutrophication. The freshwater fraction was obtained from Shiklomanov and Rodda (2003) as 5.9% (see SI2). The coastal waters fraction was obtained from SAU (2014) as 11%[25]. When subtracting global natural P and N flows (9 and 30Tg/year[8] respectively) global carrying capacities for freshwater and marine eutrophication were found to be $5.78*10^9$ kg P eq/yr and $2.02*10^{11}$ kg N eq/yr respectively. The resulting NR_{Global} was then 0.84 kg P eq/pers/year for freshwater and 29 kg N eq/pers/year for marine waters. See S2 for calculations.

Spatial concerns

Three issues are worth highlighting: 1) the fate model applied in Struijs et al. (2009), CARMEN, calculates the change in concentration as a result of an emissions increase using river catchments (freshwater) and coastal areas (marine water) for Europe. Our simple scaling does not take into account that European environmental conditions governing removal mechanisms of N and P cannot expected to be representative for the world as a whole (for instance removal via denitrification depends on the water residence time and temperature). 2) The global average concentration threshold may hide local and regional variations, due to differences

[23]Bennett et al. (2001) estimated a global pre-industrial flow from rivers to the sea of 8Tg/year. The natural pre-industrial P flow to rivers is larger than this value due to long term freshwater sedimentation, which was estimated to be around 1Tg/year from linear extrapolation of current sedimentation flows given by Carpenter and Bennett (2011) The resulting global estimate of pre-industrial P flows to rivers of 9Tg/year was scaled to the European continent by using the relationship between the European freshwater volume and the global freshwater volume (Shiklomanov and Rodda 2003), thus assuming uniform global concentration of P in global freshwaters.

[24] Green et al. 2003 estimated a European and global pre-industrial flow from rivers to the sea of 1.17Tg/year and 21Tg/year respectively. Using the generic estimate applied in Struijs et al. (2009)) that 30% of input to freshwater is removed due to denitrification pre-industrial N-inputs to European and global freshwaters becomes 1.7Tg/year and 30Tg/year respectively.

[25] This was calculated based on a dataset dividing the coastal waters (defined as marine waters in depths of less than 200m) of the globe into 64 areas. The area (km^2) of the coastal waters belonging to the European continent represents 11% of the area of global coastal waters. It is assumed that this fraction also applies to the volume of European coastal waters compared to the total coastal volume, as an average depth of 100m in coastal waters is assumed (corresponding to a linear depth increase from 0m to the 200m boundary)

in e.g. climate and the concentration of other nutrients. 3) Emissions of nutrients will not distribute uniformly across the aquatic environment. Point 2 and 3 both entail that local or regional carrying capacities may have been exceeded even though the global average carrying capacity has not.

NR' (traditional normalisation reference)

For freshwater eutrophication NR'_{Global} was obtained from Laurent et al. (2013) for the year 2000 as 0.62 kg P eq/pers/year and NR'_{Europe} from Benini et al. (2014) for the year 2010, covering nations in EU-27, as 1.49 kg P eq/pers/year. For marine eutrophication NR'_{Global} was obtained from Laurent et al. (2013) for the year 2000 as 9.4 kg N eq/pers/year and NR'_{Europe} from Sala et al. (2015) for the year 2010, covering nations in EU-27, as 17 kg N eq/pers/year.

7 Ecotoxicity, freshwater

Threshold

We chose HC5(NOEC) as a threshold. HC5(NOEC) is the concentration at which maximum 5% of species in an ecosystem are affected above their NOEC-level (no observable effect concentration).

Translation of threshold to carrying capacity and calculation of NR

In translating the threshold to a carrying capacity expressed at the impact point in the impact pathway we used the effect factor of USEtox, modified from being based on the HC50(EC50) effect level to being based on HC5(NOEC). We refer to Bjørn et al. (2014) for details on this effect factor modification and for derivation of the so-called "chemical footprint", which expresses the (theoretical) water volume needed to dilute an emission to HC5(NOEC). The chemical footprint (m^3) can be calculated from the USEtox characterisation factor (CF_{USEtox}, [PAF]*m^3*day), the emission (E, kg) of substance i to emissions compartment j (Bjørn et al. 2014):

$$ChF = 0.81 \cdot I_{USEtox}$$

The water volume available for diluting global emissions it in theory the sum of all surface freshwater in the world, estimated at 104.580km^3 by Shiklomanov and Rodda (2003)[26]. The global carrying capacity can thus be calculated as:

$$V = 0.81 \cdot I_{USEtox} \leftrightarrow I_{USEtox} = \frac{V}{0.81} = \frac{1.05 \cdot 10^{14} m^3}{0.81}$$
$$= 1.30 \cdot 10^{14}\ [PAF] m^3 \cdot day$$

[26] Lakes: 91.000km^3, swamp water: 11.470 km^3, rivers: 2.120km^3

A European carrying capacity is calculated by multiplying the global carrying capacity by the fraction of European freshwater (Shiklomanov and Rodda 2003) of 5.9%. This gave a European carrying capacity of $7.7 \cdot 10^{12}\ [PAF]m^3 \cdot day$. After division with P NR_{Europe} and NR_{Global} are found to be $1.0*10^4$ [PAF]*m^3*day/pers/year and $1.9*10^4$ [PAF]*m^3*day/pers/year respectively. See S2 for calculations.

Spatial concerns

As for aquatic eutrophication emissions will not distribute uniformly across the aquatic environment, since both freshwater resources and emission point sources are distributed in homogenously across the globe. This means that local or regional carrying capacities may be exceeded even though this model predicts that the global average carrying capacity is not. See Bjørn et al. (2014) for an elaboration of this issue.

NR' (traditional normalisation reference)

NR'_{Global} was obtained from Laurent et al. (2013) for the year 2010 as $6.7*10^2$ [PAF]*m^3*day/pers/year and NR'_{Europe} from Benini et al. (2014) for the year 2010, covering nations in EU-27, as $8.7*10^3$ [PAF]*m^3*day/pers/year.

8 Land use, soil quality

Threshold

We applied a tolerable soil erosion of 0.85 ton/ha/year as threshold. Srebotnjak et al. (2010) suggested a tolerable soil erosion of 1 ton/ha/year based on a review of several studies aiming at identifying a "sustainability threshold" for soil erosion. Verheijen et al. (2009) provides an interval of 'tolerable soil erosion' of 0.3-1.4 ton/ha/year for conditions prevalent in Europe. The applied tolerable soil erosion of 0.85 ton/ha/year represents the median of the interval of Verheijen et al. (2009) and assumed it to be applicable as a global average as well. Tolerable soil erosion is defined as 'any actual soil erosion rate at which a deterioration or loss of one or more soil functions does not occur'. Tolerable soil erosion is thus calculated as the formation rate of new soil. Soil formation takes place by processes of mineral weathering and dust deposition.

Translation of threshold to carrying capacity and calculation of NR

The tolerable soil erosion threshold was multiplied with the terrestrial land area of Europe and the globe to obtain carrying capacity of $8.7*10^8$ and $1.3*10^{10}$ tons/year. This gave NR_{Europe} and $NR_{Globale}$ of 1.2 and 1.8 ton/pers/year. NR_{global} is compatible with Saad et al. (2013)'s world generic CFs for erosion resistance, having units ton/(ha*year). Here a bias is introduced, since the carrying capacity is based on the sum of natural and man-made soil erosion, while the indicator of Saad et al. (2013) only measures the human contribution, meaning that results from carrying capacity based normalisation will be somewhat underestimated. This underestimation is thought to be negligible considering that the natural erosion is likely to be around 1 ton/(ha*year) for most biomes and since CFs for most types of land occupation are above 10 ton/(ha*year) (Saad et al. 2013). See S2 for calculations.

Spatial concerns

The applied carrying capacity represents an average European soil erosion threshold, which varied from 0.3 to 1.4 ton/ha/year . It is unknown if this average European carrying capacity is representative for average global conditions to which CFs of Saad et al. (2013) applies. In any event the tolerable soil erosion threshold may vary significantly from place to place depending on e.g. variations in weathering rate and geological characteristics.

NR' (traditional normalisation reference)

No normalisation reference has been developed for the indicator of Saad et al. (2013) so we calculating references on our own based on inventories. NR'_{Europe} was calculated by applying CFs of Saad et al. (2013) to the underlying inventory for the year 2010 of EU-27 of Sala et al. (2015) and dividing by the population of EU-27 of 2010 (see S2). We classified the 4 elementary flows of Sala et al. (2015) to the CFs of Saad et al. (2013) as shown in Table S1. This resulted in an NR'_{Europe} of 11 ton/pers/year. For the NR'_{Globa} we considered the global inventory of Sleeswijk et al. (2008), on which the normalisation references of Laurent et al. (2013) was based. However this inventory was very crude as it only covered two elementary flows, agricultural and urban land occupation. To improve the resolution of the global inventory we assumed that the total global land occupation, expressed in m^2*years, was composed of the same 4 elementary flows as the European inventory of Sala et al. (2015) and that these 4 flows contributed to the total land occupation with the same percentage as they do in the European inventory of Sala et al. (2015). After applying CFs of Saad et al. (2013) and dividing by the global population in 2000 NR'_{Global} was estimated as 9 ton/pers/year (see S2).

Table S1: Classification of elementary flows of Sala et al (2015) to CFs of Saad et al. (2013). "unspecified" and "wetlands" flows were not classified to any CF because these flows were assigned CFs of 0 in the calculation of the normalisation reference of Sala et al (2015).

Elementary flow terms (Sala et al. 2015)	**CF classification (Saad et al. 2013)**
agriculture	Permanent and annual crops (5)
artificial areas	Urban (7.1)
forest	Forest (1)
grassland	Grassland (4.1)
unspecified	NA
wetlands	NA

9 Land use, biodiversity loss

Threshold

We have applied a threshold of 31% as share of terrestrial land that needs to be conserved, based on Noss et al. (2012), who carried out a meta-review of 13 review studies, thus covering a wide range of biomes on Earth. Based on the review Noss et al. (2012) proposed a precautionary global target of 50% land conservation. Adhering to the best estimate approach we choose the median value, 31%, of the data series (see Figure S2) and assume it to be valid for European and global conditions alike.

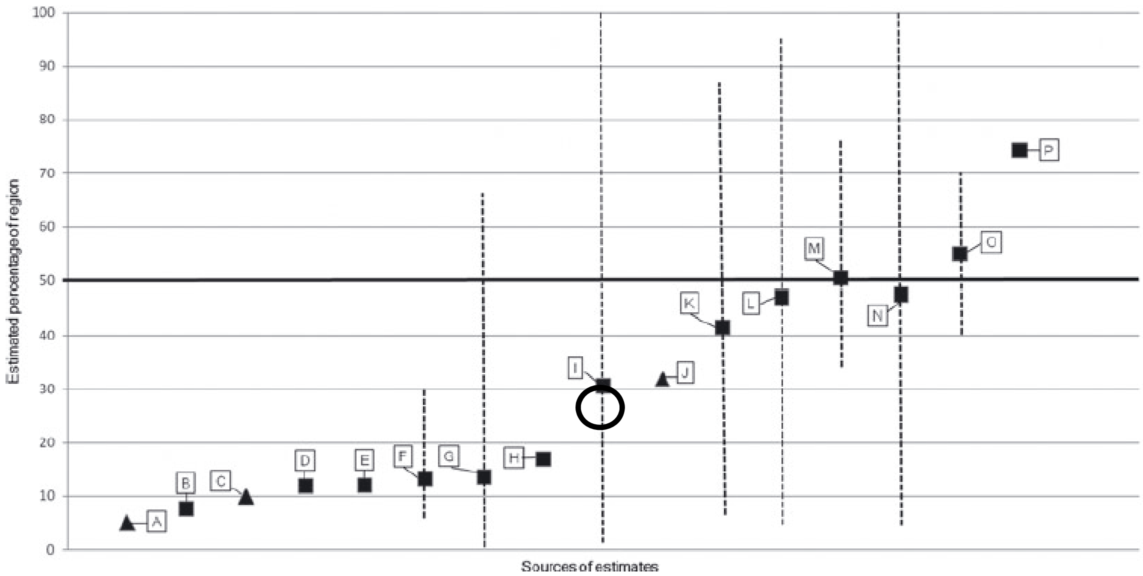

Figure S2: Estimates of the percentage of terrestrial region required to meet conservation goals on the basis of various sources (A-P) arranged from left to right in increasing order of percentage of area conserved. Triangles are political conservation targets (e.g. based on international conventions). Squares are targets derived from scientific research, reviews, and expert opinion. Vertical lines are ranges of values within published studies and points are reported means or medians of range. The circle indicates the median value of the square points (31%). Based on Noss et al. (2012)

Figure S3 shows the hypothesized bifurcation threshold behaviour of ecosystems not directly affected by land use, here hypothesized at a global value of around 50% land occupied.

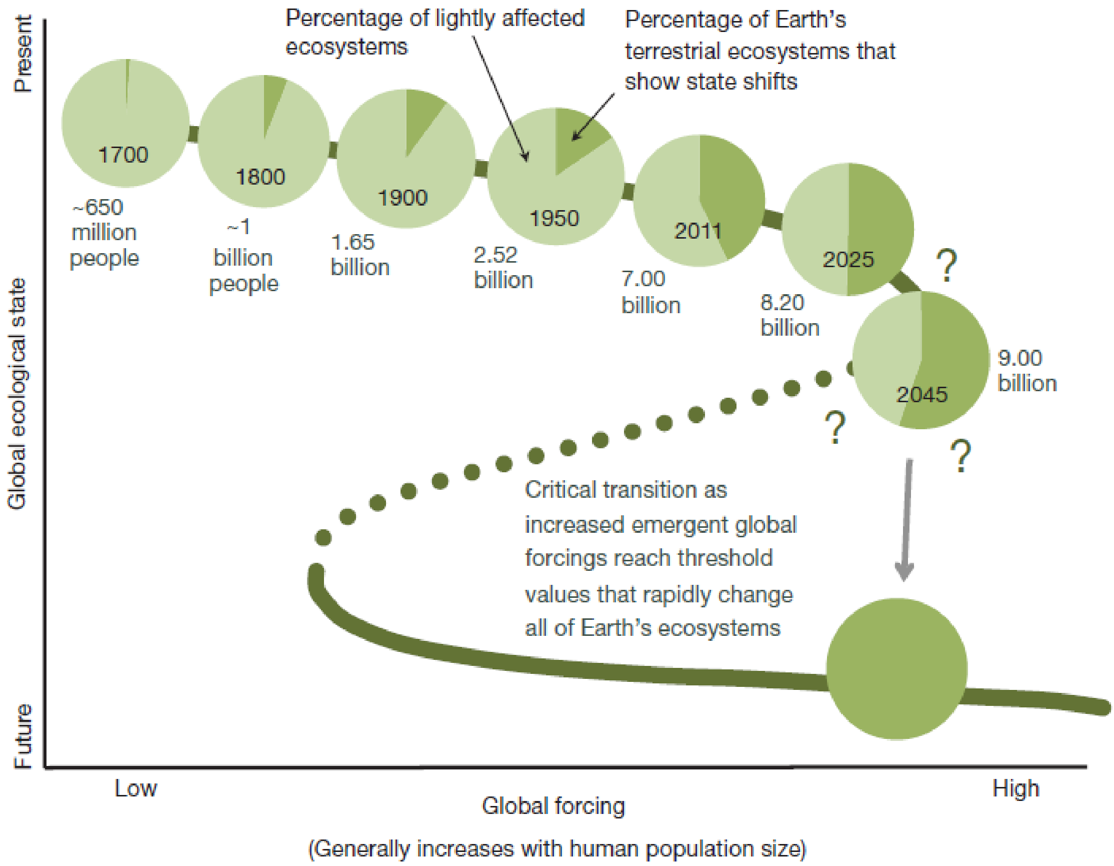

Figure S3: Conceptual figure illustrating the threshold behaviour of ecosystems not directly affected by land use, such as those neighbouring a clear cut forest. Each circle shows the estimated historical global land occupation (agricultural and urban lands) and projected future land occupation (human population below). Based on Barnovsky et al. (2012).

It must be stressed that the conservation threshold can be influenced by other stressors than land use. Thus the combined effect of land use and climate change may push ecosystems not directly affected by land use into a different state, since their resilience is lowered by each type of impact (Barnovsky et al. 2012).

Translation of threshold to carrying capacity and calculation of NR

A wildlife reserve of 31% means that 69% of terrestrial land can be used by humans, which corresponds to $1.0*10^{14}$ m^2 globally and $7.0*10^{12}$ m^2 in Europe. This gives a NR_{Global} and NR'_{Europe} of $1.5*10^4$ and $9.5*10^3$ m^2*year/pers/year. Since the threshold does not differentiate between types of land occupation, the developed NR can be applied directly to inventory data of land occupation (m^2*year). See S2 for calculations.

Spatial concerns

The targets reviewed by Noss et al. (2012) varied widely, from less than 10% to 75%, which can largely be attributed to site specific factors such as physical heterogeneity, degree of endemism and past land-use decisions, just as the location of the conserved land relative to the occupied land plays an important role. These aspects are difficult to include in LCA, where inventory flows related to land use often has no spatial information.

<u>NR' (traditional normalisation reference)</u>
No traditional normalisation references (NR') have been developed for comparison to NR since NR was designed to be used directly on inventory data of land occupation. We therefore constructed NR' on our own based on the sum of inventories. For the NR'_{Global} we calculated the sum of the global inventory of Sleeswijk et al. (2008), on which the normalisation references of Laurent et al. (2013) was based, and divided by the global population in 2000 (see S2). This resulted in an NR'_{Global} of $6.2*10^3$ m^2*year/pers/year. NR'_{Europe} was calculated by summing land occupation elementary flows of the underlying inventory for the year 2010 of EU-27 of Sala et al. (2015) and dividing by the population of EU-27 of 2010 (see S2). This resulted in an NR'_{Europe} of $7.5*10^3$ m^2*year/pers/year. Note that elementary flows related to natural land transformation (m^2) was not included, since they cover the transformation of land from one type to another and not its occupation. Note that NR'_{global} may be underestimated because the inventory of land occupation (sum of agriculture and urban land occupation) of Sleeswijk et al. (2008) only amounts to 25% of global terrestrial lands (year 2000), while Barnovsky et al. (2012) estimated that at least 43% of Earth's terrestrial ecosystems had undergone "wholesale transformation" in 2011.

10 Water depletion

<u>Threshold</u>
For the global threshold we chose a value of 2100km^3/year based on Gerten et al. (2013), which was derived as part of a refinement of the planetary boundary for global freshwater use. Gerten et al. (2013) used a spatially derived model to estimate the environmental flow requirements (EFR) needed to sustain rivers in at least a fair ecological state per 0.5°*0.5° grid cell and calculated a global range for median to maximum of 5 different calculations methods of EFR of 36-57% of blue water. These 5 EFR calculation methods varied from being based on achieving or maintaining "fair" to "good" ecological conditions (Gerten et al. 2013; Pastor et al. 2013). We assume that good ecological conditions are required for the full protection of ecosystem services, since Smakhtin et al. (2004) stated that ecosystems in good conditions may be slightly or moderately modified, but the modifications are such that they generally did not (or will not, from the management perspective) affect the ecosystem integrity. Consequently fair conditions are as-

sumed to correspond to a decrease in ecosystem services. We therefore choose the maximum global average EFR of the 5 calculations methods of 57%. When subtracting this from the estimated global accessible blue water resource (16300km^3/year) and furthermore subtracting 30% to avoid physical water stress of terrestrial ecosystems, the threshold becomes 5500km^3/year. The 30% reserved for terrestrial ecosystems can be considered a crude estimate. For the European threshold we followed Gerten et al.'s approach (2013) and started identifying the total runoff from the European continent of 3240km^3/year (Postel et al. 1995). We then subtracted remote flows, which based on Postel et al. (1995) were assumed to be 95% of the runoff from rivers in northern Europe having no damns on their main channel. Dynesius & Nilsson (1994) mapped these rivers (see Figure S1) and the total remote flow of the continent was thereby found to be 353km^3/year[27].

[27] The sum of unaffected VMAD (Virgin Mean Annual Discharge) was based on Dynesius & Nilsson (1994) (table number in parenthesis): 603m^3/s for Scandinavian medium sized river systems (table 2), 655m^3/s for Scandinavian large river systems (table 1) and 5480m^3/s for European part of Russian large river systems (table 1). Russian medium sized river systems were not mapped by Dynesius & Nilsson (1994) and their unaffected VMAD was therefore approximated based on the relationship between VMAD in Scandinavian unaffected medium sized and large river systems (1:1.09) resulting in a value of 5045m^3/s. This gives a total unaffected VMAD of 372km^3/year and thus a remote flow of 353 km^3/year (95% of former)

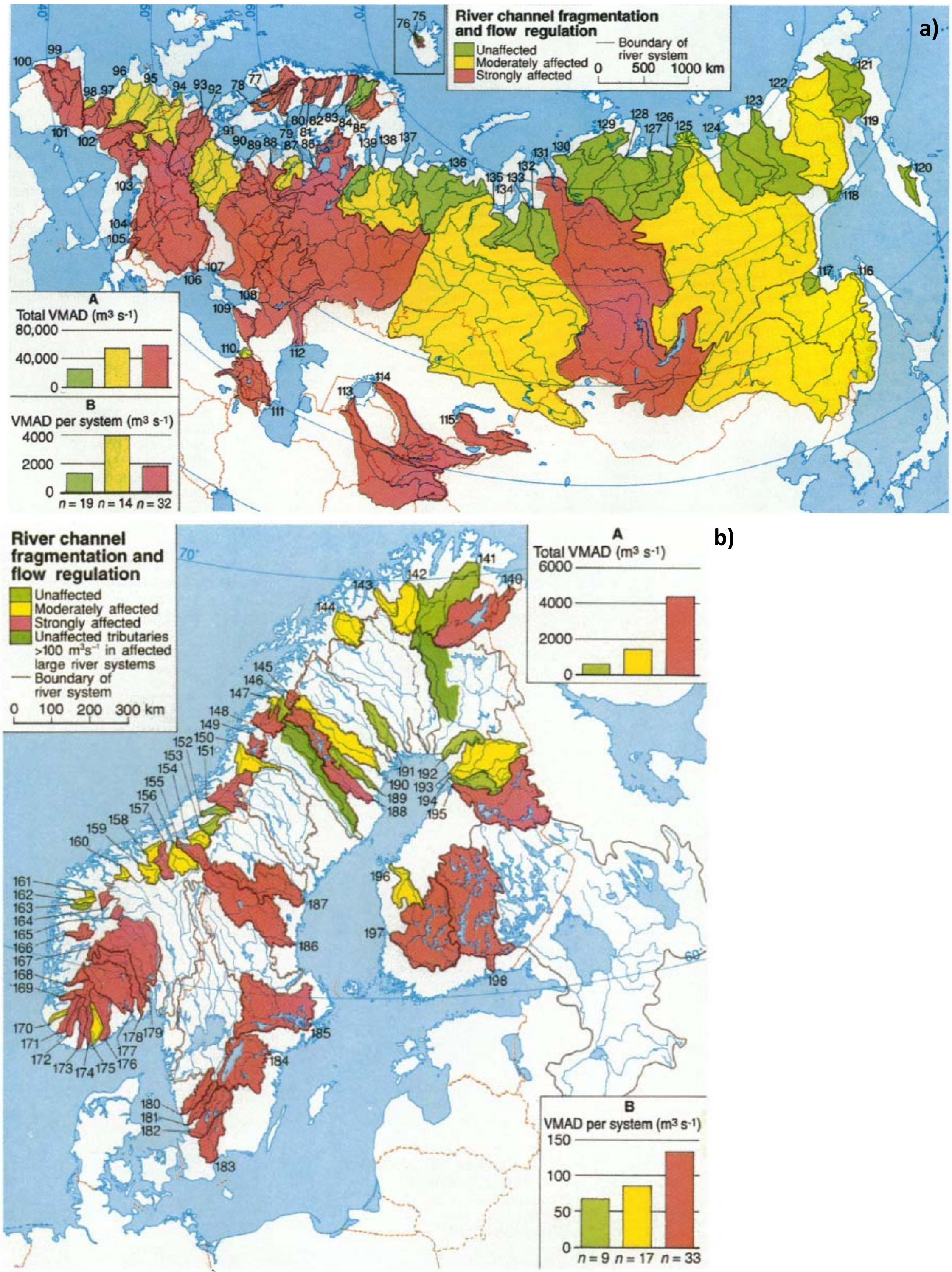

Figure S1: Impact by river channel fragmentation and flow regulation in a) Eurasia for large river systems (VMAD > 350m^3/s) and b) Scandinavia for medium sized river systems (40 < VMAD < 350m^3/s). Green basins are unaffected meaning no damns on their main channels. After Dynesius & Nilsson (1994).

Having dealt with spatial inaccessibility we now turn to temporal inaccessibility and therefore subtract high flows. The blue water flows of Europe are relatively stable throughout the year compared to the corresponding flows of e.g. Asia (Shiklomanov and Rodda 2003; Postel et al. 1995). Also European rivers are, except for those of the most north territories (Figure S1), heavily regulated by dams capable of capturing large fractions of flood runoff. The average monthly runoff for most major basins in Europe is higher for the months April, May and June (Shiklomanov and Rodda 2003). Parts of these peaks are captured by dams, but the exact fractions are difficult and time consuming to estimate. As a first estimate we calculate the temporally inaccessible blue water as all flows that exceed 120% of the annual average monthly flow, assuming that dams will capture everything else. From Shiklomanov and Rodda (2003) this gives a temporally inaccessible flows corresponding to roughly 3% of total runoff from the European continent and thus 97km^3/year. This results in an accessible blue water resource of Europe of 2790km^3/year (3240-353-97km^3/year). Since no European EFR for good conditions was available we applied the global of 57%. When subtracting this share in addition to the share reserved to terrestrial ecosystems (estimated at 30% corresponding to Gerten et al.'s (2013) global average value) the European threshold becomes 363 km^3/year.

Translation of threshold to carrying capacity and calculation of NR

In the impact pathway of water depletion a change in pressure, expressed in m^3/year water consumed, causes a change in control variable, expressed in m^3/year water availability, of similar magnitude. The threshold based on environmental flow requirements for good ecological conditions can therefore be interpreted as a pressure based carrying capacity and no conversion from threshold to carrying capacity was hence needed. When dividing the thresholds with the populations the global average and European NR become 306 and 490 m^3/pers/year respectively, which can be applied directly to inventory data related to blue water consumption, since no distinction is made between different types of consumption (e.g. lake or river water). Current LCA inventories of water use include both water withdrawal (e.g. turbine use, which is returned to the surface freshwater environment after use and therefore not consumptive), and consumption of ground water, salt water and surface freshwater sources. Only the flows classified as surface freshwater and groundwater consumption can be related to the estimated carrying capacity. The reason for not excluding groundwater is that it generally feeds into surface water and that its consumption therefore leads to occupation of environmental space for surface freshwater. See S2 for calculations.

Spatial concerns

As demonstrated by Smakhtin et al. (2004) the EFR varies from 20 to 50% globally, and the variation in the fraction reserved to terrestrial systems is also likely to show a large variation. Also freshwater resources are not distributed homogenously and neither are human demands. Therefore local or regional carrying capacities are likely to be exceeded even though the global average carrying capacity has not.

NR’ (traditional normalisation reference)

No traditional normalisation references (NR’) have been developed for comparison to NR since NR was designed to be used directly on inventory data of blue water consumption. The global and EU-27 inventories behind Sala et al. (2015) and Laurent et al. (2013) did not allowed for extracting blue water consumption from the aggregated inventories.

We therefore based NR'_{global} and NR'_{Europe} and estimates of Schiklomanov and Rodda (2003) for global and European (the continent, not the union) blue water consumption in 1995 of 2268 and 187km^3/year respectively. When dividing with the populations in 1995 NR'_{Global} and NR'_{Europe} became 395 and 256 m^3/year respectively.

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III

Is Earth recognized as a finite system in corporate responsibility reporting?

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Is Earth recognized as a finite system in corporate responsibility reporting?

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Abstract

Companies are increasingly encouraged to frame their sustainability activities and communication around ecological limits, as captured by concepts such as planetary boundaries, climate tipping points or local assimilative capacities. Ecological limits may serve as a scientific basis for defining sustainable companies and, moreover, inspire companies to align their product portfolios with emerging societal needs related to sustainable transformations. Although corporate environmental reporting is widely researched, little is known about companies' use of the ecological limits agenda in their communication.

This study presents a comprehensive review of references made to ecological limits in corporate responsibility (CR) reports. An exhaustive list of terms related to ecological limits was used to search the CorporateRegister database, which contained approximately 40.000 CR reports dating from 2000 to 2014. For every identified reference, we analyzed the context in which the ecological limit term was used in the CR report.

We found a 10-fold increase in the number of references made to ecological limits in CR reports during the period from 2000 to 2014. The number of CR reports published in this time period has also increased at a similar rate. Hence, the proportion of companies referring to ecological limits in their CR reports has over the years remained stable; roughly 5%. The most commonly invoked ecological limits were related to climate change and references to "2°C" were by far the most frequent. The vast majority of companies referring to ecological limits did so without specific references to ongoing or planned changes in their activities as a consequence of recognizing these limits. Only a very small percentage, predominately high-tech companies (31 in total), used ecological limits to define targets for resource consumption, emissions reductions and/or as a stated reason for adjusting their product portfolio. In defining targets for resource consumption or

emissions, only a few CR reports dealt explicitly with the issue of allocating resource and emission rights within ecological limits amongst companies and other actors. A longitudinal study of three companies showed that these did not directly report progress towards planned changes based on ecological limits and offered explanations as to why some companies abandoned planned changes altogether.

Our findings provide novel insights into the current use of the ecological limits agenda by companies and may be useful for actors trying to motivate companies to align their activities with the finite nature of Earth's natural systems.

Keywords
Planetary boundaries; Sustainability criteria; Sustainable transformations; CorporateRegister; Environmental Sustainability; Corporate environmental strategies

1 Introduction

An increasing number of companies is reporting on the sustainability of their business and how they are contributing to sustainable development[28], commonly defined as "…development that meets the needs of the present without compromising the ability of future generations to meet their own needs (WCED, 1987)." Sustainable development is, however, a contested term because of the subjective nature of needs and the complex task of identifying the conditions required for meeting them. For example, some promote economic growth (perhaps involving some environmental considerations) as a means to meet the needs of the world's poor, while others argue that the environmental degradation associated with economic growth will jeopardize the very meeting of human needs (Robinson, 2004). Further, sustainability reporting has been criticized as it can easily be misused by companies motivated by profit rather than a genuine interest in improving environmental and social conditions (Milne and Gray, 2012; Robinson, 2004).

Meanwhile, collective efforts to ensure sustainability are insufficient, because while meeting human needs today and in the future depends on well-functioning environments (Whiteman et al., 2013), the state of the environment is deteriorating globally (Steffen et al., 2015; WRI, 2005). Our dependency on the environment is captured in the common definition of environmental sustainability "…seek[ing] to improve human welfare by protecting the sources of raw materials used for human needs and ensuring that the sinks for human wastes are not exceeded, in order to prevent harm to humans" (Goodland, 1995). This definition

[28] See Robinson (2004) for similarities and differences in meaning and use of the two terms.

resonates with the concept of ecological limits, which holds that resource use and pollution should be restricted to certain levels to protect ecosystem functions and services critical for meeting human needs (Costanza and Daly, 1992).[29] Ecological limits have been quantified for various pollutants and resource use at different spatial scales, e.g. environmental flow requirements for freshwater use at the scale of a river basin (Hoff et al., 2014; Pastor et al., 2013), critical loads for the deposition of acidifying air emissions at the regional and ecosystem scale (Hettelingh et al., 2007), and planetary boundaries for gas emissions contributing to climate change and stratospheric ozone depletion at the global scale (Rockström et al., 2009; Steffen et al., 2015). Although ecological limits cannot be extended through technological means, technological innovations can increase the eco-efficiency of products and services, thus, allowing for larger quantities of these to be produced and consumed within ecological limits (Robinson, 2004).[30, 31]

Recently, a number of initiatives from NGOs, nonprofit organizations, think tanks, research organizations, consultancies and industry organization have encouraged companies to adopt an ecological-limit-based understanding of what constitutes a sustainable company. McElroy & van Engelen (2012), for instance, call for companies to perform context-based sustainability reporting, where context refers to carrying capacity (synonymous with ecological limit) of affected ecosystems. Context-based sustainability reporting is also encouraged in the latest G4 guideline of the Global Reporting Initiative.[32] The World Business Council for Sustainable Development's (WBCSD) Vision 2050 and Action 2020 encour-

[29] See Moldan et al., (2012) for an elaboration on the relationship between indicators of environmental sustainability and ecological limits.
[30] Geoengineering, in principle, has the potential to manipulate ecological limits related to the climate system, but this concept faces many challenges of technological, environmental, economic, political and ethical nature (Caldeira et al., 2013), which makes it unlikely to play an important role in responses to climate change.
[31] The ecological footprint indicator may get the impression that technology can in fact extend ecological limits because the biocapacity parameter (i.e. the potential yield of productive land) can be increased by, for example, the use of fertilizer and pesticides (Borucke et al., 2013; Giampietro and Saltelli, 2014). However, biocapacity is not synonymous with ecological limit because biocapacity is a characteristic of a manmade system (an cultivated area) and not of a natural system.
[32] "This involves discussing the performance of the organization in the context of the limits and demands placed on environmental or social resources at the sector, local, regional, or global level. For example, this can mean that in addition to reporting on trends in eco-efficiency, an organization may also present its absolute pollution loading
in relation to the capacity of the regional ecosystem to absorb the pollutant." (GRI, 2013). This initiative has, however, been criticized for not providing concrete guidance on this matter (Baue, 2013; GRI, 2013).

age companies to commit to the challenge of staying within ecological limits, based on the ecological footprint and planetary boundaries concepts (WBCSD, 2014, 2009). The One Planet Thinking model was developed to translate planetary boundaries to a business context (Ecofys, 2015). Other initiatives focus exclusively on climate-change and urge companies to reduce their greenhouse gas emissions in line with global reduction needs so as to avoid exceeding climatic tipping points (CDP, 2014; ClimateCounts, 2013; GreenBiz, 2014; Krabbe et al., 2015; Randers, 2012; WWF, 2013). In the light of these calls, Baue and McElroy (2013) encourage identification of the companies that use ecological limits to define corporate targets for resource use and pollution and analysis of the manner in which this is done. We argue that attention should also be given to the ecological limits concept's potential influence on changes in product portfolios, since staying within ecological limits will mean that some of today's products and services will become redundant in the future. Fossil fuels are obvious examples, while the need for new types of green-tech-related products and services will grow (SustainAbility, 1995). This means that the ecological limits concept can be expected to influence different types of companies' product portfolios in different ways.

In this study we examine how the ecological limits agenda appear in stakeholder communication in the form of corporate responsibility (CR) reports.[33] First, we estimate the share of companies referring to ecological limits in their reports based on a systematic text analysis of these reports. This is followed by a context analysis of the ecological limit references with the aim of exploring the extent to which companies present concrete ongoing or planned changes in resource use and pollution and in their product portfolios as a consequence of recognizing ecological limits and how these reporting activities develop over time. In interpreting these results we focus on 1) trends in references to ecological limits by different types of companies, 2) environmental issues covered by the ecological limits referred to and 3) companies' allocation of overall sustainable levels of impacts as defined by the ecological limits. Given that large companies are the most likely to issue CR reports, SME's are underrepresented in this study. Also, companies that do not issue CR reports in the English language have for practical reasons been omitted. Our target audience is 1) researchers wanting to understand corporate

[33] We use the term "corporate responsibility report" in the same way it is used by the CorporateRegister database (CR, 2014a). Thus a corporate responsibility report can be any type of non-financial report, such as a corporate social responsibility (CSR) report, a sustainability report, an environmental report or a so-called "integrated report".

use of the ecological limit agenda and 2) initiatives, such as those mentioned above, seeking to effectively encourage companies to adopt ecological limits.

2 Methods

The review consisted of 1) a screening; 2) a context analysis of all CR reports issued from 2000 to 2014 and included in the CorporateRegister, a database that as of November 2014 contained approximately 40.000 Anglophone reports covering 12.000 companies (CR, 2014a); and 3) an in-depth longitudinal study of a few selected companies' reports. We chose the CorporateRegister database because it is the largest regularly updated commercial database of its kind; estimated to cover at least 90% of all reporting companies going back almost two decades (CR, 2014b). Our systematic approach, essentially covering all CR reports written in English during the period, guaranteed a solid empirical basis for the analysis.

2.1 Screening

The screening was based on a list of search terms related to ecological limits. The list was developed through multiple iterations: First we applied terms used by WBCSD (2014, 2009) and Bjørn & Hauschild (2015) as search terms. This returned a number of references from the CorporateRegister database pdf search tool where each reference corresponds to a single incident of the use of the search term in a specific CR report. While conducting the context analysis of each reference (see below), synonyms for ecological limits were identified as some reports used one or more synonyms for ecological limits. The synonyms were applied as new search terms and also combined to form new synonyms. For instance, all combinations of identified synonyms for "natural" (i.e. "ecological", "environmental", etc.) and for "limit" (i.e. "constraint", "threshold", "boundary", etc.) were used as search terms. The iterative procedure was repeated until no new search terms were identified in order to maximize the chance of having identified all CR reports referring to ecological limits. The procedure applied here is very similar to the technique known as (citation) pearl growing within the field of online library searching (Hartley et al., 1990; Schlosser et al., 2006). Other librarian techniques were found not to be suitable for identifying all CR reports referring to ecological limits because they rely on the use of Boolean operators (i.e. "and", "or" etc.), which is not supported by the CorporateRegister pdf search tool. In order to ensure that identified references had a high relevance to the ecological limit agenda and to keep the amount of data manageable, search terms related to the following topics were excluded: policy targets and regulatory thresholds, carbon neutrality, the Natural Step, Cradle to Cradle, circular economy, resilience and resource scarcity. The rationale for these exclusions is explained in

S1. The resulting list of search terms consists of 286 terms presented in S2. For each search term the number of relevant references given by the pdf search tool was noted for each year since 2000. A reference was considered irrelevant when it was unrelated to ecological limits, for instance a thermostat being turned down “2 degrees” or a logistics company’s transportation “carrying capacity” being reported. The screening was carried out during November 2014.

2.2 Context analysis

The contexts in which the search terms appeared were analyzed by accessing each CR report containing one or more references to ecological limits in pdf format from the CorporateRegister database and reading the surrounding text paragraphs and any figures and tables to which the references related. Reporting companies were subsequently categorized according to whether they:

A. Referred to ecological limits without stating these as reasons for any ongoing or planned changes in activities.

B. Defined quantitative targets with deadlines for resource consumption and/or emissions based on ecological limits and:

1. Presented no strategy for how to meet targets, or
2. Presented a strategy for how to meet targets.

C. Stated ecological limits as reasons for adjusting their product portfolio and:

1. Presented ongoing adjustments, and/or
2. Presented planned future adjustments.

This categorization allowed for distinguishing between companies merely demonstrating awareness of ecological limits (group A) from companies actively using ecological limits as reasons for changes in governance (group B) or business (group C) in their stakeholder communication. Note that a single company could belong both to the B and C group. Some CR reports referred to other company documents or website content, but our analysis is based exclusively on the CR reports in the CorporateRegister database. Although we chose to focus the context analysis on companies producing physical products due to the scope of this journal, this does not mean that the analysis of references to ecological limits by service oriented companies is irrelevant, see section 4.4. Companies from the following sectors were, therefore, per default, categorized as ‘A’ group companies (despite that some of them may fit the criteria for B and C group companies): Banks, Equity Investment Instruments, Food & Drug Retailers, General Financial, General Retailers, Life Insurance, Non-equity Investment Instruments, Nonlife Insurance and Support Services. Companies within these sectors accounted for 23% of the approximately 12.000 companies in the database.

In order to further characterize the 'population' of companies actively using ecological limits in their stakeholder communication, group B and C companies were classified according to 1) sector and country of origin, 2) environmental problem(s) covered by the ecological limit(s), and, only for B companies, 3) the part of their products' life cycles covered by their performance targets, i.e. the system boundary. The B-companies' quantitative targets may pertain to products or to aggregated production (the total volume of all products produced by a company). In both instances, the corresponding system boundaries for resource use and pollution accounting may encompass the entire life cycle (or value chain) or only parts of the life cycle, such as the use stage or the industrial processes owned by the company.

2.3 Longitudinal study of three B and C companies

To analyze how ecological limits have been used in stakeholder communication over time, we selected three B and C companies as cases. To ensure sufficient time series of data, we chose companies for which the criteria for the B or C group (stating ecological limits as reasons for changes in either governance or business) were applicable for a minimum of 5 consecutive years. The companies were furthermore selected to ensure diversity in terms of sector and country of origin, system boundary applied and environmental problems covered. We focused on the development in time of two factors: 1) the presentation of planned changes based on ecological limits and 2) reporting of progress towards meeting these planned changes. The longitudinal studies expanded on that of the context analysis in two ways: 1) the targets and strategies motivated by ecological limits were evaluated relative to the other content in the CR reports to identify changes in the case companies' emphasis of these targets and strategies. 2) Reports from before and after the companies fitted the criteria for the B or C group were scanned for clues as to why the active use of ecological limits began and why it ended (for case companies not fitting the criteria for the B or C group in the most recent reporting year(s)).

3 Results

3.1 Screening

Of the 286 search terms, 93 terms returned relevant references by the CorporateRegister database pdf search tool (CR, 2014a).[34] Figure 1a shows the numbers of relevant references returned in the period 2000 – 2014 (as of 24. November 2014) across all CR reports in the database. Figure 1b shows the corresponding share of references, calculated for each year as the number of references divided by the number of CR reports published that year.

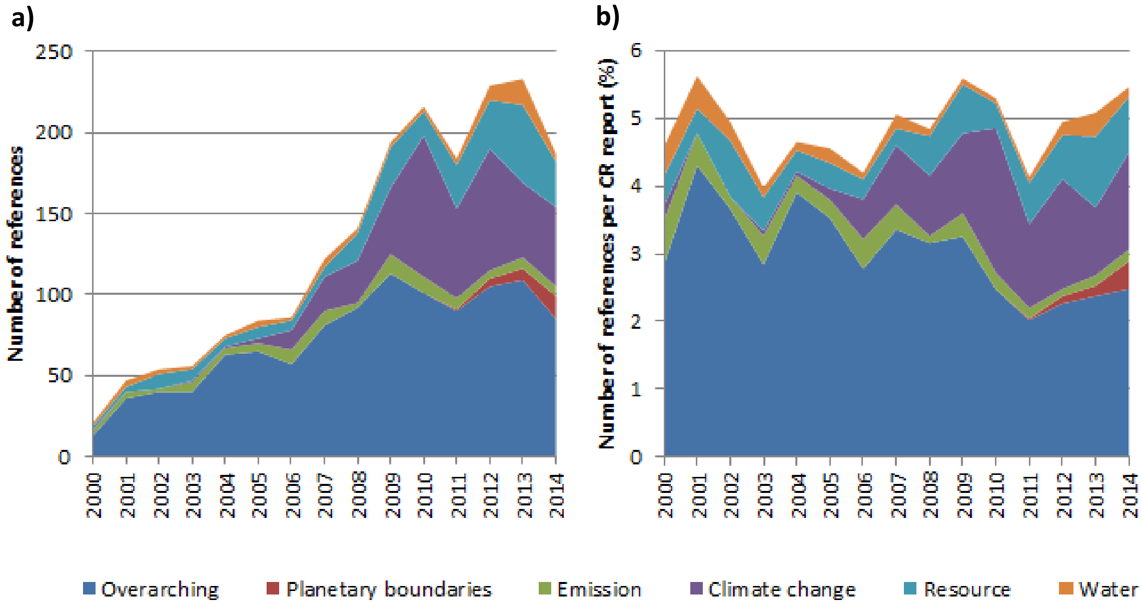

Figure 1: The absolute number of references to ecological limits (a) and the number of references per number of annually published CR reports (b) in the period 2000 - 24. November 2014 grouped into seven themes.

In Figures 1 the ecological limit terms were grouped according to the type of environmental problems they related to: "Overarching" refers to generic terms, e.g. "ecological limit" and "carrying capacity"; "planetary boundaries" refers to the concept of Rockström et al. (2009) and Steffen et al. (2015) that is concerned with the functioning of the Earth System and included terms such as "safe operating space" and "boundaries of the planet"; "Emission" refers to terms related to unspecified emission limits, such as "absorption capacity" and "critical load"; "Climate change" includes terms related to tipping points of the climate system, e.g. "2°C" and "450 ppm"; "Resource" covers terms related to resource limitations, e.g. "supply limit" and "resource constraint"; "Water" covers terms related

[34] The remaining 193 search terms all originated from combinations of previously identified search terms and either returned no references or only irrelevant references, i.e. without connection to ecological limits.

to water availability such as “Environmental flow requirement” and “water constraint (see S3 for the grouping of each search term).

Overall, most references fell into the category “overarching”, meaning that companies more frequently use generic terms for ecological limits than terms that relate to a specific environmental problem. Ecological limit terms related to “climate change” returned by far the most references of all emission related terms. Five terms returned more than 100 relevant references in 2000-2014; these were “2°C”, “carrying capacity”, “environmental constraint”, “environmental limit” and “resource constraint”. S3 shows the number of references returned for all 93 relevant search terms in the period 1995-2014.

According to Figure 1a, the number of references to ecological limits across all CR reports increased by more than a factor 10 (from 21 to 233 references) from 2000 to 2013 (the small decrease in 2014 compared to 2013 can be explained by the fact that not all CR reports published in 2014 were included in the CorporateRegister database as of 24. November 2014). However, Figure 1b shows that due to a parallel increase in the number of published CR reports, the share of references to ecological limits was found to remain relatively stable at 5% (approximately 1 reference for every 20 CR reports) throughout the entire period. In translating this share of references into share of companies that refer to the ecological limit terms two sources of bias needed to be considered: 1) Each report may refer to more than one ecological limits term (e.g. both “2°C” and “450 ppm“). Hence, some reports have been double counted in the 5% share. 2) Some companies published more than one CR report each year (for instance, one dedicated to CSR and one to sustainability). Accordingly, the annual pool of companies is smaller than the annual pool of CR reports. For the B and C companies (see below) the effects of these two sources of bias were found to approximately cancel out. We assume that B and C companies in this aspect are representative of all companies referring to ecological limits. Thus we estimate the share of companies in the database referring to ecological limits to be around 5% in any year of the 2000-2014 period. So, a short answer to the question posed in the title of this paper is “not really”.

When considering the development of each group of ecological limit terms in Figure 1, it can be seen that the number of references to climate change peaked in 2010 with a total of 87, coinciding with the publication of the Copenhagen Accord of December 2009, which recognized "the scientific view that the increase in global temperature should be below 2 degrees Celsius” (UNFCCC, 2009). References to planetary boundaries only began in 2011 following the 2009 publication of Rockström et al. (2009) and has since then increased although the total number of references are still modest (27 as of 24. November 2014).

3.2 Context analysis

3.2.1 Group A

When examining the context of each relevant reference returned by the database, it was found that an overwhelming majority (approx. 96%) of the reporting companies could be characterized as group A. These companies merely demonstrated awareness in that they referred to ecological limits without stating these as reasons for any ongoing or planned changes in activities. Typical examples of such awareness demonstration are when companies in the beginning of CR reports use an ecological limit term as part of a sustainability definition or to argue the need of sustainable development (ecological limit terms in bold): *"Sustainability includes living within **environmental limits** and ensuring a just and healthy society."* (UG, 2010)*; "As a global society we are facing enormous challenges and opportunities as we move towards nine billion people on earth, and as we get closer to our "**planetary boundaries**" on key natural processes upon which we all depend."* (Kering, 2014); Other companies referred to ecological limits to argue for the increasing importance of their products: *"Aluminium shall…be a part of the solution to bringing about growth in a way that respects the **limits of nature**... The energy is not wasted when turned into aluminium, the energy is stored in the metal, enabling it to be recycled time and again*" (NH, 2011). Many companies, especially in water supply and treatment, forestry and mining businesses, referred to their compliance with local ecological limits formalized in environmental legislation: "*…The Corporation has bulk entitlements to water from the Thomson and Maribyrnong Rivers. During the year, the **environmental flow requirements** established by these bulk entitlements were met in both rivers…."* (MWC, 2006). Common for these types of references to ecological limits is that they are not presented as a reason for changing "business as usual". This is not to imply that all companies within the A group are resisting sustainable transformations. Some companies may not perceive a need to signal changes to stakeholders if these consider existing products and production processes of such companies as compatible with (a transformation to) a sustainable society, i.e. a society not exceeding ecological limits. Yet, considering the need for sustainable transformations (Geels, 2011; Lorek and Spangenberg, 2014), it is striking that amongst companies demonstrating awareness of ecological limits only around

4%[35] actively use ecological limits in stakeholder communication as reasons for changes in governance (group B) and business (group C) .

3.2.2 Group B and C

Based on the context analysis, we categorized 31 companies as group B and/or C, see Table 1. Certain aspects of each report made them qualify to the B and/or C group. These aspects are presented for each CR report in S4 and examples are given below.

[35] The fraction is in reality a bit higher than 4% when considering the effect of classifying all companies not producing physical products (23% of all companies in the CorporateRegister database) as A companies in the context analysis (see Section 2.2).

Table 1: Group B and C companies sorted by publication year. System boundaries for product level and aggregated production levels targets are in lower case letters and capital letters respectively. Semicolons separate different system boundaries for different targets (see S4).

Name of company	Sector	Country	Publication year	Group	System boundary and level	Environmental problem
BMW AG	Automobiles & Parts	Germany	1999	C2	-	Climate change
Ricoh Company Ltd	Technology Hardware & Equipment	Japan	2005, 2006, 2007, 2008, 2009	B2	FULL LIFE CY-CLE	All covered by LCA indicators
Honda Motor Company Ltd	Automobiles & Parts	Japan	2007	C2	-	Land use
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2007, 2008, 2009, 2010, 2011, 2012, 2013, 2014	B2, C2	well-to-wheel; OWN OPERA-TIONS***	Climate change
Spier Leisure Holdings	Travel & Leisure	South Afri-ca	2008	B2	OWN OPERA-TIONS***	Climate change
Seiko Epson Corporation	Technology Hardware & Equipment	Japan	2008, 2009, 2010, 2011, 2012, 2014	B2	full life cycle	Climate change
Alstom SA	Electronic & Electrical Equipment	France	2008, 2009, 2010, 2011*, 2012	C1, C2	-	Climate change
Ford Motor Company	Automobiles & Parts	USA	2008, 2009, 2011	B2, C2	well-to-wheel; OWN OPERA-TIONS***	Climate change
Hitachi Ltd	Electronic & Electrical Equipment	Japan	2008, 2010	B2	FULL LIFE CY-CLE	Climate change
Acciona SA	Construction & Materials	Spain	2010	B2	Unclear	Climate change
Danisco A/S	Food Producers	Denmark	2010	C2	-	Climate change, water use, land use, fossil depletion, toxicity
Electrolux AB	Household Goods	Sweden	2010	B2	Unclear	Climate change
Hitachi Koki Co Ltd	Electronic & Electrical Equipment	Japan	2010**, 2011, 2012	B2	FULL LIFE CY-CLE	Climate change
Samsung SDI Co Ltd	Electronic & Electrical Equipment	Republic of Korea	2010	C2	-	Climate change
Toshiba Corporation Semi-conductor Company	General Industrials	Japan	2010	B2	full life cycle	All covered by LCA indicators

Unilever plc / NV	Food Producers	UK	2010	B2	OWN OPERATIONS***; full life cycle	Climate change
Iberdrola SA	Electricity	Spain	2011	B1	Unclear	Climate change
British Airways plc	Travel & Leisure	UK	2011, 2012, 2013	B2	Unclear	Climate change
Alcatel-Lucent	Technology Hardware & Equipment	France	2012	B2, C2	OWN OPERATIONS***	Climate change
Bridgestone Corporation	Automobiles & Parts	Japan	2012, 2013	B2	Unclear	Climate change
PTT Public Company Limited	Oil & Gas Producers	Thailand	2012, 2013	B1	OWN OPERATIONS***	Climate change
Galp Energia SGPS SA	Oil & Gas Producers	Portugal	2013	C1, C2	-	Climate change
Skretting AS	Food Producers	Norway	2013	B2	SUPPLIERS	Fishery
Novelis Inc	Industrial Metals	USA	2013, 2014	B2	FULL LIFE CYCLE	Climate change
Zhong Xing Telecommunication Equipment Company Limited	Fixed Line Telecommunications	People's Republic of China	2013, 2014	C1	-	Particulate matter, toxicity
Autodesk Inc	Software & Computer Services	USA	2014	B2, C1	OWN OPERATIONS***	Climate change
BT Group plc	Fixed Line Telecommunications	UK	2014	C2	-	Climate change
Cisco Systems Inc	Technology Hardware & Equipment	USA	2014	B1, C1	OWN OPERATIONS***	Climate change
Colgate-Palmolive Company	Personal Goods	USA	2014	B1	OWN OPERATIONS***	Climate change
Eneco Holding NV	Gas, Water & Multiutilities	The Netherlands	2014	B1	full life cycle	Climate change, fossil depletion, particulate matter
Implats	Mining	South Africa	2014	B1	Unclear	Climate change

***2010-2011 CR report was published in 2012. **2010 CR report published in 2011. *** Own operations include direct energy consumption (electricity, heat and steam) and thus corresponds to a scope 1-2 systems boundary in the terminology of the Greenhouse Gas Protocol (Ranganathan et al., 2004)**

Of the 31 companies, 23 were categorized as group B companies, because they defined quantitative targets with deadlines for their resource consumptions or emissions based on ecological limits. Of these, six companies did not present a strategy for how to meet the target (group B1), while 17 presented such a strategy (group B2). An example of the former is Colgate-Palmolive Company. Although they presented an absolute GHG reduction target (50% in 2050 compared to 2002), it was unaccompanied by a strategy for how to meet it. An example of the latter is British Airways plc. They presented a strategy for how to meet their "50 percent cut in net CO_2 emissions by 2050 relative to 2005", which included a quantitative projection for how much each element in that strategy (New aircraft technology & operational efficiency, Sustainable low-carbon fuels, Demand reduction and Purchase of emissions reductions) was expected to contribute to meeting the target. It was only possible to determine the system boundaries applied by 17 of the 23 B companies. Of these, eight, mainly Japanese producers of electronic consumer goods, applied a full life cycle boundary (either at the product level or aggregated production level). Other companies included one or more specific life cycle stages in their system boundary: Nine companies defined the boundary of their system around their own operations and their direct energy consumption (at the aggregated production level). Ford Motor Company and Nissan Motor Co applied a "Well-to-wheel" boundary (at the product level)[36], whereas Skretting AS only covered their suppliers in their targeted shift from fish-based to agricultural-based fish feed. Also, a number of companies applied different system boundaries with different quantitative targets.

Almost all ecological limits based climate change targets were derived from estimated global GHG emission reductions, or reductions from Annex 1 countries, needed by 2050 in order to avoid exceeding the 2°C threshold.[37] Many companies simply based their long term climate change target on a similar reduction percentage starting from a baseline year, thus implicitly adopting a grandfathering allocation approach, as further discussed in section 4.3. Some companies backcasted this long term target to determine milestones for near future years, e.g. 2020. Other companies used tools such as C-FACT (Autodesk, 2015) or The 3% Solution (WWF, 2013) that are specifically designed to calculate annual reduction needs required to prevent exceeding the 2°C threshold. The development of

[36] A Well-to-wheel boundary encompasses fuel production (Well-to-tank) and vehicle use (Tank-to-wheel) (JEC, 2014)

[37] Many companies referred to the statement of the 4th IPCC assessment report that "a 50 to 85% reductions of 2000 levels by 2050 would be needed to stabilize at between 445 and 490ppm (resulting in an estimated global temperature 2 to 2.4°C above the pre-industrial average)" (IPCC, 2007).

corporate sustainability targets from other ecological limits than those related to climate change appear to be arbitrary and non-transparent (see Section 3.3.3 for the example of Ricoh Company Ltd). Of the 31 companies, we categorized 13 as group C companies, because they described a process of aligning their product portfolio to ecological limits. Of these, five companies presented ongoing adjustments (C1), while ten companies presented planned adjustments (C2). Two presented both (C1 and C2). One of these two, Galp Energia SGPS SA, for example, presented the 450 ppm scenario of IEA as basis for its climate change strategy, which involved a change in business in the form of ongoing as well as planned increase in the provision and utilization of biofuels and other renewable energy sources.

In the B and C groups the number of Japanese companies is disproportionally large and other companies are mainly from the US or Europe (note that CR reports written in other languages than English were not covered by our study). With respect to coverage of environmental issues, the most common issue referred to is that of climate change with 27 of the 31 companies referring to related ecological limits such as "2°C" and "450 ppm". Ecological limits for fossil energy depletion, land use, resource use, water use, particulate matter and fisheries were only referred to by a few companies. Ricoh Company Ltd and Toshiba Corporation Semiconductor Company were the only companies that attempted to define reduction targets based on ecological limits for all indicators commonly covered in a Life Cycle Assessment (LCA), see section 3.3. With regards to sectors, "Automobile & Parts" was the most widely represented (5 companies), while there were 4 companies from each of the two sectors "Electronic & Electrical Equipment" and "Technology Hardware & Equipment". Of the 31 B- and C-companies, 22 have only fit the criteria since 2010. Although a very low number of companies at present actively uses ecological limits as reasons for changes in governance or business in stakeholder communication, an increasing trend can thus be observed.

3.3 Longitudinal study of three group B and C companies

Table 1 shows that only four companies fit the criteria for the B or C group for a minimum of 5 consecutive years (being one of the selection criteria for the longitudinal study, see Section 2.3). In order to obtain the largest possible diversity in sector, country of origin, system boundary applied and environmental problems covered, we selected from those the following three companies: Alstom SA, Nissan Motor Co Ltd and Ricoh Company Ltd (in the following referred to by the first word in the names only). The main commonality between these companies is that they all referred to and acted upon ecological limits related to climate change. Ricoh, in addition, addressed ecological limits related to all other environmental impacts commonly quantified by an LCA. In the following we analyse how each company developed over time with regards to two aspects: 1) the presentation of planned changes based on ecological limits and 2) reporting of progress towards meeting these planned changes.

3.3.1 Alstom

Alstom has in its 2008-2012 reports made references to ecological limits related to climate change using terms such as “environmental constraint”, “450 ppm” and “2°C”, and has acted upon these limits by describing ongoing (C1) and planned (C2) adjustments related to its product portfolio. Alstom’s argument for taking this approach rather than reducing internal emissions and resource use is that “Alstom’s main contribution to environmental protection lies in the technologies it offers.”(Alstom, 2008). Concretely, the planned product portfolio adjustments have varied over the years. In the beginning of the 2008-2012 period the focus was on carbon capture and storage (CCS). In later years CR reports emphasized planned renewable energy technologies and thermal conversion technologies adaptable to the fluctuating conditions caused by the increasing proportion of renewable energy production. In 2013 and 2014 Alstom no longer made references to ecological limits, and in the 2014 report (the latest covered by this study) there is, perhaps incidentally, no reference to CCS, at all. With regards to on-going adjustments, Alstom continuously described some of its new products, such as “CO_2 capture ready” power plant designs, as being compatible with an anticipated short-term transformation of the energy sector.

In terms of progress towards planned adjustments in product portfolio, the reports do not provide a direct indicator such as “share of revenue generated by product portfolio aligned to ecological limits”. The company did, however, since 2012 report on the “cumulative annual, avoided CO_2“ enabled by Alstom technologies in use worldwide when compared to “Business-as-Usual” scenarios. The cumulative avoidance due to Alstom’s renewable technologies has increased significantly since 2002. Based on this indicator it is, however, difficult to evaluate whether the total annual avoidance caused by Alstom products is sufficient in the light of overall reduction demands needed to satisfy the “450 ppm” or ”2°C” limits. In 2010, Alstom noted a cumulative saving of 0.189 Gt CO_2, which corresponds to 2% of the annual reduction requirement of the global power generation sector according to an IEA reduction scenario, designed to meet the 2°C target (both pieces of information were presented in the 2012 report). Whether these 2% were “enough” may be judged by considering Alstom’s global market share of the power generation sector, but such a comparison is not made in the CR reports. Although Alstom stops referring to ecological limits in 2013 and 2014, they continued reporting on avoided cumulative emissions. Rather than being driven by a need to communicate alignment to and performance against ecological limits, Alstom appears to be driven by regulation and high environmental concerns in general, as exemplified by the 2012 CEO statement: *"Environmental concerns – and the regulations that go along with them – have been and will continue to be a growth driver for us. They spur demand for higher-tech products and more complex services."* (Alstom, 2012).

3.3.2 Nissan

Amongst the case companies Nissan has fitted the criteria for the B or C group for the longest time, 8 years, and it is the only company that still matched these criteria in their most recent report (2014). Throughout the years, Nissan has consistently referred to "2 degree" and they have used ecological limits to derive an initial governance target of a 70% reduction of "Well-to-wheel" CO_2 emissions from new vehicles in 2050, compared with levels in 2000 (qualifying Nissan for the B group). In 2010, this target was increased to 90% (same target- and base year). The reason for this change is presumably that the original 70% target was based on not exceeding a CO_2 concentration of 550 ppm. The upwards revision of the target to 90% was likely in recognition of the updated IPCC estimate (IPCC, 2007), which stipulated that CO_2 concentrations should in fact stay below 450 ppm to avoid exceeding the 2 degree target. In 2012, Nissan announced a supplementary GHG reduction target for corporate activities of 80% by 2050 compared to 2000. This has a more direct link to the state of the climate than the "Well-to-wheel" target, because it is independent of the total travel distance of the vehicles sold. However, the Well-to-wheel target appears to be an increasing priority, given that it was mentioned in the CEO statements of the 2013 and 2014 reports, contrary to earlier years. Nissan also used the "2 degree" ecological limit as a stated reason for planned adjustments in product portfolio (qualifying it for the C2 group), e.g. "*the spread of new types of electricity-powered vehicles, such as hybrid, full-cell and electric vehicles*" and the company stressed that "*The 70% reduction target cannot be met even with these technologies, though, unless renewable energy is used to power the motors*." (Nissan, 2007).

Throughout the 8-year period, Nissan has not reported the direct progress towards the "Well-to-wheel" target in terms of "% reductions of 'Well-to-wheel' CO_2 emission from new vehicles compared with levels in 2000", although such an indicator would seem practical given Nissan's knowledge of emissions figures for their different car models and their sales figures. Instead, Nissan has, from 2012 and onwards, reported on its progress towards a number of 2016 sub-targets that are linked to the 2050 “Well-to-wheel” target, albeit somewhat indirectly. It is, for instance, unclear how meeting the 2016 sub-target of “20 million cumulative sales of CVT [continuously variable transmission]-equipped units” will constitute towards the 2050 target. With respect to its planned transition to engines directly or indirectly powered by renewable fuels, Nissan has in earlier reports focused on variants of electric vehicles (basic, plugin, hybrid) with improved battery technology, fuel-cell vehicles and vehicles running 100% on biofuels. In all cases, short-term targets and progress towards these were presented. These planned product portfolio adjustments were in the most recent report (2014) maintained, except for the original proposal of vehicles running 100% on biofuels, which was no longer mentioned. In recent reports emphasis is given to more systemic issues, e.g. in the 2014 report Nissan presented the need for collaborating with other car manufactures, governments, etc. in establishing charging infrastructure. Furthermore, emphasis was put on the positive role Nissan can play by integrating its fleet of electrical vehicles in the grid (e.g. allowing charging of batteries when renewable electricity generation is peaking and feeding electricity back into the grid when demands exceed production).

3.3.3 Ricoh

Ricoh has been characterized as a B2 company in 2005-2009, because they in that period have had a consistent reduction target of 87.5% for the ”integrated environmental impact” of the full life cycle of their products at the aggregated production level for 2050 compared to 2000, based on not exceeding the “tolerable impact” of an “ideal society.” Integrated environmental impact is expressed in an Environmental Load Unit (ELU), which is a composite metric covering all impact categories of the EPS (environmental priority strategies) LCA impact assessment method (Bengt, 1999). The referring to “tolerable impact” began in the 2004 report and in the ensuing year the 87.5% target was first presented without a clear scientific rationale. In 2009, IPCC reports were stated as inspiration for the 87.5% target, even though the IPCC is not concerned with ecological limits unrelated to climate change. The 87.5% target was maintained in the 2014 report, but this time restricted to GHG emissions and resource use – and the target was no longer motivated by an ecological limit or mentioned in the CEO statement (as was the case in the 2009 report).

Throughout the 2005-2009 period, Ricoh (like Nissan) established a number of sub-targets for 2007 (2005, 2006 and 2007 reports) and 2010 (2009 and 2010 reports), reportedly based on the 87.5% target, and reported progress towards meeting these sub-targets. Sub-targets mostly covered measures to reduce emissions from own operations, the development of in-use energy-saving products and improvement of product recycling and recycled content. As for Nissan, the link between these sub-targets and the overall target (87.5% reduction) is rather weak. Towards the end of every report, Ricoh presented the life cycle environmental impact at the aggregated production level for the two financial years prior to publication. Combining this information from reports published in 2005-2009, it can be observed that the impact actually increased from 2003 to 2008.[38] This means that Ricoh in the period seem to have made negative progress towards meeting the 87.5% reduction target, and this may explain why the presentation of the target in the 2014 report was more cautious (i.e. no longer motivated by ecological limits or mentioned in CEO statement) and why 2011 was the last year, so far, Ricoh reported its total life cycle environmental impact.

4 Discussion and conclusions

We begin our discussion by firstly addressing the three focus points presented in the introduction. Secondly, we propose how future research can improve our understanding of the topic. Thirdly, we put the findings into perspective by providing recommendations to existing initiatives seeking to encourage companies to adopt ecological limits in their management and reporting practices. Before embarking on the discussion, it must be noted that while our study reveals the number of companies engaging with ecological limits in stakeholder communication and the manners of this engagement, they do not directly reveal why companies engage in this behavior, let alone relevant decision-making processes within companies. CR reports do not offer a "true" representation of companies, but can be seen as part of companies' response to existing or anticipated external pressure from e.g. suppliers, customers, policy-makers, public opinion, competitors and social movements (Fernandez-Feijoo et al., 2014; Penna and Geels, 2012). Yet, in the discussions below we offer some explanations of the reasons behind the observed trends, while acknowledging that other explanations exist.

4.1 Trends in references

The most striking result of our study is that very few companies use ecological limits as stated reasons for changing their products. This has only been the case for 31 of the database's pool of approximately 9000 companies producing physical products. This seems to confirm the finding of CDP (2009), which, based on interviews of directors and managers within relevant departments of the world's

[38] An increase or unchanged impact was observed between all neighboring years reported in each report. However, due to occasional changes in calculation method, the increase of impact in 2008 compared to 2003 cannot be quantified based on the reports.

100 largest companies, concluded that “Company target setting is motivated by market forces, not scientific requirements.” The recent attention given to ecological limits by NGOs and business organizations alike may lead to more companies considering scientific requirements in future target setting. Yet, many companies may perceive a long-term commitment to ecological limits based targets as a risk. This is because companies are used to regularly adjusting targets and strategies in response to unforeseen changes in e.g. raw material prices, demands of products or rapid technological developments. Such unforeseen changes are, scientifically speaking, not valid reasons for adjusting targets and strategies motivated by ecological limits, although the changes can make it harder (or easier) to meet the targets and strategies. Abandoning or easing reduction targets (originally) based on an ecological limit could therefore be interpreted by critical stakeholders as a clear sign of abandoning the ambition of becoming a sustainable company. Ricoh may be an example of a company trying to gradually abandon or adjusting its 87.5% reduction goal, which it maintained in CR reports over several years but later disconnected from its original ecological limits framing and removed from the prominent position in CEO statements.

Another noteworthy trend is that nearly half of the B- and C-companies belong to the Automobiles & Parts sector (5 companies) and the technologically advanced Electronic & Electrical Equipment or Technology Hardware & Equipment sectors (8 companies). The relatively large number of car companies in these categories may, perhaps, be attributed to the tightening of regulation related to fuel economy in different countries and regions. For instance, the US Corporate Average Fuel Economy (CAFE) standards regulate the fleet fuel economy of automakers and tightens over time (Al-Alawi and Bradley, 2014). The CAFE standards were mentioned by Ford in their 2008 sustainability report together with a reduction goal, aligned with the 450 ppm climate threshold (Ford, 2008). For companies in the high-tech industry (e.g. Seiko Epson Corporation, Alstom SA and Cisco Systems Inc), regulation related to energy efficiency may also play a role. An additional reason for their relatively high representation in the B and C groups may be found in the rapid technological development within this industry which enables 1) dramatic increases in eco-efficiency per unit of service (e.g. LCD monitors generally use much less electricity than cathode ray tube monitors) and 2) flexibility to pursue new business opportunities created by sustainable transformations, e.g. smart grid technologies for low-carbon energy systems. Geographical factors may also play a role. As such the predominance of Japanese companies amongst B- and C-companies may be explained by the country’s historical focus on energy efficiency caused by its lack of domestic energy resources (EIA, 2014).

4.2 Coverage of environmental issues

Why are ecological limits related to climate change much more frequently mentioned in CR reports than ecological limits related to other environmental issues? Firstly, climate change has long been subject to much debate and the issue figures prominently on the political agenda worldwide. Consider, for example, the many policy documents in which the 2°C target, proposed by IPCC, has been adopted in. Secondly, the universality of the 2°C target and relatively high scientific certainty of global GHG emission reduction requirements means that they can be translated into company-scale emission reduction requirements, irrespective of the geographical setting of companies. Thirdly, monitoring the manageable number of existing GHGs is relatively simple, and CO_2 emissions can be predicted relatively precisely, based on fossil fuels consumption. By comparison, most other ecological limits are regional or local and may also vary naturally over time, which makes their translation into corporate level sustainability targets more challenging. Methods and tools for this translation are currently in their infancy (Bjørn and Hauschild, 2015; Ecofys, 2015) and years may pass before they reach a level of maturity that allow them to be as convenient to use as the ones for climate change (see e.g. WWF(2013). The fact that two B companies (Ricoh and Toshiba) simply applied the emission reduction percentage derived from the 2°C target to all LCA impact categories indicates that some companies want to cover more than just climate change in their ecological limits based targets, but find it difficult to do so scientifically in practice. In addition we found many companies in the A group referring to ecological limits concepts related to water use such as Environmental Flow Requirement (Figure 1 and Table S2), but not a single company presenting a quantified reduction target for water use based on this ecological limits concept. Given the spatial and temporal variability in water availability, operating with dynamic targets for each operation site, possibly including suppliers, would be highly impractical to communicate to stakeholders. Instead, companies tend to commit qualitatively to taking regional or local water limitations into account by, for instance, conducting regular formal meetings with stakeholders of the concerned watersheds. For instance, the mining company Teck wrote in their 2013 CR report that their approach to water management involves "*collaborating with our COIs* [communities of interest] *to ensure the fair allocation of water*." Note that such a commitment is relatively vague and difficult to hold companies accountable for compared to e.g. a quantitative commitment to reduce GHG emissions in line with a climate target.

Another explanation for why there are so few references to other ecological limits than those related to climate change could be that critical stakeholders perceive some of the other environmental problems as 'solved' or sufficiently controlled by regulation, which gives companies little reason to aim for emissions below the legal threshold. Considering that SO_X and NO_X emissions from combustion processes are well regulated in most developed countries and the little public attention currently given to these emissions, this may explain why no company referred to ecological limits related to acidification and eutrophication.

4.3 Allocation

In a world of limited resources and assimilative capacity for pollutants, figuring out how to share these in a reasonable manner is of paramount importance. Yet few B companies were explicit when it came to this issue of allocation. Instead, most of them implicitly adhered to the so-called grandfathering principle, where future emission rights are based on (a lenient granting of) historical emissions. In practice, this means applying the same impact reduction percentage to the corporate level as what is needed by all sources of impacts combined to not exceed an ecological limit, compared to some base year. Although the grandfathering principle may appear intuitively appropriate it can be seen as unfair for two reasons: 1) Companies who historically have done little to reduce emissions will be allowed relatively high emissions at the expense of environmental frontrunners, 2) It does not preclude companies from outsourcing some of their activities and pollution instead of reducing emissions by technical means, if they only include their own operations in the systems boundary. The grandfathering principle is also encouraged by "The 3% Solution" initiative (WWF, 2013), in which US companies are asked to reduce absolute emissions of GHGs with on average 3.2% per year from a 2010 baseline year until 2020.[39]

Alternative allocation principles are proposed by other initiatives encouraging companies to define targets based on ecological limits: The GreenBiz initiative (2014) suggests introducing a universal GHG emission target relative to contribution to global GDP. This brings a different unfairness into the picture, because differences in industry characteristics are not taken into account. For instance, companies within the service sector can easily appear sustainable, while dramatic GHG reductions for companies in energy and raw material-intensive sectors are required. A third principle was proposed by ClimateCounts (2013) which combined the grandfathering- and "contribution-to-GDP" principles through the use

[39] This reduction need was based on meeting an IPCC 2°C-pathway in which developed countries by 2020 reduce GHG emissions by 25-40% below 1990 levels.

of baseline year emissions to define an initial company-specific GHG reduction pathway, which is continuously adjusted based on changes in companies' revenues. A fourth principle was taken by CDP (2014) which, as part of their "Science based targets" initiative, used the sector-specific reduction pathways of the International Energy Agency (IEA), designed to achieve the 2°C target, to construct a tool for companies to calculate GHG reduction needs, taking into account expected changes in production output. Other legitimate institutions whose reduction pathways could form the basis for allocation between companies are IPCC, states and municipalities. No matter what allocation principle is applied, an additional concern is that for environmental issues with long-lasting effects, such as climate change, current and future reduction needs are functions of past emissions. As such, companies responsible for large historical emissions, e.g. many based in developed countries, can be seen as obliged to commit to the greatest reductions. This is reflected in the allocation adopted by The 3% Solution and ClimateCounts initiatives[40] (ClimateCounts, 2013; WWF, 2013), but not in the one adopted by the GreenBiz and "Science based targets" initiatives (CDP, 2014; GreenBiz, 2014).

In the end, any allocation will inevitably lead to the perception of one or more parties being treated unfairly. This may explain why companies in our study largely refrained from dealing explicitly with the issue in their CR reports, which tend to reflect the dominant "win-win"-discourse (i.e. the belief that the economy of a company, the environment and all social actors can benefit from an action and that no tradeoffs exists) (Bamburg, 2015). Many companies, especially when reporting on resource limits related to wood, agricultural products and fish, instead framed the ecological limits issue qualitatively: For instance, furniture manufacturer Knoll Inc was aiming to only source FSC certified wood (Knoll, 2007), which involves harvesting wood "*at or below a level which can be permanently sustained*" (FSC, 2012). "*Maintain or improve soils by preventing degradation*" was part of The Coca-Cola Company's guidance for "sustainable sourcing" of agricultural products (Coca-Cola, 2013a, 2013b). Walmart Stores Inc required "*its seafood suppliers to become third-party certified as sustainable*" because an "*estimated three quarters of the world's fisheries are at or beyond sustainable limits*", meaning a ban on overfished species (Walmart, 2013).[41] While this focus on local or specific sustainable practices is certainly important, it tends

[40] Both initiatives base company reduction needs on IPCC proposed reduction needs for developed countries, and not on global reduction needs.
[41] These three companies and others who framed ecological limits qualitatively were all categorized as group A companies in this study.

to divert attention from the fact that the Earth is a finite system. As illustrated by the Ecological Footprint method (Borucke et al., 2013), there is a limit to how many acres of forest and agricultural areas can be sourced sustainably. Similarly, a shift from sourcing overfished species to other less threatened species is likely to increase the pressure on these species that therefore may become at risk for being overfished as well. Thus, the allocation issue cannot be avoided when actors commit to collectively staying within ecological limits.

4.4 Proposals for future research

The growing pool of CR reports is an increasingly rich source of information on how companies navigate the sustainability agenda outwardly. Our study found a surprisingly small number of companies that, judged by CR reports, acted upon the recognition of ecological limits. Companies may, however, present ambitious reduction targets or plans to change their product portfolios to accommodate the needs of a sustainable transformation of societies without referring to ecological limits: Of the Carbon Disclosure Project's list of organizations that have *"committed to GHG emissions reduction targets that limit global warming to below 2°C"* (CDP, 2015) 17 companies, producing physical products, as of February 2015, do not figure in our list of B companies (Table 1), while 5 companies do. This is because these 17 companies do not refer to "2°C" or any other climate change-related ecological limit when presenting their commitment (an examination of the most recent CR reports of these 17 companies confirmed this). In benchmarking the ambitiousness of companies' environmental commitments against ecological limits (regardless of whether references to these are made in CR reports), the newly established Pivot Goals database of corporate sustainability targets may be helpful (J Gowdy Consulting, 2015) and inspiration can be sought in the study of CDP (2009), which found that the targets of the world's 100 largest companies insufficiently contributed to avoiding dangerous climate change.

To improve the understanding of companies' adaptations of the ecological limits concept, future research may draw upon the broader literature on drivers and barriers for implementation of environmental strategies in companies (Bey et al., 2013). Specifically, the role of public policies as a driver of company adaptation of ecological limits deserves attention. Also resource scarcity is a potential driver for companies' recognition and acting upon ecological limits: Although not necessarily related to ecological limits[42], resource scarcity occurs partly because Earth is physically finite. Due to the economic impact of rising prices, resource scarcity could act as an important driver for the broader adaptation of ecological limits and this mechanism deserves further attention.

Although the scope of this study has been restricted to companies producing physical products, there are other types of companies, particularly within the finance and retail industries, that can be associated with notable environmental effects and thereby contribution to exceeding various ecological limits. Future studies of the relationship between companies in the finance industry and ecological limits should consider the implications that divestment[43], can have for the shift from fossil fuels to renewables. The relationship between retailers and ecological limits is also worth studying, since retailers can influence consumer demands by the types of products that they sell and how these products are presented in the stores. Also retailers, if large enough, can have a substantial influence on the sustainability behavior of their suppliers, as exemplified by Walmart's sustainability ranking of suppliers (Gunther, 2013).

Outside the scope of this study was also companies' use of the circular economy (EAF, 2014) and resilience[44] (RAI, 2014) concepts in their CR reports. The relationship between companies' use of these increasingly popular concepts and (ecological limits framed) environmental sustainability in stakeholder communication make up relevant future research themes: It is important to identify conflicts in the simultaneous pursuit of environmental sustainability and resilience, because the former focuses on preventing ecological degradation by aligning business activities with ecological limits, while the latter is to some extent concerned with accommodating and adjusting to changes in environmental, social and economic conditions caused by ecological degradation. Also the question of whether companies attempt to use circular economy to legitimize not engaging with ecological limits in sustainability strategies and stakeholder communication deserves attention, considering that even an ideal circular economy that is growing indefinitely is at odds with ecological limits (Bjørn and Hauschild, 2013; Townsend, 2014).

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[42] The depletion of mineral deposits does not directly endanger any ecosystems, although emissions following this process might, which can be taken into account by emissions related ecological limits.

[43] Divestment is the reduction of assets in the fossil fuel sector based on an anticipated political will to prevent exceeding climatic tipping points.

[44] The Resilience Action Initiative has provided the following definition: "Resilience is the capacity of business, economic and social structures to survive, adapt and grow in the face of change and uncertainty related to disturbances, whether they be caused by resource stresses, societal stresses and/or acute events" (RAI, 2014).

Methodologically, further studies of CR reports may utilize text analysis software to enable the identification of certain clusters of words associated with themes of interest, see e.g. Liew et al. (2014) and Sengers et al. (2010). This may be combined with a more qualitative approach to studying the extensive database of CR report. For example, text analysis software could be used to examine relationship between ecological limits terms and terms related to company-external drivers, such as regulation, raw material prices, media and civil society. The outcome could guide in-depth reading of selected reports, which could offer explanations as to why some companies refer to ecological limits and why references sometimes is used to frame corporate targets and strategies. Such an extended analysis of stakeholder communication could be complemented by a study of internal corporate processes related to ecological limits via, for example, interviews with relevant managers and employees in a selected group of companies. Also, future research should pay attention to companies that do not communicate in English as these are likely to be situated in emerging economies that are predicted to substantially impact the global environment.

4.5 Taking the limits seriously – some recommendations

Recent initiatives encouraging companies to voluntarily adopt ecological limits can play a crucial role in increasing the currently very small number of companies that engage with ecological limits. Our study provides three prime recommendations for these recent initiatives.

Firstly, while climate change is recognized as a major threat to humanity, it is important that companies are urged to take a holistic approach in reporting their performance and targets in the context of ecological limits. If only climate change is considered, then there is a risk of burden shifting i.e. decreasing GHG emissions at the expense of increases of other environmental burdens, such as land use, water use or emissions of toxins (Laurent et al., 2012). While ecological limits for regional and local environmental issues can, for reasons given above, be challenging to incorporate in environmental strategies and reporting, this should not be an excuse for neglecting them. The WBCSD (2014; 2009) and One Planet Thinking (Ecofys, 2015) initiatives are currently at the forefront when it comes to considering other issues than climate change, and are, hence, important sources of inspiration. See also Bjørn and Hauschild (2015) for an attempt to operationalize ecological limits for use in LCA indicators.

Secondly, it is important that companies are encouraged to explicitly state the system boundary that they have chosen for their resource use and emission accounting and to argue why this boundary was chosen. The existing initiatives are currently split between encouraging a boundary encompassing a company' own operations and energy supply, while others encourage taking a full life cycle perspective. There is probably no single system boundary on which it is meaningful for all companies to base their reporting. Instead, sector-specific recommendations could be given. For instance, in light of the significant environmental impact of agriculture and the fact that farmers themselves usually do not report on sustainability issues, companies in the food producing and textile sectors should be encouraged to include supply chains in their system boundaries to align agricultural impacts (such as forest clearing, land erosion and emissions of pesticides and nutrients) with ecological limits of local ecosystems. The same goes for manufacturing companies that have outsourced large parts of their production to suppliers, who do not report on sustainability issues.

Thirdly, we find it problematic that none of the recent initiatives appears to ask companies to reflect upon the role of their products and their functions in a societal transformation towards sustainability. Past eco-efficiency increases have been insufficient in decoupling increases in environmental impacts from economic growth (PricewaterhouseCoopers, 2014) and future efforts should, therefore, not assume that eco-efficiency increases alone can bring about the necessary industrial transformations (Huesemann, 2004). In our view, changes in *how* things are produced must be augmented by changes in *what* is being produced and, let's face it, a transformation of the economic system in which companies are embedded. Asking companies to question how their products help meeting needs (as opposed to wants) of current and future generations and to reconsider business models may seem like a futile endeavor, but the size of the challenge should not be an excuse for willful blindness.

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Supporting information
Supporting information is available online and contains S1) rationale for exclusion of certain search themes, S2) list of search terms, S3) number of relevant references per search term and S4) qualifying aspects for each of the 31 B and C companies.

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Supporting information for:

Is Earth recognized as a finite system in corporate responsibility reporting?

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S1. Rationale for exclusion of certain search terms

Search terms exclusively related to policy targets and regulatory requirements were not included, although such terms may be informed by ecological limits.[45] The reason for this is that it would have been very time demanding to first identify all the policy targets and regulatory requirements that companies refer to in CR reports and then screen these targets for references to ecological limits.

Due to the focus on ecological limits other concepts designed to fulfill conditions of environmental sustainability, such as carbon neutrality, the Natural Step and Cradle to Cradle and Circular Economy, were not included in the screening. These approaches fall outside the scope of this study because they entail long term visions of nature being practically undisturbed by man (Bjørn and Hauschild, 2013; Robèrt et al., 2013).[46] On the contrary ecological limits are quantified tolerable disturbances of nature.

Terms related to resilience were also excluded. Resilience shares characteristics with ecological limits, mainly the focus on non-linearity in a system's response to increasing stress (Rockström et al., 2009). An important difference, and reason for exclusion of resilience, is however that a system's resilience can be manipulated by man, e.g. by changing species composition or management of a piece of land. Resilience is therefore not an absolute natural limit imposed by nature (Walker et al., 2012).

Finally, terms related to resource scarcity were excluded because, although scarcity is a function of stocks and flows of productive land, minerals and fossil fuels and freshwater, it is also a function of technical parameters determining the capacity of resource harvesting infrastructure (water supply plants, mines, oil wells and yield-determining agricultural management), geopolitics (resource tariffs and bans or limits imposed on export) and demand from other users (a resource, no matter how small, is never scarce if no one demands it).

[45] For instance some national governments have adopted climate targets based on recommendations from the Intergovernmental Panel on Climate Change (IPCC) based on the 2°C climate thresholds. E.g. the so-called "factor 4" objective in France aims to reduce CO_2 emissions by 75% by 2050 compared to the 1990 level (Boissieu, 2006) .

[46] Carbon neutrality is often associated with offsetting, which has recently been criticized for not necessarily leading to the claimed reductions in greenhouse gas emissions (Dhanda & Hartman 2011; Pinkse & Busch 2013).

S2. List of search terms

Table S1 shows the terms related to ecological limits that were applied as search strings and the number of hits each term returned (not sorted for relevance)

Table S1: All 286 applied search terms and hits returned by Corporate Register database (not corrected for relevance)

	boundar	capability	capacity	constraint	limit	space	threshold	tolerance
absorption	0	19	97	0	0	0	0	0
absorptive	0	0	5	0	0	0	0	0
assimilation	0	0	4	0	0	0	0	0
assimilative	0	5	12	0	0	0	0	0
biodiversity	5	2	9	2	2	1	0	0
biological	0	0	15	1	13	0	0	0
biophysical	0	0	1	2	3	0	0	0
bio-physical	0	0	0	0	1	0	0	0
dilution	0	0	7	1	119	0	0	0
Earth	0	0	0	0	11	2	0	0
ecological	3	0	8	16	46	15	5	1
ecosystem	1	0	3	1	2	0	0	0
environmental	17	38	83	277	288	33	9	4
natural	19	1	34	5	41	285	1	3
Planetary	21	0	0	0	2	1	2	0
productive	0	44	442	0	1	11	0	0
regenerative	0	5	30	0	0	0	0	0
renewable	0	0	108	0	3	0	0	0
resource	3	29	93	270	93	2	1	0
self-purifying	0	1	10	0	0	0	0	0

of	Earth	ecosystems	nature	resources	the Earth	the ecosystem	the environ-ment	the planet	the resour-ces
boundaries	1	0	3	0	1	0	6	1	1
capacity	1	20	12	2	31	11	46	5	2
constraint	0	0	0	1	0	0	0	0	0
constraints	0	0	2	1	0	0	1	2	0
limit	0	0	0	0	0	0	6	0	0
limits	0	0	3	1	10	1	12	24	3
threshold	0	0	0	0	0	0	0	0	0
thresholds	0	0	0	0	0	0	0	0	0
tolerance	0	0	0	0	0	0	0	0	0

	damage	disturbance	impact
acceptable	0	0	0
allowable	0	0	4
tolerable	8	0	12

Unique terms

accommodation capacity	26
biocapacity	17
bio-capacity	7
buffer capacity	18
buffering capacity	14
carrying capacity	483
global Hectare	23
critical load	31
overshoot	69
Safe limit	82
safe operating space	3
tipping point	164

Climate

2 degrees	213
2°C	410
350 ppm	35
350ppm	6
400 ppm	30
400ppm	15
450 ppm	56
450ppm	8
carbon budget	99
two degrees	158

Land

Land contraint	6
Land limitation	4

Water

Environmental flow requirement	28
Environmental water requirement	8
water constraint	37
water limitation	6

S3. Relevant references per search term

Table S2 shows the number of relevant references returned by the Corporate Register database for all 94 search terms returning relevant hits. The bottom row shows the number of reports published each year.

Table S2: Number of relevant references returned for each search term 1995-2014 as of 24.11.2014

Category	Search term	Number of relevant hits	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Climate change	2 degrees	61	0	0	0	0	0	0	0	0	0	0	0	2	3	5	5	13	7	10	10	6
Climate change	2°C	197	0	0	0	0	0	1	0	0	1	0	2	10	11	9	17	31	20	42	26	27
Climate change	350 ppm	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	1	2	1	0
Climate change	450 ppm	54	0	0	0	0	0	0	0	0	0	0	0	0	1	4	10	16	12	6	3	2
Climate change	350ppm	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Climate change	450ppm	7	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	2	1	2	0
Emission	absorption capability	4	0	0	0	0	0	0	1	0	0	1	0	0	0	1	0	0	0	1	0	0
Emission	absorption capacity	38	0	0	0	0	0	0	1	2	1	2	2	5	2	0	4	6	5	2	2	4
Overarching	allowable impact	4	0	0	0	0	0	0	1	0	0	1	0	0	0	1	1	0	0	0	0	0
Emission	assimilation capacity	4	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	2	0
Emission	assimilative capability	5	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	0	1	0
Emission	assimilative	12	0	0	0	0	1	0	0	0	1	0	2	2	0	0	4	0	0	1	1	0

	capacity																					
Overarching	biocapacity	17	0	0	0	0	0	0	0	1	0	0	0	0	0	1	4	3	1	3	2	2
Overarching	Bio-capacity	6	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	0
Overarching	Biodiversity constraint	2	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0
Overarching	Biological capacity	10	0	0	0	0	0	0	0	0	0	1	0	1	1	4	0	0	0	1	2	0
Overarching	Biological limit	3	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	0	0	0
Overarching	biophysical capacity	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Overarching	biophysical constraint	2	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0
Overarching	biophysical limit	3	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0	0
Overarching	bio-physical limit	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Overarching	boundaries of the earth	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Planetary boundaries	boundaries of the planet	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Resource	boundaries of the resources	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Emission	Buffer capaci-ty	2	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Emission	buffering ca-pacity	6	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	2	1	0	0	1
Overarching	capacity of ecosystems	20	0	0	0	0	0	2	0	0	2	1	1	2	3	2	1	2	0	1	3	0
Overarching	capacity of nature	10	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	1	0	0	0
Overarching	capacity of the Earth	18	0	0	0	0	0	0	0	2	0	2	0	2	0	1	3	1	1	2	2	2
Overarching	capacity of the ecosystem	11	0	0	0	1	1	0	0	0	0	0	2	0	1	0	2	0	0	1	1	2

Overarching	capacity of the environment	44	0	0	0	0	1	2	3	3	3	2	4	2	3	2	5	3	1	3	4	3
Overarching	capacity of the planet	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0
Overarching	carrying capa-city	186	1	0	1	1	2	1	14	14	7	16	19	9	12	12	18	12	12	12	18	5
Resource	constraint of resources	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Overarching	constraints of nature	2	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0
Resource	constraints of resources	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Planetary boundaries	constraints of the planet	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Emission	critical load	12	0	0	0	1	1	2	2	0	2	0	0	0	3	0	1	0	0	0	0	0
Emission	dilution capa-city	2	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0
Overarching	ecological boundar	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Overarching	ecological capacity	8	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	1	1	2	2	0
Overarching	ecological contraint	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Overarching	ecological limit	50	0	0	0	0	0	0	1	1	3	4	1	0	5	1	5	8	5	4	8	4
Overarching	ecological threshold	5	0	0	0	0	0	0	0	0	1	0	0	0	1	0	1	0	1	1	0	0
Overarching	ecological tolerance	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Overarching	ecosystem capacity	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	1
Overarching	ecosystem constraint	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Overarching	ecosystem limit	2	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0

Overarching	environmental boundar	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0
Overarching	environmental capacity	32	0	0	0	0	0	0	2	2	1	2	1	2	4	1	4	5	4	0	2	2
Overarching	environmental constraint	101	0	0	0	0	0	4	4	4	3	5	2	4	7	7	12	7	12	13	10	7
Water	Environmental flow require-ment	28	0	0	0	0	0	1	3	3	1	2	3	2	2	1	1	1	1	1	4	2
Overarching	environmental limit	318	0	0	0	1	2	1	7	6	14	16	18	17	26	32	30	23	25	35	33	32
Overarching	environmental space	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Overarching	environmental threshold	9	0	0	0	0	2	0	1	0	0	1	0	0	0	1	0	0	3	0	0	1
Water	Environmental water requi-rement	8	0	0	0	0	1	1	1	0	1	0	1	0	1	1	0	0	0	0	1	0
Overarching	global hectare	23	0	0	0	0	0	0	0	1	0	2	2	0	3	5	4	1	1	3	0	1
Resource	land constraint	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0
Resource	land limitation	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0
Overarching	limit of the environment	5	0	0	0	0	0	0	0	0	1	1	0	1	0	0	0	2	0	0	0	0
Overarching	limits of na-ture	3	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	1	0	0
Overarching	Limits of the Earth	6	0	0	0	0	0	0	0	0	0	0	1	1	1	1	0	0	1	0	0	1
Overarching	limits of the ecosystem	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Overarching	limits of the environment	12	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	4	2	1	1	1
Overarching	limits of the planet	33	0	0	0	0	0	0	1	0	0	0	1	0	2	3	0	4	5	2	5	10
Resource	limits of the resources	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0

Overarching	Natural capa-city	15	0	0	0	0	0	0	0	0	1	3	2	2	1	1	1	1	1	1	1	0
Overarching	Natural cons-traint	3	0	0	0	0	0	0	0	0	0	0	1	0	0	2	0	0	0	0	0	0
Overarching	Natural limit	23	0	0	0	0	0	0	0	1	2	1	0	2	4	0	1	5	2	1	0	4
Overarching	Natural tole-rance	3	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	1	0
Planetary boundaries	planetary boundar	18	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	7	8
Planetary boundaries	Planetary limit	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Planetary boundaries	Planetary threshold	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Resource	productive capability	19	0	0	0	0	0	0	1	3	3	0	1	0	1	1	1	1	1	2	2	2
Resource	Productive space	4	0	0	0	0	0	0	0	1	0	1	1	0	0	1	0	0	0	0	0	0
Resource	regenerative capability	3	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1
Resource	regenerative capacity	29	0	0	0	0	0	2	0	1	2	2	2	2	1	1	5	1	1	3	3	3
Resource	renewable capacity	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Resource	renewable limit	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Resource	resource capa-city	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	1	0
Resource	resource cons-traint	119	0	0	0	0	0	0	2	2	0	1	2	1	1	8	11	11	17	17	31	15
Resource	Resource limit	41	0	0	0	0	0	0	0	1	1	1	1	3	2	3	6	2	4	5	8	4
Overarching	Safe limit	39	0	0	0	0	1	2	1	0	0	0	2	2	1	3	5	6	3	6	6	1
Planetary boundaries	safe operating space	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Emission	self-purifying capability	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1

Emission	self-purifying capacity	10	0	0	0	0	0	1	0	0	1	1	0	0	2	1	2	1	0	1	0	0
Resource	supply limit	7	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	2	0	0	2
Overarching	tipping point	24	0	0	0	0	0	0	0	0	0	0	0	1	0	4	2	5	3	2	3	4
Overarching	Tolerable damage	7	0	0	0	0	0	0	0	0	0	0	1	0	1	1	1	1	0	1	1	0
Overarching	tolerable im-pact	12	0	0	0	0	0	1	0	0	0	2	2	2	1	2	2	0	0	0	0	0
Climate change	two degrees	92	0	0	0	0	0	0	0	0	0	1	1	0	6	7	7	25	13	14	4	14
Water	water constra-int	29	0	0	0	0	0	0	0	0	0	0	0	0	2	1	2	2	1	7	11	3
Water	water limitati-on	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	1	0	0
Total		**1933**	**1**	**0**	**1**	**4**	**13**	**21**	**47**	**53**	**56**	**73**	**84**	**85**	**122**	**141**	**194**	**212**	**183**	**225**	**232**	**186**
CR reports published		**39682**	**10**	**18**	**52**	**93**	**237**	**459**	**836**	**1092**	**1408**	**1613**	**1843**	**2049**	**2414**	**2913**	**3473**	**4081**	**4450**	**4630**	**4586**	**3425**

S4. Qualifying aspects of B and C companies

Table S3 shows the rationale for including each company in the B or C group for the different years.

Table S3: Rationale for including each company in the B or C group for every year. System boundaries for product level and aggregated production levels targets are in lower case letters and capital letters respectively.

Name of company	Sector	Country	Publication year	B1	B2	C1	C2	System boundary and level	Rationale
Acciona SA	Construction & Materials	Spain	2010		X			Unclear	The company has made a 2010-2013 Strategic Plan with a reduction target that "would account for 0.5% of the total reduction in 2013 needed to attain the International Energy Agency's proposal to stabilize global CO_2 concentrations at around 450 ppm."
Alcatel-Lucent	Technology Hardware & Equipment	France	2012		X		X	OWN OPERATIONS***	Motivated by e.g. "resource constraints" the company has committed to reduce "absolute carbon footprint from our operations by 50% by 2020 from its 2008 baseline", is engaging with the ""GreenTouch™ Consortium initiated by Bell Labs, to make communications networks 1,000 times more energy efficient than they are today." and plans to engage in more activities "enabling a low-carbon economy".
Alstom SA	Electronic & Electrical Equipment	France	2008				X	-	The company is developing CCS products: "In the medium term, Alstom should be in a position to offer power plants that are fully compatible with environmental constraints related to global warming."
Alstom SA	Electronic & Electrical Equipment	France	2009			X		-	"Alstom offers its customers a "CCS Ready" plant concept. This concept takes into account the needs of customers who purchase plants today that will ensure they are not financially penalized when the technology becomes available."
Alstom SA	Electronic & Electrical	France	2010			X		-	"Alstom offers its customers a "CO_2 capture Ready" plant concept. This concept takes into account the needs of cus-

	Equipment								tomers who purchase plants today that will ensure they are not financially penalized when the technology becomes available."
Alstom SA	Electronic & Electrical Equipment	France	2012			X	X	-	Based on reduction needs of the global power sector, calculated by IEA, to avoid exceeding the 2°C threshold, the company presents ongoing and planned technological developments in power generation, such as CCS, increased flexibility to aligned with larger fraction of renewables in grid, and renewable energy technologies.
Autodesk Inc	Software & Computer Services	USA	2014		X	X		OWN OPERATIONS***	Has developed a methodology (C-FACT) that calculates company GHG reduction targets "in line with global scientific and policy climate stabilization targets, and in proportion to their relative contribution to the economy". They use this methodology on their own company and have made it open source so that other companies can use it.
BMW AG	Automobiles & Parts	Germany	1999				X	-	"It would be irresponsible to test the carrying capacity of the ecosystem to its limits. BMW is therefore already making preparations for a large-scale fuel changeover, i.e. replacing hydrocarbon fuels as vehicle fuel and establishing on the market vehicles with almost zero emissions."
Bridgestone Corporation	Automobiles & Parts	Japan	2012		X			Unclear	Committed to reducing "overall CO_2 emissions by at least 50%" by 2050. Also presents 2020 goals, back-casted for the 2050 vision, for CO_2 reduction in companies operations and "after-use" (35%) and use stage (25% by improving rolling efficiency).
Bridgestone Corporation	Automobiles & Parts	Japan	2013		X			Unclear	Committed to reducing "overall CO_2 emissions by at least 50%" by 2050. Also presents 2020 goals, back-casted for the 2050 vision, for CO_2 reduction in companies operations and "after-use" (35%) and use stage (25% by improving rolling efficiency).
British Airways plc	Travel & Leisure	UK	2011		X			Unclear	Based on the 2 degree target "By 2050, emissions must be reduced by at least 50% to meet this objective, and this is the basis for our emissions targets."
British Airways plc	Travel & Leisure	UK	2012		X			Unclear	Based on the 2 degree target "aviation emissions must be reduced by 50% by 2050."

British Airways plc	Travel & Leisure	UK	2013		X			Unclear	Based on the 2 degree target the company aims to "Reduce net carbon dioxide emissions through a cap on emissions from 2020 (Carbon Neutral Growth), and a 50 percent cut in net CO_2 emissions by 2050 relative to 2005."
BT Group plc	Fixed Line Tele-communications	UK	2014				X	-	"Our vision is to use our products and people to help society live within the constraints of the planet's resources. Our 2020 3:1 goal is to help customers reduce carbon emissions by at least three times the end-to-end carbon impact of our business." and "By December 2020, we will reduce our CO_2 emission intensity by 80% against 1996/97 levels"
Cisco Systems Inc	Technology Hardware & Equipment	USA	2014	X		X		OWN OP-ERA-TIONS***	Based on the 2 degree target the company's "GHG goal is a 40 percent reduction in absolute emissions by FY17 (FY07 baseline)". Furthermore creates new "market opportunities" from the "enabling effect" of network technologies by "of-fering low-carbon ways to avoid business travel" and "Providing connected energy management."
Colgate-Palmolive Com-pany	Personal Goods	USA	2014	X				OWN OP-ERA-TIONS***	Based on the 2 degree target the company has a "commit-ment to reduce carbon emissions on an absolute basis by 25 percent compared to 2002, with a longer-term goal of a 50 percent absolute reduction by 2050 compared to 2002."
Danisco A/S	Food Producers	Denmark	2010				X	-	Identifies four critical global challenges in the light of Earth's limited carrying capacity and presents a plan on how to align product portfolio to these, e.g. involving the switch from petroleum to bio-based energy sources, chemicals and materials.
Electrolux AB	Household Goods	Sweden	2010		X			Unclear	Motivated by the 2 degree target the company aims to ex-ceed the EU target of reducing greenhouse gas emissions by 20% in 2020 compared to 1990: "The Group's targets ex-ceed these goals. In its new savings target for operations, the Group intends to attain this already in 2012."
Eneco Holding NV	Gas, Water & Multiutilities	The Net-herlands	2014	X				full life cycle	One planet thinking approach aims to define sustainable levels of impact per kWh electricity supplied, taking plane-tary boundaries and fair shares into account.
Ford Motor Company	Automobiles & Parts	USA	2008		X			well-to-wheel	Based on an internal model "addressing how light-duty transport could contribute to meeting 450 ppm to 550 ppm stabilization pathways" the company "is targeting a 30 per-

									cent reduction in U.S. and EU new vehicle CO_2 emissions, relative to the 2006 model year baseline, by 2020". An action plan for meeting the target was developed.
Ford Motor Company	Automobiles & Parts	USA	2009				X	-	"because of energy costs, climate change concerns, infrastructure constraints and resource limits, business as usual will not work….we are using our mobility expertise to forge partnerships among Ford, municipal governments and utilities aimed at building markets for electric vehicles."
Ford Motor Company	Automobiles & Parts	USA	2011		X		X	OWN OPERATIONS***	The company presents a "global technology migration path" that outlines planned changes in product portfolio, e.g. implementation of hybrid technology, and makes a commitment to reduce "facility CO_2 emissions by 30 percent by 2025 on a per-vehicle basis.”
Galp Energia SGPS SA	Oil & Gas Producers	Portugal	2013			X	X	-	In response to challenge of achieving World Energy Outlook 2012 450 ppm scenario the company makes several targets related to biofuel share of portfolio and present ongoing activities to reach targets.
Hitachi Koki Co Ltd	Electronic & Electrical Equipment	Japan	2011		X			FULL LIFE CYCLE	Motivated by the 450 ppm target the company aims to reduce annual CO_2 emissions by 100 million tons by fiscal 2025 through eco-efficiency improvements of products and services via eco-design.
Hitachi Koki Co Ltd	Electronic & Electrical Equipment	Japan	2012		X			FULL LIFE CYCLE	Motivated by the 450 ppm and 2 degree target the company aims to reduce annual CO_2 emissions by 100 million tons by fiscal 2025 through eco-efficiency improvements of products and services via eco-design.
Hitachi Ltd	Electronic & Electrical Equipment	Japan	2008		X			FULL LIFE CYCLE	Motivated by the 450 ppm target the company plan to "contributing to the IPCC target by helping to reduce CO_2 emissions from the use of Hitachi Group products worldwide by 100 million tons before 2025" through eco-efficiency improvements of products and services via eco-design.
Hitachi Ltd	Electronic & Electrical Equipment	Japan	2010		X			FULL LIFE CYCLE	Motivated by the 450 ppm target the company aims to reduce annual CO_2 emissions by 100 million tons by fiscal 2025 through eco-efficiency improvements of products and services via eco-design.
Honda Motor Company Ltd	Automobiles & Parts	Japan	2007				X	-	Presents supply limits of corn feedstock for conventional bio-ethanol product and describe developed 2nd generation

									technology (cellulose and hemicellulose).
Iberdrola SA	Electricity	Spain	2011	X				Unclear	Based on the 2 degree target the company is "targeting a 30% reduction in its emissions by 2020 as compared to 2007."
Iberdrola SA	Electricity	Spain	2011	X				Unclear	Based on the 2 degree target the company has "has set a target to reduce the intensity of its emissions by 30 % in 2020 as compared to 2007."
Implats	Mining	South Af-rica	2014	X				Unclear	Based on the WBCSD 2050 vision (living well within the limits of the planet) the company has committed to "Con-tribute to 2050 goal of 50% reduction in global CO_2 emis-sions on 2005 levels"
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2007		X		X	well-to-wheel	Based on the 2 degree threshold the company presents target for "well-to-wheel" CO_2 emissions for new vehicles" of 70% in 2050 compared with levels in 2000. "The 70% reduction target cannot be met even with these technologies, though, unless renewable energy is used to power the motors. Strengthening coordination with the energy sector will thus be essential."
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2008		X		X	well-to-wheel	Based on the 2 degree threshold the company presents target for "well-to-wheel" CO_2 emissions for new vehicles of 70% in 2050 compared with levels in 2000. "Nonetheless, the 70% CO_2 reduction target cannot be met even with these technologies unless renewable energy is used to power the motors and/or recharge the batteries. It will therefore be key to strengthen coordination with the energy sector."
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2009		X		X	well-to-wheel	Based on 450ppm and 550 ppm scenarios the company pre-sents target for "well-to-wheel" CO_2 emissions for new ve-hicles" of 70% in 2050 compared with levels in 2000. "Over the longer term, it is unlikely that the 70% CO_2 reduction target can be met without the spread of electric-powered vehicles, such as electric and fuel-cell vehicles, and the use of renewable energy as a source of power for them."
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2010		X		X	well-to-wheel	Based on 450ppm (550ppm?) scenario the company presents target for "well-to-wheel" CO_2 emissions for new vehicles" of 90% (note: higher than 70% target defined in 2009 report) in 2050 compared with levels in 2000. "If the 90% emissions

									reduction target is to be met, there will have to be greater use of electric-powered vehicles, such as electric and fuel-cell vehicles, over the longer term."
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2011		X		X	well-to-wheel	Defines 2050 reduction target (90% compared to 2000) based on 2 degree target. Over the long term the company sees the need to "bring about widespread use of electric and fuel-cell vehicles, making use of renewable energy sources to provide the power they need"
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2012		X		X	well-to-wheel	Defines 2050 reduction target (90% compared to 2000) based on 2 degree target. "Over the long term, we need to increase the adoption of electric vehicles and fuel-cell electric vehicles and to make use of renewable energy to power these technologies while each country and region moves toward more renewable energy sources. We are advancing technological development on the basis of this future scenario."
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2013		X		X	well-to-wheel	Defines 2050 reduction target (90% compared to 2000) based on 2 degree target. "Over the long term, we need to increase the adoption of electric vehicles and fuel-cell electric vehicles (EVs and FCEVs) and to make use of renewable energy to power these technologies while each country and region moves toward more renewable energy sources. We are advancing technological development on the basis of this future scenario."
Nissan Motor Co Ltd	Automobiles & Parts	Japan	2014		X		X	well-to-wheel; OWN OPERATIONS***	Defines 2050 reduction target (90% compared to 2000) based on 2 degree target. "Over the long term, we need to increase the adoption of electric vehicles and fuel-cell electric vehicles (EVs and FCEVs) and to make use of renewable energy to power these technologies while each country and region moves toward more renewable energy sources. We are advancing technological development on the basis of this future scenario." "Nissan has also calculated that it needs to reduce CO_2 emissions from its corporate activities by 80% by 2050 compared with levels in 2000. Accordingly, it plans to continue its energy efficiency measures, leverage the power storage ability of lithium-ion batteries and expand

									its use of renewable energy."
Novelis Inc	Industrial Metals	USA	2013		X			FULL LIFE CYCLE	Based on the existence of a "upper safe limit for absolute concentration of carbon dioxide (CO_2) in the Earth's atmosphere" the company aims to reduce absolute emissions by 50% across entire value chain. The company plans to achieve this by a combination of increasing recycling content and energy efficiency within own operations.
Novelis Inc	Industrial Metals	USA	2014		X			FULL LIFE CYCLE	Based on the existence of a "upper safe limit for absolute concentration of carbon dioxide (CO_2) in the Earth's atmosphere" the company aims to reduce absolute emissions by 50% across entire value chain. The company plans to achieve this by a combination of increasing recycling content and energy efficiency within own operations.
PTT Public Company Limited	Oil & Gas	Thailand	2012	X				OWN OPERATIONS***	"Our long-term greenhouse gas emission target is to reduce 15 percent of Scope 1 and 2 emissions by 2020 against our business as usual protection in 2011. The target has been designed to be in-line with the shared vision of the global communities and scientific research to prevent the global average temperature increase to below 2 degrees Celsius."
PTT Public Company Limited	Oil & Gas Producers	Thailand	2013	X				OWN OPERATIONS***	2020 goal (15% reduction compared to business as usual) for GHG emissions "designed to be in line with" the 2 degree target.
Ricoh Company Ltd	Technology Hardware & Equipment	Japan	2005		X			FULL LIFE CYCLE	Defines 2007 and 2010 goals for climate change, resource use and impacts from chemical substances based on backcasting of 2050 goal for "tolerable impact" (unclear how it was derived).
Ricoh Company Ltd	Technology Hardware & Equipment	Japan	2006		X			FULL LIFE CYCLE	Defines 2007 and 2010 goals for climate change, resource use and impacts from chemical substances based on backcasting of 2050 goal for "tolerable impact" ("Advanced nations need to reduce their environmental impact to one-eighth the fiscal 2000 levels by 2050").
Ricoh Company Ltd	Technology Hardware & Equipment	Japan	2007		X			FULL LIFE CYCLE	Defines 2013 and 2010 goals for climate change, resource use and impacts from chemical substances based on backcasting of 2050 goal for "tolerable impact" ("Advanced nations need to reduce their environmental impact to one-eighth the fiscal 2000 levels by 2050").

Ricoh Company Ltd	Technology Hardware & Equipment	Japan	2008		X			FULL LIFE CYCLE	Defines 2013 and 2010 goals for climate change, resource use and impacts from chemical substances based on back-casting of 2050 goal for "tolerable impact" ("Advanced nations need to reduce their environmental impact to one-eighth the fiscal 2000 levels by 2050").
Ricoh Company Ltd	Technology Hardware & Equipment	Japan	2009		X			FULL LIFE CYCLE	Defines 2020 and 2050 goals for climate change, resource use and impacts from chemical substances based on back-casting of IPCC reduction recommendations (87.5% in 2050, also used for non-climate change related goals).
Samsung SDI Co Ltd	Electronic & Electrical Equipment	Republic of Korea	2010				X	-	Based on the 450 ppm limit the company presents various planned product portfolio changes related to energy technologies.
Seiko Epson Corporation	Technology Hardware & Equipment	Japan	2008	X				full life cycle	"Recognizing that the Earth's carrying capacity is limited and believing that everyone must share responsibility for reducing environmental impacts equally, Epson is aiming to reducing CO_2 emissions by 90% across the lifecycle of all products and services by the year 2050."
Seiko Epson Corporation	Technology Hardware & Equipment	Japan	2009	X				full life cycle	"Recognizing that the Earth's carrying capacity is limited and believing that everyone must share responsibility for reducing environmental impacts equally, Epson is aiming to reducing CO_2 emissions by 90% across the lifecycle of all products and services by the year 2050."
Seiko Epson Corporation	Technology Hardware & Equipment	Japan	2010		X			full life cycle	"Recognizing that the Earth's carrying capacity is limited and believing that everyone must share responsibility for reducing environmental impacts equally, Epson is aiming to reducing CO_2 emissions by 90% across the lifecycle of all products and services by the year 2050." A back-casting approach is outlines.
Seiko Epson Corporation	Technology Hardware & Equipment	Japan	2011		X			full life cycle	"Recognizing that the Earth's carrying capacity is limited and believing that everyone must share responsibility for reducing environmental impacts equally, Epson is aiming to reducing CO_2 emissions by 90% across the lifecycle of all products and services by the year 2050." A back-casting approach is outlines.
Seiko Epson	Technology	Japan	2012		X			full life	No explicit state based climate target is defined, but Epson is

Corporation	Hardware & Equipment							cycle	committed to reducing CO_2 emissions by 90% across the lifecycle of all products and services by the year 2050 based on the premise that "everyone must share responsibility for reducing environmental impacts equally."
Seiko Epson Corporation	Technology Hardware & Equipment	Japan	2014		X			full life cycle	Based on the CO_2 absorption capacity in 2050 according to IPCC and the projected ratio of Japan's population, the company concludes that Japan needs to reduce its emissions by 90% (no baseline year) and defines similar company reduction target.
Skretting AS	Food Producers	Norway	2013		X			SUPPLIERS	Undergoing transition from basing feed for carnivorous species on fishmeal and fish oil to basing it on agricultural products to avoid contributing to exceeding sustainable limits of fish catch and defines short term milestones for meeting this target.
Spier Leisure Holdings	Travel & Leisure	South Africa	2008		X			OWN OPERATIONS***	"Spier's entire carbon emissions footprint will be zero. This will be achieved by conservation and completely shifting all energy requirements to renewable sources."
Toshiba Corporation Semiconductor Company	General Industrials	Japan	2010		X			full life cycle	Calculate global average eco-efficiency improvement at the product level in 2050 required to respect limit of the environment and use this as product level target in 2050 and designsmilestones for 2015 and 2025.
Unilever plc / NV	Food Producers	UK	2010		X			OWN OPERATIONS***; full life cycle	"we will meet the United Nations' requirement to reduce GHGs by 50-85% by 2050 in order to limit global temperature rise to two degrees". Further commits to "halve the environmental footprint of the making and use of our products" by 2020 and presents a plan on how to meet the two targets."
Zhong Xing Telecommunica-tion Equipment Company Limited	Fixed Line Telecommunications	People's Republic of China	2013			X		-	Based on the Internet of Things concept the company has developed a real time environmental monitoring network to be used by e.g. EPAs and offers smart grid solutions to remedy the exceeding of environmental capacity.
Zhong Xing Telecommunica-tion Equipment Company Limited	Telecommunication Services	People's Republic of China	2014			X		-	Based on the Internet of Things concept the company has developed a real time environmental monitoring network to be used by e.g. EPAs and offers smart grid solutions to remedy the exceeding of environmental capacity.

***2010-2011 CR report was published in 2012. **2010 CR report published in 2011. *** Own operations include direct energy consumption (electricity, heat and steam) and thus corresponds to a scope 1-2 systems boundary in the terminology of the Greenhouse Gas Protocol (Ranganathan et al., 2004)**

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IV

Modifying life cycle assessment to measure absolute environmental sustainability

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Modifying life cycle assessment to measure absolute environmental sustainability

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Abstract

Environmental monitoring indicates that progress towards the goal of environmental sustainability in many cases is slow, non-existing or negative. Indicators that use environmental carrying capacity references to evaluate whether anthropogenic systems are, or will potentially be, environmentally sustainable are therefore increasingly important. Such absolute indicators exist, but suffer from shortcomings such as incomplete coverage of environmental interferences, varying quality of inventory data and varying or insufficient spatial resolution. The purpose of this article is to demonstrate that Life Cycle Assessment (LCA) can potentially reduce or eliminate these shortcomings.

We developed a generic mathematical framework for the use of carrying capacity as environmental sustainability reference in spatially resolved life cycle impact assessment models and applied this framework to the LCA impact category terrestrial acidification. In this application carrying capacity was expressed as acid deposition (eq. mol $H^{+}\cdot ha^{-1}\cdot year^{-1}$) and derived from two complementary pH related thresholds. A geochemical steady-state model was used to calculate a carrying capacity corresponding to these thresholds for 99,515 spatial units worldwide. Carrying capacities were coupled with deposition factors from a global deposition model to calculate characterisation factors (CF), which expresses space integrated occupation of carrying capacity (ha·year) per kg emission. Principles for calculating the entitlement to carrying capacity of anthropogenic systems were then outlined, and it was demonstrated that a studied system can be considered environmentally sustainable if its indicator score (carrying capacity occupation) does not exceed its carrying capacity entitlement. The developed CFs and entitlement calculation principles were applied to a case study evaluating emission scenarios for

personal residential electricity consumption supplied by production from 45 US coal fired electricity plant.

Median values of derived CFs are 0.16-0.19 ha·year·kg^{-1} for common acidifying compounds. CFs are generally highest in Northern Europe, Canada and Alaska due to the low carrying capacity of soils in these regions. Differences in indicator scores of the case study emission scenarios are to a larger extent driven by variations in pollution intensities of electricity plants than by spatial variations in CFs. None of the 45 emission scenarios could be considered environmentally sustainable when using the relative contribution to GDP or the grandfathering (entitlement proportional to past emissions) valuation principles to calculating carrying capacity entitlements. It is argued that CFs containing carrying capacity references are complementary to existing CFs in supporting decisions aimed at simultaneously reducing environmental interferences efficiently and maintaining or achieving environmental sustainability.

We have demonstrated that LCA indicators can be modified from relative to absolute indicators of environmental sustainability. Further research should focus on quantifying uncertainties related to choices in indicator design and on reducing uncertainties by achieving consensus on these choices.

Keywords:

LCA; Terrestrial acidification; Carrying capacity; characterisation factors; entitlement

1 Introduction

During the last decades the number of sustainability indicators and their use in decision-making has greatly increased (Hak et al., 2012; Singh et al., 2012). Many such indicators rank the sustainability of anthropogenic systems. For instance Switzerland ranked highest and Somalia lowest in the 2014 Environmental Performance Index of countries (Hsu et al., 2014). Another example is Greenpeace's Guide to Greener Electronics (2012b;2012a), which ranks 16 large electronics companies. Here we term indicators used for ranking *relative environmental sustainability indicators* (RESI) because indicator scores of studied anthropogenic systems are relative because they are evaluated by comparison to indicator scores of one or more reference systems, chosen specifically to match the nature or function of the studied system. While RESI can reveal how the sustainability performance of system X compare to that of a chosen reference system, it cannot evaluate whether system X can be considered sustainable on an absolute scale

(Moldan et al., 2012). This limitation is very problematic considering that the state of the environment is declining by and large (Steffen et al., 2015; WRI, 2005). Therefore the global economy and its subsystems are in fact drifting further away from the goal of environmental sustainability, originally defined as "seek[ing] to improve human welfare by protecting the sources of raw materials used for human needs and ensuring that the sinks for human wastes are not exceeded, in order to prevent harm to humans" (Goodland 1995).

This shortcoming of RESI may be addressed by supplementing RESI by indicators containing reference values of environmental sustainability (Moldan et al., 2012). We term such indicators *absolute environmental sustainability indicators* (AESI) because the environmental sustainability references are absolute, since they are based on characteristics of natural systems independent of the study. While ranking of products or systems is also possible in AESI, the environmental sustainability of a system can additionally be evaluated on an absolute scale, i.e. answering the question "is system X environmentally sustainable or not?" Figure 1 illustrates the difference and complementarity between RESI and AESI.

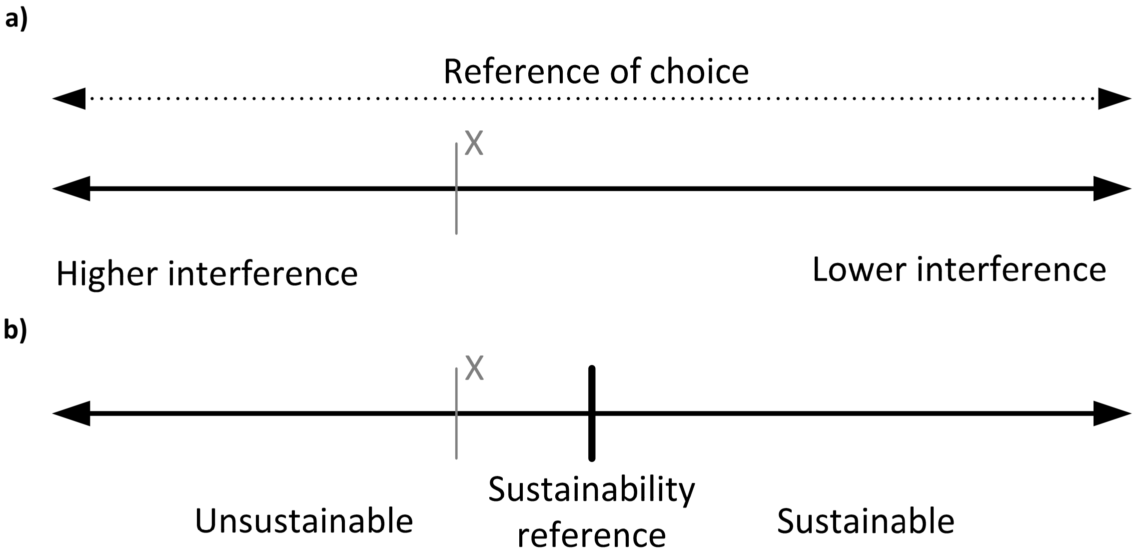


Figure 1: The concepts of relative (a) and absolute (b) environmental sustainability indicators. The ranking of the hypothetical system X depends on the chosen reference(s) (a). System X is environmentally unsustainable because its environmental interference is higher than the sustainability reference (b).

The concept of carrying capacity (Sayre, 2008) can be applied in AESI to operationalize and quantify references for environmental sustainability as defined by Goodland (1995). Following Bjørn and Hauschild (2015) we define carrying capacity as "the maximum sustained environmental interference a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert." Here we use "environmental interference" as a generic term for anthropogenic changes to any point in an impact pathway (from emission or resource use to ultimate damage). It follows that total environmental interferences on natural systems, whether caused by resource uses or emissions, can be considered environmentally sustainable if their level is below the affected eco-system's carrying capacity.

"Footprinting" indicators, that use carrying capacity as sustainability reference value, can be characterized as AESI. The popular ecological footprint indicator expresses demands on nature in units of "global hectares" and compares this to land availability (termed "biocapacity") to facilitate an evaluation of whether demands are environmentally sustainable (Borucke et al., 2013). This has inspired other footprint indicators such as the well-established water footprint (Hoekstra and Mekonnen, 2012) and first generation chemical footprints (Bjørn et al., 2014; Zijp et al., 2014). Existing footprinting indicators, however, have weaknesses such as: 1) the incomplete coverage of all environmental interferences that are threatening environmental sustainability, 2) the varying data sources which are generally crude for assessments at the product scale (Huijbregts et al., 2008; Kitzes et al., 2009), 3) the variations in spatial resolution amongst footprints[47], which can be a source of bias due to the potentially high spatial variability of carrying capacity (Bjørn and Hauschild, 2015), and 4) the inconvenience for users that each indicator is made available by means of a unique software tool. We believe that the life cycle assessment (LCA) method has the potential to overcome these weaknesses of current AESI.

LCA aims to cover all relevant environmental interferences over the life cycle (from raw materials to waste management) of a product (or other anthropogenic systems). LCA requires a life cycle inventory (LCI), which compiles the physical inputs and outputs (resource uses and emissions) of a product during its life cycle, and is commonly based on product system specific data supplemented by a common life cycle inventory database of unit processes (e.g. the average electricity generation of a country). LCA uses characterisation factors (CFs), which express the relationship between the resource uses or emissions of a LCI and measures of resulting environmental interference. CFs are obtained from mathematical representations of cause effect-chains that can be spatially resolved and allow the conversion of a LCI into indicator scores for a number of mutually exclusive and col-

[47] The ecological footprint normalises land demands in the unit "global hectares", which means that indicator results are unaffected by spatial differences in yield, while water- and chemical footprints are spatially resolved to varying extents.

lectively exhaustive “impact categories” such as climate change, eutrophication and eco-toxicity.

The characteristics of LCA make it potentially suitable for reducing or eliminating the listed weaknesses of current AESI. However LCA indicators can be characterized as RESI: Indicator scores are typically used to rank the environmental performance of functionally comparable product systems or scenarios, based on their potential to, via their emissions or resource uses, create a small change in the level of environmental interferences. This small change is either calculated as a marginal change in the known existing level of environmental interference or as an approximated linear change in interference within the zone between 0 and a chosen level of interference (see S1 for a conceptual figure of the two approaches) (Hauschild and Huijbregts, 2015). LCA indicators therefore generally do not include carrying capacity as sustainability reference values (Castellani and Sala, 2012). To harness the potentials of LCA in AESI, LCA indicators need to be modified to quantifying occupations of carrying capacity instead of quantifying small changes in levels of environmental interferences. The overall purpose of this article is to provide an initial contribution to this development.

This article aims to 1) develop a generic mathematical expression for calculating spatially resolved occupation of carrying capacity for any emissions based LCA impact category, 2) use this method tentatively on the terrestrial acidification LCA impact category, 3) demonstrate the applicability of the method in a case study, , 4) compare the relevance and complementarity of AESI and RESI in decision support.

2 Methods

2.1 Definitions and interpretations

To support the operationalization of carrying capacity (defined as “the maximum sustained environmental interference a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert”) we introduce two definitions: 1) control variable: “a numerical indicator of the structure and/or functioning of a natural system.”; 2) Threshold: “the maximum value of a control variable a natural system can withstand without experiencing negative changes in structure and/or functioning that are difficult or impossible to revert.” The carrying capacity is generally closer to the cause in an impact pathway than the threshold from which it is derived. Carrying capacity is static because it is calculated from a situation where a control variable value equals a threshold value at steady state (Bjørn and Hauschild, 2015). Note that the definitions of threshold and carrying capacity leave room for interpretation (what are negative changes and at what point do these become difficult to revert?). This interpretative flexibility is intentional as it reflects the ambiguity in the definition of environmental sustainability of Goodland (1995) with respect to preventing “harm to humans”: Humans may be physically harmed by a reduction of material eco-system services (e.g. access to clean water) caused by severe environmental degradation. According to some, humans may also be harmed culturally and spiritually by effects on or disappearance of a single vulnerable species caused by just minor environmental degradation. Environmental sustainability can thus be interpreted anthropocentrically or eco-centrically (or somewhere in between), which can greatly influence the choice of threshold and resulting quantification of carrying capacity. The sensitivity of AESI scores to this interpretation of environmental sustainability and other choices is analysed in Bjørn et al. (2015).

2.2 Characterisation framework

In LCA characterisation factors (CF) are multiplied with each inventoried emission or resource use (Q) of pollutants or resource (x) that contribute to a given impact category and the products are summed to calculate the indicator score (IS) for that impact category:

$$IS = \sum_x CF_x \cdot Q_x \quad (1)$$

By integrating carrying capacity as sustainable reference value in CFs, indicator scores can be expressed as occupation of carrying capacity. We propose this integration by dividing spatially resolved conventional CF constituents by carrying capacity (CC) for any emissions based indicator (aim 1):

$$CF_{x,i,k} = \sum_j \frac{FF_{x,i,k,j} \cdot XF_{i,j} \cdot EF_{i,j}}{CC_j} \quad (2)$$

Here CF ($ha*year*kg_{emitted}^{-1}$) is the characterisation factor for substance x emitted within spatial unit i into environmental compartment k (air, soil or water). FF is a fate factor linking an emission of pollutant x within i into k to its fate typically expressed as a change in concentration or mass in the receiving spatial unit j. XF is an exposure factor which accounts for the fraction of pollutant x that species of concern in j are exposed to. EF is an effect factor, which calculates the effect increase on these species in j from an increased exposure of x. CC is the carrying capacity in j. The metric of CC depends on the metrics of FF, XF and EF and differs from one impact category to another. Note that equation 2 applies to indicators of effects on species. If indicator scores are expressed closer to the cause of these effects the denominator should only contain FF or FF·XF. When following equation 1 by multiplying CFs with emissions (kg) the indicator score is expressing the carrying capacity occupation in a unit of ha·year, which indicates an area in which carrying capacity for a given impact category is occupied for a time. If the time frame during which pollutants are emitted is known, the indicator score can be expressed in a unit of ha, which resembles that of the ecological footprint method (Borucke et al., 2013).

Note that our proposed framework is only compatible with indicators for which FF, XF or EF are of a linear nature, i.e. that calculate the approximated linear environmental change from an emission within the zone between 0 and a chosen level of interference (see S1). Our proposed framework is not compatible with marginal CF components because these are derivatives of estimated existing levels of environmental interference, while carrying capacity should be independent of existing levels of environmental interference (Bjørn and Hauschild, 2015).

2.3 Application to terrestrial acidification

We demonstrate the calculation of proposed characterisation factors for the LCA impact category terrestrial acidification, for which no AESI currently exists (aim 2). The spatial derivation was based on the only existing global deposition model of Roy et al. (2012) having a 2.0°x2.5° resolution (i.e. composed of 13,104 grid cells).

2.3.1 Choice of control variable and threshold

As a basis for carrying capacity two complementary thresholds of the control variable "soil solution pH" were chosen. The first threshold was based on a deviation of natural pH corresponding to the point where the numerical decrease in pH starts increasing for every additional quantity of deposition. At this point the functioning of the soil ecosystem starts changing as the carbonate buffering system is weakening and additional depositions will bring the system close to its chemical pH threshold.[48] Based on a screening of pH curves modelled with the geochemical steady-state model PROFILE (Warfvinge and Sverdrup, 1992) we found that a pH decrease of 0.25, compared to natural pH, generally corresponded well with this point where pH starts responding non-linearly to additional depositions (see S2). The second threshold was required to take into account naturally acidic soils for which the critical factor threatening ecosystem structure is not pH decrease, but rather the mobilisation of toxic aluminium (III) from the buffering

[48] We did not choose the steepest point of the chemical pH threshold as basis for carrying capacity because this point is often 2 pH units or more below natural pH, which represents a pH decrease that few species can tolerate (Azevedo et al., 2013) and can therefore not be considered as reference for environmental sustainability.

of acid depositions through reaction with aluminium oxides and hydroxides from clay particles (Sparks, 2002). This buffering process occurs in the pH interval 2.8-4.2 and we therefore chose pH 4.2, below which aluminium (III) starts to mobilize, as the second threshold.[49] In other words, we interpreted environmental sustainability, with regards to the interference of acidifying compounds with natural soils, to correspond to a situation where natural buffer systems are not weakened and aluminium (III) is not mobilized.

2.3.2 Calculation of carrying capacity

The carrying capacity was, inspired by the critical loads concept (Spranger et al., 2004), expressed as a critical deposition of acidifying compounds (eq.$\cdot$ha$^{-1}\cdot$year^{-1}, where 1 eq refers to 1 mol H^+-eq.). The carrying capacity was derived for 99,515 spatial units, covering the global terrestrial area (Roy et al., 2012a), by running PROFILE in 9 steps gradually increasing depositions of SO_X above natural levels for each spatial unit until a change of 0.25 pH units or an absolute pH value below 4.2 was reached. Natural depositions were modelled based on Tegen and Fung (1994) and Bey et al. (2001) as described in Roy et al. (2012b). The design of the 9 steps is explained in S2. We found that 10% of spatial units were for at least one deposition step affected by a non-convergence error in PROFILE. For these cells the carrying capacity was approximated by neighbouring cells using a kriging function, see S4. Area-weighted averages of the carrying capacities of the 99,515 spatial units of PROFILE were used to estimate the carrying capacities of the 13,104 grid cells of the deposition model of Roy et al. (2012). CFs were then calculated according to equation 2 using atmospheric fate factors (FF, $keq_{deposited}*kg_{emitted}^{-1}$) of Roy et al. (2012)[50] and excluding XF and EF in the denominator because CC is expressed as a critical deposition:

$$CF_{x,i} = \sum_j \frac{FF_{x,i,j}}{CC_j} \quad (3)$$

2.4 Carrying capacity entitlement

Our CFs can in principle be used to evaluate whether a society as a whole is environmentally sustainable because the indicator score, expressing the area equivalent of fully occupied carrying capacity, from all activities of the society can be compared to the actual area of the relevant ecosystem. An individual system em-

[49] Our choice of an absolute threshold of 4.2 pH units is in good agreement with a proposal within the critical loads framework that a pH of 4 could be used to calculate critical loads for forest soils (Spranger et al., 2004).
[50] The fate factors of Roy et al. (2012) were expressed in $kg_{deposited}*kg_{emitted}^{-1}$. For this study $kg_{deposited}$ was converted to $keq_{deposited}$ by division by the molecular weight of the emissions and multiplication by the electrical charges of their corresponding ions, following Posch et al. (2008).

bedded in society, such as a product, a person or company, can in turn be considered environmentally sustainable if it does not occupy more of the total carrying capacity than it can be considered entitled to. Carrying capacity entitlement is a normative concept because it depends on the perceived value of a studied system relative to those of "competing systems" that rely on occupying carrying capacity in the same area where the studied system occupies carrying capacity. Therefore environmental sustainability references for individual anthropogenic systems embedded in society are inherently normative. Below we outline three steps in deriving and applying these environmental sustainability references

2.4.1 Identify competing systems

Ideally competing systems would be identified by combining a source-receptor fate model with a spatially differentiated emission inventory covering all anthropogenic systems of society in a chosen reference year: The fate model would first identify the spatial units affected by emissions of the studied system. The fate model would then identify all the systems of the societal total emission inventory whose emissions affect the spatial units previously identified. These systems would be labeled competing systems because they rely on occupying parts of the same carrying capacity as the studied system for their functioning. Note that the group of competing systems is potentially unique for each affected spatial unit (of which there may be thousands). This is impractical to operate with and therefore three simplifications are introduced: 1) a cut-off criterion is established whereby only spatial units receiving above a specified share of emissions from the studied system (e.g. 0.1%) are considered (the territory of these spatial units are termed $T_{affected}$ and its area is termed $A_{affected}$), 2) all emissions that occur within $T_{affected}$ are, in this part of the AESI, assumed to occur in the spatial unit where the emission from the studied system occurs and thus assumed to have the same fate, 3) it is assumed that no emissions within $T_{affected}$ leave $T_{affected}$ and that no emissions from outside enters. These three simplifications are visually presented in Figure 2.

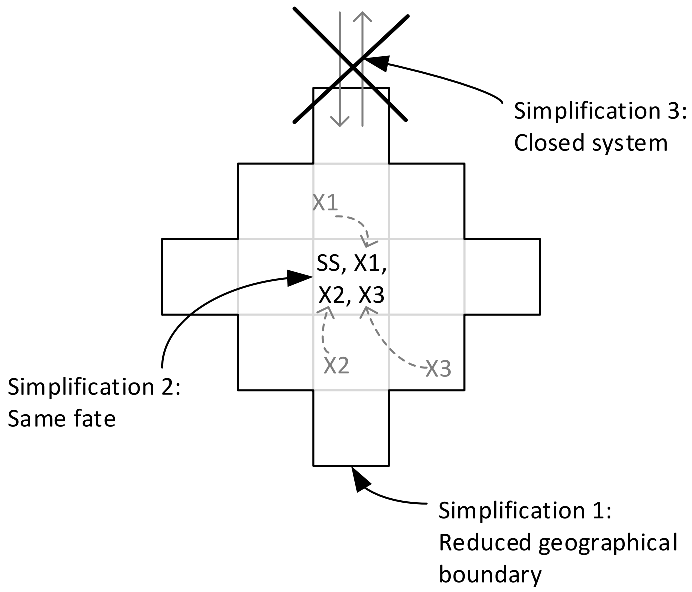


Figure 2: Illustration of three simplifications for identifying competing systems (X1-X3) of a studied system (SS) located in the middle grid cell and affecting 13 grid cells above an arbitrary emission distribution threshold. These 13 grid cells make up $T_{affected}$ and have the area $A_{affected}$. The dotted arrows indicate a change in location of X1-X3.

The consequence of the simplifications is that only one carrying capacity entitlement needs to be calculated for each emission location of a studied system and that the group of competing systems is the same for all anthropogenic systems within $T_{affected}$. The simplifications can be defended in situations where potential competing systems are rather homogenously distributed in space and have emissions of similar magnitude. When this is not the case it may be more appropriate to follow the ideal approach outlined above to identifying competing systems.

2.4.2 Quantify relative value of studied system

The perceived value of a studied system relative to identified systems competing for carrying capacity in the same territory may be quantified using different valuation principles, such as 1) relative contribution to GDP, or 2) "grandfathering" where the relative value of a system is considered proportional to its relative indicator score in a chosen past reference year (i.e. if total carrying capacity was exceeded in the reference year, the indicator scores of all systems in that reference year should be reduced by the percentage that is needed to reduce the total indicator score below the total carrying capacity. The perceived relative value of a studied system may be expressed as a value factor (VF) between 0 and 1 of the total value (i.e. the sum of the perceived value of the studied system and those of competing systems).

2.4.3 Calculate carrying capacity entitlement and compare to AESI score

The time-integrated area in which carrying capacity can be entitled to a studied system ($A_{entitled}$, in ha·year) can be calculated by multiplying $A_{affected}$ for the studied system by the duration of the emissions (t) and the value factor (VF) for each emissions location (i):

$$A_{entitled\_i} = A_{affected\_i} \cdot t_i \cdot VF_i \tag{4}$$

If $A_{entitled}$ exceeds the AESI score of a studied system for one or more emission locations (i) the studied system cannot be considered environmentally sustainable.

2.5 Case study

We applied the derived CFs to emissions caused by the electricity production from one randomly selected coal fired electricity plant in 45 states of contiguous United States[51] in 2010. For each of the electricity plants we calculated an emission inventory corresponding to the residential electricity consumption of an average inhabitant in the concerned state in the year 2010. The case study provided a vehicle for demonstrating the use of the proposed indicator for terrestrial acidification on 45 scenarios of realistic residential electricity consumption in a hypothetical situation where this is entirely supplied by coal (aim 3).[52] We use the term "scenario" to stress that we are not attempting to model the actual situation. The case study also allows for discussing the relevance of LCA-supported AESI compared to using LCA to rank environmental performance (aim 4).

State specific annual per capita residential electricity consumption was obtained from the US Department of Energy (DoE, 2015) and used to define the quantities of electricity produced (P) by each of 45 power plants (i) to meet demand for an average inhabitant. Power plant specific emissions intensities (EI) expressing emissions of SO_X and NO_X (x) per kWh of generated electricity were obtained from the eGRID database of the US EPA (2014), which contains data on a total of 541 US coal fired electricity plants in 45 states.[53] EI was multiplied by P to obtain the emissions (Q) of SO_X and NO_X per power plant (i). Indicator scores (IS) for each power plant were hence, following equation 1, calculated as:

[51] The contiguous United States consists of the 48 adjoining U.S. states plus Washington, D.C. (federal district).

[52] In reality residential electricity use is supplied by various energy technologies that, due to an integrated federal grid, may be located far away (i.e. in another state) than the location of consumption.

[53] The states of Maine, Rhode Island and Vermont were not covered by the eGRID database of coal fired electricity plants, presumably because they have none.

$$IS_i = \sum_x CF_{i,x} \cdot Q_x = \sum_x CF_{i,x} \cdot P_i \cdot EI_{i,x} \quad (5)$$

Here $CF_{i,k}$ is the characterisation factor derived for pollutant x (SO_X or NO_X) for the grid cell in which power plant i is located.

Indicator scores were evaluated by comparing them to carrying capacity entitlements established following the simplified approach outlined above: We used the fate model of Roy et al. (2012) to identify spatial units receiving depositions caused by emissions of the different power plants. This model predicts that all 13,104 grid cells of the global model receives a share of an emission from any of the power plants (Roy et al., 2012b). However, most grid cells receive a very small share. For identifying competing systems we therefore used a cut-off value of 0.1% deposition of an emission. This resulted in an affected territory ($T_{affected}$)for each i in which around 70% of an emission deposits (depending on the pollutant and i).[54] $A_{affected}$ (the area of $T_{affected}$) for all i and both pollutant are approximately equivalent to the area of the entire contiguous United States. Since all power plants are located in contiguous United States there is a great geographical overlap between $T_{affected}$ of the 45 emission scenario locations. This overlap justified the additional simplification of assigning the terrestrial area of contiguous United States, 765,300,400ha (USCB, 2012), a common $T_{affected}$ for all i. Competing systems for all i are consequently all systems that emit acidifying compounds to air within the contiguous United States.

In quantifying the value factors (VF) of the 45 studied emissions scenarios two alternative valuations were applied to explore the sensitivity of case study outcomes to value judgment. The first valuation was based on the relative contribution to GDP, estimated by dividing personal or household expenditure on a studied product or service by pre-tax income. In 2009 (no data for 2010) an average US household spent 2.0% of its pre-tax income on residential electricity (ACCCE, 2014). The relative contribution to GDP valuation principle thus grants residential electricity consumption a value of 0.02 relative to all other anthropogenic systems. The alternative valuation was based on the grandfathering principle, according to which US residential electricity consumption is entitled to maintain its past share of total environmental interferences. In 2010 38% of US total electricity consumption was consumed by the residential sector (IEA, 2012), meaning that 38% of environmental interferences from total electricity consumption could be attributed to the residential sector. We could not obtain the share of environmental interference with respect to terrestrial acidification taken up by to-

[54] The remaining share of an emission, on average 30%, deposits on grid cells receiving less than 0.1% of the emission and accumulates in high altitude, near the stratosphere.

tal electricity consumption of the total US environmental interference. We therefore approximated this share by the corresponding share in EU27, where in 2010 23% of total environmental interferences was presumably taken up by electricity production[55]. Our use of the grandfathering valuation principle thus grants residential electricity consumption in the US a tentative value of 9% (38% of 23%) relative to all other anthropogenic systems.

Since both valuation principles were applied to average residential electricity consumption in the US, the value factors for the 45 scenarios are the same (i.e. not calculated specifically for each emissions scenario, although this is in theory possible) and can be calculated by dividing the nationwide relative values with the US population (312,245,116 in 2010 (UNDESA, 2012)). $A_{entitled}$ was subsequently calculated for the alternative valuation principles following equation 4:

Relative contribution to GDP:

$$A_{entitled} = A_{affected} \cdot t \cdot VF = 765{,}300{,}400\text{ha} \cdot 1 year \cdot \frac{0.02}{312{,}245{,}116} = 0.049 ha \cdot year \tag{6}$$

Grandfathering:

$$A_{entitled} = A_{affected} \cdot t \cdot VF = 765{,}300{,}400\text{ha} \cdot 1 year \cdot \frac{0.09}{312{,}245{,}116} = 0.21 ha \cdot year \tag{7}$$

The two alternative $A_{entititlted}$ were compared to the indicator scores of the 45 scenarios to evaluate which of them could be considered environmentally sustainable. We then compared the spatial variation in each of the components of equation 5, including the CF components of equation 3, to analyse the sensitivity of indicator scores of the 45 scenarios to each of these components. As a basis for discussing the relevance of AESI compared to RESI we furthermore compared the CFs of the 45 power plant locations with corresponding CFs of Roy et al. (2014).

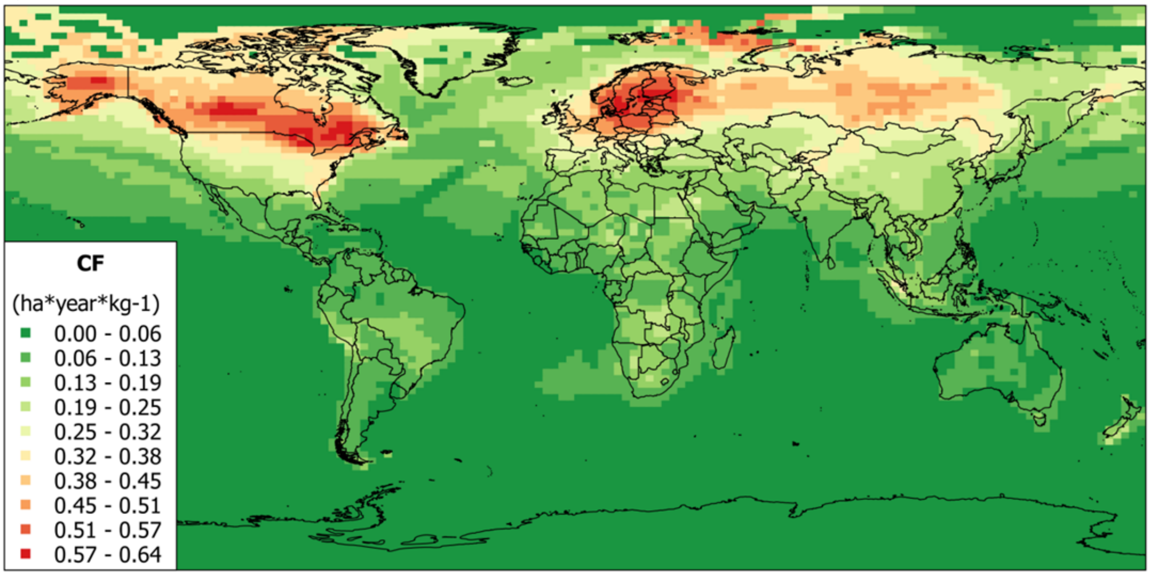
[55] Environmental interferences were calculated using the tentative CFs for terrestrial acidification developed in this study (average of the 45 emission locations) on the emission inventory for EU27 of EMEP (2015). The sector "Combustion in energy and transformation industries (stationary sources)" of the EMEP inventory was assumed to cover electricity production only.

3 Results

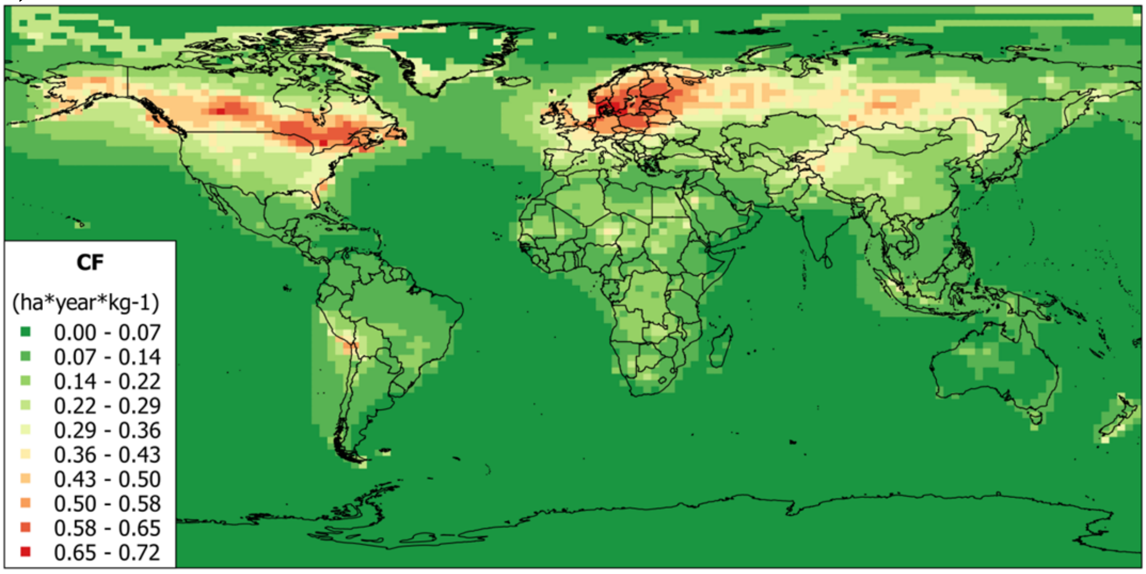
3.1 Carrying capacities and characterisation factors

Estimated carrying capacities (CC) ranged from less than 100 eq.·ha^{-1}·year^{-1} to more than 4000 eq.·ha^{-1}·year^{-1} with a median value around 500 eq.·ha^{-1}·year^{-1}. The global distribution is shown in S5. Numerical CFs for all 13,104 grid cells for NO_X, SO_X and NH_X are available in a spreadsheet in S6, from which they may be exported to LCA software such as GaBi (Thinkstep, 2015) or Simapro (PRé, 2015) and thereby linked to LCI databases such as EcoInvent (2015). CFs for SO_X ranged from less than 0.0054 ha·year·kg^{-1} (10th percentile) to more than 0.41 ha·year·kg^{-1} (90th percentile) with a median value of 0.16 ha·year·kg^{-1} (when excluding CFs for locations in the open sea, which are generally close to 0). In absolute terms the median CF for SO_X can be interpreted as 1 kg SO_X emitted occupying the carrying capacity of 0.048 hectares (corresponding to a square with 22m sides) for 1 year. Figure 3 shows the distribution of CFs for all global locations of NO_X, SO_X and NH_X.

a) NOx



b) SOx



c) NHx

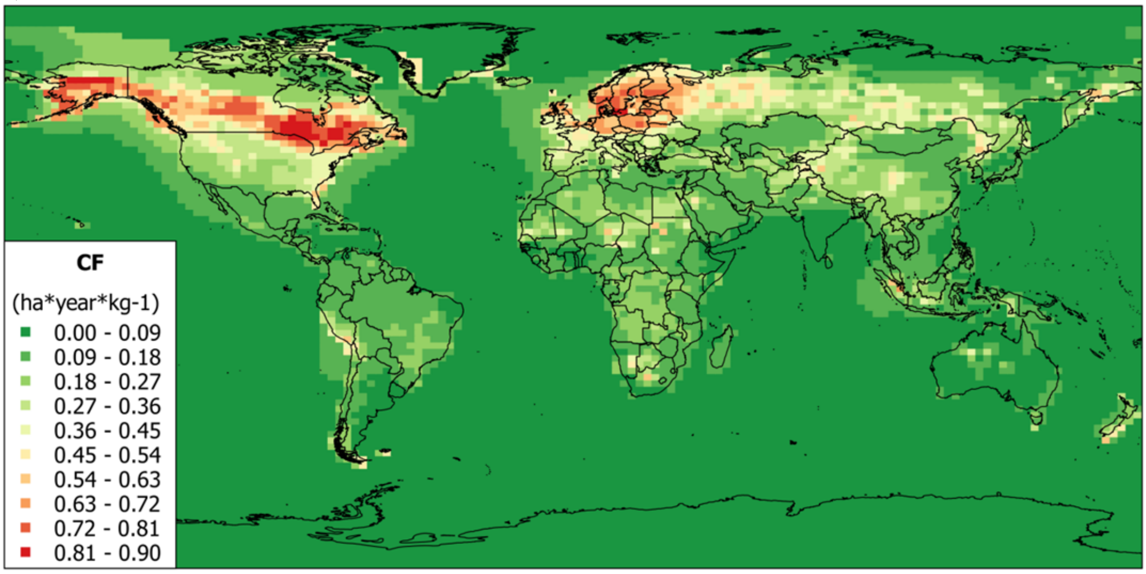


Figure 3: Global distribution of CFs for NO_X (a), SO_X (b) and NH_X (c)

It can be seen that CFs are generally highest in Northern Europe, Canada and Alaska, which is caused by the relatively low carrying capacity of soils in these regions (see S5). The highest CFs for NO_X, SO_X and NH_X corresponds to emission locations in Canada (latitude 55°; longitude -112.5°), Denmark/Sweden (latitude 55°, longitude 12.5°) and Alaska (latitude 65°, longitude -157.5°) respectively. It can also be seen that local differences in CFs (e.g. between neighbouring cells) are lowest for NO_X, higher for SO_X and highest for NH_X. This is because the share of an emission that deposits in or close to the emission cell is largest for

NH_X, smaller for SO_X and smallest for NO_X.[56] In other words, local differences in carrying capacity have a much larger influence on CFs for NH_X than for NO_X. This observation was also made by Huijbregts et al. (2000) for the spatial pattern of European CFs based on the critical loads concept (Spranger et al., 2004).

3.2 Case study

Table 1 shows the input parameters for equation 5 and indicator scores for the 45 emission scenarios.

[56] The deposition patterns vary between emissions cells due to meteorological variations. Yet, a strong tendency of deposition shares close to the emission of NH_X being largest, of SO_X being smaller, and of NO_X being smallest was observed in deposition model of P.-O. Roy et al. (2012). E.g. for an emissions cell in Minnesota 35% of a NH_X emission deposits within the emission cell and 42% within the emission cell and the four neighboring cells, while the corresponding numbers for SO_X are 20% and 26% and for NO_X are 8% and 15% respectively (see also Figure 3).

Table 1: Input parameters for equation 5, indicator scores and comparison to two carrying capacity entitlements for 45 scenarios in the reference year 2010.

State	Plant name	Per capita annual residential electricity consumption (kWh), 2010	Rank	Emissions intensities (kg/MWh), NO_X, 2010	Rank	Emissions intensities (kg/MWh), SO_X, 2010	Rank	CF, NO_X (ha*year/kg)	Rank	CF, SO_X (ha*year/kg)	Rank	Indicator score (ha·year)	Rank
Alabama	Barry	7425	1	0.50	37	1.11	26	0.23	38	0.24	37	2.81	29
Arkansas	White Bluff	6584	8	1.31	18	2.36	22	0.24	36	0.24	34	5.85	19
Arizona	Coronado	5060	23	1.83	16	1.70	24	0.16	44	0.17	44	2.92	28
California	Stockton Cogen	2337	45	0.14	45	0.68	35	0.13	45	0.12	45	0.23	45
Colorado	Rawhide	3587	37	0.73	30	0.35	39	0.31	25	0.36	6	1.28	39
Connecticut	Bridgeport Station	3655	36	0.70	31	0.94	30	0.38	8	0.34	10	2.16	32
Delaware	NRG Energy Center Dover	5295	20	2.32	9	5.24	9	0.35	13	0.31	19	12.87	10
Florida	Big Bend	6489	11	0.48	38	0.96	29	0.34	17	0.44	3	3.85	25
Georgia	Bowen	6338	12	0.28	41	0.30	40	0.33	22	0.32	16	1.20	40
Iowa	Walter Scott Jr Energy Center	4572	29	0.59	34	1.09	27	0.31	26	0.27	26	2.29	31
Idaho	Amalgamated Sugar LLC Nampa	5180	21	3.53	4	11.60	4	0.28	30	0.27	28	21.26	5
Illinois	John Deere Harvester Works	3783	35	3.80	3	20.56	2	0.33	19	0.28	24	26.89	2
Indiana	Sagamore Plant Cogeneration	5402	19	2.58	6	11.00	5	0.30	27	0.25	31	18.87	7
Kansas	Tecumseh Energy Center	5014	24	1.34	17	3.17	16	0.27	32	0.24	36	5.64	20
Kentucky	Ghent	6703	7	0.57	35	0.82	31	0.30	28	0.27	27	2.64	30
Louisiana	Dolet Hills	7190	2	0.91	27	4.10	10	0.20	40	0.21	39	7.56	15
Massachusetts	Salem Harbor	3266	42	0.87	29	4.01	11	0.33	21	0.29	23	4.68	23
Maryland	Morgantown Generating Plant	5002	25	0.24	42	0.67	36	0.33	18	0.31	18	1.43	37
Michigan	Belle River	3511	38	0.99	25	2.74	18	0.40	5	0.34	9	4.72	22
Minnesota	Virginia	4231	33	1.85	14	1.34	25	0.54	1	0.55	1	7.36	16

Missouri	Southwest Power Station	6222	14	0.70	32	2.61	21	0.26	33	0.25	30	5.16	21
Mississippi	Henderson	6793	5	5.81	2	6.43	8	0.24	36	0.24	34	20.11	6
Montana	Lewis & Clark	4591	28	2.16	10	2.71	20	0.39	7	0.32	17	8.08	12
North Carolina	Mayo	6502	10	0.35	39	1.00	28	0.37	12	0.35	8	3.09	26
North Dakota	Antelope Valley	6518	9	1.86	13	2.12	23	0.41	4	0.34	11	9.67	11
Nebraska	Platte	5523	17	1.93	12	3.81	13	0.26	34	0.24	33	7.93	14
New Hampshire	Schiller	3408	40	1.18	24	3.88	12	0.47	2	0.46	2	8.03	13
New Jersey	Chambers Cogeneration LP	3444	39	0.55	36	0.82	32	0.35	13	0.31	19	1.53	36
New Mexico	Four Corners	3270	41	2.53	7	0.72	34	0.19	42	0.19	42	2.05	33
Nevada	TS Power Plant	4295	32	0.20	43	0.19	45	0.20	39	0.20	41	0.33	44
New York	AES Greenidge LLC	2627	44	0.93	26	0.75	33	0.40	6	0.36	5	1.70	35
Ohio	Muskingum River	4522	30	1.21	22	13.36	3	0.37	9	0.33	12	22.91	4
Oklahoma	Hugo	6300	13	0.89	28	2.82	17	0.19	41	0.20	40	4.67	24
Oregon	Boardman	4909	26	1.97	11	3.44	15	0.29	29	0.26	29	7.13	17
Pennsylvania	G F Weaton Power Station	4345	31	1.29	19	2.73	19	0.37	9	0.33	12	5.97	18
South Carolina	US DOE Savannah River Site (D Area)	7085	4	12.90	1	36.24	1	0.35	15	0.35	7	120.97	1
South Dakota	Big Stone	5672	16	3.46	5	3.52	14	0.42	3	0.37	4	15.66	8
Tennessee	Bull Run	7109	3	0.29	40	0.21	43	0.32	23	0.31	21	1.11	41
Texas	Oak Grove	5431	18	0.62	33	0.56	37	0.17	43	0.18	43	1.10	42
Utah	Huntington	3183	43	1.23	21	0.46	38	0.24	35	0.24	32	1.31	38
Virginia	Altavista Power Station	6038	15	1.27	20	0.19	44	0.35	16	0.33	15	3.04	27
Washington	Transalta Centralia Generation	5178	22	1.20	23	0.27	41	0.27	31	0.23	38	1.99	34
Wisconsin	Nelson Dewey	3918	34	2.35	8	10.25	6	0.33	19	0.28	24	14.47	9
West Virginia	Kammer	6711	6	1.85	15	8.55	7	0.37	9	0.33	12	23.48	3
Wyoming	Wygen III	4835	27	0.20	44	0.26	42	0.32	24	0.29	22	0.67	43

3.2.1 Absolute interpretation of results

Indicator scores varied 2 orders of magnitude from a minimum of 0.23 ha·year to a maximum of 121 ha·year for a power plant located in California and South Carolina respectively. This means that the equivalent production of annual residential electricity use in 2010 occupies carrying capacities of between 0.23 ha and 121 ha of land for 1 year depending on the scenario. These areas are abstract because they cannot be empirically observed as special pieces of land somehow dedicated to absorbing acidifying emissions. Instead results should be interpreted as space integrated carrying capacity occupation, which is driven by carrying capacities in grid cells on which large shares of emissions deposit. Note that indicator results hold no information on the extent to which an emission occupy the carrying capacity of the individual grid cells that are affected by its depositions.[57] Table 1 shows that none of the 45 scenarios could be considered environmentally sustainable when using any of the two valuation principles because these require indicator scores to be below 0.049 ha·year (relative contribution to GDP principle) or 0.21 ha·year (grandfathering principle). The scenario in California would, however, only require a slight reduction in indicator score (0.02 ha·year) to be considered environmentally sustainable from the application of the grandfathering perspective. Note that some of the scenarios may be considered environmentally sustainable by the use of other valuation principles than the two used in this study. If, for example, value factors had instead been derived from relative contribution to meeting human needs, a relatively high carrying capacity would perhaps be entitled to residential electricity, since it enables people to meet essential needs, such as heating and cooking (although residential electricity certainly can be used for meeting less essential needs too).

3.2.2 Spatial variations

Since the indicator score is directly proportional to all input parameters (equation 5), results are equally sensitive to variations of all input parameters, i.e. a doubling of any parameter will lead to a doubling of indicator results. From Table 1 it can be seen that the input parameter showing the strongest relative variation in the case study is the emission intensity (factors of almost 200 and 100 difference from smallest to largest for SO_X and NO_X respectively) The cause of this variation is likely differences in flue gas cleaning systems, and for SO_X also differences in the sulfur content of the coal (Henriksson et al., 2014). By contrast the state specific annual per capita residential electricity consumption (P) varies by a

[57] In a hypothetical example where carrying capacities of 4 grid cells of 1ha are each occupied by 10%, 20%, 80% and 130% from depositions of an emission, the aggregated result would be 2.4ha (0.1*1 ha+0.2*1 ha+0.80*1 ha+1.3*1 ha).

factor of 3, while CFs vary by a factor of 5 and 4 for SO_X and NO_X. Variations in P and CF thereby have negligible contributions to the observed 2 orders of magnitude variations in indicator scores of the 45 scenarios. In other words, to achieve a low carrying capacity occupation it is more important to be supplied by a power plant with low emission intensities than for the emissions of the power plant to deposit in areas with high carrying capacity or to reduce residential electricity consumption, although the latter is the only factor that the consumer can easily influence. The power plant located in South Carolina had by far the highest emission intensities of both SO_X and NO_X, which is the reason that the highest indicator score was observed for the scenario in this state (see Table 1). The power plant located in California had the 5th lowest average emissions intensity of the two pollutants. In combination with the lowest CF for both pollutants and the lowest residential electricity consumption this explains why the scenario of California had the lowest indicator score (see Table 1).

With regards to the sensitivity of CFs to input parameters, equation 3 in turn shows that CFs are highest when depositions concentrate around receiving cells with low carrying capacities. This explains why the lowest CFs for both pollutants corresponds to the location of the California power plant for which the majority of depositions happens on grid cell with quite high carrying capacities. On the other hand the highest average CF is for the power plant in Minnesota for which the majority of depositions happens on grid cell with quite low carrying capacities, see Figure 4.

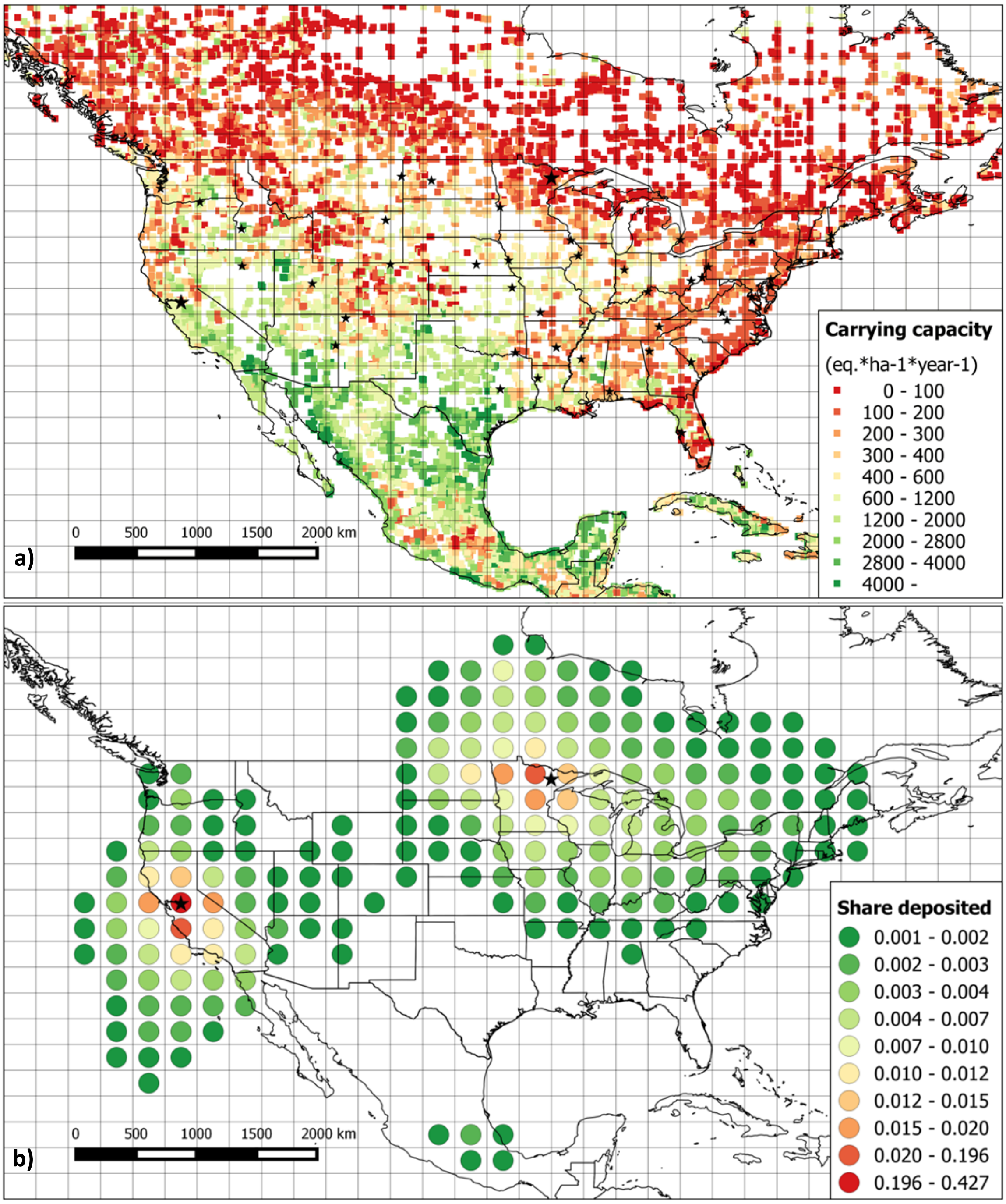
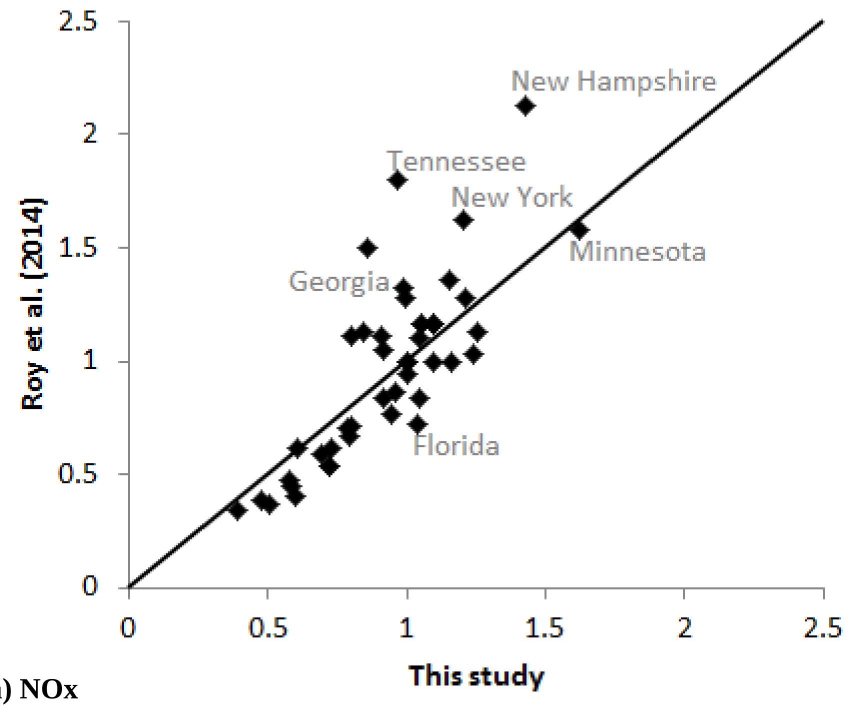


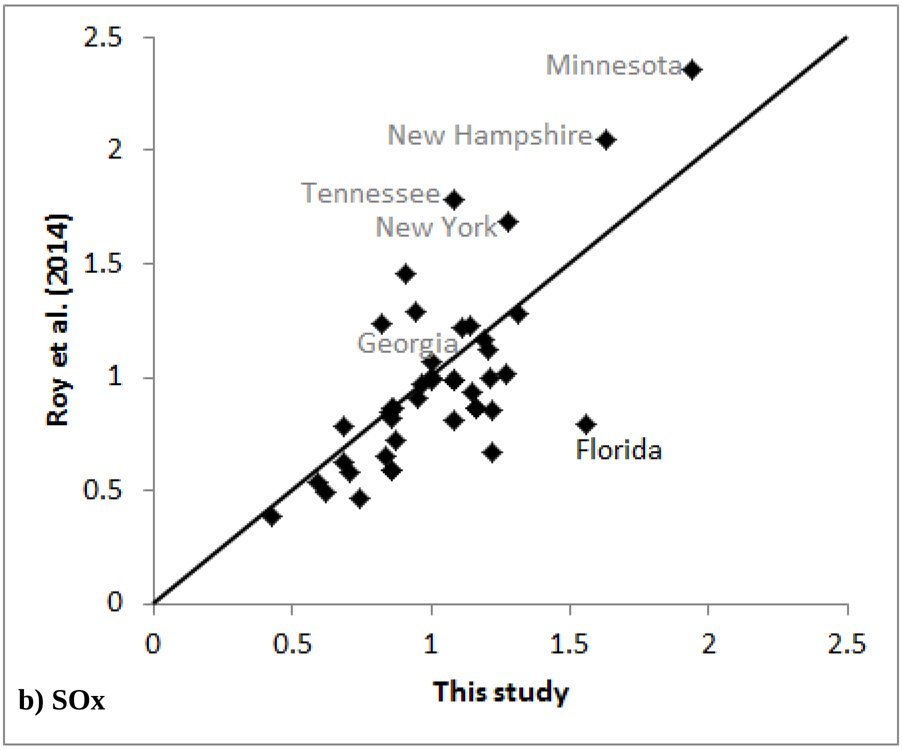
Figure 4: Maps of North America containing a) carrying capacities and power plants (stars), and b) deposition shares on cells receiving more than 0.1% of SO_X emissions from the power plants in California and Minnesota (enlarged stars).

3.2.3 Comparison with alternative CFs

Our CFs express carrying capacity occupation per kg emission and are calculated as acid deposits divided by a pH-based carrying capacity integrated over space (see equation 3). In contrast, the CFs of Roy et al. (2014) express the marginal increase in concentration of H^+-ions in soil solution, compared to modelled existing concentrations, per kg emission. These CFs are calculated as acid deposits multiplied by a so-called soil sensitivity factor which represents the change in existing soil H^+ related to a change in acid deposits integrated over space. Our CFs and the CFs of Roy et al. (2014) use the same fate factors for calculating acid deposits (Roy et al., 2012b) and thus differ only in the use of carrying capacity versus soil sensitivity factor. In Figure 5 we compare the two sets of CFs for the 45 power plant locations. Each set of CF is normalized to the CF of the power plants in Illinois, which ranks approximately in the middle of the 45 CFs for all pollutants and both studies.



a) NOx



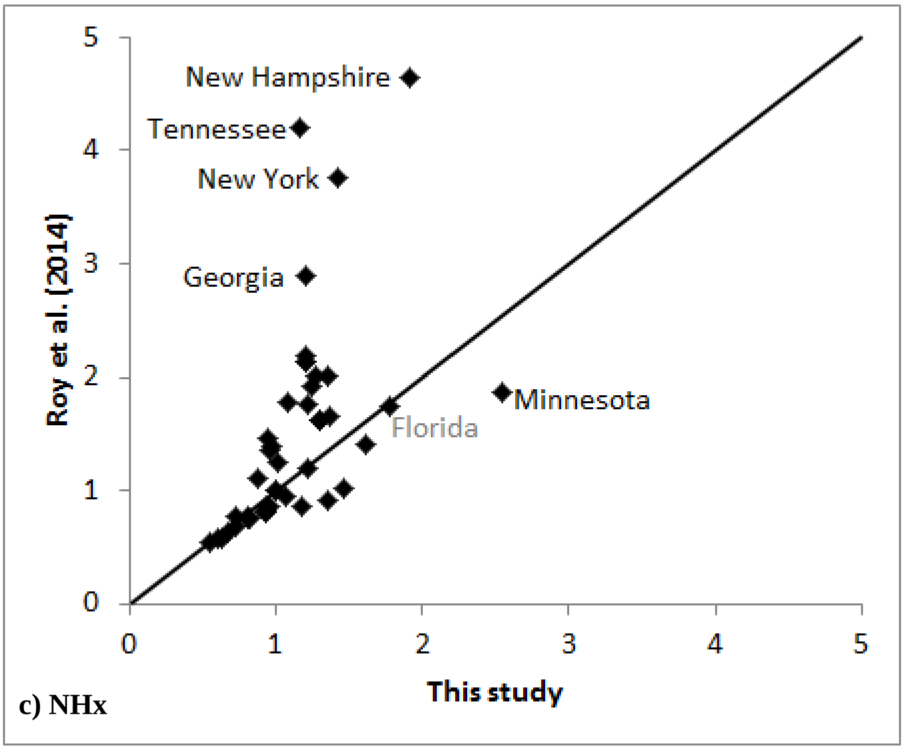


Figure 5: CFs of this study plotted against CF of Roy et al. (2014) for the 45 power plant locations for NO_X, SO_X and NH_X. Each set of CF is normalized to the CF of the power plants in Illinois. State names are written for outliers (in grey across pollutants). CFs above the 1:1 line are relatively higher for Roy et al. (2014) than for this study and vice versa.

It can be seen that there is some agreement between the two sets of CFs for all pollutants, although the agreement appears lower for NH_X than the other pollutants. The partial agreement can be explained from the common fate factors. Difference in agreement amongst the three pollutants can be explained from differences in deposition patterns of pollutants: Due to the relatively large shares of depositions of NH_X close to the emission cell (see footnote 10) fewer grid cells receive large shares of an NH_X emissions than for emissions of SO_X and NO_X. Differences between the relative values of carrying capacities and soil sensitivity factors in individual receiving cells will thus have the largest effect for NH_X CFs. The range of CFs for the 45 power plant locations is for all pollutants larger for Roy et al. (2014) than for this study. This trend, which is strongest for NH_X (Figure 5c), can be explained from the high range of global soil sensitivity factors of 11 orders of magnitude compared to the range of carrying capacities in this study of just 2 orders of magnitude (see S5).

Two types of outliers can be seen on the plots of Figure 5. For the first type CFs in this study are relatively high, while CFs of Roy et al. (2014) are relatively low. This is the case for the CFs of Minnesota for NH_X and CFs of Florida for SO_X. In these cases the high CFs of this study are driven by relatively low carrying capacities in the grid cells receiving large shares of deposition. By comparison corresponding CFs of Roy et al. (2014) are moderate or low for Minnesota and Florida because soil sensitivity factors are moderate or low in the area receiving large shares of deposition. The observed discrepancies between soil sensitivity factors and carrying capacities can be explained from the fact that for some soils a relatively small acid deposition reduces the modelled natural pH by 0.25, while a marginal increase in acid deposition, compared to the modelled existing deposition, leads to a low marginal pH decrease. See Figure S7b for a conceptual pH curve that illustrates this point. This discrepancy between carrying capacity and soil sensitivity factor occur for some soils that have low carrying capacities and for which the background acid deposition is relatively small. This is the case for the parts of the US Midwest and Canada that receive large shares of the depositions from the emission cell of the Minnesota power plant. In these scarcely populated areas modelled background depositions of the three pollutants are 1-2 orders of magnitude lower than those of the most densely populated part of the US East Coast (data not shown).

Outliers of the second type, i.e. low CFs of this study and high CFs of Roy et al. (2014), can be observed in Figure 5c for NH_X for the grid cells of the New Hampshire, New York, Georgia and Tennessee power plants. In these cases the high CFs of Roy et al. (2014) are driven by high soil sensitivity factors in the emission cell and neighboring grid cells. These factors are high because modelled existing depositions are, due to high modelled existing depositions, somewhere in the steep interval of the pH curves of the soils, meaning that marginal increases in deposition can create high reductions in pH in these grid cells. See Figure S7c for a conceptual pH curve. Due to the large variation of soil sensitivity factors (see above), high factors in just a few of the grid cells receiving relatively high shares of an emission can to a very large extent drive CF values of Roy et al. (2014). By comparison the CFs of this study for the grid cells of the New Hampshire and New York power plants are no more than moderate in spite of low to moderate carrying capacities in the vicinity of the emission grid cell, because the power plants are close to the sea, meaning that relatively high shares of emissions deposits on water.

4 Discussion

We have demonstrated the feasibility of modifying LCA indicators to AESI. Thereby we have shown that LCA can potentially solve some of the problems associated with current AESI, such as incomplete coverage of impact categories, varying quality of inventory data, varying or insufficient spatial resolution and the inconvenience to users of needing different software tools for accessing and using AESI. With point of departure in the experiences from the case study, this section discuss differences and complementarities between LCA based RESIs and AESI in decision support (aim 4) and proposes a research agenda for the support of AESI by LCA.

4.1 Decision support related to absolute environmental sustainability

The main characteristic of AESI is that they allow for the assessment of environmental sustainability of systems in absolute terms. This information can be useful on many levels. It may for instance quantitatively inform various emission reduction scenarios designed by e.g. municipalities, nations and supranational organizations with the purpose of achieving environmental sustainability. AESI can thus play similar roles as greenhouse gas emissions reduction scenarios, designed to prevent e.g. a temperature increase of 2°C (IPCC, 2013; Vuuren et al., 2011), that have been adopted at different governmental levels. Also AESI may support individuals motivated to learn what it takes to have an environmentally sustainable life style, i.e. one that is associated with environmental interferences that do not exceed the carrying capacity entitled to an individual person.

4.2 Decision support related to ranking

For a given impact category the ranking of systems or scenarios obtained by an AESI will in principal be identical to the ranking obtained by a RESI (relative environmental sustainability indicator) when the impact pathway model of the

RESI is based on a linear approach (see the introduction section and S1). This is because the relationship between RESI and AESI CFs in such cases will be the same across pollutants and locations. There will therefore be no conflict between RESI based on the linear approach and AESI when used to support decisions where environmental performances of alternative solutions are part of the decision criteria. However, when the impact pathway model of a RESI is based on a marginal approach (see the introduction section and S1)there may be discrepancies in the relationships between AESI and RESI CFs across pollutants and locations, and thus in the ranking of systems or scenarios. This was observed to some extent in the case study when comparing the AESI developed in this study to the marginal based RESI of Roy et al. (2014) (see Figure 5). Thus, if the aim is to oppose reductions in soil solution pH, as quantified by Roy et al. (2014), the optimal solution may be different than the one corresponding to the aim of achieving the lowest possible carrying capacity occupation. Given these discrepancies between AESI and marginal based RESI, which type of indicator should ideally be used to support decisions related to environmental sustainability? The answer, we will argue in the next sub-section, is neither of the two, but both combined.

4.2.1 Risk of sub-optimization

If either marginal based RESIs or AESI are used in isolation there is a risk of sub-optimal decision support. In the case of marginal based RESIs Huijbregts et al. (2011) argued that quantifying marginal changes in environmental interferences can be misleading in cases where changes are small, but existing levels of environmental interferences are unacceptably high. For the impact category terrestrial acidification this may be the case for receiving cells in which existing depositions are so high that the corresponding existing pH is at the lower buffering zone of a pH curve (see Figure S7d and S7e). At this zone additional depositions of hydrogen ions are effectively buffered through reaction with aluminium oxides and hydroxides from clay particles. In such cases RESI based CFs will be low and marginal emission increases will thus seem relatively unproblematic although the state of the soil ecosystems is highly degraded by existing depositions. Another case of sub-optimal decision support is when marginal changes are small and existing levels of environmental interferences are low, i.e. far from exceeding thresholds (see Figure S7a). Although a small marginal increase in existing levels of environmental interferences can here seem unproblematic for environmental sustainability this conclusion is not scalable. The marginal approach thus suffers from a freeriding bias, i.e. only “the drop that spills the cup” is blamed for the crossing of a threshold. This is especially problematic in situations where the combined environmental pressure is increasing, which has for example been the case in large parts of China during the last couple of decades. In such situations CFs based on marginal RESIs will potentially be highly time dependent.

Decisions made only with the aid of AESI can also be suboptimal. For instance they may lead to choices that favour systems whose emissions end up in spatial units with high carrying capacity. Such choices can be suboptimal because they do not consider emissions of existing or future anthropogenic systems that, combined with the additional emissions, risk to exceed carrying capacities in these spatial units. An ideal quantification of entitlement would eliminate this risk of sub-optimization because it would take into account existing and potential competing systems, but the risk is quite real considering the difficulties of carrying out an ideal quantification of entitlement (see Section 2.4).

4.2.2 Combining marginal based RESI and AESI to avoid sub-optimization

The differences between the AESI and marginal based RESI are not only technical, but in fact also ethical: The CFs for terrestrial acidification developed in this study are compatible with decision making grounded in rule based ethics according to which a decision is considered “good” if it follows one or more prescribed rules that may be either universal or situation-dependent (Ekvall et al., 2005). In AESI the rule is that a decision should, whenever possible, lead to anthropogenic systems that do not occupy more carrying capacity than they can be considered entitled to. If this is not possible within the decision space, the rule is that a decision should lead to the lowest possible carrying capacity occupation amongst alternatives. Thus if all societal decisions were to follow these rules a transition towards environmental sustainability would in principle happen.[58] In contrast, the decision-making that the marginal RESI of Roy et al. (2014) supports is grounded in consequential ethics, according to which a decision is “good” if its consequences are better than those of alternative(s) (Ekvall et al., 2005). The rule and consequential based ethics are conflicting in cases where following the prescribed rule(s) does not lead to the best consequences and vice versa.[59]

In real life, decisions are unlikely to be based entirely on either rule or consequential ethics, because decisions are often taken in consensus processes and because individuals rarely 100% adhere to a specific ethical mindset (Hofstetter,

[58] Note that the only way to guarantee that total carrying capacity is not exceeded by the combined environmental interferences of all anthropogenic systems is to (somewhat oxymoronically) ensure that the same valuation principle is used to calculate carrying capacity entitlement of all systems.

[59] Consider the hypothetical situation where a person has the option of saving 5 lives by taking 1 (innocent) life. Doing this would lead to the best consequence, compared to inaction, but would also violate the rule of not killing an (innocent) person (Thomson, 1976).

1998). Therefore the different ethical perspectives of marginal based RESI and AESI can be seen as complementary rather than competing. In the case study, our AESI was used to evaluate the sustainability of the 45 scenarios absolutely and to point to the scenario associated with the lowest carrying capacity occupation. The RESI oriented CFs of Roy et al. (2014) could on the other hand point to the scenario associated with the lowest marginal increase in environmental interferences. Both types of information are valuable in decision processes, which aim to simultaneously reduce existing levels of environmental interferences efficiently and maintain, or take steps towards achieving, environmental sustainability of society as a whole and of its individual anthropogenic systems.

4.3 Research agenda on AESI in a life cycle perspective

This study is intended primarily as a proof of concept and its theme must be expanded upon in future research for the proposed modification of LCA to measure environmental sustainability in absolute terms to be useful in decision support. Below we outline a few key challenges that deserve academic attention.

The designs of AESI are associated with several choices, to which indicator scores may show different degrees of sensitivities. In our modification of the LCA indicator for terrestrial acidification to AESI the choices of control variable, threshold value and the use of PROFILE to translate the threshold into carrying capacities all have potentially high contribution to uncertainty in indicator scores and efforts to reduce this uncertainty should be made (see S9 for an elaboration). Similar choices are unavoidable in any AESI. It is therefore important for indicator designers to 1) be aware of these choices and communicate them explicitly to users, so they can be considered in the decision support along with the indicator scores, 2) to quantify the sensitivity of indicator scores to changes in choices, and 3) to use these quantifications to effectively reduce overall uncertainties in indicator scores. As most choices are, at least partially, related to value judgement, consensus processes involving e.g. environmental scientists, indicator designers and indicator users may be feasible for reducing overall uncertainties.

Uncertainties in LCIs also deserve attention when using AESI. Because many current societies cannot be considered environmentally sustainable a key use of AESI is to support transitions towards environmentally sustainable societies. Such transitions per definition involve large changes in technologies. For example, environmental interferences from energy use are expected to change considerably in many countries over the next decades. As a result, environmental interferences of many product systems will also change in the future. It is therefore important to carefully evaluate, and if necessary modify, existing LCI unit processes in absolute environmental sustainability assessments, which aims to capture the effects of future technological transformations (Miller and Keoleian, 2015).

A core characteristic of LCA is that it covers a comprehensive set of impact categories. In this context a relevant question is how to aggregate AESI scores from different impact categories. One option is to simply add the scores since they can be expressed in the same metric (ha·year) for all impact categories. However, a weighting step may be required as the consequences of exceeding carrying capacities can vary in severity between impacts categories. Some factors influencing the severity of exceedance are the social and/or economic consequences, the spatial extent and the time required for reversion of damage. In addition, care should be taken when attempting to aggregate indicator scores across impact categories, since the interaction between different types of environmental interferences within a specific territory is complex and not well understood. For some combinations of impact categories additivity between carrying capacity occupations may be a good assumption. In other cases, however, a territory that has its carrying capacity 100% occupied for one impact category may have unoccupied carrying capacity for other impact categories[60], which means that simply adding indicator scores across impact categories would overestimate the actual area equivalent of carrying capacity occupation. Another challenge related to aggregating indicator scores is the need for absolute sustainability references for the LCA impact categories that are not related to ecosystems, i.e. those related to human health impacts and depletion of non-renewable resources. Carrying capacity does per definition not apply to such impact categories, but other more normative sustainability references may be quantified (McElroy et al., 2008).

Another key challenge is how to integrate a carrying capacity entitlement module in LCA software that is relevant and requires only a manageable data input by the software user. Ideally the user should only have to choose a valuation principle and define the duration of environmental interventions (t) of each emission location. The software would then calculate $T_{affected}$ and $A_{affected}$, identify competing systems and subsequently calculate VF to arrive at the carrying capacity entitlement (see equation 4) for each emission location and compare this to the corre-

[60] This situation will for example occur when carrying capacities are derived from a threshold of affected species and when the species that are most sensitive to one type of environmental interferences (e.g. acidification) are different than the species that are most sensitive to another type (e.g. chemicals with eco-toxicity potentials).

sponding indicator score. This would require the software to be equipped with a fate model, calculating $T_{affected}$ and $A_{affected}$ for each emission location, and to be linked to a complete spatially derived emission inventory that contains information needed to calculate VF, such as contribution to GDP, for each of its anthropogenic systems. For many emissions in a typical product life cycle location and duration (t) will be partly or completely unknown. The AESI should therefore be equipped with a meaningful default choice for location and duration that is compatible with the calculation of carrying capacity entitlement.

Supporting Information

Supporting information is available online and contains methodological details and elaboration of results and discussions.

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Supporting information for:

Modifying life cycle assessment to measure absolute environmental sustainability

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1 Linear and marginal approaches in LCA indicators

Figure S1.1 shows the two different approaches to calculating small changes in environmental interference from small changes in emissions and resource use.

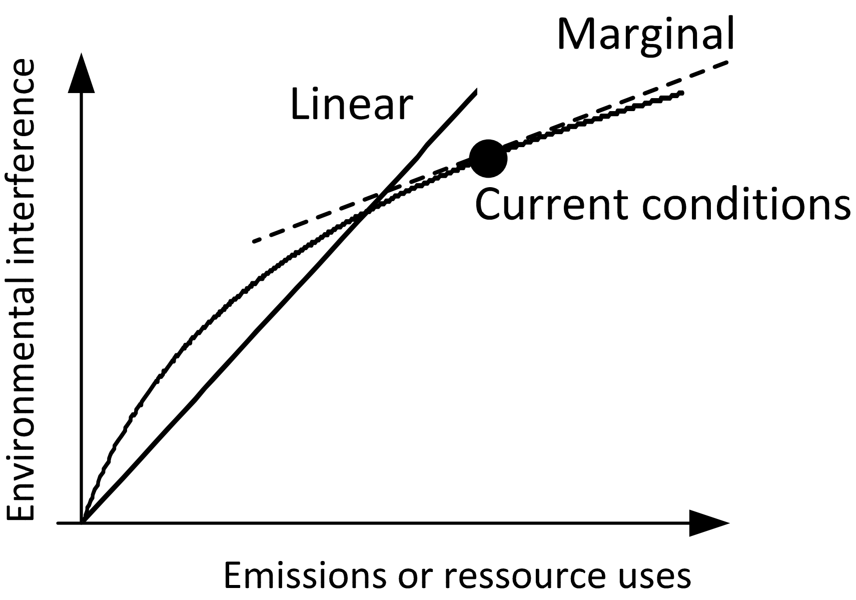
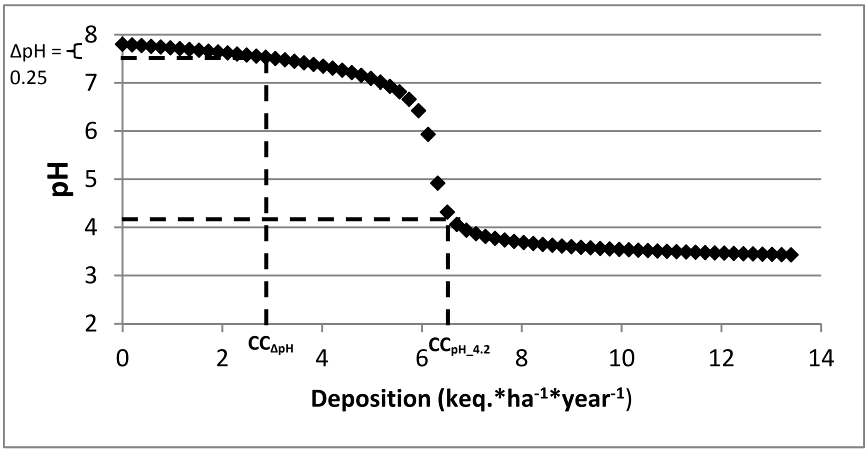


Figure S1.1: Linear and marginal approach in LCA indicators for a cause-effect curve.

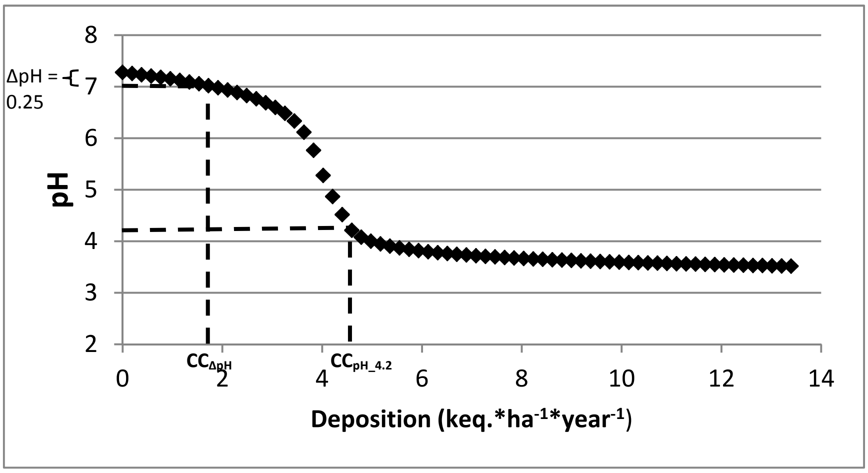
2 pH thresholds

To determine a pH threshold the pH for 70 random grid cells was simulated using PROFILE in a sequence of 71 steps. In the first step only grid specific natural depositions, from e.g. lightning, eruptive and non-eruptive volcanoes, were modelled based on Tegen & Fung (1994) and Bey et al. (2009). In the subsequent 70 steps the average background deposition of SO_X (approx. 0.1 keq/ha/year) was increased by a factor of 5 for each step so that the average background deposition of SO_X increase was by a factor 350 at the final step 70.

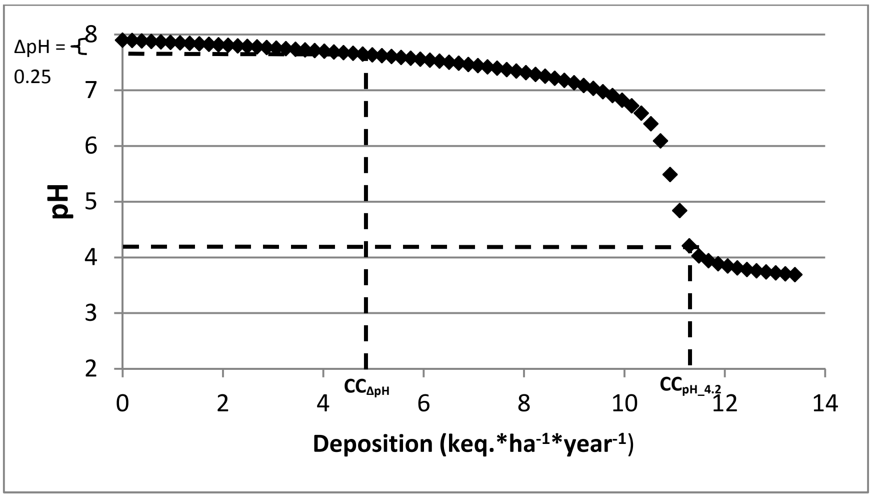
Figure S2.1 shows the simulated pH variations for three representative receiving grid cells according to an increase of deposition above the natural deposition. Depositions corresponding to a pH decrease of 0.25 and an absolute minimum pH of 4 are indicated.



a) grid cell #129



b) grid cell #999



c) grid cell #6659

Figure S2.1: pH variations of receiving environment grid cell according to an increase of deposition of SOX above natural emissions for three representative receiving grid cells. Carrying capacities (CC) corresponding to a pH decrease of 0.25 and an absolute minimum pH of 4 are indicated.

3 Design of deposition steps

Carrying capacity (eq.*ha^{-1}*$year^{-1}$) was calculated for the 70 random grid cells presented in S2 based on the 71 deposition steps. 1 eq refers to 1 mol H^+-eq. From this the distribution presented in Figure S3.1 was obtained.

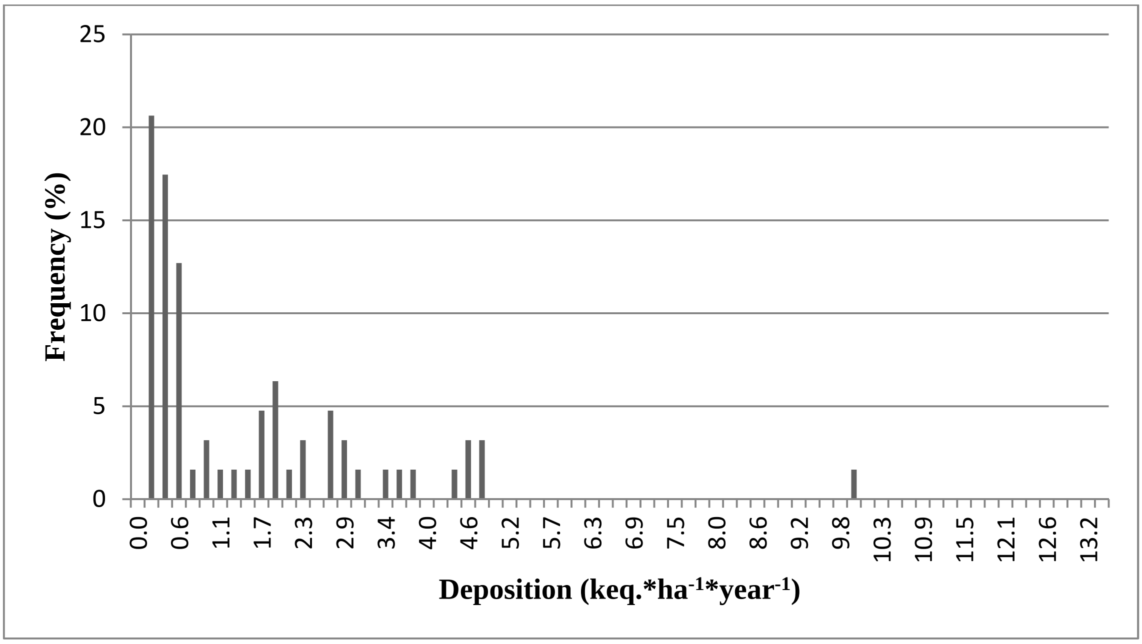


Figure S3.1: Threshold distribution for the 71 steps increasing deposition above natural emissions by a factor of 0 to 350 . 1 keq refers to 1000 mol H+-eq.

It appear that the distribution of carrying capacities in all grid cells may be best be described by a log normal distribution, since the highest frequency of carrying capacities are just above 0 and a long tail in the distribution can be observed as depositions are increased. In designing the deposition steps we aimed for a uniform distribution of grid cell carrying capacities, in other words ≈10% falling into each interval. We did not carry out more than 9 steps due to the computational capacity required to model pH for 99,515 cells in each deposition step. This lead to the carrying capacity intervals and values used in CF calculations shown in table S3.1:

Table S3.1: Deposition intervals

Step	Carrying capacity interval	Carrying capacity used for CF calculations
#	$eq*ha^{-1}*year^{-1}$	$eq*ha^{-1}*year^{-1}$
1	<100	50
2	100-200	150
3	200-300	250
4	300-400	350
5	400-600	500
6	600-1200	900
7	1200-2000	1600
8	2000-2800	2400
9	2800-4000	3400
NA	>4000	5000

4 Kringing function

The function is presented in a Matlab script below

```
%%% Prepare an excel sheet with latitude and longitude coordinates in column 1
and 2, and CC_min, CC_max and CC_default in column 3, 4 and 5

%%% Flag the non-convergence error by the number 1E8

%%% Load the excel sheet

file=xlsread('pathname',1);

%%% Identifies erroneous cells

X=find(file(:,3)==1E8);

Y=find(file(:,4)==1E8);

Z=find(file(:,5)==1E8);

it=1;

while it<=size(X,1)

   %%% identify the areas that are closest to the ones that you need to correct

   U=find(file(:,1)>file(X(it),1)-0.5 & file(:,1)<file(X(it),1)+0.5 &...

      file(:,2)>file(X(it),2)-0.5 & file(:,2)<file(X(it),2)+0.5);


   eval(it)=size(U,1);

   ver=find(file(U,5)<1E8);

   verif(it)=size(ver,1);


   comp=1;

   while verif(it)<1


      U=find(file(:,1)>file(X(it),1)-comp & file(:,1)<file(X(it),1)+comp &...
```

```
        file(:,2)>file(X(it),2)-comp & file(:,2)<file(X(it),2)+comp);


      ver=find(file(U,5)<1E8);

      verif(it)=size(ver,1);

      comp=comp+1;

   end


  garde1=file(U,3); % TMin

  garde2=file(U,4); % Tmax

  garde3=file(U,5); % Tmoyen

  p=find(garde1<1E8);

  q=find(garde2<1E8);

  r=find(garde3<1E8);


  %%% calculate the median without the cells without the ones which are erro-
neous

  file(X(it),3)=median(garde1(q));

  file(Y(it),4)=median(garde2(q));

  file(Z(it),5)=median(garde3(q));

   it=it+1

end

ok=zeros(99515,1);

ok(X)=1;


final=[file,ok];
```

5 Additional results

Figure S5.1 shows the global distribution of carrying capacity. By comparison the soil sensitivity factors of Roy & Desche (2012) for NO_X, SO_X and NH_X are shown in Figures S5.2-S5.4.

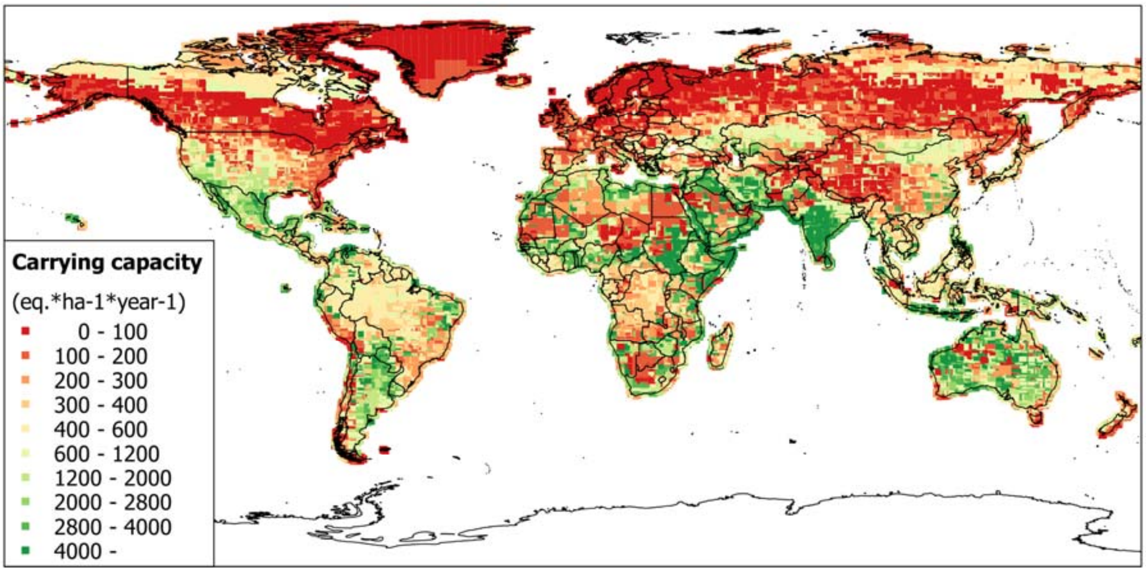


Figure S5.1: Carrying capacity.

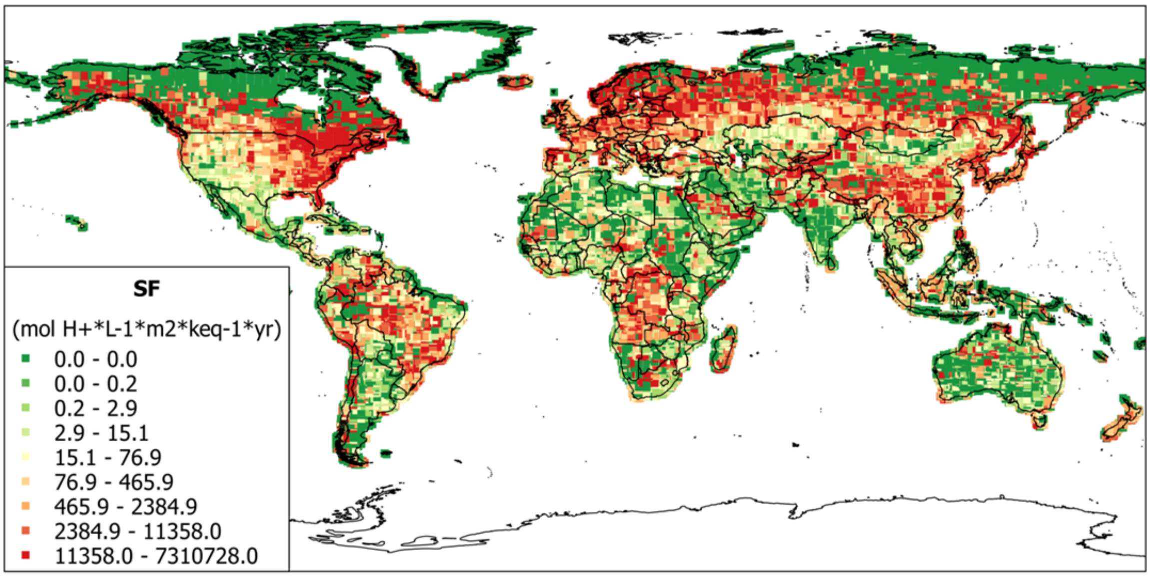


Figure S5.2: Soil sensitivity factors of Roy & Desche (2012) for NOX.

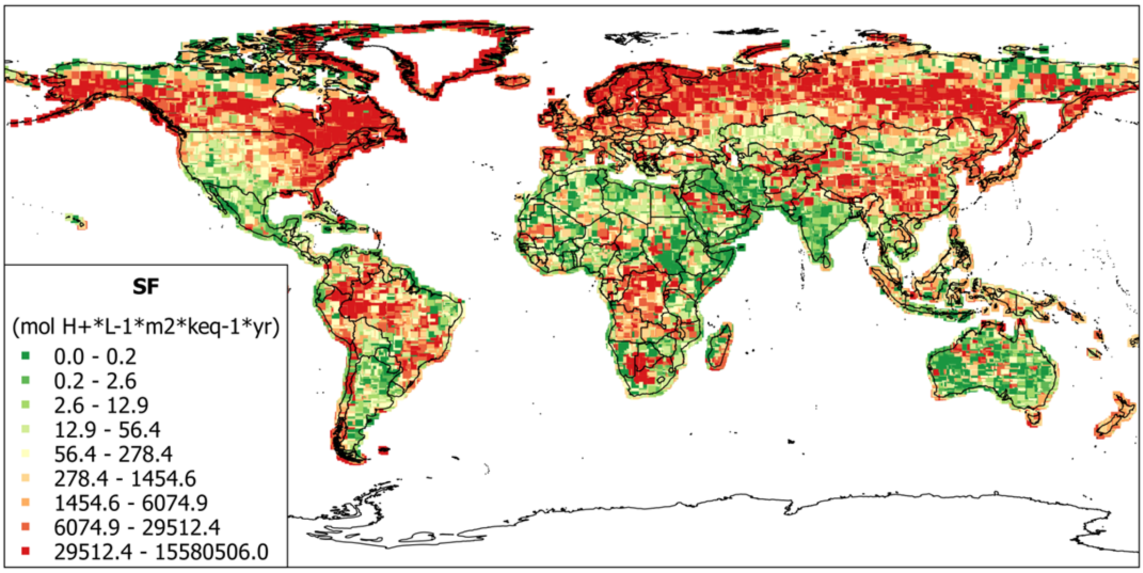


Figure S5.3: Soil sensitivity factors of Roy & Desche (2012) for SOX.

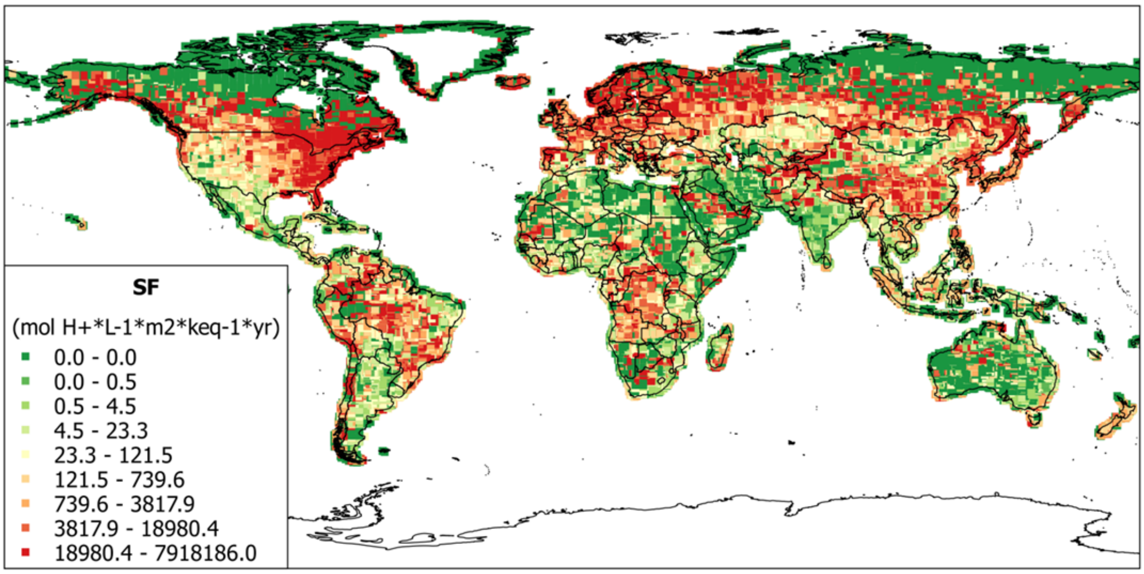


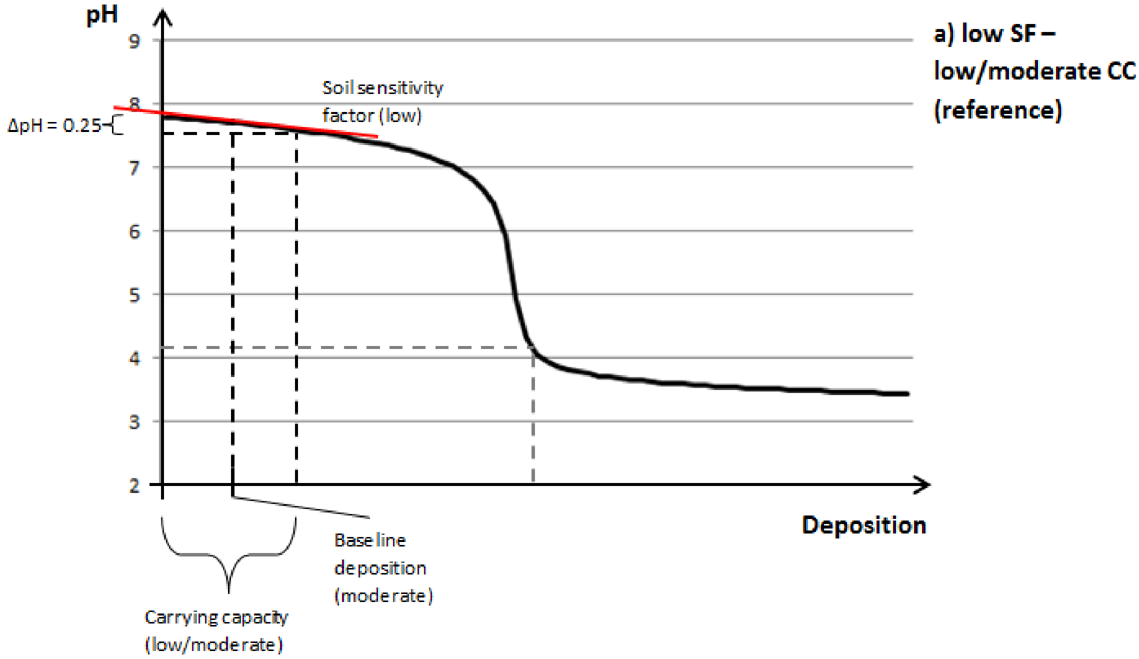
Figure S5.4: Soil sensitivity factors of Roy & Desche (2012) for NHX.

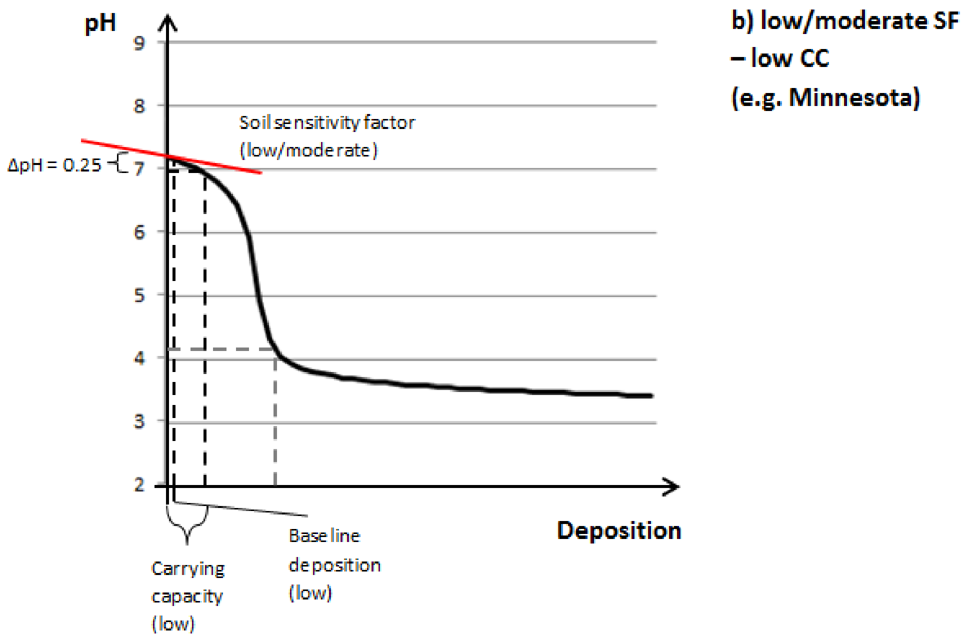
6 Characterisation factors

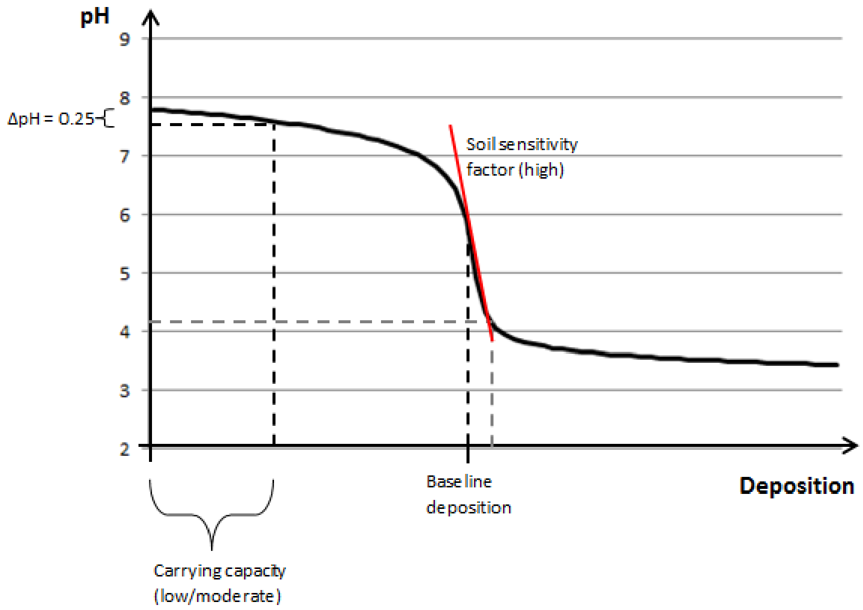
See Excel sheet for CFs for SO_X, NO_X and NH_X. The GIS coordinates correspond to the lower left corner of grid cells.

7 Conceptual pH curves

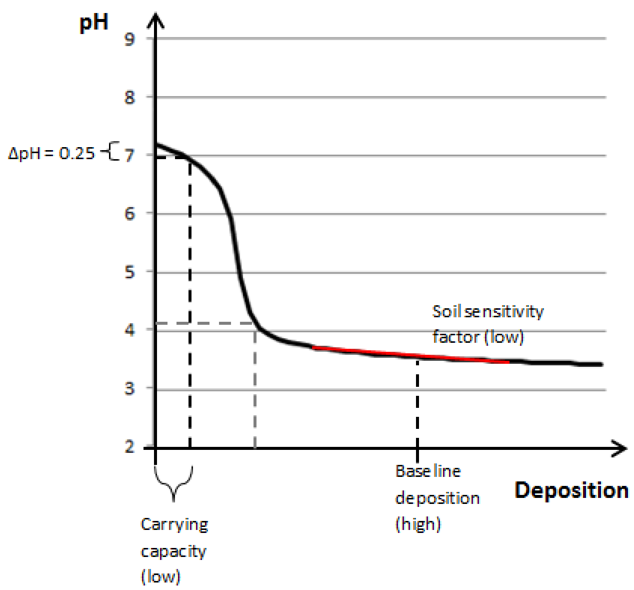
Figure S7 shows conceptual pH curves related to the derivation of soil sensitivity factors and carrying capacities for 5 cases, which varies with respect to natural pH (manmade deposition = 0) and level of modelled existing deposition.







c) high SF – low/moderate CC (e.g. New Hampshire)



d) low SF – low CC

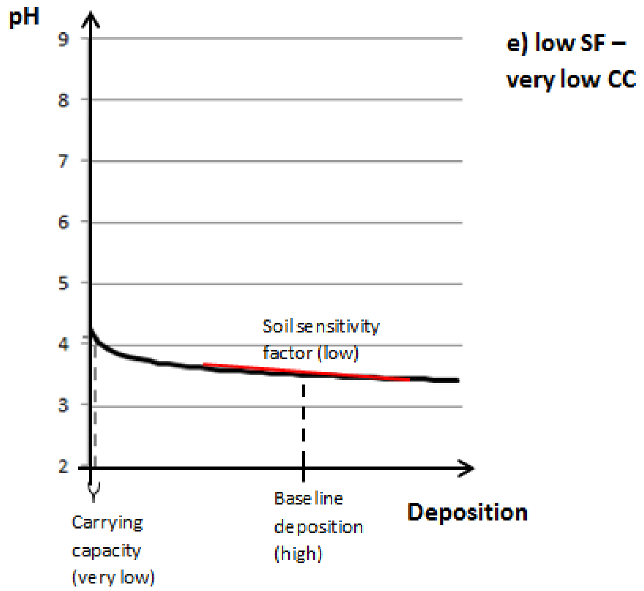


Figure S7. Response in pH to deposition for 5 cases combining values of natural pH and baseline depositions. Soil sensitivity factors (SF) and carrying capacities (CC) are categorized accordingly.

8 Key choices in the AESI for terrestrial acidification

In our modification of the indicator of Roy et al. (2014) we chose two complementary threshold values based on the two points of the pH curve where the carbonate buffering system starts weakening and where the mobilisation of aluminium starts to occur. As environmental sustainability references other pH related threshold values could be applied, for example by taking the pH sensitivity of vegetation into account, as proposed in the critical loads concept (Spranger et al. 2004). We could also have applied a control variable more directly related to the sensitivities of ecosystems, such as "potentially disappeared fraction of species" (PDF), which is a common damage indicator in LCA. In this case a corresponding threshold value of a sustainable minimum level of species diversity should be chosen. The change in indicator score from changing choices of control variable and threshold value is important to quantify in the effort of managing and reducing overall uncertainties in indicator scores.

We furthermore calculated a substance generic carrying capacity from simulation of pH responses to increasing depositions of SO_X. However depositions of similar quantities of H^+ equivalents can cause different responses in pH for nitrogen containing pollutants (NO_X and NH_X) than for SO_X due to the effect of nitrogen uptake processes in vegetation across soils. To reduce the uncertainty introduced by calculating substance generic carrying capacity, simulations of pH response to stepwise increasing depositions of NO_X and NH_X should be carried out in the same manner as they were done for SO_X here.

Thirdly, due to the approach of determining carrying capacities from simulated pH responses to stepwise increases of deposition, the range of carrying capacity values was in fact determined by the carrying capacity values assigned to grid cells for which threshold were crossed at the first deposition step and grid cells for which thresholds were not crossed at deposition step 9. In this study the former was assigned a value of 50 eq*ha^{-1}*year^{-1} (the middle of the 0-100 eq*ha^{-1}*year^{-1} interval in which the actual carrying capacity lies according to PROFILE) and the latter an arbitrary value of 5000 eq*ha^{-1}*year^{-1} (the deposition at step 9 was 4000 eq*ha^{-1}*year^{-1}). The sensitivity of CFs to the assignment of minimum and maximum carrying capacities could be easily tested. If large uncertainties should be reduced by obtaining more realistic minimum and maximum carrying capacity values from additional simulations in PROFILE.

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V

Environmentally sustainable or not? Managing and reducing indicator uncertainties

Bjørn, A., Richardson K., & Hauschild, M. Z.

To be resubmitted to *Ecological Indicators*.

Environmentally sustainable or not? Managing and reducing indicator uncertainties

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Abstract

There is a growing interest in ecological indicators that compare measures of environmental interferences to safe limits due to their potential use in sustainability assessments. Such environmental sustainability indicators evaluate whether the environmental interference of a studied anthropogenic system is environmentally sustainable or the opposite. It is important for the evaluation that indicator scores are relatively certain. The purpose of this study is to develop guidance for how to manage and reduce potentially important sources of uncertainties in indicator scores.

We use the Baltic Sea region as a case study to adapt and analyse five recently developed indicator sets that compare environmental interferences with the nitrogen and phosphorous cycles to safe limits in aquatic environments. Twelve universal concerns are identified, for which choices need to be made in the design of environmental sustainability indicators. These choices influence indicator scores and each concern is therefore a potential source of uncertainty in indicator scores. Indicator scores are calculated for all practically possible combinations of choices made for each concern in each of the five adapted indicator sets.

Concerns such as “Control variable”, “Modelling of safe limit” and “Spatial coverage and resolution” are largely related to scientific understanding, while concerns such as “Ecosystem of focus”, “Ecological boundary value” and “Aggregation of indicator scores” depend on value judgements. Indicator scores related to phosphorous range 3 orders of magnitude between and 2 orders of magnitude within a given indicator. This means that environmental interferences are evaluated to be anywhere from a factor 100 below to a factor 10 above a safe limit across indicators, depending on the combination of choices made for the twelve concerns. For indicators related to nitrogen, scores range by factors of 59 and 21 between and within indicators. Variations in choices are found to be potentially important sources of uncertainty in indicator scores for all twelve identified concerns.

In the effort to reduce uncertainties in environmental indicators in general, scientific choices that are not in accordance with the state-of-the-art should be excluded and where needed, more research should be devoted to improving the scientific state-of-the-art for these concerns. For concerns related to value judgement, the range of choices can be reduced by adhering to established societal norms. It is neither practically nor theoretically possible to eliminate all uncertainties. Indicators should therefore be transparently designed so choices made and the effects of these are visible to users and decision makers.

Keywords:
Indicator comparison; eutrophication; uncertainty management

1. Introduction

The planetary boundaries concept (Rockström et al., 2009; Steffen et al., 2015) and various footprinting methods (Borucke et al., 2013; Galli et al., 2012) have successfully communicated the importance of developing and applying ecological indicators that compare measures of environmental impact, caused by anthropogenic resource use or emissions, to "safe limits" at various scales. Such indicators can be used in sustainability assessments when the non-exceedance of safe limits is considered to be a precondition for environmental sustainability. This precondition can be inferred from the often cited environmental sustainability definition offered by Goodland (1995): "…seek[ing] to improve human welfare by protecting the sources of raw materials used for human needs and ensuring that the sinks for human wastes are not exceeded, in order to prevent harm to humans". Throughout this study, we use the term "environmental interference" for anthropogenic changes to any point in an impact pathway and refer to indicators that that compare environmental interferences to safe limits as "environmental sustainability indicators". Such indicators could constitute a central quantitative element in operationalising any forthcoming Sustainable Development Goals (SDGs). In fact, if environmental sustainability indicators are not adopted, there is a danger that the social SDGs (i.e. ending poverty) will never be achieved due to the social consequences of environmental degradation resulting from exceeding various safe limits (Griggs et al., 2013).[61]

As for all types of indicators, environmental sustainability indicators do not provide objective insights into reality in all of its complexity. The representations of reality that they provide are influenced by how a number of concerns are dealt with in the design of the indicator. In Table 1 we have identified 12 concerns for which a choice needs to be made in the design of environmental sustainability indicators for any type of environmental stressors (resource uses or emissions). We consider the 12 concerns to be mutually exclusive and collectively exhaustive.

[61] Consider the high susceptibility of people in poverty stricken regions to the effects of climate change and water depletion.

Table 1: Presentation and classification of 12 concerns in the design of environmental sustainability indicators

Concern	Explanation	Classification
1. Ecosystem of focus	As a stressor can affect several ecosystems, an ecosystem of focus needs to be chosen.	Value judgement
2. Goal	Ecosystems are complex and can deliver many types of services (MEA, 2005). A goal specifying the ecosystem characteristics that should be protected as a condition for environmental sustainability must be formulated.	Value judgement
3. Control variable	To measure the degree to which the goal is met a relevant control variable must be chosen.	Scientific understanding
4. Basis for the ecological boundary	An ecological boundary is a value of the control variable that demarcates whether or not the goal is met. Ecological boundaries can be established using different approaches (Dearing et al., 2014) and a choice of approach must therefore be made.	Scientific understanding/value judgement
5. Ecological boundary value	A single value must be chosen from the range of numerical ecological boundary values established.	Value judgement
6. Location of safe limit in the impact pathway	To facilitate the comparison of ecological boundaries to measures of environmental interferences ecological boundaries are translated to metrics of safe limits. Safe limits are generally expressed at an earlier point in the impact pathway than the ecological boundaries. This point must match the point, or one of the points, for which environmental interferences of the assessed system is expressed.	Assumed user preference
7. Modelling of safe limit	In the translation of ecological boundaries to safe limits an impact pathway model is needed. A choice needs to be made between available models which can vary in structure and parameters.	Scientific understanding
8. Quantifying environmental interferences	An approach related to monitoring, modelling or a combination of the two must be chosen to quantify environmental interferences of the anthopogenic system assessed by the indicator.	Scientific understanding
9. Spatial coverage and resolution	Spatial variations in impact pathway and safe limits within chosen geographical boundaries can be captured to a larger or smaller extent depending on the choice of spatial resolution.	Scientific understanding/ assumed user preference
10. Temporal coverage and resolution	Some environmental interferences vary within the chosen time frame and, due to natural dynamics, so do ecological boundaries, impact pathways and hence safe limits. These variations within the considered time frame can be captured to a larger or smaller extent depending on the choice of temporal resolution.	Scientific understanding/ assumed user preference
11. Aggregation of indicator scores	A choice must be made as to how to aggregate estimated degrees of safe limit exceedance across spatial and temporal units to a single indicator score for the entire ecosystem within the geographical boundary.	Value judgement
12. Basis for allocation	When different anthropogenic systems are causing environmental interferences within the geographical boundary a choice must be made on how to allocate "environmental interference entitlements" within safe limits between these systems to judge the environmental sustainability of each of them.	Value judgement

Indicator scores will show potentially high sensitivities to variations in choices for these concerns. This sensitivity is important to understand as it can potentially inhibit the usability of environmental sustainability indicators in decision making if, for example, scores of two alternative indicators disagree on whether a specific anthropogenic system is sustainable (i.e. safe limit exceeded by environmental interferences or not). In studies presenting environmental sustainability indicators, sensitivity analyses are commonly carried out, but all combinations of choices for all 12 concerns identified in Table 1 are apparently never systematically considered. In addition, the influences of the 12 concerns identified in Table 1 on variations in indicator scores between different indicators used to evaluate the same environmental stressors have, to our knowledge, never been systematically studied. The fragmented attention given to sensitivities of environmental sustainability indicators means that the uncertainties of indicator scores are not fully known or understood. The current study was initiated in light of the increasing interest in environmental sustainability indicators. Specifically, we saw that there is a need for systematic sensitivity studies within and between indicators in order to provide guidance on how to systematically define indicators and effectively reduce uncertainties in indicator scores. In this study, we also attempt to identify causes of uncertainty that cannot be eliminated practically or theoretically and to provide guidance on how to manage these sources of uncertainty.

As study objects we chose five recently proposed environmental sustainability indicator sets that relate environmental interferences of the nitrogen (N) and phosphorous (P) cycles to limits identified as being “safe”. The 5 indicators were adapted by varying choices for the 12 concerns in the assessment of recent environmental interferences in the Baltic Sea region. This case was chosen because the Baltic Sea has suffered from widespread eutrophication for decades despite the existence of policies targeted to improve the situation. This means that a rich body of monitoring and modelling data exists from which realistic environmental interferences can be quantified and indicator scores compared. We restricted the analysis to environmental interferences from P and N in aquatic ecosystems but note that environmental interferences related to effects of ammonia and nitrogen oxides on terrestrial biodiversity, of nitrate on groundwater quality and of nitrous oxide on climate change also can be evaluated using environmental sustainability indicators, as proposed by de Vries et al. (2013). Due to the focus on aquatic ecosystems, we use the more specific term “assimilative capacity” instead of “safe limit” in the analysis of the 5 adapted indicators. Adopting the definition of carrying capacity presented by Bjørn & Hauschild (2015), we define assimilative capacity for N and P as “the maximum sustained environmental interference in natural cycles an aquatic ecosystem can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert". Two consecutive research questions guide our interpretation of the sensitivity analysis: 1) Which of the 12 concerns identified in Table 1 contribute most to uncertainty in scores of the 5 compared environmental sustainability indicators? 2) Building on these findings - what general guidance can be given on how to reduce and manage uncertainties of environmental sustainability indicators?

2. Methods

2.1. Case study

The Baltic Sea is a brackish water body with a surface area of 417,600km^2 encompassed by the Scandinavian Peninsula and the mainland of northern Europe. It is bordered by Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Russia and Sweden (termed contracting parties in the HELCOM treaty for monitoring and managing the pollution of the Baltic Sea) and further includes parts of Belarus, Czech Republic, Norway, Slovakia and Ukraine in its catchment (termed non-contracting parties) (HELCOM, 2013a). Eutrophication has been a widespread problem in the Baltic Sea in recent decades following a sharp increase in inputs of N and P from the 1950s, mainly caused by an increase in riverine inputs, which are tightly connected to run-off from agriculture (HELCOM, 2013a). The case study was designed to include the entire Baltic Sea region, defined as the Baltic Sea including its catchment area, which covers 1,720,270km^2 (HELCOM, 2013a). For the quantification of total environmental interferences, the reference year used was 2012, or the most recent year where data is available (see 3.1.8). The population of the Baltic Sea region was 85.64 million in 2006. Arable land, a significant source of N and P input to the Baltic Sea, covers approximately 20% of the catchment area. This is a relatively high percentage compared to other regions of the world (HELCOM, 2011).

2.2. Covered indicators and adaptation to case study

The Baltic Sea region case study was used to apply and analyse indicators presented in five previous studies: 1) Those of the original planetary boundaries for the N and P cycle of Rockström et al. (2009), 2) the proposal of a modified planetary boundary for P of Carpenter & Bennett (2011), 3) the proposal of a modified planetary boundary for N of de Vries et al. (2013), 4) the "grey water footprint" for N and P of Liu et al. (2012) and 5) the most recent indicator set for N and P inputs to the Baltic Sea of the HELCOM project (Gustafsson and Mörth, 2015;

HELCOM, 2013b; Savchuk et al., 2012).[62] We did not include the updated planetary boundaries approach presented in Steffen et al (2015) as this relies heavily on the approach of Carpenter & Bennett (2011) for P and de Vries et al. (2013) and can, therefore, not be considered to be independent of these.

The sensitivity of indicator scores to a change in choices for different concerns was, to some extent, explored in all of the studies where the five indicators were originally presented. We contacted corresponding authors for indicators of de Vries et al. (2013), Liu et al. (2012) and HELCOM to access the complete indicator scores of these sensitivity analyses beyond what was presented in their studies for spatial units falling within the Baltic Sea region. These indicators were adapted by introducing different choices for the aggregation of indicator scores for spatial units (concern 11). We further adapted the global indicators of Rockström et al. (2009) and Carpenter & Bennett (2011) to the case study. This adaptation was made by quantifying total environmental interferences for the Baltic Sea region (concern 8) and introducing an allocation module to the indicators used to allocate global assimilative capacities to the environmental interferences of the Baltic Sea region (concern 12).

2.3. Sensitivity study

By contacting authors and adapting the indicator sets, we ended up with a total of 96 and 64 linear combination of choices related to P and N, respectively.[63] Each linear combination represents a specific choice made for each of two, three or four concerns, depending on the indicator.[64] These linear combinations were plotted to examine the variation of indicator scores within and between the indicators. To investigate the contributions of individual choices to variations within indicator scores, we calculated the range, $R_{i,j,k}$, of indicator scores:

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[62] HELCOM (2013a) presented ecological boundaries (termed targets) and indicators. Savchuk et al. (2012) documented the impact pathway model used to translate these ecological boundaries into assimilative capacities (termed maximum allowable inputs). Gustafsson & Mörth (2015) present these assimilative capacity estimates and propose their allocation to countries in the catchment.

[63] The numbers of linear combinations for each indicator related to phosphorous were: 6 for Rockström et al. (2009), 36 for Carpenter & Bennett (2011), 36 for Liu et al. (2012) and 18 for HELCOM. The numbers of linear combinations for each indicator related to nitrogen were: 6 for Rockström et al. (2009), 4 for (de Vries et al., 2013), 36 for Liu et al. (2012) and 18 for HELCOM.

[64] In theory, we could have examined 12 linear combinations for each indicator corresponding to the 12 concerns but, as we did not have access to the mathematical codes for five of the indicators, this was not practically possible.

$$R_{i,j,k} = \frac{MAX_c(IS_{c,i,j,k})}{MIN_c(IS_{c,i,j,k})}$$

For each indicator, k, the numerator and denominator identify, respectively, the maximum and minimum indicator scores, $IS_{i,j,k}$, amongst all choices, c, for concern i when choices for the other concerns are fixed at combination j. A high value of $R_{i,j,k}$ thus indicates that indicator scores are highly sensitive to the choice made for concern i, when choices for other concerns are fixed at combination j. $R_{i,j,k}$ was calculated for all j and a $R_{i,k}$ range across j was plotted.

3. Results

3.1. Comparison of indicator choices for 12 concerns

In the following, the choices for the 12 concerns covered in this study for the five indicators and their adaptations with respect to concerns 8, 9, 11 and 12 are presented. Table 2 contains a summary. For concerns related to the point of expression of a parameter in the impact pathway, choices covered by each adapted indicator are shown in Figure 1, which is based on the so-called DPSIR framework (Smeets and Weterings, 1999). Parameters were here classified to the driver point when expressed as application of P or N as fertilizer, the pressure point when expressed as emission of P or N and the state point when expressed as a change in an indicator value for the state of the environment, such as concentration of P or N. None of the parameters of this study were classified to the impact point, which covers impacts on ecosystems as indicated by e.g. a decrease in species diversity.

Table 2: Comparison of reviewed indicators. Bold numbers indicate that more than one choice was covered by the sensitivity analysis of this study. Concerns for which choices have been adapted for this study are given in italics.

Indicator / Concern	Adaptation of Rockström et al. (2009) (P; N)	Adaptation of (Gustafsson and Mörth, 2015; HELCOM, 2013b; Savchuk et al., 2012)	Adaptation of Liu et al. (2012)	Adaptation of Carpenter & Bennett (2011)	Adaptation of de Vries et al. (2013)
1. Ecosystem of focus	The Earth system.	The Baltic Sea.	River.	Freshwater.	Agricultural runoff to surface water.
2. Goal	Avoid a major oceanic anoxic event (P); maintain resilience of ecosystems (N).	The Baltic Sea unaffected by eutrophication.	None stated.	Avert widespread eutrophication.	Preventing aquatic ecosystems from developing eutrophication or acidification.
3. Control variable	Inflow of P to ocean; anthropogenic fixation of N.	choices which combines five control variables (winter dissolved inorganic N and P concentrations, summer secchi depth, summer concentration of Chl α, oxygen concentration) and expert judgement (see S2 for details).	Concentration of **1**) particulate N and P, **2**) dissolved organic N and P, **3**) dissolved inorganic N and P, **4**) all.	Total P concentration.	Dissolved inorganic N concentration.
4. Basis for the ecological boundary (classification according to Dearing et al. (2014) in italic)	Threshold based on geological model of P cycle and anoxic response (Handoh and Lenton, 2003) (*type IIIb*); expert judgement (*unclassified*).	Upper limit in 95% confidence interval for time series in "pre-eutrophication period" identified via a change-point analysis (*type IIb*). For control variables with insufficient data: average of simulated value for the year 1900 and measured values around 1970.	Review of national policy targets (Laane 2005) (*type Ia*).	**1**) Pre-industrial river concentration (Bennett et al., 2001) (*type Ib*), **2**) water quality criterion (Carlson 1977) (*type Ia*).	Review of policy targets (Laane, 2005; Liu et al., 2012) and of ecological and toxicological effects of inorganic N pollution (Camargo and Alonso, 2006) (*type Ia*).

5. Ecological boundary value	**1**) high, **2**) low.	None stated.	**1**) high, **2**) best estimate, **3**) low.	None stated.	**1**) high, **2**) low.
6. Location of assimilative capacity in the impact pathway	Pressure (P flow to ocean); Driver (N fixation).	Pressure (N and P flows to ocean)	Pressure (N and P flows to rivers).	**1**) P flow to freshwater (pressure point), **2**) P flow to soil (driver point).	**1**) Driver (N_2 fixation), **2**) Pressure (N losses)
7. Modelling of assimilative capacity	No calculation necessary as ecological boundary and assimilative capacity are identical.	Based on the impact pathway model of Savchuk et al. (2012) the loads reduction needed to get below ecological boundaries are modelled (Gustafsson and Mörth, 2015).	Tolerable river load of N and P (kg/year) is calculated per watershed based on 1) the basin discharge, 2) the maximum concentration, and 3) the natural concentration.	Linear, steady state mass balance model is used to calculate, for the two ecological boundaries, assimilative capacities for flows of P to soil and freshwater (Tg/year). Three assimilative capacities estimates are given, reflecting **1**) low, **2**) medium, and **3**) high estimates of current P flow to the sea.	Tolerable runoff of N (kg/year) is calculated per grid cell based on modelled current runoff and the relationship between the ecological boundary concentration and the modelled current concentration (Bouwman et al., 2006).
8. Quantifying environmental interferences of assessed system	*HELCOM monitoring data (P); statistics on fertilizer application (N).*	Current riverine and direct loads are estimated mainly from measurements. Gaps are filled out by extrapolations. Modelled depositions based on Granat et al. (2001) and an EMEP model (Bartnicki et al., 2008).	NEWS impact assessment model estimates current river exports based on human activities and natural conditions (Mayorga et al., 2010).	*Statistics on fertilizer application (P flow to soil), HELCOM monitoring data and retention factors (P flow to freshwater).*	IMAGE impact assessment model estimates current N runoff based on human activities and natural conditions (Bouwman et al., 2006).

9. Spatial coverage and resolution	*Baltic Sea region, site-generic.*	Baltic Sea region, 17 sub basins (ecological boundary site-specific).	*Baltic Sea region, 33 rivers (ecological boundary site-generic).*	*Baltic Sea region, site-generic.*	*Baltic Sea region, 0.5°×0.5° (ecological boundary site-generic).*
10. Temporal coverage and resolution	Environmental interferences established annually (2008-2010 average for N; 2012 for P).	Environmental interferences established annually (1997-2003 average). Ecological boundaries seasonally specified.	Environmental interferences established annually (2000).	Environmental interferences established annually (2008-2010 average for flow to freshwater and 2012 for flow to soil).	Environmental interferences established annually (2000).
11. Aggregation of indicator scores	None needed because indicator is site-generic.	***1****) average,* ***2****) average, spatial unit indicator scores below 1 fixed as 1, and* ***3)*** *share of spatial units with indicator scores above 1.*	***1****) average,* ***2****) average, spatial unit indicator scores below 1 fixed as 1, and* ***3)*** *share of spatial units with indicator scores above 1.*	None needed because indicator is site-generic.	***1****) average.* ***2****) average, spatial unit indicator scores below 1 fixed as 1, and* ***3)*** *share of spatial units with indicator scores above 1.*
12. Basis for allocation	***1****) GDP,* ***2****) cropland area,* ***3****) population.*	No allocation needed as indicator is spatially resolved.	No allocation needed as indicator is spatially resolved.	***1****) GDP,* ***2****) cropland area,* ***3****) population.*	No allocation needed as indicator is spatially resolved.

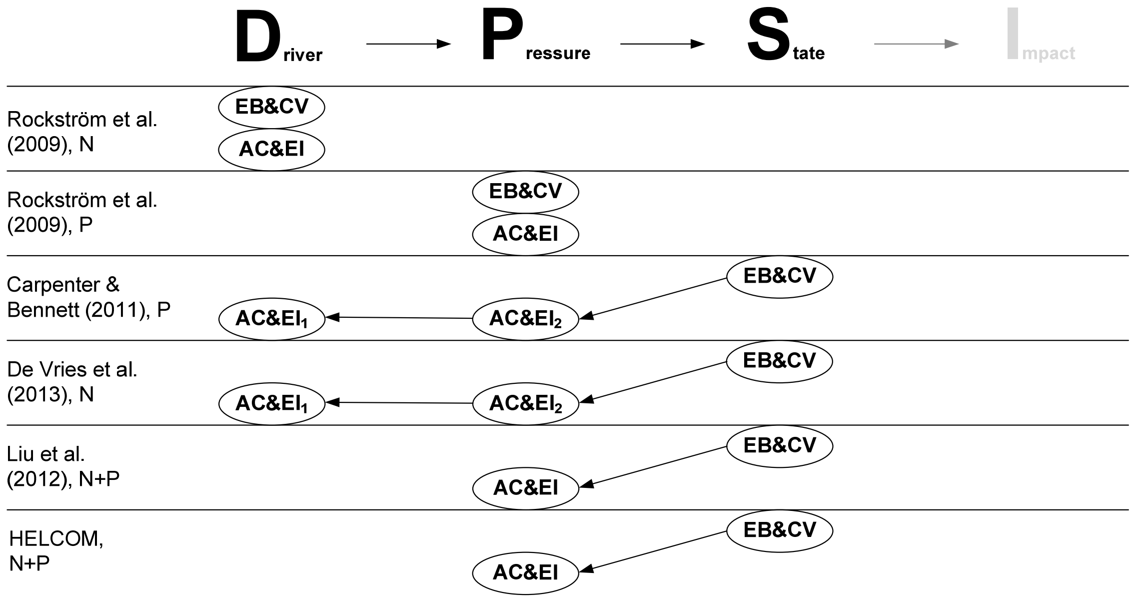


Figure 1: Choices covered by the adapted indicators for the expression of ecological boundaries and control variables (EB&CV) and assimilative capacities and environmental interferences (AC&EI) in a generic impact pathway based on Smeets & Weterings (1999). For Carpenter & Bennett (2011) and de Vries et al. (2013), two choices are covered for AC&EI (see also Table 1).

3.1.1. Ecosystem

The indicators covered one specific ecosystem, ranging from agricultural ditches (de Vries et al., 2013) through rivers and freshwater (Carpenter and Bennett, 2011; Liu et al., 2012) to the open ocean (HELCOM). The only exception is the indicator of Rockström et al. (2009) where the ultimate concern is the Earth system rather than specific ecosystems.

3.1.2. Goal

One specific goal was addressed by each indicator. Goals related to the avoidance of eutrophication were given by Carpenter & Bennett (2011), de Vries et al. (2013) and HELCOM, whereas Rockström et al. (2009) chose for P the avoidance of major anoxic events in the ocean and for N maintaining the resilience of ecosystems. In contrast, Liu et al. (2012) did not state a goal, possibly because their indicator uses policy targets as the basis for ecological boundaries, see below. Thus, all indicators are concerned with the overall state of ecosystems as a precondition for environmental sustainability rather than the protection of single species, functions or processes.

3.1.3. Control variable

For most indicators, a single control variable was used by the authors. However, Liu et al. (2012) and HELCOM employed more than one control variable (concentrations of different forms of N and P and, for the indicator of HELCOM, Chl α, oxygen[65] and summer secchi depth). Liu et al. (2012) presented N and P indicator scores for three control variables in addition to a combination of all three control variables (i.e. a worst case scenario). By comparison, HELCOM did not present indicator scores for individual control variables but chose to combine five control variables in six ways, also involving expert judgment.[66] The choice of control variables made by Rockström et al. (2009) was unique because these are expressed at the beginning of the impact pathway (driver point for N in the form of anthropogenic N fixation and pressure point for P in the form of inflow to oceans), rather than at the state point as was the case for all the other indicators, see Figure 1.

3.1.4. Basis for the ecological boundary

Dearing et al. (2014) provided a useful classification of ecological boundaries into four types, each of which was further classified into two sub-types.[67] Amongst the indicators covered by this study, those of Carpenter & Bennett (2011), de Vries et al. (2013) and Liu et al. (2012) can be classified as type Ia or Ib, meaning that the ecological boundaries are either based on (Ia) environmental limits, i.e. quality targets or political targets, to some extent informed by science, or (Ib) distance from a pre-historic baseline. Note that despite the linear characterization of Dearing et al. (2014), these type I ecological boundaries may be based on a suspicion of thresholds (type III ecological boundaries). The ecological boundaries of HELCOM are based on a critical deviation from naturally dynamic measurement points of the Baltic Sea within a period of negligible eutrophication (1900-1945). These boundaries can, therefore, be classified as type IIb, i.e., envelope of variability for non-linear trends. The ecological boundary for P of Rockström et al. (2009) can be classified as type IIIb, i.e., threshold of abrupt hysteretic change, as it is based on the relationship between the dynamics of estimated pre-historic inputs of P, caused by ice age dynamics, and fossil indications of hysteretic[68] type

[65] The state variable oxygen concentration is expressed in the form of oxygen debt, defined as depth-integrated deviation between oxygen concentration and saturation (HELCOM, 2013b).

[66] Expert judgment was used to evaluate the ecological relevance of each control variable for the different sub-basins. For example, choice 4 (termed case 3 in Gustafsson & Mörth (2015)) combined the four control variables secchi depth, oxygen, dissolved organic P and dissolved organic N, but exceptions were made for three sub-basins due to the low ecological relevance of some of the four control variables at these locations as judged by experts.

[67] The four types are: I) linear trends - a) environmental limits, b) distance from a baseline or background/low impact state; II) nonlinear trends - a) rate of change, b) envelope of variability; III) thresholds - a) abrupt non-hysteretic changes, b) abrupt hysteretic change; IV) early warning signals - a) shifts in magnitude and frequency, b) variability metrics (Dearing et al., 2014).

[68] When a hysteretic type threshold is crossed, negative effects are difficult to counteract because the ecosystem has entered a new quasi stable state enabled by the initiation of new positive feedback mechanisms. In practice, this means that a reduction in environmental interference

transitions from oxygen rich to anoxic seabed. This transition triggered by high P inputs is characterized as a “major anoxic event” (Handoh and Lenton, 2003; Rockström et al., 2009). The ecological boundary for N of Rockström et al. (2009) cannot be classified according to the scheme of Dearing et al. (2014), as the boundary of 75% reduction of current anthropogenic N_2 fixation was based on expert judgement . The difficulty of establishing a science based ecological boundary for N may reflect the fact that Rockström et al. (2009) chose maintaining resilience of ecosystems as a (rather abstract) goal (concern 2). Note that the original planetary boundary for N was changed in the updated planetary boundaries study (Steffen et al., 2015) and now draws heavily on the analysis made by de Vries et al. (2013).

3.1.5. Ecological boundary value

Most indicators provided at least two values ranging from low to high in response to the uncertainty of ecological boundary values. Carpenter & Bennett (2011) and HELCOM provided just one unspecified choice.

3.1.6. Location of assimilative capacity in the impact pathway

All assimilative capacities were expressed at the driver or pressure point of the impact pathway, which are closer to the cause of eutrophication than the state point at which control variable were typically expressed, see Figure 1. Only in Carpenter & Bennett (2011) and de Vries et al. (2013) were indicators chosen to express assimilative capacities both at the driver and pressure points, which was aligned with current interventions that were also expressed at these two points in the studies. Note that we in this analysis excluded the original choice covered by Carpenter & Bennett (2011) of expressing assimilative capacity as stock of P in soil. This was done because of the high uncertainties on estimates of natural and anthropogenically introduced P in soils (MacDonald et al., 2011; Yang et al., 2013) and because of the high sensitivity of indicator scores to estimates of natural P in soil.[69]

3.1.7. Modelling of assimilative capacity

To translate ecological boundaries into assimilative capacities, different environmental impact pathway models were used for the indicators (the only exception

of a similar magnitude as the increase in environmental interferences, that previously caused the threshold to be exceeded, is not sufficient to bring the system back to its original state. Hysteresis has been observed for e.g. the response of shallow lakes to changes in phosphorous loadings (Scheffer et al., 2001).

[69] Indicator scores are very sensitive to estimates of natural P in soils given that annual anthropogenic flows of P to soil are around 3 orders of magnitude lower than estimates of natural P stocks in soils (Carpenter and Bennett, 2011; MacDonald et al., 2011; Yang et al., 2013).

being Rockström et al. (2009), which was in need of no such translation as the ecological boundaries and assimilative capacities are identical in their P and N indicators, see Figure 1). These models are generally steady state based multimedia models that, for example, calculate the emission that corresponds to the numerical value of an ecological boundary at steady state. This calculation is either based on modelling of how much emissions must be decreased in order to get below ecological boundaries (de Vries et al. 2013; HELCOM), or on modelling how much natural or pre-industrial emission levels might be increased without ecological boundaries being exceeded (Carpenter and Bennett, 2011; Liu et al., 2012). The critical emissions calculated from ecological boundary values were then used directly as assimilative capacity, if this is expressed at the pressure point (see above and Figure 1). If assimilative capacity is expressed at the driver point, the critical emissions were translated into a rate of fertilizer application, under the assumption of a given fertilizer use efficiency. The indicator of Carpenter & Bennett (2011), considered 3 estimates of current environmental interferences in the form of P flow to the sea, and each of these resulted in specific translations of ecological boundary values to values of assimilative capacity. The BALTSEM model applied by HELCOM is more sophisticated than the models of the other indicators as it is dynamic and covers variations in meteorological forcing, seasonal variations of parameters, and water circulation patterns (Gustafsson and Mörth, 2015; Savchuk et al., 2012). As some of the control variables of HELCOM (Chl α and oxygen concentration and summer secchi depth) are affected both by loads of N and P, infinite sets of assimilative capacities for N and P could, in theory, be calculated. To arrive at one set, a cost function was applied to estimate an optimum.

3.1.8. Quantifying environmental interferences of assessed system

de Vries et al. (2013) and Liu et al. (2012) calculated environmental interferences from impact pathway models relying on drivers such as agriculture, combustion processes and waste water outlets. In contrast, HELCOM, which focused on the Baltic Sea only, estimated environmental interferences primarily on the basis of measurements (HELCOM, 2013a). In the adapted indicators of Rockström et al. (2009) and Carpenter & Bennett (2011), we calculated environmental interferences within the Baltic Sea region from the aforementioned HELCOM measurements, combined with estimates of P and N inputs in fertilizer (Eurostat 2015) (see S1 for details).

3.1.9. Spatial coverage and resolution

Indicators were adapted to cover the Baltic Sea region (except in the case of the HELCOM indicators which was originally designed for that region).The spatial resolution of the indicators varied widely. The N and P indicators of Rockström et al. (2009) are spatially generic, which reflects the choice of the Earth system (concern 1). Carpenter & Bennett's (2011) indicator is also spatially generic, meaning that freshwater ecosystems (chosen in concern 1) are treated as globally homogenous in the underlying models. de Vries et al. (2013) and Liu et al. (2012) use site-generic ecological boundaries but their indicators have a finer resolution with respect to environmental interferences by means of a grid cell structure and division of compartments into main rivers. HELCOM's was the only indicator to be both spatially resolved with respect to environmental interferences and ecological boundaries by division of the Baltic Sea into sub-basins.

3.1.10. Temporal coverage and resolution

For all indicators, environmental interferences (concern 11) were estimated annually. Thus, potential seasonal variations caused by e.g. variations in application of fertilizer are neglected. Reference years varied from 2000 to 2012. The chosen ecological boundaries employed were generally seasonal worst case scenarios representing the most sensitive time of the year in the response of control variables to environmental interferences, although HELCOM specified seasons for most of their ecological boundaries (e.g. summer secchi depth and winter DIN and DIP).

3.1.11. Aggregation of indicator scores

For the spatially derived indicators, two choices were introduced in the adaptation of the indicators to average indicator scores from spatial units into a single score: 1) Averaging[70], 2) Averaging, but with the adjustment that non-exceeded spatial units were assigned a score of 1 (meaning that environmental interferences are considered to occupy 100% of assimilative capacities). The rational for the latter choice is that non-exceeded assimilative capacity at one site in reality cannot offset exceeded assimilative capacity at another site. The two averaging related choices of aggregation provide information about the degree of assimilative capacity exceedance. Nevertheless, inspired by Liu et al. (2012), we introduced a third approach in which the share of spatial units where the assimilative capacity is estimated to be exceeded is used as an aggregated indicator score. As none of

[70] For the HELCOM indicator set, we did not have access to the indicator score for spatial units where assimilative capacity was not exceeded because a value of 1 was assigned to these, in agreement with the second choice of aggregation covered in this study. To correct for this, we increased the assimilative capacities by 10% for these relevant spatial units based on expert judgement of Gustafsson (2015).

the indicators are temporally resolved, there was no need to aggregate indicator scores across time.

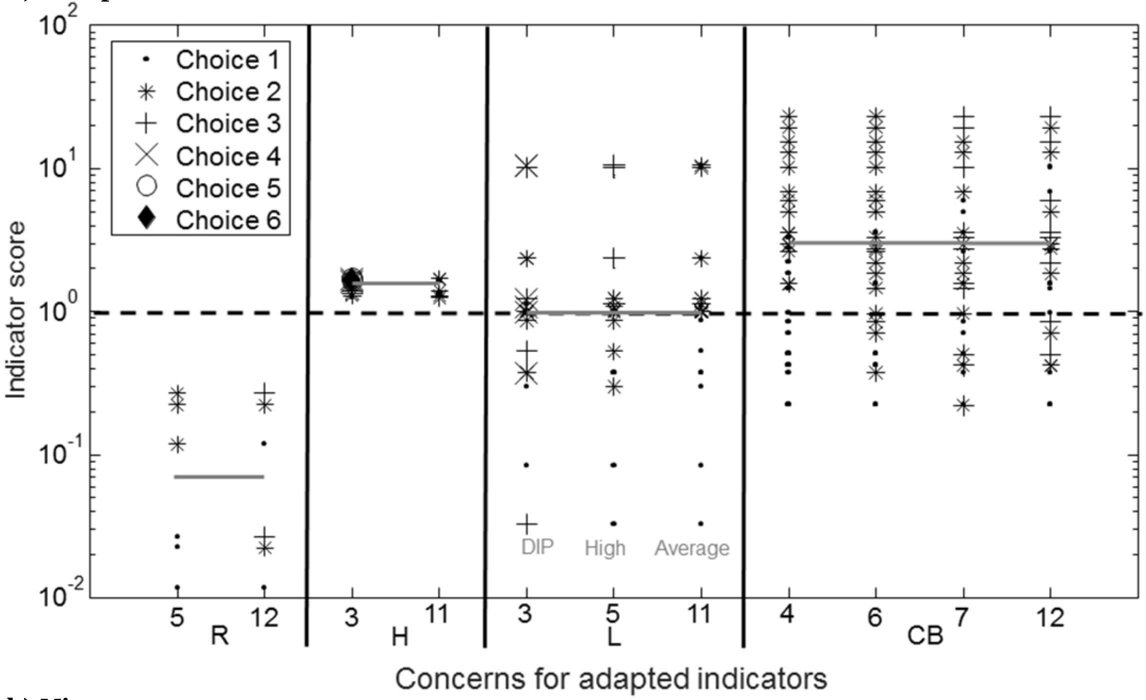
3.1.12. Basis for allocation

Three choices for allocation basis of global assimilative capacity to assimilative capacity for environmental interferences within the Baltic Sea region were introduced in the adaptation of Rockström et al. (2009) and Carpenter & (Bennett 2011): Population, cropland area and GDP (see S1 for details).

3.2. Comparison of indicator scores

Indicator scores for all linear combinations of choices are plotted in Figure 2. Exceptions are linear combinations involving the choice of expressing aggregated indicator scores as shares of spatial units having their assimilative capacity exceeded (choice 3 for concern 11). The latter are, instead, presented in S3 due to the different meaning of their indicator scores compared to averaging related choices of aggregation. In Figure 2, linear combinations represent a combination of choices pertaining to two, three or four concerns, depending on the study. Linear combinations are, therefore, presented as two to four “dots” in the horizontal plane where each dot represents an individual choice for a specific concern. Specifications of choices 1-6 for the different concerns and indicators are given in Table 2. For example, as indicated in Figure 2 with grey font, the lowest linear combination in Figure 2a for the adapted P indicator of Liu et al. (2012) of 3.3% occupation of assimilative capacity is composed of a combination of choice 3 (dissolved inorganic phosphorous) for concern 3 (control variable), choice 1 (high) for concern 5 (ecological boundary value) and choice 1 (averaging) for concern 11 (aggregation).

a) Phosphorous



b) Nitrogen

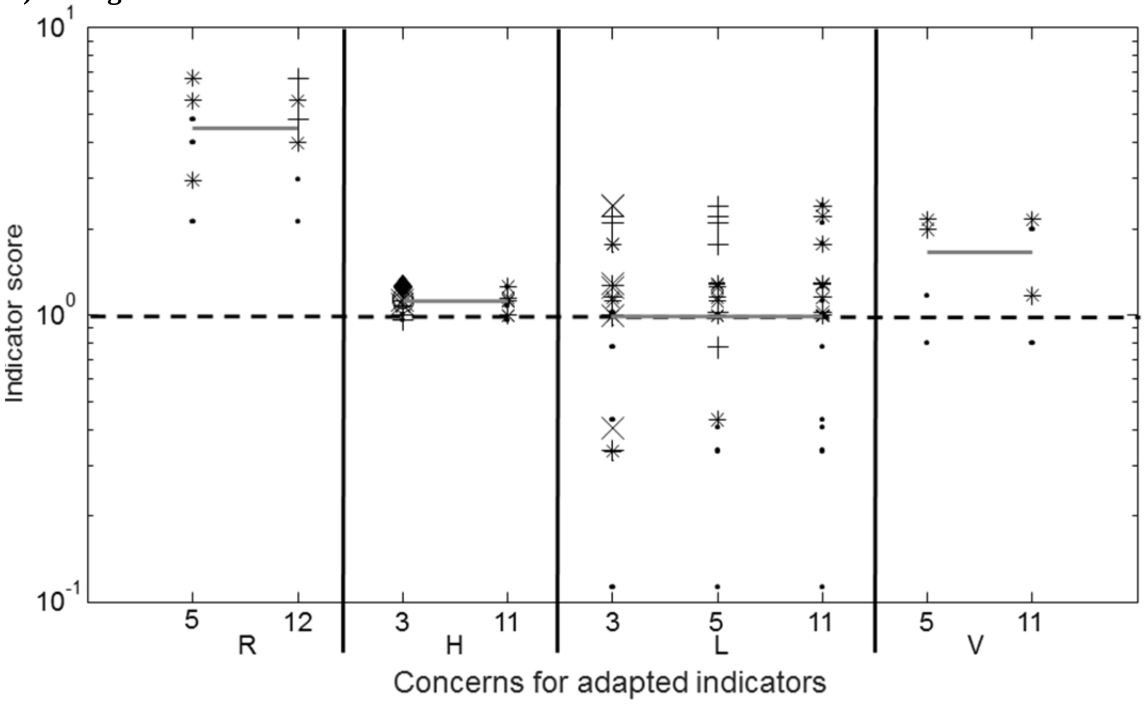


Figure 2: Indicator scores for linear combinations of the adaptation of Rockström et al. (2009) (R), HELCOM (H), Liu et al. (2012) (L), Carpenter & Bennett (2011) (CB) and de Vries et al. (2013) (V) for averaging related aggregation (concern 11) for phosphorous (a) and nitrogen (b). Horizontal grey lines indicate the median value of linear combinations for each indicator. See Table 2 for specification of choices. The location of assimilative capacity (concern 6) is not covered in our adaptation of de Vries et al. (2013) because indicator scores of the choices are identical. Note that the y-axis in Figure 2a spans more values than that of Figure 2b.

3.2.1. Total variation and variations within indicators

Figure 2a illustrates that the estimated occupation of assimilative capacity for ecosystems in the Baltic Sea region with respect to phosphorus interferences ranges by 3 orders of magnitude across indicators and linear combinations: The lowest indicator score is given by the adapted environmental sustainability indicator of Rockström et al. (2009) as a 1.2% occupation of assimilative capacity for the combination of a high ecological boundary value (choice 1 for concern 5) and allocation based on GDP (choice 1 for concern 12). In contrast, the adaptation of the indicator of Carpenter & Bennett (2011) gives the highest indicator score of 2300% of assimilative capacity occupied (22 times exceedance[71]) for the combination of an ecological boundary based on a water quality criterion (choice 2 for concern 4), assimilative capacity expressed as P to soil (choice 2 for concern 6), a high estimate of the parameter P flow to sea in the impact pathway model (choice 3 for concern 7) and allocation based on population (choice 3 for concern 12). With respect to nitrogen environmental interferences, Figure 2b shows that estimation of assimilative capacity occupation ranges only a factor 59 across indicators: The lowest indicator score is 11% for the adaptation of Liu et al. (2012) for the combination of particulate N as the control variable (choice 1 for concern 3), high ecological boundary value (choice 1 for concern 5) and aggregation based on averaging (choice 1 for concern 11). In contrast, the highest indicator score is 670% (5.7 times exceedance) for the adaptation of Rockström et al. (2009) for the combination of a low ecological boundary value (choice 2 for concern 5) and allocation based on population (choice 3 for concern 12).

Within indicators, it seems that the highest variation in indicator scores occurs for indicators with a relatively high number of linear combinations. For example, in Figure 2a for P, the variations of indicator scores for the adaption of Carpenter & Bennett (2011) (36 linear combinations) and Liu et al. (2012) (24 linear combinations) both extend over two orders of magnitude while the variations of indicator scores for the adapted indicators of HELCOM (12 linear combinations) and Rockström et al. (2009) (6 linear combinations) are a factor 1.35 and 23 respectively. This trend can be explained from the fact that the range of indicator scores for linear combinations will always increase when expanding the linear combination with additional concerns.

[71] Note that carrying capacity occupation of 2300% corresponds to 22 and not 23 times exceedance because 200% occupation of carrying capacity corresponds to 1 time exceedance.

3.2.2. Indicator differences

Comparing the range of indicator scores between indicators in Figure 2a for P interferences, it can be seen that the adapted indicator of Rockström et al. (2009) consistently (i.e. for all linear combinations) concludes that assimilative capacity is not exceeded, while that of HELCOM consistently concludes the opposite while those of Liu et al. (2012) and Carpenter & Bennett (2011) deliver no consistent conclusion (50-63% of linear combinations exceed 1). By comparison, Figure 2b shows that for N, only the adaptation of Rockström et al. (2009) makes a consistent conclusion (assimilative capacity exceeded), whereas the linear combinations of other adapted indicators distribute on both sides of the 100% assimilative capacity occupation mark (50-83% are above 1). Considering the median indicator scores of linear combinations for the adapted indicators, it can be seen that for P indicators, they rank as Carpenter & Bennett (2011) > HELCOM > Liu et al. (2012) > Rockström et al. (2009) and for N indicators, the rank is Rockström et al. (2009) > de Vries et al. (2013) > HELCOM > Liu et al. (2012) (see also Figure S1).[72]

Thus, for the three adapted indicator sets covering both N and P interferences, the ranking of the median score between adaptations of HELCOM and Liu et al. (2012) is the same, while the median indicator score of the adaptation of Rockström et al. (2009) ranks highest amongst N indicators and lowest amongst P indicators. This discrepancy between the adapted N and P indicators of Rockström et al. (2009) may be explained from the fact that the goals (concern 2) of these indicators are quite different from each other and those of the other indicators: The goal of the P indicator of avoiding major oceanic anoxic events can be seen as being less strict than the goals of the other indicators which are all related to avoiding eutrophication because oceanic anoxic events do not occur without extensive eutrophication. On the other hand, the goal of the Rockström et al. (2009) N indicator of maintaining resilience of ecosystems can be seen as being more strict than the eutrophication oriented goals of the other indicators as the resilience of ecosystems may be affected by loss of species and genotypes adapted to nutrient poor conditions at levels of environmental interferences lower than levels associated with the negative effects of eutrophication.

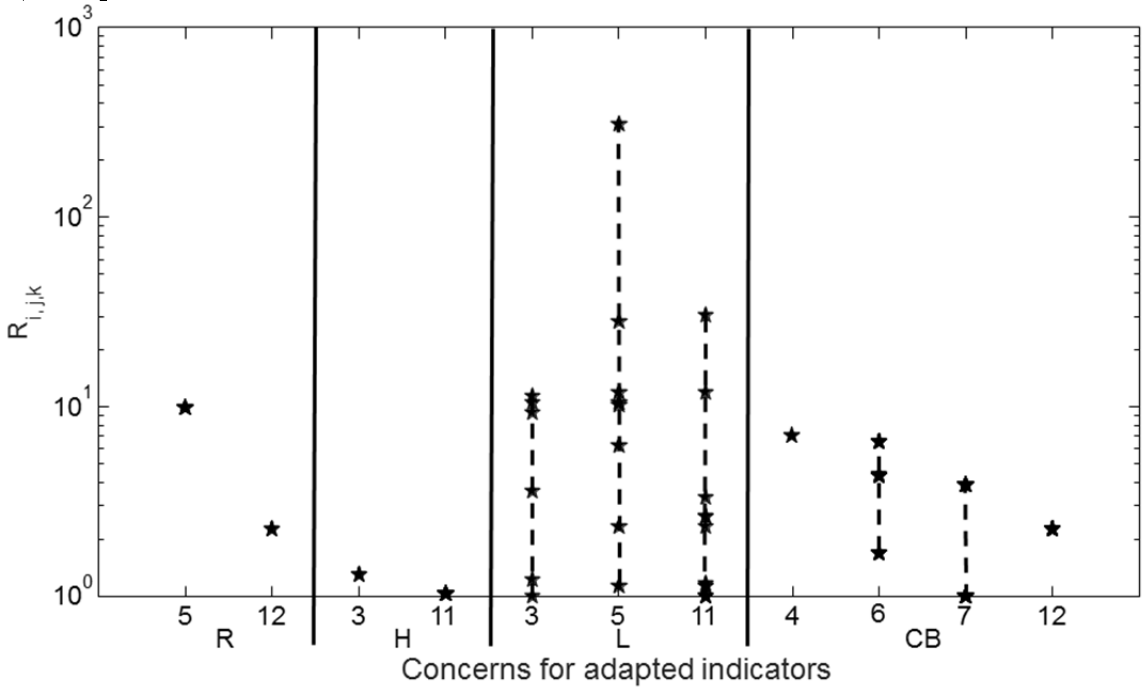
3.2.3. Sensitivity to different concerns

Figure 2 gives an initial impression of the concerns for which variations in choices lead to large variations in indicator scores. It is, for example, evident from the

[72] The same rankings were obtained from the geometric mean of the indicator scores. The arithmetic mean was not considered due to the nature of the populations of indicator scores.

linear combinations of the adaptation of Rockström et al. (2009) in Figure 2a that P indicator scores are more sensitive to changes in concern 5 (ecological boundary value) than to changes in concern 12 (allocation).To further explore the sensitivity of indicator scores to different concerns, we plotted the range, $R_{i,j,k}$, defined in section 2.3, for the indicator scores where aggregation is based on averaging (choice 1 and 2 for concern 11) for P and N in Figure 3.

a) Phosphorous



b) Nitrogen

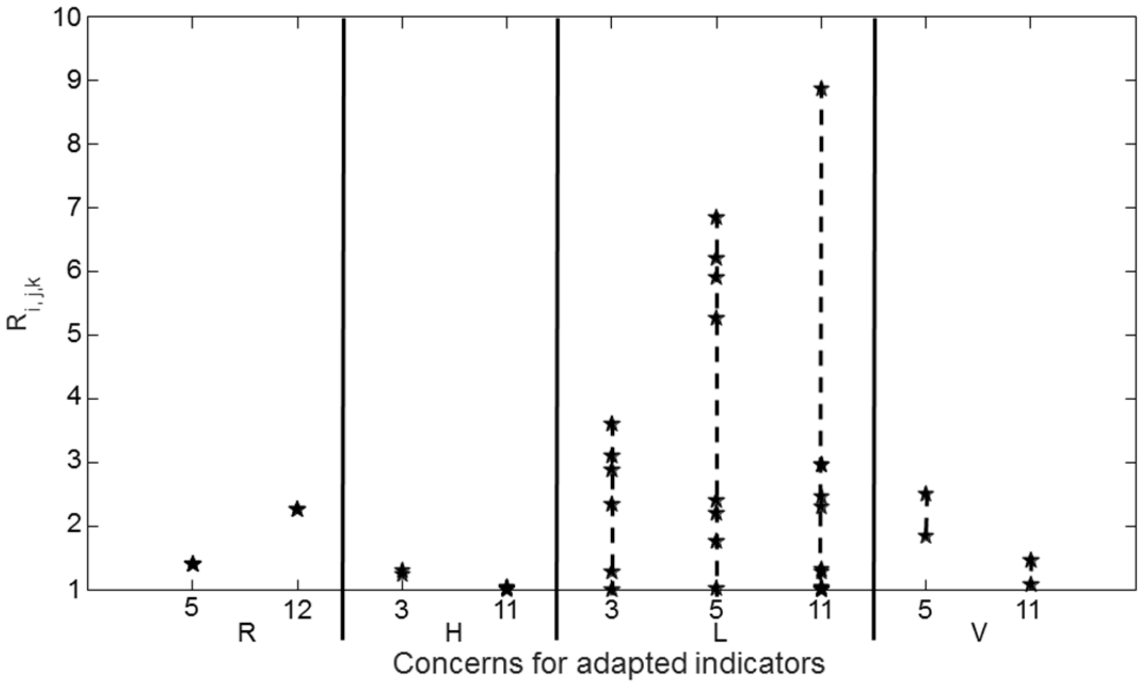


Figure 3: Plots of $R_{i,j,k}$ for the adaptation of Rockström et al. (2009) (R), HELCOM (H), Liu et al. (2012) (L), Carpenter & Bennett (2011) (CB) and de Vries et al. (2013) (V) for phosphorous indicators (a) and nitrogen indicators (b). i refers to the tested concern, j to a fixed combination of choices for other concerns and k to an adapted indicator. Vertical stippled lines indicate the range between the lowest and highest $R_{i,j,k}$ for one concern. Note that the $R_{i,j,k}$ scale of Figure 3a is logarithmic.

A high $R_{i,j,k}$ indicates a wide range between then smallest and largest indicator score obtained when varying choices for concern i, while keeping choices for other concerns fixed at combination j for adapted indicator k. In Figure 3, $R_{i,j,k}$ is generally higher for P indicators than for N indicators. This is in agreement with Figure 2, where a larger variation in indicator scores can be observed for P indicators than for N indicators. Figure 3 shows that $R_{i,j,k}$ for some indicators varies quite a lot across j (the combination of choices for other concerns). This is especially evident for concern 5 (ecological boundary value) for the adapted P indicator of Liu et al. (2012) (Figure 3a) from which $R_{i,j,k}$ varies from a value of 1.13 to 308 across j. This shows that, for this indicator, the sensitivity of indicator scores to changes in choice of ecological boundary value is highly contingent upon the fixed combination of choices made for other concerns (j): For a j composed of DIP as control variable (choice 3 for concern 3) and averaging as aggregation (choice 1 for concern 11), a change between the three choices of ecological boundary value (concern 5) can change indicator scores by up to a factor of 308. For other j, the same change between the three choice of ecological boundary value can only change indicator scores by only 13% (a factor 1.13). This example illustrates that, in many cases, the indicator score is neither inherently sensitive nor insensitive to variation in choices for certain concerns. Instead, many indicator scores are potentially sensitive to variations in choices for a given concern, depending on choices made for other concerns (j).

Overall, Figure 3 shows that a common strategy for the effective reduction of uncertainties for the 5 adapted indicators should prioritize reduction of the range of choices related to control variable (concern 3), ecological boundary value (concern 5) and aggregation of indicator scores (concern 11) and that the choice of allocation (concern 12) should receive less attention. In the next section, we use these results to develop guidance on the reduction and management of uncertainties for indicators in general.

4. Discussion

The results of the indicator analysis using the Baltic Sea region as a case study showed that the range of indicator scores within and between environmental sustainability indicators typically spanned at least one and up to three orders of magnitude. The results also show that these wide ranges of indicator scores in many cases prevented clear conclusions as to the environmental sustainability of assessed anthropogenic systems. This is problematic from a decision-making perspective as it creates confusion regarding whether or not current environmental interferences need to be reduced (and if so, the extent of this reduction) in order to achieve conditions that can be considered environmentally sustainable. For example, an environmental sustainability indicator associated with the proposed SDG #2 on promoting sustainable agriculture (UN, 2014) is not useful if uncertainties in indicator scores are so large that it is unclear whether a given country or region should decrease nitrogen and phosphorous interferences to be environmentally sustainable or whether there is, in fact, room for increased environmental interferences through e.g. agricultural intensification. The reduction of indicator uncertainties is, therefore, of great importance for the use of environmental sustainability indicators in decision making.

With respect to the prioritization of concerns in the reduction of uncertainty, this study indicates that there are no obvious “low hanging fruits”: In our comparison of environmental sustainability indicators for N and P, the sensitivity of indicator scores to changes in choices for a given concern depends to a large extent on the indicator and on the combination of choices made for other concerns (j) (see Figure 3). In addition, concerns that from Figure 3 appear to be insignificant sources of uncertainty in indicator scores may, potentially, be large sources of uncertainty for indicator scores for indicators beyond the 5 compared. The range of choices of allocation (concern 12), for example, can be much wider than the range covered in this study, especially if the object of study is at the micro-scale as in the case of product systems.[73] This means that the concern of allocation cannot generally be ruled out as being important in the effort to reduce uncertainties in scores of environmental sustainability indicators. We conclude, therefore, that all 12 concerns need consideration in the pursuit of uncertainty reduction for any environmental sustainability indicator. Below, we discuss how to reduce the range of choices for each concern and outline how designers of indicators and users can manage the remaining choices. Note that the role of the user of environmental sustainability indicators may vary from passively extracting indicator scores to actively inputting data on current interventions and making choices for the other concerns, if the indicator allows it. In the following, we differentiate between these types of users.

4.1 Ecosystem

The protection of all ecosystems is potentially important for environmental sustainability and the choice of ecosystem of concern becomes, therefore, essentially a value judgement (see Table 1). This means that it is not desirable to decrease the range of ecosystems of concern for a given stressor. Instead, designers of indicators should make it clear to users what ecosystem(s) is covered. This is especially important in cases where a chosen ecosystem is known to be less sensitive to the identified stressors than ecosystems not considered.

4.2 Goal

As in the case of the ecosystem of concern, the value laden choice of goal is also closely related to an understanding of environmental sustainability. The definition of Goodland (1995) has a strong anthropocentric focus in that the ultimate goal of environmental sustainability is “to prevent harm to humans”. The translation of this ultimate goal to a goal at the ecosystems level is not straight forward: One could argue that goals should be centred on the protection of eco-system services (MEA, 2005), in which case the loss of species or ecosystem processes, seemingly irrelevant for ecosystem services, could be considered unproblematic. One could, however, also argue that the prevention of harm to humans indirectly requires protection of biodiversity because biodiversity is generally needed to ensure resilient ecosystems, i.e. ecosystems that are, in short, able to cope with or adapt to sudden changes in surroundings (Scheffer et al., 2001). Resilience is, for example, a central element in the theoretical foundation of the planetary boundaries concept (Rockström et al., 2009; Steffen et al., 2015) which argues that non-

[73] Consider the wide range of opinions on how much of the atmosphere’s assimilative capacity for greenhouse gas emissions that should be allocated to privately owned cars.

exceedance of planetary boundaries represents a "safe operating space for humanity" and, thereby, is a precondition for sustainability.

Furthermore, alternative definitions of environmental sustainability may consider the environment valuable not only in instrumental terms for preventing harm to humans but in its own right (Dryzek, 2005). From this perspective any species lost, or even a decrease in a species population, could be considered environmentally unsustainable. Due to this variation in the understanding of environmental sustainability, we do not consider it desirable to reduce the range of goal choices. Rather, we encourage designers of indicators to clearly communicate the link between their choice of goal and their understanding of environmental sustainability to users so these aspects are transparent when indicators are used as decision support.

4.3 Control variable

The concern of control variable is mainly related to scientific understanding of the ecosystem variable that is most suitable for measuring the extent to which the chosen goal has been met. In combination, the original five indicator sets used in this study applied 10 different control variables (see Table 1). This seems like an unnecessarily high number considering that the goals of most of the five indicators are related to the prevention of negative effects of eutrophication. The choice of an appropriate control variable in a given case should, however, not only consider the ability of different variables to represent relevant characteristics of ecosystems but also on the availability of monitoring data. The sensitivity analysis of section 3.2 showed that the concern of control variable is associated with a potentially high contribution to uncertainty in indicator results. To reduce the range of choices, two guidelines can be provided: 1) Designers of indicators should use the state-of-the art scientific understanding to guide them in their choice of control variables, 2) More research should be directed towards improving state-of-the-art, including monitoring data.

4.4 Basis for ecological boundary

The choice of ecological boundaries is partly related to scientific understanding and partly to value judgement. Regarding the former, a scientific basis as outlined by Dearing et al. (2014), should underpin the establishment of any ecological boundary. The range of choices can thus be shortened by, for example, excluding boundaries solely based on expert judgement. Even as scientific understanding improves, there will inevitably remain an element of value judgement in the choice of an ecological boundary, as there is not necessarily a single objectively correct science based ecological boundary pertaining to each case of an ecosystem’s response to different stressors (Dearing et al., 2014). Given its potentially large influence on indicator scores, it is important that the strategy used in choosing an ecological boundary is made clear to users and decision makers. Changing, for example, the choice of ecological boundary in the indicator of Carpenter & Bennett (2011) from being based on a (regulatory) environmental limit (type Ia) to being based on a pre-industrial reference condition (type Ib) was found to change indicator scores seven-fold (see Figure 3a).

4.5 Ecological boundary value

This concern is related to value judgement in the face of incomplete scientific understanding of ecological boundaries. A possible way of reducing the range of choices (i.e. from low to high numerical ecological boundary value) is to frame the choice around risk. Decision makers are used to balancing the utility of an action with the risk introduced by that action. For example, the costs of additional traffic safety measures are routinely compared to the expected reductions in risk of traffic accidents from these measures when deciding whether to adopt the measures. Likewise, if decision makers are presented with the risks (%) of crossing an ecological boundary, the cost of preventing its crossing and the expected negative consequence of the crossing for different choices of ecological boundary values, a reasonable range of choices may be established in a consensus process. Such a process might, for example, lead to the removal of the highest and the lowest ecological boundary values included by Liu et al. (2012), which currently span a factor 15 and, thus, a great reduction of uncertainties in indicator score (see Figure 3a).

4.6 Location of safe limit in impact pathway

The concern of where to locate a safe limit in the impact pathway is related to assumed preferences of active users regarding the format of quantified environmental interferences. Ideally, the choice of where to locate safe limit in an impact pathway should not create uncertainty in indicator scores because the fraction of a safe limit taken up by environmental interferences should not depend of the point in the impact pathway where these are expressed. This is, indeed, the case for the indicator of de Vries et al. (2013), which is why we did not include the two choices of location in the analysis in section 3.2. Discrepancies in indicator scores, however, occur when an impact pathway model predicts a different relationship between the points in the impact pathway than the relationship obtained from monitoring data. This explains why choosing between the driver and pressure level for Carpenter & Bennett (2011) changed scores of the adapted indicator by up to a factor 7. An obvious way to reduce this uncertainty is to disregard choices of safe limit locations that are located closer to the driver point in the impact pathway (i.e. further to the left in Figure 1) than estimation of environmental interferences from monitoring data. In the case of our adaptation of the indicator of Carpenter & Bennett (2011), monitoring data was available for P input to Baltic Sea from which environmental interferences could be expressed as P input to freshwater rather accurately. The additional uncertainty of expressing assimilative capacities at the driver point (P input to soil) is, therefore, avoidable for this indicator and others for which environmental interferences can be established directly from monitoring data at the pressure point. In addition, a safe limit should be expressed in a metric that is more sensitive to human environmental interferences than to natural conditions. Expressing the safe limit for P with regard to avoiding freshwater eutrophication, for example, as mass of P in soil is not recommended because indicator scores will almost completely depend on uncertain estimates of natural P content in soils (see section 3.1.6).

4.7 Modelling of safe limit

This concern is related to scientific understanding because impact pathway models may differ in their assumptions about important processes in an impact pathway. There is a trade-off between model uncertainty and parameter uncertainty in any impact pathway model (van Zelm and Huijbregts, 2013): Increasing the number of processes covered reduces model uncertainties because the model more closely mirrors reality but, at the same time, also increases parameter uncertainty because of the increasing need for, often uncertain, input parameters required to model these additional processes. An optimum where the combination of model and parameter uncertainty is lowest should, therefore, be the aim. This would ideally reduce the range of choices of impact pathway models to just one optimum impact pathway model and the identification of this optimum could be facilitated by improved scientific understanding of the impact pathway and improved estimates of the parameters involved. However, as the optimum model may vary from case to case due to variations in quality of input parameters, seeking consensus amongst model developers may be a more practical way of reducing the range of choices for this concern. Such a consensus process led to the reduction of variations in model indicator scores from up to 13 to no more than 2 orders of magnitude for life cycle impact assessment models concerning human toxicity and freshwater ecotoxicity (Rosenbaum et al., 2008).

4.8 Quantifying environmental interferences of assessed system

This concern is entirely related to scientific understanding of the environmental interferences of the assessed anthropogenic system(s). Environmental interferences are for many types of environmental problems relatively well known and, therefore, not expected to be a great source of uncertainty in indicator scores.

Nevertheless, for environmental interferences comprised of many different stressors, the monitoring of each stressor or class of stressors should be improved for the sake of lowering uncertainties. In cases where the direct measurements of emissions are impractical, emission figures may be derived from solving mass balances.

4.9 Spatial coverage and resolution

This concern is related to scientific understanding and to the assumed preference of active users regarding the spatial information of quantified environmental interferences required by them. With respect to coverage, there is generally no need for an indicator that covers a larger area than the area relevant for the studied environmental interferences. Indicators, such as those employed by HELCOM, tailored to a specific region are likely to give more accurate indicator scores than e.g. global indicators because region specific processes in the impact pathway tend to be better represented in regional models. The representation of reality is generally improved with higher spatial resolution but the cost is a more complicated model that may be less accessible and useful to active users due to the higher spatial resolution required of the quantified environmental interferences. In addition, an appropriate resolution also depends on the spatial heterogeneity of the environmental interference with regard to the fate of pollutants, thresholds and sources of environmental interferences as well as the extent to which corresponding input parameters exist (e.g. ecological boundary values for each spatial unit given by the resolution in question). As the choice of spatial resolution, to a large extent, depends on user preferences that may vary, it might be feasible to make available different versions of the same environmental sustainability indicator reflecting different spatial resolutions.

4.10 Temporal coverage and resolution

This concern is also related to scientific understanding and to assumed preferences of active users. With respect to temporal coverage, it is unlikely that variations in time frame for environmental interferences will lead to large uncertainties in indicator scores. For example, the environmental interferences quantified for the HELCOM indicators only changed around 10% from reference period 1997-2003 to reference period 2008-2010 (HELCOM, 2011). In this case, changing the choice of reference period would thus lead to negligible uncertainty in indicator scores (results not shown). It should, therefore, generally not be a top priority to obtain estimates of environmental interferences from the most recent time period, unless emission patterns of the studies systems are known to change rapidly. Much of the guidance regarding spatial resolution also applies to temporal resolution: The representation of reality generally increases with temporal resolution but the cost may be a more complicated model that may be less accessible and useful to active users due to the higher temporal resolution required of the quantified environmental interferences. Whether the temporal resolution should be e.g. annual, monthly or hourly also depends on the temporal heterogeneity of the environmental interference with regard to fate of pollutants, thresholds and sources of environmental interferences and the extent to which corresponding input parameters exist (e.g. hourly ecological boundary values). In line with the guidance for the previous concern, it might be feasible to make available different versions of the same environmental sustainability indicator reflecting different temporal resolutions (e.g. from hourly to annual).

4.11 Aggregation of indicator scores

This concern is related to value judgement regarding the most appropriate way of establishing a single indicator score across all spatial and temporal units. The sensitivity analysis of section 3.2 showed that this concern has a potentially high contribution to uncertainty in indicator results. This uncertainty can be reduced by excluding the aggregation choice of averaging indicator scores of each spatial unit across all units covered. The rational for the exclusion is that a safe limit that is not exceeded in one location can, in reality, not offset an exceeded safe limit in another location. The same can be said with regard to averaging over temporal units. Aggregations should, therefore, rather be based on averaging, while fixing indicator scores for spatial and temporal units below 1 as 1 or be based on the share of spatial and temporal units exceeded (choice 2 and 3 in Table 1). We stress that it should still be possible for users to access indicator scores for each spatial unit so these may be communicated to decision-makers.

4.12 Allocation

Although the different allocation choices covered in the analysis above did not lead to large differences in indicator score (Figure 3), the concern of allocation has the potential to influence indicator scores strongly due to its root in value judgement. In the effort of reaching some level of consensus on fair allocation to different systems, inspiration can be sought in broadly accepted norms such as the UN’s Universal Declaration of Human Rights. It might be possible to establish the consensus for consumption oriented assessments that carrying capacity should be shared equally amongst the people depending on it. Also the sector specific GHG reduction scenarios of the IPCC, IEA and individual countries’ and municipalities’ climate strategies could act as policy references for how to allocate carrying capacity between products belonging to different sectors and geographic regions. Nevertheless, if an environmental sustainability indicator includes an allocation module, it is recommended that active users can make choices between different allocations to best fit the decision context that the indicator is to support.

In conclusion, for concerns related to scientific understanding, the range of choices may be reduced by only considering those fulfilling scientific standards and by improving these standards through research and international scientific consensus building. For concerns depending on value judgements, the range of choices can be reduced by considerations of established societal norms (e.g. related to risk acceptance or human rights). We believe these means can greatly reduce existing variations in indicator scores that we have demonstrated can be as high as 3 orders of magnitude for the 5 indicator sets compared in this study. We stress that it is neither practically nor theoretically possible to eliminate all uncertainties. It is, therefore, crucial that indicators are transparently designed so users can understand the concrete choices made and, in some cases, have the freedom to make choices of their own, and communicate the reasoning behind and effect of these choices to decision makers.

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Supporting information
Supporting information is available online and contains additional information on the adaptation of the indicators and additional results.

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Supporting information for:

Environmentally sustainable or not? Managing and reducing indicator uncertainties

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1. Allocation of global assimilative capacity and estimations of current interventions for Rockström et al. (2009) and Carpenter & Bennett (2011)

1.1. Allocation of global assimilative capacity

To use the two indicators on the Baltic Sea region case we introduced an allocation module to the indicators (concern 12) and tested three choices for allocating global assimilative capacity to activities within the Baltic Sea based on: population, agricultural area and GDP.

1.1.1. Population

According to HELCOM (2013) the population of the Baltic Sea region was 85.64 million in 2006. When divided by the estimated global population in 2006 of 6.59 billion (UN, 2012) a population based allocation of global assimilative capacity of **1.30%** is calculated.

1.1.2. Cropland area

Cropland is here understood as the sum of arable land and permanent crop land, where the former involve crops that are sown and harvested within the same agricultural year and the latter involves perennial crops. Cropland management often involves fertilizer inputs and since fertilizer is a major source of nitrogen and phosphorous emissions cropland area can be used as an allocation basis. We estimated the cropland area from extraction of a Baltic Sea region land cover map of HELCOM (2015) to be $2.43*10^7$ hectares. When dividing this number by the estimated global cropland area of $1.56*10^9$ha in 2012 (FAO, 2015) an agricultural area based allocation of global assimilative capacity of **1.56%** is calculated.

1.1.3. GDP

According to NIB (2012) the GDP (purchasing-power-parity (PPP) adjusted) of the Baltic Sea region was $1826 billion in 2011. According to NIB (2012) the population of the Baltic Sea region was 57.3 million in 2011. The discrepancy between this estimate and the 2006 estimate of 85.64 million by HELCOM (2013) is thought to mainly be caused by different differences between definitions of the Baltic Sea region, rather than actual differences in population between the years 2006 and 2011. We therefore corrected the GDP number by multiplying with a correction factor defined as the HELCOM (2013) population estimate (85.64 million) divided by the NIB (2012) estimate (57.3 million). This give a corrected GDP for the Baltic Sea region as defined by HELCOM (2013) of $2729 billion in

2011. When dividing this number by the estimated 2011 global GDP (also PPP adjusted) IMF (2014) of $92651 billion a GDP based allocation of global assimilative capacity of **2.95%** is calculated.

1.2. Estimations of current interventions

1.2.1. Rockström et al. (2009), anthropogenic fixation of nitrogen

For the nitrogen AESI of Rockström et al. (2009) the assimilative capacity is expressed as anthropogenic fixation of nitrogen, which includes nitrogen fixated via the Haber–Bosch process for use as inorganic fertilizer and nitrogen fixated by combustion processes. While the Haber–Bosch is an ultimate cause of environmental interventions from nitrogen the spatial distribution of these interventions is a function of the distribution of fertilizer application. We therefore estimate current interventions for this indicator as the sum of nitrogen applied as fertilizer in the Baltic Sea region and the nitrogen fixated via combustion processes in the Baltic Sea region.

Nitrogen applied as fertilizer in the Baltic Sea region in 2012 was estimated from Eurostat (2015). The Eurostat figures cover total fertilizer use in the countries. To estimate the share of this total use that fell within the Baltic Sea region in 2012 we calculated the share of each country's cropland area that fell within the Baltic Sea region by dividing the cropland are of each country falling within the Region by the total agricultural area of each country, both extracted from the GIS data of HELCOM (2015) (see above), and multiplying by the total fertilizer use per country, see Table S1. The fertilizer use of Belarus, Russia and Ukraine was not covered by Eurostat and was therefore approximated by multiplying the average fertilizer application intensity of covered Eastern European countries by cropland areas of these countries falling within the Baltic Sea region. From this we obtained total input of N as fertilizer in the Baltic Sea region in 2012 of **2578kt/year** respectively, around half of which was applied in Poland (see table S1).

Table S1: Estimation of total inputs of N and P as inorganic fertilizer in Baltic Sea region in 2012

Country	**Total N (tons/year)**	**Total P (tons/year)**	**Total cropland area (km^2)**	**Cropland area within Baltic Sea region (km^2)**	**Total N intensity ($tons/km^2/year$)**	**Total P intensity ($tons/km^2/year$)**	**Estimated input of N (kt/year)**	**Estimated input of P (kt/year)**	**Share N (%)**	**Share P (%)**
Estonia	30000	2500	2109	2106	14.2	1.2	30	2	1.16	0.86
Belarus	NA	NA	90297	28894	11.8*	1.1*	342	32	13.27	11.05
Czech Republic	349000	18500	17259	1687	20.2	1.1	34	2	1.32	0.62
Denmark	187000	13000	22382	14325	8.4	0.6	120	8	4.64	2.87
Finland	139000	10500	2037	1901	68.2	5.2	130	10	5.03	3.38
Germany	1640500	108000	128519	14209	12.8	0.8	181	12	7.03	4.12
Latvia	65000	8500	5074	5074	12.8	1.7	65	9	2.52	2.93
Lithuania	147000	15500	19719	19662	7.5	0.8	147	15	5.69	5.33
Norway	94500	8500	7571	312	12.5	1.1	4	0	0.15	0.12
Poland	1094500	162000	122699	122605	8.9	1.3	1094	162	42.42	55.85
Russia	NA	NA	228871	17516	11.8*	1.1*	207	19	8.05	6.70
Slovakia	128000	10500	17214	253	7.4	0.6	2	0	0.07	0.05
Sweden	148000	10500	8047	7471	18.4	1.3	137	10	5.33	3.36
Ukraine	NA	NA	326950	7202	11.8*	1.1*	85	8	3.31	2.75
Total							**2578**	**290**		

*** approximated as average fertilizer application intensity of Eastern European countries**

Emissions of nitrogen oxides (nitrogen fixated via combustion) in 2012 were estimated based on Shamsudheen & Bartnicki (2014), who calculated total emissions for HELCOM Contracting Parties based on EMEP data (Shamsudheen & Bartnicki, 2014).[74] We supplemented these estimates by estimates of non-contracting parties in the Baltic Sea region (Belarus, Czech Republic, Norway, Slovakia and Ukraine) from the same EMEP source (Gauss, Shamsudheen, Benedictow, & Klein, 2014) that Shamsudheen & Bartnicki (2014)used. The obtained emission figures overestimate total emissions within the Baltic Sea region since the territory of each HELCOM country extends beyond the region. To correct for this we multiplied emissions estimates of each country by the share of countries territory that falls within the Baltic Sea region, thus assuming that emissions are spatially homogenous within each country, see Table S2. The total emissions estimate was **470kt/year**, which is around a factor 5 lower than the total input of N (see Table S1) and was dominated by emissions of Poland. Total anthropogenic fixation of nitrogen in the Baltic Sea region was thus estimated as

Table S2: Estimation of total emissions of NO_X from Baltic Sea region

Country	**Area within Baltic Sea region (km^2)**	**Total area (km^2)**	**Total emissions (kT/year)**	**Estimated emissions within Baltic Sea area (kT/year)**
Estonia	44748	45408	9.7	9.6
Belarus	88465	207565	52.1	22.2
Czech Republic	7268	78843	64.3	5.9
Denmark	28984	43503	35.3	23.5
Finland	300349	338398	44.7	39.7
Germany	28231	357472	387.4	30.6
Latvia	64479	64548	10.7	10.7
Lithuania	64900	64931	17.7	17.7
Norway	14882	325701	50.0	2.3
Poland	310573	311788	248.7	247.7
Russia	326498	17098242	901.2	17.2
Slovakia	253	49046	24.7	0.1
Sweden	435970	450040	39.9	38.7
Ukraine	12188	597574	183.8	3.7
Total				**469.6**

[74] Shamsudheen & Bartnicki (2014) also reported emissions of ammonia, but we did not include those here to avoid double counting, since ammonia emission are primarily caused by application of fertilizer, which is already accounted for.

1.2.2. Rockström et al. (2009), phosphorous flow to ocean

For the phosphorous AESI of Rockström et al. (2009) the assimilative capacity is expressed as inflow of

phosphorus to ocean. The latest estimates for phosphorus input to the Baltic Sea are from 2010 HELCOM (2013), but we base our calculation on the annual average estimate of 33.1kt/year for 2008-2010 (HELCOM, 2013). We chose this approach because annual loads can vary quite a lot due to variations of run-off caused by variations in precipitation. The HELCOM estimate only covers contracting parties. We approximated the additional phosphorous input from non-contracting parties by calculating the share of applied phosphorous as fertilizer within the region from contracting parties that was discharged to the Baltic Sea by waterways (8%) and multiplying it with the quantity of phosphorous applied by non-contracting parties within the region (42 kt/year). This resulted in an estimate of 5.3kt for non-contracting parties and a total estimated input of **38.4kt/year**, including atmospheric depositions.

1.2.3. Carpenter & Bennett (2011), phosphorous flow to soil

For estimating current interventions for the phosphorous AESI of Carpenter & Bennett (2011) based on phosphorous flow to soil we used an identical approach as for the estimation of inorganic nitrogen applied as fertilizer for the AESI of Rockström et al. (2009), see Table S1. This resulted in an estimation of **290kt/year.**

1.2.4. Carpenter & Bennett (2011), phosphorous flow to freshwater

The phosphorous flow to freshwater was calculated by applying country specific estimates of phosphorous retention from HELCOM (2011) to the estimated total phosphorous flow to the Baltic Sea, established in S1.2.2. Table S3 shows the calculations for each country. As we did not have estimates of retention for Russia and non-contracting parties we applied the average retention factor of 30.7% for the remaining contracting parties. This resulted in an estimate of phosphorous flow to freshwater within the region of **56.3kt**/year for 2008-2010.

Table S3: Estimation of phosphorous flow to freshwater based on inputs to the Baltic Sea and retention factors

Country	Phosphorous retention, 2006 (%)	Normalized input, 2008-2010 (kt/year)	Phosphorous flow to freshwater, 2008-2010 (kt/year)	Share of total (%)
Denmark	-0.9**	1.7	1.7	3.1
Estonia	50.7	0.6	1.3	2.3
Finland	23	3.2	4.2	7.4
Germany	31.5	0.5	0.8	1.3
Latvia	18	2.8	3.4	6.1
Lithuania	43.9	1.8	3.3	5.8
Poland	46.5	10.7	19.9	35.4
Russia	30.7*	6.3	9.1	16.2
Sweden	33	3.3	4.9	8.8
Non-contracting parties*	30.7*	5.3	7.7	13.6
Total		**36.4**	**56.3**	

*** Average of contracting parties (except Russia) used.**

**** The negative value for Denmark is caused by presently low oxygen levels in the bottom waters of eutrophied lakes that result in the leaching of phosphorus from bottom sediments (HELCOM, 2011).**

2. Control variable choices for the adaptation of HELCOM

Table S4 details the 6 combinations of control variables for the adaptation of HELCOM

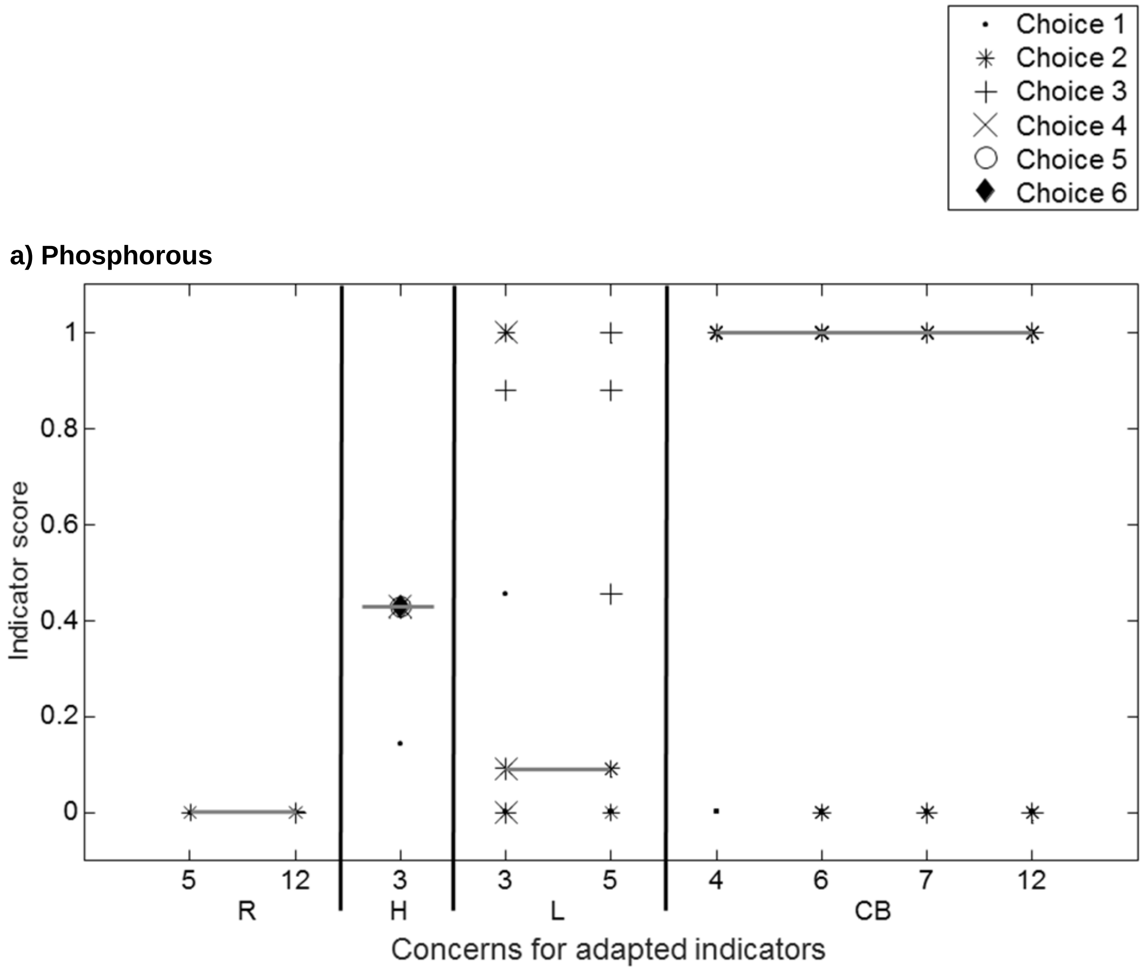
Table S4: 6 choices for control variable combinations in the adaptation of HELCOM

choice 1	choice 2	choice 3	choice 4	choice 5	choice 6
Summer secchi depth, O_2-debt	Summer secchi depth, O_2-debt, winter DIP adjusted according to low N input	Summer secchi depth, O_2-debt, winter DIP adjusted according to present N input	Summer secchi depth, O_2-debt, winter DIP, winter DIN (except 3 sub-basins)	Summer secchi depth, O_2-debt, winter DIP, winter DIN (except 3 sub-basins), HEAT3.0 adjustment (including summer Chl α)	Summer secchi depth, O_2-debt, winter DIP, winter DIN

3. Additional results

Outcomes for all linear combinations of choices involving the choice of expressing aggregated outcomes as shares of spatial units having their assimilative capacity exceeded (choice 3 for concern 11).

a) Phosphorous



b) Nitrogen

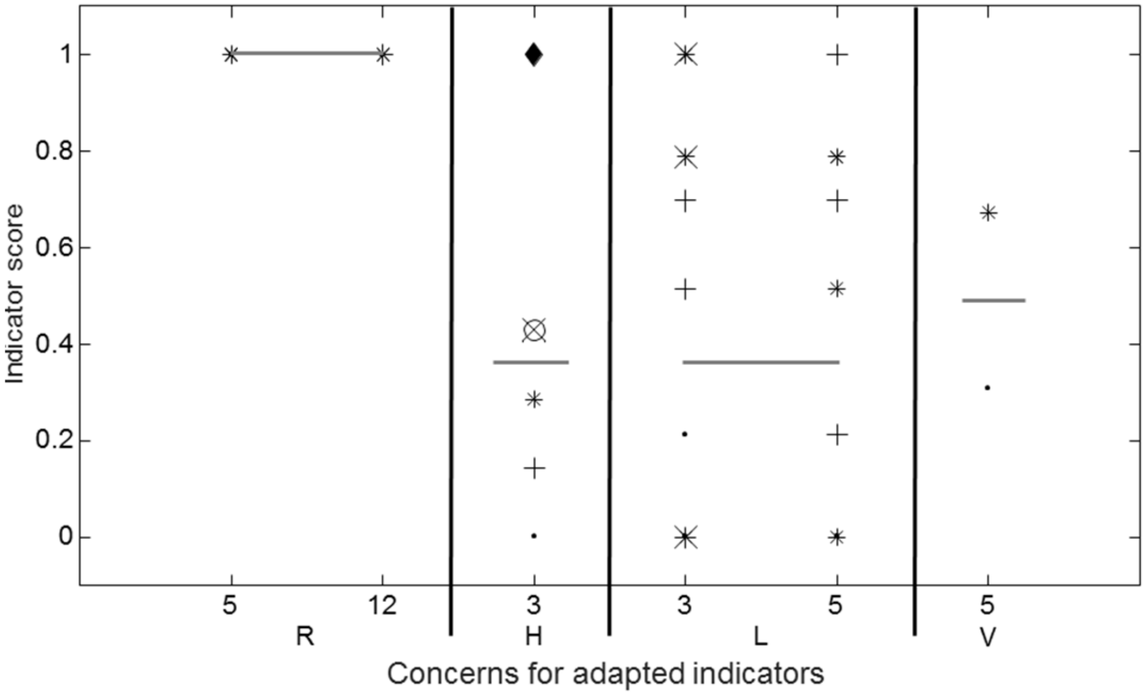


Figure S1: Indicator scores for linear combinations of the adaptation of Rockström et al. (2009) (R), HELCOM (H), Liu et al. (2012) (L), Carpenter & Bennett (2011) (CB) and de Vries et al. (2013) (V) for share of exceeded spatial units as aggregation (concern 11) for phosphorous (a) and nitrogen (b). Horizontal grey lines indicate the median value of linear combinations for each indicator. See Table 2 for specification of choices.

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VI

Strengthening the link between life cycle assessment and indicators for absolute sustainability to support development within planetary boundaries

Bjørn, A., Diamond, D., Owsianiak, M., Verzat, B., & Hauschild, M. Z.

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Strengthening the link between life cycle assessment and indicators for absolute sustainability to support development within planetary boundaries

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The insufficiency of eco-efficiency

Life cycle assessments (LCA) are increasingly used by industry to communicate improvements of environmental performance in a scientifically defendable way. Typically, studies compare new product designs with "last year's model" or a market reference to document that the eco-efficiency of a company's product portfolio is gradually improving or to show that the company is ahead of its competitors in terms of eco-efficiency performance. In both cases the signal to stakeholders is that companies are doing "their share" to foster sustainability. However, while the environmental performance of individual products is being improved, humanity is generally moving further away from a state of environmental sustainability.[1] The reason for this seeming contradiction is that improvements in eco-efficiency are insufficient to offset increasing levels of consumption. For example PricewaterhouseCoopers calculated that the current global eco-efficiency improvement with respect to greenhouse gas (GHG) emission of 0.9% per year needs to increase to 6.2% per year and remain at that level until the year 2100 for emission volumes to be aligned with the IPCC RCP2.6 reduction pathway designed to curb a global temperature increase of 2°C.[2] How can the current LCA practice of assessing environmental performance relative to a reference product be improved to support decisions on the path to environmental sustainability? How can we ensure that LCA is not used to legitimize a business as usual situation of incremental and insufficient eco-efficiency improvements?

Carrying capacity as absolute sustainability reference

To change current practice, LCA indicators need carrying capacity as a reference to compare environmental interventions from a product system to sustainable levels of interventions. Carrying capacities are derived from inherent thresholds in nature's response to, for example, increasing concentrations of pollutants or use of resources. Staying below thresholds is a precondition for environmental sustainability because it safeguards ecosystem services and the biodiversity levels that are required for resilient socio-ecological systems and thus for development within planetary boundaries. With carrying capacity as a reference, LCA may support *absolute environmental sustainability indicators* (AESI). Such indicators are absolute, because carrying capacities are independent of the product system assessed. Initial steps in this direction were recently taken by Bjørn and Hauschild,[3] who developed carrying capacity references for the normalisation step of LCA. These references allow translating an LCA midpoint indicator score to the corresponding fraction of carrying capacity occupied in person equivalents, making it possible to quantify the share of personal carrying capacity taken up, for example, by food consumption or transportation. This type of analysis is similar to ecological footprint analysis where available land is compared to land area needed to supply resource uses and assimilate emissions of product systems.[4] Using LCA combined with carrying capacity based normalisation has an advantage over the ecological footprint method in that it covers a much broader spectrum of environmental interferences, rooted in the strong methodological development activities in the field of life cycle impact assessment, and is linked to comprehensive inventory databases of unit processes.

Carrying capacity entitlement is key

Beyond scientific technicalities of the impact assessment, the question of carrying capacity entitlement is central because a product can only be considered sustainable if it does not exceed the carrying capacity to which it is entitled. Entitlements are, of course, normative due to the diversity of perspectives on what constitutes needs (and wants) in life. A product's entitlement can also depend on the geographical context and can evolve through time. For instance, some may perceive bottled water to be entitled near zero carrying capacity when consumed in a developed country with reliable access to safe tap water. Bottled water is consequently likely to be assessed as unsustainable in this context. In a developing country, however, this assessment could be different because of the common lack of reliable, safe, publically accessible drinking water. The normative nature of entitlement poses a challenge when combined with the science-based approach of LCA. Yet, we are optimistic that some degree of consensus on entitlements could be obtained: Just as UN's Universal Declaration of Human Rights is broadly accepted, we believe that it is possible to agree upon a rule that carrying capacity should be shared equally amongst people, so that all people can potentially meet their needs. The sector specific GHG reduction scenarios of IPCC, IEA and individual nations' and municipalities' climate strategies could also serve as policy references for how to allocate carrying capacity entitlement between products belonging to different sectors and geographic regions.

The road ahead

The timing of developing AESI is certainly ripe. United Nations is currently developing sustainable development goals for the planet, goals that will be calling for sustainability indicators. In 2012, the World Business Council for Sustainable Development (WBCSD), representing 200 large companies with a combined annual revenue of $US 7 trillion, announced that they are working with planetary boundaries researchers to bridge the gap between business and science. Recently the Dutch energy utility Eneco took the first steps in bridging this gap by using the "One Planet Thinking Model", which is based on linking LCA indicators to the planetary boundaries concept. Preliminary results show that Eneco must improve its eco-efficiency (intervention per kWh produced electricity) for the impact categories fossil depletion and climate change by factors of 2 and 15 respectively to be considered environmentally sustainable.[5] While this is a positive example we do not expect that all companies will find it appealing to adopt AESI in stakeholder communication considering the obvious conflict between the dictum of continuous economic growth versus the need to stay within finite carrying capacity entitlements. Yet developing a comprehensive basket of AESI will leave foot-dragging companies with one less excuse for avoiding to face actual sustainability challenges. We believe that modifying the already widely adopted LCA framework from assessing sustainability in relative terms to assessing it in absolute terms can and must play a major role in this development.

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Many indicators quantify environmental impacts of anthropogenic systems, such as products, infrastructure and companies. Most of these indicators are relative, meaning that they are designed to answer questions like "which alternative has the lowest environmental impact?" Often the limitations in such a relative approach are ignored and the best performing alternative is referred to as "sustainable." This means that anything essentially can be presented as sustainable when compared to alternatives that have a worse environmental indicator score. This is highly problematic, considering that the generally declining state of the environment tells us that efforts targeting environmental sustainability are collectively insufficient. New types of indicators are therefore needed that can evaluate whether something is environmentally sustainable in an absolute sense or how much its environmental impacts must decrease for this to be true. In this thesis I evaluate existing indicators of absolute environmental sustainability and their use in industry. In addition, I propose and demonstrate different ways of integrating measures of carrying capacity of ecosystems as references of environmental sustainability in the indicator framework of Life Cycle Assessment (LCA). This integration allows the well-established and widely used LCA tool to measure absolute environmental sustainability.

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