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Abbreviations

ADP	Abiotic depletion potential
AEP	Aquatic ecotoxicity potential
BDP	Biotic depletion potential
BSI	British Standards Institution
CC	Carbon content
CF_{class}	Classical carbon footprint
CF_{clim}	Climate-related carbon footprint
CFI	Composite footprint index
CO_2	Carbon dioxide
CSR	Carbon sequestration rate
EC	European Commission
EFA	Ecological footprint analysis
EIA	Environmental impact assessment
EIOA	Environmental input–output analysis
EPA	Environmental Protection Agency
EQF	Equivalence factor
ESA	Environmental sustainability assessment
ESD	Environmental sustainability distance
ESR	Environmental sustainability ratio
ESRI	Index of environmental sustainability ratio
EU	European Union
EW-MFA	Economy-wide material flow accounts
FAO	Food and Agriculture Organization of the United Nations
F–B ESA	Footprint–boundary environmental sustainability assessment
GDP	Gross Domestic Product
GFN	Global Footprint Network
GHG	Greenhouse gas
GTAP	Global Trade Analysis Project
GWP	Global warming potential
ICNW	Inventory-characterization-normalization-weighting
IOA	Input–output analysis
IPCC	Intergovernmental Panel on Climate Change
IPOF	Impact-oriented footprint
ISO	International Organization for Standardization
IVOF	Inventory-oriented footprint
LB	Local boundary
LCA	Life cycle assessment

LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LDF	Land disturbance factor
LF_{class}	Classical land footprint
$LF_{disturb}$	Disturbance-related land footprint
LOP	Land occupation potential
MEA	Millennium ecosystem assessment
MFA	Material flow analysis
MF_{class}	Classical material footprint
MF_{scarc}	Scarcity-related material footprint
MRIO	Multi-regional input–output
NB	National boundary
NFA	National Footprint Accounts
NPP	Net Primary Productivity
ODP	Ozone depletion potential
OEF	Organization Environmental Footprint
PBF	Planetary boundaries framework
PCF	Product carbon footprint
PD	Population density
PEF	Product Environmental Footprint
PLUM	Product Land Use Matrix
PRB	Population Reference Bureau
RB	Reginal boundary
RDF	Resource depletion footprin
RDP	Resource depletion potential
RWR	Renewable water resources
Sb	Antimony
SFA	Substance flow analysis
TAP	Terrestrial acidification potential
UNEP	United Nations Environment Programme
WDF	Water depletion footprint
WF_{class}	Classical water footprint
WFN	Water Footprint Network
WF_{scarc}	Scarcity-related water footprint
WTA	Withdrawal-to-availability
WWF	World Wildlife Fund
ZSL	Zoological Society of London

Chapter 1

General introduction

1.1. The origin of environmental footprints

Significant changes to the Earth's environment have been witnessed over the past decades, with consequences that are undesirable or even disastrous for humanity (Barnosky et al., 2012; Scheffer et al., 2009). This has led to a transition from the Holocene—a geological epoch that spanned a long period of time—to the Anthropocene—a new era in which human disturbance is greatly eroding the stability and resilience of the Earth system (Crutzen, 2002; Steffen et al., 2011). In striving to preserve the planet as a pleasant place for living and as a source of human welfare in this challenging era, there is a great need for novel approaches to modeling anthropogenic effects that are the key to identifying the driving forces of contemporary environmental change.

Ecological footprint analysis (EFA) was originally introduced and advocated to evaluate the effects of anthropogenic activities on urban sustainability (Rees, 1992). It compiled, on an area basis, the inputs of biological resources and the outputs of carbon emissions (i.e., the ecological footprint) and compared to the regenerative and assimilative capacity of urban ecosystems (i.e., the biocapacity), indicating whether or not the situation remains sustainable (Rees, 1997). At the human–environment interface, six types of land use on which human disturbance is most likely to place have been taken into account, including cropland, grassland, fishing ground, woodland, built-up land, and carbon uptake land. These relate to six ecosystem services respectively: plant-based food production, animal-based food production, fish-based food production, timber production, living space supply, and carbon dioxide (CO_2) sequestration.

In view of the success in raising public awareness of environmental issues and in evoking effective policy actions, Wackernagel and Rees (1997) have implemented an extension to the methodological application of the EFA, particularly pinpointing nation-wide economy. In the latest edition of the National Footprint Accounts (NFA), the ecological footprint is defined as the area of biologically productive space required to produce the resources consumed and to absorb the waste generated, considering the prevailing technology and resources management practices (Borucke et al., 2013). The biocapacity, which can be probably traced back to attempts to quantify human carrying capacity (e.g., Cohen, 1995; Ehrlich, 1982), is conceived in such a way that it can provide a region-specific threshold value for the ecological footprint of a given population. The comparison of the ecological footprint and biocapacity makes it possible to contrast sustainable and unsustainable consumption or production in an explicit manner.

1.2. The development of environmental footprints

Despite the worldwide popularity gained in the past two decades, the EFA is found incapable of capturing all aspects of human disturbance to the biosphere (Goldfinger et al., 2014), let alone to the atmosphere and hydrosphere. For this reason, a growing number of footprint-style indicators have been developed in order to complement the EFA in different dimensions. Examples in the environmental domain include the water footprint (Hoekstra and Hung, 2002), the energy footprint (Stöglehner, 2003), the carbon footprint (Wiedmann and Minx, 2008), the chemical footprint (Hitchcock et al., 2011), the phosphorus footprint (Wang et al., 2011), the biodiversity footprint (Lenzen et al., 2012), the nitrogen footprint (Leach et al., 2012), the land footprint (Weinzettel et al., 2013), the material footprint (Wiedmann et al., 2015), the resource footprint (Huysman et al., 2014), and so on.

In keeping with the ongoing expansion of the footprint family (Fang et al., 2014; Galli et al., 2012), the underlying methodologies are currently undergoing rapid development. In addition to the NFA, life cycle assessment (LCA) (Weidema et al., 2008), input–output analysis (IOA) (Hertwich and Peters, 2009), material flow analysis (MFA) (Schoer et al., 2012) and emergy-based (Zhao et al., 2005) and exergy-based methods (Chen and Chen, 2007) have proved useful in calculating different environmental footprints at scales ranging from single products, processes, organizations, industries, nations, even to the whole human economy. Recently emerging hybrid approaches that take advantage of both LCA and IOA have shown great potential for meso-level footprint accounting for which well-established methods are lacking. As a whole, the choices of appropriate methods are playing a central role in quantitative footprint studies.

The footprint family concept has attracted considerable interest as it offers the scientific community an opportunity to achieve simultaneous measurement of various environmental footprints with implications for trade-off issues (e.g., De Meester et al., 2011; Ridoutt et al., 2014). By structuring a specified footprint family based on LCA (De Benedetto and Klemeš, 2009) or on multi-regional input–output (MRIO) models (Ewing et al., 2012), for instance, current accounts for selected footprints have been conceptually integrated into single unified frameworks that would allow for greater transparency and consistency. A further step towards policy-relevant research is to develop an integrated footprint family that is supposed to encompass the complexity of some highly heterogeneous environmental issues, such as climate change (carbon footprint), water use (water footprint), land use (land footprint), and material use (material footprint).

1.3. Debates on environmental footprints

Since the first emergence of the ecological footprint, the term "footprint" has stimulated scientific debates, representing important steps in the ongoing discourse on sustainability (e.g., Hoekstra and Wiedmann, 2014; Kitzes et al., 2009). Of particular concern is what actually counts as a footprint. Efforts have been made to lay the foundation for a widely accepted footprint definition (e.g., Hammond, 2007; Pelletier et al., 2014). At least five competing criteria are evidently available in the literature for definition of footprints:

a) Footprints are indicators that express their results in an area-based metric unit. Clearly the ecological footprint is the most obvious example of this. Many other well-known indicators, like the carbon footprint and the water footprint, would fall outside the scope by this definition.

b) Footprints are indicators that are intended for easy communication of results to the general public, policy makers, or other decision-makers. Indicators that aim to inform scientists and engineers, for instance, would in that case not be considered as a footprint. An example of such indicators is the eutrophication potential.

c) Footprints are indicators that include a supply chain or that even take a full life cycle perspective, i.e., indicators that only look at impacts of a country, company, for instance, without including at least the upstream impacts would be disqualified from the footprint family with this categorization.

d) Footprints are indicators that apply to the macro, economy-wide scale, in contrast to, for instance, LCA that studies one functional unit (e.g., kg) of a product.

e) Footprints are indicators that have the word "footprint" in their name. This criterion is based on an accidental nomenclature but many footprint users implicitly stick to. It excludes, for instance, those studies that account for water footprint but refer to this as virtual water or embodied water.

It is necessary to recognize further that a single footprint indicator may differ from another that has the same name but a different logic. For instance, the difference between volumetric water footprint and scarcity-based water footprint has been a subject of intense debate (Berger and Finkbeiner, 2013). This highlights the importance of categorization in systematizing the footprint family—a topic that has grown in interest in recent years (Čuček et al., 2012). In addition, while integrating different footprint results, by using weighting factors, into a single composite metric is appealing from a user-perspective, the scientific robustness and certainty of such a step are disputable, as any weighting schemes inevitably involve subjective judgments and are therefore prone to a lot of uncertainty (Huppes et al., 2012). The core of this challenge is the trade-offs between aggregate and disaggregate measures of environmental footprints (Fang and Heijungs, 2015).

1.4. The relation to life cycle assessment

Discussions on the close connection between environmental footprints and LCA have spawned an enormous literature (e.g., Hoekstra, 2015; Lenzen, 2014). A large number of pilot studies have provided concrete evidence of how LCA frameworks, in particular life cycle impact assessment (LCIA), allow various footprint indicators to be suited for environmental impact assessment (EIA) (e.g., Castellani and Sala, 2012; Huijbregts et al., 2008). Given that the footprint community has indeed learned and borrowed much from LCA knowledge, some LCA experts argue that all footprint accounts should be exclusively on an LCA basis (Ridoutt et al., 2015), in the hope that the much broader appeal of footprint indicators could, in turn, facilitate the diffusion of life cycle thinking, particularly for non-LCA experts (Ridoutt and Pfister, 2013; Weidema et al., 2008). In that case, LCA is meant to replace or supersede all other footprinting methods and to be the "golden standard".

However, the relationship between environmental footprints and LCA is not as simple as it may seem. While being similar in many key elements, environmental footprints differ from LCA in that they can be operationalized in contexts where there is no clear life cycle or even without an LCA. This can be exemplified by the case of the NFA calculations, where data gaps constitute a major challenge to the use of LCA for nation-wide economy (Hellweg and Milà i Canals, 2014). Moreover, there are certain important types of issues for which environmental footprints are desirable while an LCA is not or only partially suitable, as proved by the risk of double counting in large-scale LCA (Lenzen et al., 2007), and vice versa, some typical impact categories (e.g., ozone depletion, ionizing radiation) are out of the scope of the footprint family in its present form. These discrepancies observed suggest that environmental footprints may preferably have a different orientation from LCA.

1.5. The relation to planetary boundaries

With the latest scientific knowledge, the planetary boundaries framework (PBF) defines the global-scale safe operating space for humanity by determining the difference between the current and threshold values for several environmental issues (Rockström et al., 2009; Steffen et al., 2015). The PBF, in this sense, has much in common with the EFA because both of them are well suited for monitoring the extent to which humanity is approaching or exceeding biophysical limits. Yet, this does not mean that there is no difference between the PBF and the EFA. Their differing optimal scales present one example, in which the former is primarily limited to the global scale with the aim of supporting global sustainability goals and pathways, whereas the latter has a far wider range of applications and thus can be applied to environmental sustainability assessment (ESA) at sub-global scales (e.g., cities, nations).

In contrast to the ecological footprint, many footprint indicators, especially those based on LCA, do not include a comparison to any reference conditions, although this is increasingly being perceived as essential for ESA (e.g., Hoekstra, 2015; Moldan et al., 2012). For this reason, allocating planetary boundaries to the national- or regional-scale shares in comparison to environmental footprints may be a novel way to enhance the policy relevance of footprint studies (Fang et al., 2015). A main challenge arises from the fact that not all environmental issues are likely to show explicit threshold behaviors at sub-global scales. Meanwhile, lots of well-grounded footprint models would allow the PBF to have more accurate and reliable estimates of environmental pressures or impacts due to human activities at multiple scales. All this calls for a deeper understanding of the synergies between environmental footprints and planetary boundaries.

1.6. Research questions

The aims of the thesis are to provide novel insights into the ongoing discourse on environmental footprints and to bring clarity to a number of important theoretical and methodological issues which pose substantial barriers to the development of footprint indicators. On the basis of the brief introduction to the background, we identify the following research questions that will be answered in this thesis:

RQ1: Does it make sense to bring together different environmental footprints into a unified framework?

RQ2: How to make use of a selection of environmental footprints to constitute a truly integrated footprint family?

RQ3: Is life cycle assessment a necessity for accounting for environmental footprints?

RQ4: What are the complementarities of environmental footprints and planetary boundaries?

RQ5: How to allocate planetary boundaries to nations and how does this relate to nation-specific environmental sustainability assessment?

1.7. Outline of the thesis

In accordance to the research questions presented, this work of thesis comprises five thematic chapters (**Chapter 2–6**), together with an introductory chapter (**Chapter 1**) and a concluding chapter (**Chapter 7**), as illustrated in Figure 1.1.

Chapter 2 develops a conceptual framework for a footprint family that consists of the ecological, energy, carbon and water footprints. A comparative analysis of the four environmental footprints is performed, with emphases on their characteristics in some key aspects. By evaluating the performance of the footprint family on data availability, coverage complementarity, methodological consistency and policy relevance, the four footprint indicators are found to be complementary in assessing human pressures or impacts associated with natural resource extractions and hazardous emissions. The present footprint family captures a broad spectrum of sustainability issues and is thus able to offer policy makers a more complete picture of human-induced environmental change particularly at the national level than single footprints.

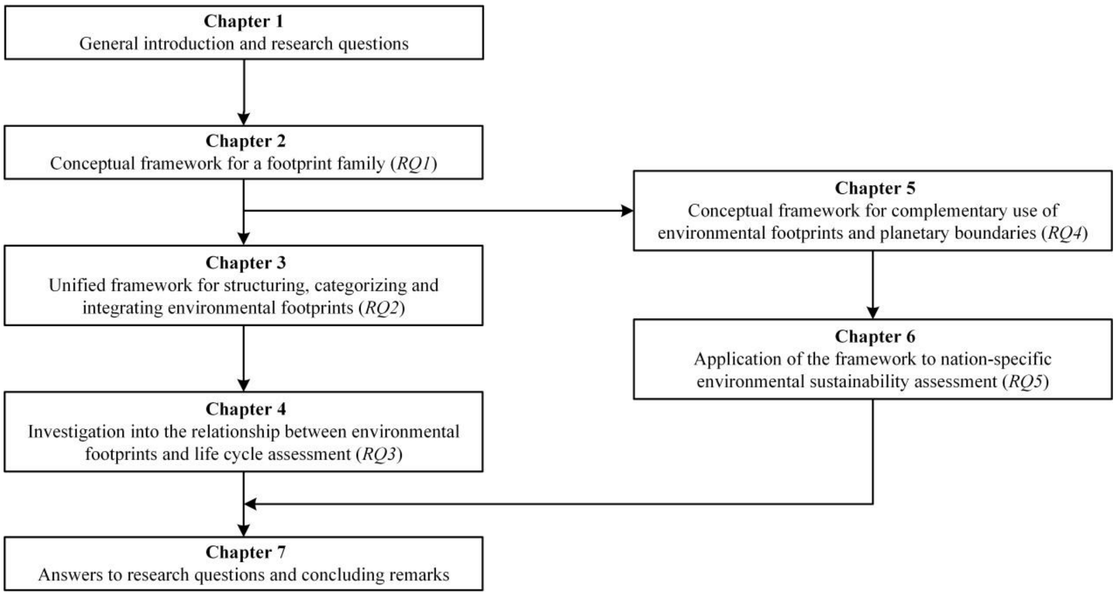

Figure 1.1. Outline of the thesis.

Chapter 3 investigates the roles of inventory schemes and impact characterization schemes in shaping footprint indicators, and concludes that within a single footprint, environmental exchange (extractions and emissions) is addressed either at the inventory level or at the impact assessment level. As such, a two-category framework is proposed, whereby existing environmental footprints can be classified into the inventory-oriented category and the impact-oriented category. While both categories have been found simultaneously in each of the carbon, water, land and material footprints in the literature, a truly integrated footprint family could be achieved only if all environmental footprints involved are impact-oriented. A unified framework for characterization, normalization and weighting of the impact-oriented footprints is established, with the aim of assisting policy makers in modeling the overall environmental impacts of single products, organizations, nations, or even the whole human economy.

Chapter 4 explores the tangled relationship between environmental footprints and LCA from different angles. On the one hand, footprint indicators could benefit from the use of LCIA elements. The important contribution of life cycle characterization modeling to the scientific foundation of the carbon footprint is discussed. With examples of the carbon and material footprints, it is strongly evident that the procedures for inventory aggregation can be improved by substitution of science-based characterization factors for arbitrary weighting factors. On the other hand, an analysis of several limitations of LCA in footprint accounting is conducted. It is demonstrated that narrowing environmental footprints down to an LCA context could create blind spots, where either inventory analysis and impact characterization are difficult to handle due to lack of data, or double counting of impacts occurs due to the inherent limits of LCA at the meso level.

Chapter 5 uncovers the complementary linkages between environmental footprints and planetary boundaries in support of ESA. By presenting a set of consensus-based estimates of the regenerative and absorptive capacity on the global scale, the PBF is found able to benchmark environmental footprints against reference conditions and, in reverse, many well-grounded footprinting methods could provide the PBF with more accurate and reliable estimates of contemporary anthropogenic interference. A framework for the complementary use of environmental footprints and planetary boundaries is therefore proposed, where sustainability gap is referred to as means to understand the difference between current magnitudes of human disturbance and finite biophysical thresholds. The footprint–boundary (F–B) ESA framework makes sense as it represents an important shift in focus, from EIA to ESA.

Chapter 6 conducts an empirical analysis of the F–B ESA framework, with a particular focus on its application at the national level. By using the latest datasets available, the planetary boundaries for carbon emissions, water use and land use are allocated to 28 selected countries in comparison to the respective national environmental footprints. The environmental sustainability ratio (ESR)—an internationally comparable indicator that communicates the sustainability gap in relative terms—allows one to map the transgression or reserve of nation-specific environmental boundaries for the 28 countries, visualizing how far countries are from their respective environmental boundaries. Multiple regression analysis shows that the worldwide unsustainability of carbon emissions is largely driven by economic development, while resource endowments play a central role in explaining national performance on water and land use.

Chapter 7 presents a synthesis of the answers to the research questions, followed by a general discussion and a research agenda for future work.

References

Barnosky, A. D., Hadly, E. A., Bascompte, J., Berlow, E. L., Brown, J. H., Fortelius, M., Getz, W. M., Harte, J., Hastings, A., Marquet, P. A., Martinez, N. D., Mooers, A., Roopnarine, P., Vermeij, G., Williams, J. W., Gillespie, R., Kitzes, J., Marshall, C., Matzke, N., Mindell, D. P., Revilla, E., Smith, A. B., 2012. Approaching a state shift in Earth's biosphere. Nature 486, 52–58.

Berger, M., Finkbeiner, M., 2013. Methodological challenges in volumetric and impact-oriented water footprints. Journal of Industrial Ecology 17, 79–89.

Borucke, M., Moore, D., Cranston, G., Gracey, K., Iha, K., Larson, J., Lazarus, E., Morales, J. C., Wackernagel, M., Galli, A., 2013. Accounting for demand and supply of the biosphere's regenerative capacity: the National Footprint Accounts' underlying methodology and framework. Ecological Indicators 24, 518–533.

Castellani, V., Sala, S., 2012. Ecological Footprint and Life Cycle Assessment in the sustainability assessment of tourism activities. Ecological Indicators 16, 135–147.

Chen, B., Chen, G. Q., 2007. Modified ecological footprint accounting and analysis based on embodied exergy—a case study of the Chinese society 1981–2001. Ecological Economics 61, 355–376.

Cohen, J. E., 1995. Population growth and Earth's human carrying capacity. Science 269, 341–346.

Crutzen, P. J., 2002. Geology of mankind. Nature 415, 23–23.

Čuček, L., Klemeš, J. J., Kravanja, Z., 2012. A review of Footprint analysis tools for monitoring impacts on sustainability. Journal of Cleaner Production 34, 9–20.

De Benedetto, L., Klemeš, J., 2009. The Environmental Performance Strategy Map: an integrated LCA approach to support the strategic decision making process. Journal of Cleaner Production 17, 900–906.

De Meester, S., Callewaert, C., De Mol, E., Van Langenhove, H., Dewulf, J., 2011. The resource footprint of biobased products: a key issue in the sustainable development of biorefineries. Biofuels, Bioproducts and Biorefining 5, 570–580.

Ehrlich, P. R., 1982. Human carrying capacity, extinctions, and nature reserves. Bioscience 32, 331–333.

Ewing, B. R., Hawkins, T. R., Wiedmann, T. O., Galli, A., Ertug Ercin, A., Weinzettel, J., Steen-Olsen, K., 2012. Integrating ecological and water footprint accounting in a multi-regional input–output framework. Ecological Indicators 23, 1–8.

Fang, K., Heijungs, R., 2015. Investigating the inventory and characterization aspects of footprinting methods: lessons for the classification and integration of footprints. Journal of Cleaner Production 108, 1028–1036.

Fang, K., Heijungs, R., De Snoo, G. R., 2014. Theoretical exploration for the combination of the ecological, energy, carbon, and water footprints: overview of a footprint family. Ecological Indicators 36, 508–518.

Fang, K., Heijungs, R., De Snoo, G. R., 2015. Understanding the complementary linkages between environmental footprints and planetary boundaries in a footprint–boundary environmental sustainability assessment framework. Ecological Economics 114, 218–226.

Galli, A., Wiedmann, T., Ercin, E., Knoblauch, D., Ewing, E, Giljum, S., 2012. Integrating ecological, carbon and water footprint into a "Footprint Family" of indicators: definition and role in tracking human pressure on the planet. Ecological Indicators 16, 100–112.

Goldfinger, S., Wackernagel, M., Galli, A., Lazarus, E., Lin, D., 2014. Footprint facts and fallacies: a response to Giampietro and Saltelli (2014) "Footprints to Nowhere". Ecological Indicators 46, 622–632.

Hammond, G., 2007. Time to give due weight to the "carbon footprint" issue. Nature 445, 256–256.

Hellweg, S., Milà i Canals, L., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science 344, 1109–1113.

Hertwich, E. G., Peters, G. P., 2009. Carbon footprint of nations: A global, trade-linked analysis. Environmental Science & Technology 43, 6414–6420.

Hitchcock, K., Panko, J., Scott, P., 2012. Incorporating chemical footprint reporting into social responsibility reporting. Integrated Environmental Assessment and Management 8, 386–388.

Hoekstra, A. Y., 2015. The sustainability of a single activity, production process or product. Ecological Indicators 57, 82–84.

Hoekstra, A. Y., Hung, P. Q., 2002. Virtual Water Trade: A Quantification of Virtual Water Flows between Nations in Relation to International Crop Trade. Value of Water Research Report Series

(No. 11), UNESCO-IHE Institute for Water Education, Delft, The Netherlands.

Hoekstra, A. Y., Wiedmann, T. O., 2014. Humanity's unsustainable environmental footprint. Science 344, 1114–1117.

Huijbregts, M. A. J, Hellweg, S., Frischknecht, R., Hungerbühler, K., Hendriks, A. J., 2008. Ecological footprint accounting in the life cycle assessment of products. Ecological Economics 64, 798–807.

Huppes, G., Van Oers, L., Pretato, U., Pennington, D. W., 2012. Weighting environmental effects: analytic survey with operational evaluation methods and a meta-method. The International Journal of Life Cycle Assessment 17, 876–891.

Huysman, S., Schaubroeck, T., Dewulf, J., 2014. Quantification of spatially differentiated resource footprints for products and services through a macro-economic and thermodynamic approach. Environmental Science & Technology 48, 9709–9716.

Kitzes, J., Galli, A., Bagliani, M., Barrett, J., Dige, G., Ede, S., Erb, K., Giljum, S., Haberl, H., Hails, C., Jolia-Ferrier, L., Jungwirth, S., Lenzen, M., Lewis, K., Loh, J., Marchettini, N., Messinger, H., Milne, K., Moles, R., Monfreda, C., Moran, D., Nakano, K., Pyhälä, A., Rees, W., Simmons, C., Wackernagel, M., Wada, Y., Walsh, C., Wiedmann, T., 2009. A research agenda for improving national ecological footprint accounts. Ecological Economics 68, 1991–2007.

Leach, A. M., Galloway, J. N., Bleeker, A., Erisman, J. W., Kohn, R., Kitzes, J., 2012. A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. Environmental Development 1, 40–66.

Lenzen, M., 2014. An outlook into a possible future of footprint research. Journal of Industrial Ecology 18, 4–6.

Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L., Geschke, A., 2012. International trade drives biodiversity threats in developing nations. Nature 486, 109–112.

Lenzen, M., Murray, J., Sack, F., Wiedmann., T., 2007. Shared producer and consumer responsibility—theory and practice. Ecological Economics 61, 27–42.

Moldan, B., Janoušková, S., Hák, T., 2012. How to understand and measure environmental sustainability: Indicators and targets. Ecological Indicators 17, 4–13.

Pelletier, N., Maas, R., Goralczyk, M., Wolf, M. A., 2014. Conceptual basis for development of the European Sustainability Footprint. Environmental Development 9, 12–23.

Rees, W. E., 1992. Ecological footprint and appropriated carrying capacity: What urban economics leaves out. Environment and Urbanization 4, 121–130.

Rees, W. E., 1997. Urban ecosystems: the human dimension. Urban Ecosystems 1, 63–75.

Ridoutt, B., Fantke, P., Pfister, S., Bare, J., Boulay, A.-M., Cherubini, F., Frischknecht, R., Hauschild, M., Hellweg, S., Henderson, A., Jolliet, O., Levasseur, A., Margni, M., McKone, T., Michelsen, O., Milà i Canals, L., Page, G., Pant, R., Raugei, M., Sala, S., Saouter, E., Verones, F., Wiedmann, T., 2015. Making sense of the minefield of footprint indicators. Environmental Science & Technology 49, 2601–2603.

Ridoutt, B. G., Page, G., Opie, K., Huang, J., Bellotti, W., 2014. Carbon, water and land use footprints of beef cattle production systems in southern Australia. Journal of Cleaner Production 73, 24–30.

Ridoutt, B. G., Pfister, S., 2013. Towards an integrated family of footprint indicators. Journal of Industrial Ecology 17, 337–339.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F. S., Lambin, E. F., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., De Wit, C. A., Hughes, T., Van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P. K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R. W., Fabry, V. J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J. A., 2009. A safe operating space for humanity. Nature 461, 472–475.

Scheffer, M., Bascompte, J., Brock, W. A., Brovkin, V., Carpenter, S. R., Dakos, V., Held, H., Van Nes, E. H., Rietkerk, M., Sugihara, G., 2009. Early-warning signals for critical transitions. Nature 461, 53–59.

Schoer, K., Weinzettel, J., Kovanda, J., Giegrich, J., Lauwigi, C., 2012. Raw material consumption of the European Union – Concept, calculation method, and results. Environmental Science & Technology 46, 8903–8909.

Steffen, W., Persson, Å., Deutsch, L., Zalasiewicz, J., Williams, M., Richardson, K., Crumley, C., Crutzen, P., Folke, C., Gordon, L., Molina, M., Ramanathan, V., Rockström, J., Scheffer, M., Schellnhuber, H. J., Svedin, U., 2011. The Anthropocene: From global change to planetary stewardship. Ambio 40, 739–761.

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S. R., De Vries, W., De Wit, C. A., Folke, C, Gerten, D., Heinke, J., Mace, G. M., Persson, L. M., Ramanathan, V., Reyers, B., Sörlin, S. Planetary boundaries: Guiding human development on a changing planet. Science 2015, 347, DOI: 10.1126/science.1259855.

Stöglehner, G., 2003. Ecological footprint — a tool for assessing sustainable energy supplies. Journal of Cleaner Production 11, 267–277.

Wackernagel, M., Rees, W. E., 1997. Perceptual and structural barriers to investing in natural capital: Economics from an ecological footprint perspective. Ecological Economics 20, 3–24.

Wang, F., Sims, J. T., Ma, L., Ma, W., Dou, Z., Zhang, F., 2011. The phosphorus footprint of China's food chain: implications for food security, natural resource management, and environmental quality. Journal of Environmental Quality 40, 1081–1089.

Weidema, B. P., Thrane, M., Christensen, P., Schmidt, J., Løkke. S., 2008. Carbon footprint: A catalyst for life cycle assessment. Journal of Industrial Ecology 12, 3–6.

Weinzettel, J., Hertwich, E. G., Peters, G. P., Steen-Olsen, K., Galli, A., 2013. Affluence drives the global displacement of land use. Global Environmental Change 23, 433–438.

Wiedmann, T., Minx, J., 2008. A definition of 'carbon footprint'. In: Pertsova, C. C. (Ed.), Ecological Economics Research Trends. Nova Science Publishers, Hauppauge, New York, USA, pp. 1–11.

Wiedmann, T. O., Schandl, H., Lenzen, M., Moran, D., Suh, S., West, J., Kanemoto, K., 2015. The material footprint of nations. Proceedings of the National Academy of Sciences of the United States of America 112, 6271–6276.

Zhao, S., Li, Z., Li, W., 2005. A modified method of ecological footprint calculation and its application. Ecological Modelling 185, 65–75.

Chapter 2

Theoretical exploration for the combination of the ecological, energy, carbon, and water footprints

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Abstract

Over the past two decades, a continuously expanding list of footprint-style indicators has been introduced to the scientific community with the aim of raising public awareness of how humanity exerts pressures on the environment. A deeper understanding of the connections and interactions between different environmental footprints is required in an attempt to support policy makers in the measurement and choice of environmental impact mitigation strategies. Combining a selection of footprints that address different aspects of environmental issues into an integrated system is, therefore, a natural step. This chapter starts with the idea of developing a footprint family from which most important footprints can be compared and integrated. On the basis of literature review in related fields, the ecological, energy, carbon, and water footprints are employed as selected indicators to define a footprint family. A brief survey is presented to provide background information on each of the footprints with an emphasis on their main characteristics in a comparative sense. Although the four footprints differ in more aspects than only in the impacts that are addressed, the footprint family has proved effective in making use of them in a complementary way. We evaluate the performance of the footprint family in terms of data availability, coverage complementarity, methodological consistency, and policy relevance and propose solutions and suggestions for further improvement. The key conclusions are that the footprint family, which captures a broad spectrum of sustainability issues, is able to offer a more complete picture of environmental complexity for policy makers than single footprints and, in particular, in national-level studies. The research provides new insights into the distinction between environmental impact assessment and sustainability evaluation, properly serving as a reference for multidisciplinary efforts in estimating planetary boundaries for global sustainability.

2.1. Introduction

Over the past decades, our Earth has witnessed a significant shift from local environmental issues to global environmental change associated with an irreversible decline in natural capital stocks and ecosystem services on a global scale (Oosthoek and Gills, 2005). In striving to monitor the pressures humanity exerts on the environment, an integrated system where different impact categories can be measured through a set of appropriate indicators is needed (Giljum et al., 2011). The indicators of environmental footprints have the potential to constitute a series of integrated systems with the purpose of providing a complete picture of environmental complexity (Ridoutt and Pfister, 2013).

The concept of "footprint" originates from the idea of ecological footprint which was formally introduced to the scientific community in the 1990s (Rees, 1992, 1996; Wackernagel and Rees, 1996, 1997; Wackernagel et al., 1999a, 1999b). Since then, many different footprint-style indicators have been created and became complementary to the ecological footprint during the last two decades including the energy footprint (Wackernagel and Rees, 1996), the water footprint (Hoekstra and Hung, 2002), the emergy footprint (Zhao et al., 2005), the exergy footprint (Chen and Chen, 2007), the carbon footprint (Wiedmann and Minx, 2008), the biodiversity footprint (Yaap et al., 2010), the chemical footprint (Panko and Hitchcock, 2011), the phosphorus footprint (Wang et al., 2011), the nitrogen footprint (Leach et al., 2012), and so on.

Nowadays, environmental footprints have become colloquial and ubiquitous for researchers, consultants and policy makers, and the implications for sustainability and human well-being have been investigated from different perspectives with an increasing interest in similarities, differences, and interactions between some selected footprints. Nevertheless, there is not yet a completely satisfactory and generally accepted footprint that can solely represent the overall impacts of human activities as the "golden standard" indicator (Huijbregts et al., 2010; Rees, 2002). Therefore, it seems to be a natural step to move toward an integrated system of footprint indicators. Following Galli et al. (2012), we refer to this as, namely, the "footprint family". The concept of a footprint family has only been preliminarily applied in that a very limited number of papers have dealt with it. Further research is thus required to improve transparency, consistency and scientific robustness of this topic.

The aim of this chapter is to further operationalize the footprint family concept. To that end, it starts from a review of the existing literature on the combination of different environmental footprints. We then elaborate on a specific footprint family with the most important, potential members and present a brief survey of those footprints with reference to their definitions, developments, and applications. The main characteristics of each footprint are summarized through a comparison of key issues with particular emphasis on methodological options at different scales. This is followed by a performance evaluation of the footprint family in different respects. The remainder of this chapter proposes suggestions for improving the footprint family and draws conclusions.

2.2. Review of the literature on combining environmental footprints

This section is intended to provide background on the criterion for selecting footprints, not as a complete review. The term "footprint family" was first advocated simultaneously and independently by Giljum et al. (2008) and Stoeglehner and Narodoslawsky (2008). Subsequently, a landmark work on this topic is being undertaken by the OPEN: European Union (EU) Project within the Seventh Framework Program, integrating the ecological, carbon, and water footprints into a footprint family in collaboration with an environmentally extended multiregional input–output (MRIO) model (Galli et al., 2012, 2013).

Some other studies have discussed similar topics without mentioning the term "footprint family". For instance, De Benedetto and Klemeš (2009) designed a composite footprint indicator as a single measure for the sustainability of a given option. Niccolucci et al. (2010) developed an integrated footprint-based approach for environmental labeling of products. Herva et al. (2011) reviewed a series of environmental indicators and proposed the ecological and carbon footprints to be the most appealing indicators for enterprises. Čuček et al. (2012a) presented a comprehensive overview of the environmental, social, and economic footprints that can be used to measure the three pillars of sustainability. Steen-Olsen et al. (2012) used a MRIO model to quantify the total pressures due to consumption in the EU by calculating the carbon, water, and land footprints. Fang et al. (2013) presented a critique on some of these integration schemes.

A review of the existing literature that compares or integrates multiple footprints is shown in Table 2.1. As we see, the environmental pillar of sustainability is much better covered than the social and economic pillars, so we restrict the discussion of footprints in the environmental domain. The social and the economic footprints (Čuček et al., 2012a), for instance, are outside the scope of our analysis.

Table 2.1. Review of the literature that compares or integrates different footprint indicators.

Reference	Ecological	Carbon	Water	Energy	Nitrogen	Biodiversity	Land	Phosphorus	Waste	Material	Emission	Social	Economic
Chakraborty and Roy, 2013		×		×									
Cranston and Hammond, 2012	×	×		×									
Čuček, 2012a	×	×	×	×	×	×	×					×	×
Čuček, 2012b		×			×								
Curry and Maguire, 2011	×	×											
De Benedetto and Klemeš, 2009		×	×	×							×		
Del Borghi et al., 2013		×	×										
Ewing et al., 2012	×		×										
Fang et al., 2013	×	×	×	×									
Galli et al., 2012	×	×	×										
Galli et al., 2013	×	×	×										
Giljum et al., 2011	×	×	×										
Hanafiah et al., 2010	×			×	×			×					
Hanafiah et al., 2012	×			×		×							
Herva et al., 2011	×	×	×										
Herva et al., 2012	×			×									
Hoekstra, 2009	×		×										
Hoekstra and Wiedmann, 2014	×	×	×							×			
Hubacek et al., 2009	×		×										

Jess, 2010	×			×					×				
Lenzen et al., 2012	×	×	×										
Li et al., 2007	×			×									
Moran et al., 2013	×	×	×			×				×			
Page et al., 2012		×	×										
Steen-Olsen et al., 2012		×	×				×						
Tukker et al., 2014		×	×				×						
Xue and Landis, 2010		×			×								
Total	18	17	15	10	4	3	2	1	1	1	1	1	1

2.3. Elaboration of a footprint family

A footprint family consists of a number of members, each of which is a single-dimensional footprint. In this section, we discuss the most important potential members with an emphasis on their characteristics in a comparative sense.

2.3.1. Selection of footprint indicators

The composition of a footprint family may vary depending on the relevance of the impact categories addressed (Ridoutt and Pfister, 2013). In principle, any two or more footprint indicators could be considered as a footprint family. However, given the fact that there are a variety of footprint indicators that measure individual or collective pressures or impacts arising from production and consumption, we have to choose some of them to define a specific footprint family which is expected to address some significant anthropogenic impacts on major ecosystem compartments of the Earth-system.

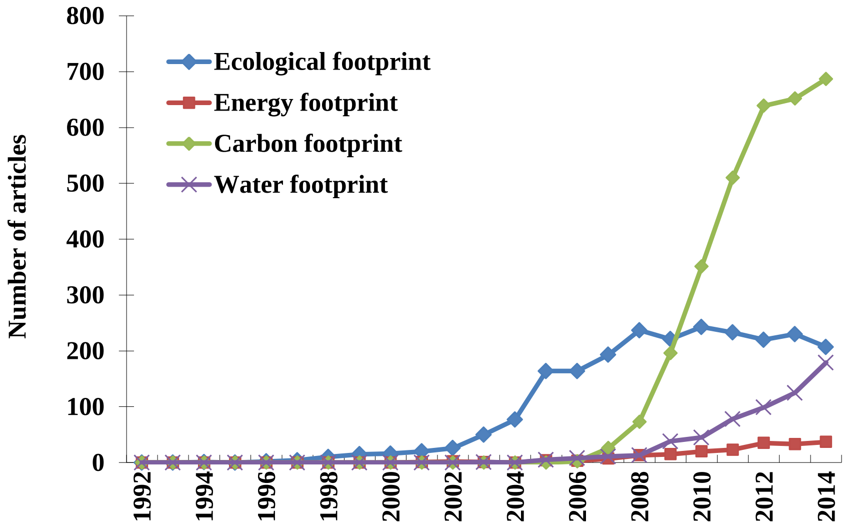

Figure 2.1. Number of articles with the topic of "ecological footprint", "energy footprint", "carbon footprint", and "water footprint", respectively, delivered by global scholars from 1992 to 2012 by searching the Web of Science on 2015-08-24.

As is clear from Table 2.1, the ecological, energy, carbon, and water footprints rank as the most important footprint indicators in the existing literature. This is partially because they are in close relation with the four worldwide concerns over threats to human society: food security, energy security, climate security, and water security (Mason and Zeitoun, 2013). Their frequencies of occurrence in scientific literature show a similar trend of increasing popularity during the period of 1992–2002 (Figure 2.1). The total number of articles with the topic of "ecological footprint", "energy footprint", "carbon footprint", and "water footprint" reaches 598, 50, 839 and 172, respectively, while the number of papers for the rest of the footprints listed in Table 2.1 is only 39. Notably, the statistics of the carbon footprint began to soar in 2007 and have even exceeded that of the ecological footprint since 2009. In this chapter, we make a selection of footprint indicators by combining the ecological, energy, carbon, and water footprints as potential members into a footprint family.

2.3.2. Survey of selected footprints

2.3.2.1. Ecological footprint

The ecological footprint is well-known as an effective communication tool for raising public awareness of the environmental impacts resulting from production and consumption. It was originally defined as a measure of the amount of ecologically productive land and water required to supply a specific activity with resources consumed and carbon dioxide (CO_2) generated (Monfreda et al., 2004; Wackernagel and Rees, 1996). Recently, it has been shifted to measure the land use that is required for population activities taking place on the biosphere within a given year while considering the prevailing technology and resource management of that year (Bastianoni et al., 2012; Borucke et al., 2013).

The methodology of the ecological footprint is included in a standardization process directed by the Global Footprint Network (GFN). National Footprint Accounts (NFA)—the most widely used approach for the ecological footprint practitioners and users—is developed and maintained by the GFN. The NFA has been applied in more than 200 countries, serving as an essential database for the Living Planet Report that is prepared by a number of the world's top organizations (WWF, 2012). A detailed research agenda for improving the NFA is established as a way of reaching consensus on priorities for implementing ecological footprint standards (Kitzes et al., 2009).

2.3.2.2. Energy footprint

Ferng's (2002) initiative highlights the importance of the energy footprint independent of the ecological footprint in energy scenario analysis by using an input–output analysis (IOA) based framework. Over the past years, an expanding list of researchers has chosen to concentrate exclusively on the topic of the energy footprint (e.g., Palmer, 1998; Stöglehner, 2003; Wiedmann, 2009a). It was previously introduced as a sub-indicator of the ecological footprint, representing the amount of forest area that would be required to absorb CO_2 emissions from fossil fuel combustion and electricity generation through the use of sequestration values for a world-average forest (Wackernagel and Rees, 1996). Depending on updated data obtained from the Intergovernmental Panel on Climate Change (IPCC, 2001), the calculation of the energy footprint has been revised with a fraction of approximately 30% of the total anthropogenic emissions for ocean uptake (Borucke et al., 2013; Monfreda et al., 2004). Recently, some researchers argue for a redefinition of the energy footprint as the sum of all area used to sequestrate CO_2

emissions from the consumption of non-food and non-feed energy (Čuček et al., 2012a; GFN, 2009).

2.3.2.3. Carbon footprint

The carbon footprint has become tremendously popular over the last few years, giving rise to a wide range of discussions in the scientific community. It is defined as the amount of CO_2-equivalent emissions caused directly and indirectly by an activity (Wiedmann and Minx, 2008), or as the total amount of greenhouse gas (GHG) emissions over the life cycle of a process or product (BSI, 2008). The usefulness of the carbon footprint differs from the energy footprint and has been justified in two respects: it takes into account non-CO_2 emissions (e.g., CH_4, N_2O) of which the global warming potentials (GWPs) are much higher than that of CO_2 (IPCC, 2007; Weidema et al., 2008), and the carbon footprint makes it easy to allocate the responsibility for global warming to consumers (Wiedmann and Minx, 2008; Wright et al., 2011). It is widely accepted that the life cycle assessment (LCA) is a useful tool for calculating the carbon footprint, especially at the product level (Wiedmann and Minx, 2008). Some standards such as the PAS 2050 (BSI, 2008) and ISO 14067 (Wiedmann, 2009b) have been or are being established on a life cycle basis.

2.3.2.4. Water footprint

The water footprint is another booming indicator that has gained worldwide popularity in recent years. It is defined as the cumulative virtual water content of all products and services consumed by individuals or communities within a given region (Hoekstra, 2009; Hoekstra and Hung, 2002). The real-water content of a product, in most cases, could be much less than the virtual-water content (Chapagain and Hoekstra, 2008). The water footprint can thus be termed in a way similar to that of the embodied energy, namely, the "embodied water" (Chambers et al., 2000). Two principal methods have been applied to water footprint accounting: the bottom-up and top-down approaches (Feng et al., 2011; Hoekstra, 2009; Van Oel et al., 2009). The bottom-up approach belongs to process analysis using detailed descriptions of individual production processes and, conversely, the top-down approach resembles IOA which is an economic approach adopted in economic and environmental domains (Feng et al., 2011). The international water footprint standards are simultaneously under development by the Water Footprint Network (WFN) (Hoekstra et al., 2011) and the International Organization for Standardization (ISO 14046) (Ridoutt and Huang, 2012).

2.3.3. Comparison of selected footprints

Based on a review of original articles on the fundamentals and applications of environmental footprints over the past two decades, here, we examine the differences and similarities of the ecological, energy, carbon and water footprints by listing key issues

Table 2.2. Main characteristics of the ecological, energy, carbon, and water footprints.

Item	Ecological footprint	Energy footprint	Carbon footprint	Water footprint
Conceptual roots	Carrying capacity	Ecological footprint	GWP	Virtual water
Research stressors	Biologically productive land for supporting resources consumption	Forest for absorbing energy-related GHG emissions	GHG emissions from products or activities	Freshwater for supporting products or activities
Footprint components	Cropland, grassland, woodland, fishing ground, built-up land, carbon uptake land	Fossil fuel, hydroelectricity, nuclear, etc.	CO_2, CH_4, N_2O, etc.	Blue water, green water, gray water
Metric units	Area-based (gha, ha, etc.)	Area-based (gha, ha, etc.)	Mass-based (kg, t, etc.)	Volume-based (m^3, L, etc.)
Calculation methods	NFA, NFA-NPP, IOA, LCA, bottom-up, top-down, emergy, exergy, MEA	NFA, NFA-PLUM, IOA, bottom-up, top-down	IOA, LCA, hybrid approach	IOA, LCA, bottom-up, top-down
Data availability	High	Medium	Medium	Low
Methodological standardization	Medium	Low	High	High
Weighting accuracy	Medium	Low	High	Low
Resultant interpretation	High	Low	Medium	Medium
Geographical specification	Medium	Medium	Low	High
Global comparability	Medium	High	High	Low
General applicability	High	Low	High	High

MEA: millennium ecosystem assessment; NPP: net primary productivity; PLUM: product land use matrix.

in Table 2.2 Despite sharing the term "footprint" in their names, the four footprints differ in more aspects than only in the impacts that are addressed, such as conceptual roots, research stressors, footprint components, metric units, calculation methods, and so on. We elaborate on the key differences item by item.

2.3.3.1. Roots and stressors

a) The ecological footprint is built upon a tradition of seeking alternatives to the appropriated carrying capacity which is related to the maximum population size that can be supported by a given set of resources (Dietz and Neumayer, 2007; Ehrlich, 1982). It is intended to deal with the question of how much area of biologically productive space is required to produce consumed resources and to absorb generated waste.

b) The origin of the energy footprint can be traced back to a subset of the ecological footprint (Wackernagel and Rees, 1996) in reply to the question of how much forest area is required to absorb CO_2 emissions from energy consumption. The energy footprint, in many cases, makes up a dominant proportion of the entire amount of ecological footprint at a regional or global level (Kitzes et al., 2009).

c) The carbon footprint is rooted in the indicators of GWP which represents the quantities of GHGs that contribute to global warming when considering a specific time horizon such as 100 years (Wiedmann, 2009b). It is concerned with the question of how much (CO_2-eq.) weight of the total GHG emissions is over the life cycle of products or activities.

d) The water footprint is derived from the concept of virtual water equal to the volume of freshwater used to produce a commodity (Allan, 1998; Hoekstra, 2009). It aims to address the question of how much volume of the cumulative virtual water is needed to provide products or activities.

2.3.3.2. Components and units

a) The ecological footprint is built upon a tradition of seeking alternatives to the appropriated carrying capacity which is related to the maximum population size that can be supported by a given set of resources (Dietz and Neumayer, 2007; Ehrlich, 1982). It is intended to deal with the question of how much area of biologically productive space is required to produce consumed resources and to absorb generated waste.

b) The energy footprint can be classified into concrete components such as the fossil fuel footprint, the hydroelectricity footprint, and the nuclear footprint (Browne, 2009; Čuček et al., 2012a; Stöglehner, 2003), all of which are expressed as the area of forest that is necessary to compensate for human-induced CO_2 (Van den Bergh and Verbruggen, 1999). The unit of measurement can be gha or local hectares with a specific carbon sequestration estimate (Walsh et al., 2010).

c) The components of the carbon footprint include a variety of GHGs such as CO_2, CH_4, and N_2O. The use of GHG characterization factors as determined weightings dependent on the 100-yr GWP has a broad base of acceptance (Ridoutt and Pfister, 2013) because the greater the GWP of a GHG, the more responsibility it should take on for increasing global temperatures. The metric is expressed in CO_2-equivalent mass units such as kilogram or ton, indicating the impact unit of time-integrated radiative forcing.

d) The water footprint consists of three components in volume-based units: the blue, green and gray water footprints. The blue and green footprints correspond to the demand for freshwater resources (from surface or ground water, and from rain, respectively), while the gray footprints refer to the waste assimilation by water and is defined as the volume of freshwater required to assimilate the load of pollutants based on the existing ambient water quality standards (Chapagain and Hoekstra, 2008; Hoekstra, 2009). Although Hoekstra (2009) has declared that no weighting is involved, equal weighting is also a form of weighting. It fails to distinguish the green, blue, and gray water components as they should have held different values for the environment (Hoekstra, 2009). The blue water's opportunity cost, for instance, is thought to be higher than the green water (Chapagain et al., 2006a).

2.3.3.3. Methods and scales

An important aspect to consider is that an environmental footprint measures the pressure or impact of X on Y, where Y stands for land appropriation, climate change, virtual water, etc., and X stands for a product, an organization, a country, etc. In other words, the object and scale of study act as an important contributor to the methodological options of a given footprint. A full understanding of all of the possible methods for the four footprints on different scales is, therefore, needed.

Using the language of LCA, we may distinguish a phase of goal and scope definition where the object and scale ("X") are defined, a phase of inventory analysis where the emissions and/or resource extractions are determined, and a phase of impact assessment where the emissions and/or extractions are further modeled into impacts (on "Y"). Figure 2.2 illustrates the ideas. Table 2.3 categorizes the literature into the two dimensions of object/scale ("X") and impact/footprint ("Y").

a) As noted above, the mainstream researchers of the ecological footprint place particular emphasis on the assessment at the national level, and modifications have been continuously supplemented to the NFA (Borucke et al., 2013; Wiedmann and Barrett, 2010). In addition to the typical NFA, several common tools for quantifying the ecological footprint at the macro-scale have been implemented including the

NFA-NPP, IOA, MEA, emergy analysis, and exergy analysis. For instance, IOA can facilitate the calculation of a national ecological footprint of different categories of final demand (Tukker et al., 2009). A micro- or meso-scale ecological footprint often benefits from the component-based (bottom-up) or compound-based (top-down) approaches. Moreover, since LCA is more comprehensive in terms of coverage of impact categories but neglects the limits of renewable capacity supplied by the biosphere (i.e., biocapacity), studies have recommended to give priority to combining the ecological footprint and LCA in a complementary way so that more robust and detailed sustainability assessment could be achieved (Castellani and Sala, 2012; Huijbregts et al., 2008).

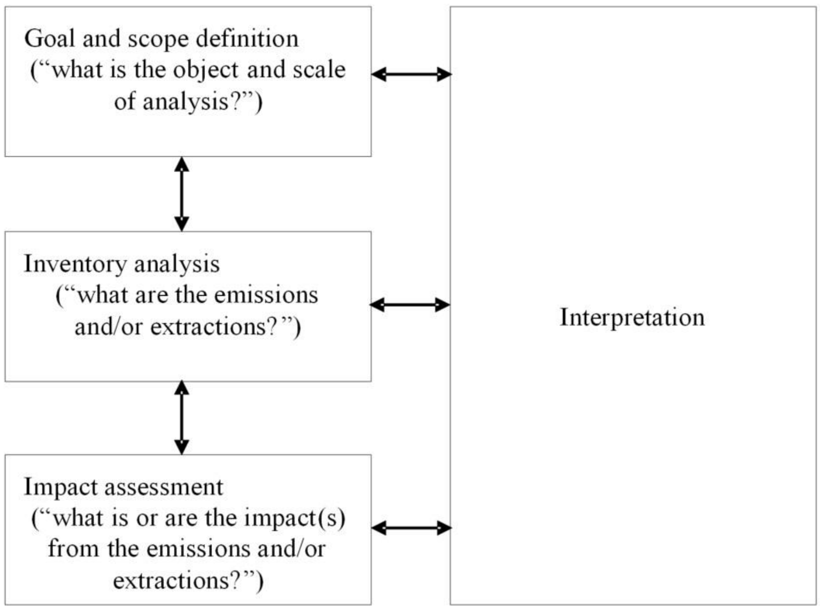

Figure 2.2. The four phases of the implementation of an impact assessment in the framework of LCA.

b) The NFA and its variation, NFA-PLUM, are the most common options for the organizational- or national-level energy footprint analysis. However, unlike the ecological footprint, many of the alternative approaches mentioned above are not available for the energy footprint. Moreover, from a methodological perspective, there is a great need to clarify which key features distinguish the energy footprint from the carbon footprint. It is our conviction that the most crucial difference between them is that the energy footprint takes a further step in translating the amount of CO_2 emissions into the amount of biologically productive land and water required to absorb these emissions than does the carbon footprint (Fang et al., 2013).

c) Given the widespread acceptance of LCA in carbon footprinting, it has almost become the preferred method for measuring the product-level carbon footprint (Wiedmann and Minx, 2008). However, limitations hampering the application of LCA have been observed in terms of boundary settings and responsibility sharing among the organizational or national carbon footprints, with the particular argument that a full life cycle perspective does not allow both producers and consumers to evaluate their footprints without double counting (Lenzen et al., 2007; Peters, 2010). Currently, there are a growing number of studies on hybrid approaches for companies or sectors, which combine the strength of both LCA and IOA in order to cover a broader scope of organizations and their activities (Heijungs and Suh, 2002).

d) In the water footprint analysis the bottom-up and top-down approaches both serve as major supporting tools. The former is accumulated item by item by multiplying all products and activities consumed by the respective water needs for those commodities. In some cases, it appears to be more reliable than the latter (Van Oel et al., 2009) which is, on the contrary, calculated as the total use of freshwater plus the gross virtual-water import minus the gross virtual-water export (Hoekstra, 2009), resembling the NFA of the ecological footprint. Furthermore, the NFA has been argued to be a special case of generalized physical input–output formulation (Kitzes et al., 2009). Therefore, it seems reasonable to make a deduction from a broad point of view that the NFA and IOA are designed in a similar way.

2.3.3.4. Other characteristics

a) A strength of the ecological footprint lies in its spatial dependence for easy interpretation as it translates the demand for biological resources and energy by the population into an easily interpretable area-based unit (Cranston and Hammond, 2012; Kitzes and Wackernagel, 2009). In addition, the ecological footprint benefits from its counterpart biocapacity, which is the ability of the Earth to supply the ecosystem services that humanity consumes (Borucke et al., 2013), as the comparison between the two metrics enables us to differentiate sustainability evaluation from the environmental impact assessment of a given activity (Castellani and Sala, 2012; Kates et al., 2001). The ecological footprint, however, has come under intense criticism for several reasons such as controversial hypotheses, a weak analytical basis, and an aggregate calculation system (e.g., Fiala, 2008; Kitzes et al., 2009; Van den Bergh and Grazi, 2010; Vogelsang, 2002).

b) In comparison with the ecological footprint, the usefulness of the energy footprint becomes apparent as it establishes a delicate connection between atmospheric carbon emissions and terrestrial carbon sinks. However, objections to the basic methodology never stop. A noticeable critique of its scientific robustness can be attributed to the failure to capture energy-related emissions other than CO_2 and to reduce the uncertainty in the estimation of carbon sequestration rate (Kitzes et al., 2009; Van den Bergh and Verbruggen, 1999; Venetoulis and Talberth, 2008), even though some of the concerns have been addressed through a series of modified models (e.g., De Benedetto and Klemeš, 2009; Kitzes et al., 2009; Lenzen and Murray, 2001).

c) The carbon footprint is booming with a much broader appeal than alternative indicators and LCA (Weidema et al., 2008). Nevertheless, criticisms toward the carbon footprint remain. A prominent one is the view expressed by some observers that the huge demand for detailed data compromises the quality of outcome, especially in those situations where extremely limited data for use at micro- or meso-scale accounts lead to underestimation (Chakraborty and Roy, 2013; De Benedetto and Klemeš, 2009). Another criticism is that the lack of consideration of carbon sequestration land runs the risk of disregarding the terrestrial feedback processes such as abrupt degradation of forest or changes in the distribution of vegetation and oceanic fluxes that further affect the global carbon cycle, which may have subsequent detrimental impacts on climate (Fang et al., 2013).

d) Freshwater is a highly site-specific resource cycling throughout the planet (Herva et al., 2011; Kitzes et al., 2009); it requires tracking down the origin of consumer products at the place of production (Hoekstra, 2009). As a consequence, the water footprint is unlikely to be as globally expressed as the ecological footprint but, rather, to be a geographically explicit indicator that show not only the volume of water use but also the locations with emphasis on the distinction between internal and external water footprints (Hoekstra, 2009). On the other hand, the water footprint seems to be more vulnerable to data constraints than the ecological footprint.

Table 2.3. Summary of approaches applied to the studies of ecological, energy, carbon, and water footprints from the literature.

Object/scale	Ecological footprint	Energy footprint	Carbon footprint	Water footprint
Product/ food/ energy	LCA (Huijbregts et al., 2008)	Bottom-up approach (Šantek et al., 2010)	LCA (Carballo-Penela and Doménech, 2010)	LCA (Ridoutt and Pfister, 2010)
	Bottom-up approach (Simmons et al., 2000)	Top-down approach (Stöglehner, 2003)	Hybrid approach (Virtanen et al., 2011)	Bottom-up approach (Chapagain et al., 2006b)
	Top-down approach (Mamouni Limnios et al., 2009)			Top-down approach (Van Oel et al., 2009)
Organization/ sector/ trade/ building	IOA (Hubacek and Giljum, 2003)	IOA (Ferng, 2002)	IOA (Huang et al., 2009)	IOA (Steen-Olsen et al., 2012)
	Bottom-up approach (Simmons et al., 2000)	Top-down approach (Chen and Lin, 2008)	LCA (Sanyé et al., 2012)	Bottom-up approach (Van Oel et al., 2009)
	Top-down approach (Bastianoni et al., 2007)	NFA (Browne et al., 2009)	Hybrid approach (Berners-Lee et al., 2011)	Top-down approach (Van Oel et al., 2009)
	Emergy (Zhao et al., 2013)	NFA-PLUM (Wiedmann, 2009a)		
Region/ nation/ world/	IOA (Lenzen et al., 2012)	NFA (Chen and Lin, 2008)	IOA (Hertwich and Peters, 2009)	IOA (Zhao et al., 2009)
	NFA (Wackernagel and Rees, 1996)			Bottom-up approach (Hoekstra and Mekonnen, 2012)
	NFA-NPP (Venetoulis and Talberth, 2008)			
	MEA (Burkhard et al., 2012)			
	Emergy (Zhao et al., 2005)			
	Exergy (Chen and Chen, 2007)			

When the authors do not specify the approaches used in their research, we roughly categorize them into two broad types: the bottom-up approach and the top-down approach.

2.4. Discussion

The proposed footprint family allows for a broader assessment of significant environmental impacts. In order to better understand its role in tracking human pressure on the planet, an evaluation of the footprint family will be conducted by testing the performance in crucial respects, and key priorities for further development will be suggested as well.

2.4.1. Evaluation of the footprint family

2.4.1.1. Data availability

In the case of cross-national databases, the ecological footprint accounts for over 150 nations during 1961–2008 have been consistently released by the GFN with a separate column for the energy footprint (e.g., Borucke et al., 2013; Ewing et al., 2010; GFN, 2009). A database of the carbon footprint for 73 nations is available as well (Hertwich and Peters, 2009). There is also an average of the data of the water footprint for 140 nations between 1996 and 2005 (Hoekstra and Mekonnen, 2012). Recently, the advanced online database, EUREAPA (2011), has been constructed based on the support of the Global Trade Analysis Project (GTAP) version 7, in which detailed datasets on the ecological, carbon, and water footprints of 27 EU member states and additional 18 nations or regions with 57 sectors are all available for the base year of 2004.

In contrast to the well-documented data at the national level, data for the micro-scale products and the meso-scale organizations are quite limited. For instance, the access to the database of a company which comprises detailed statistics of resources used and wastes discharged within the defined boundary, in many cases, is not available. Data availability has become a limiting factor in the extension at the sub-national level. A consequence is that potential applications of the footprint family would be restricted to the national level. Nevertheless, this is constrained by a lack of available dedicated personnel and financial resources rather than a lack of understanding or willingness to promote data collection and quality (Kitzes et al., 2009).

2.4.1.2. Coverage complementarity

Each of the four environmental footprints is designed for addressing environmental issues with different emphases and, hence, is a complement to and not a substitute for others. The carbon and water footprints have been found to be able to benefit the ecological footprint by providing core information on natural capital use in relation to human consumption of the atmosphere and hydrosphere, respectively (Galli et al., 2012). For instance, the fishing component of the ecological footprint denotes the area of sea space required to sustain the harvested aquatic species, while the water footprint includes the volume of evaporated and polluted water associated with the activity of fishing and aquaculture (Borucke et al., 2013; Hoekstra, 2009). Thus, the ecological and water footprints can be seen as two complementary indicators in terms of fishing-related impact assessment. Another example is the combination of the carbon and water footprints which can be profitably implemented as dual single-issue indicators allowing for trade-offs without the risk of problem shifting both in the case of potable water distribution (Del Borghi et al., 2013) and in the case of fresh tomato production (Page et al., 2012).

However, significant double counting of the footprint family has been discerned in terms of carbon emissions and sequestration. A partial solution is to exclude the carbon uptake land from current ecological footprint accounting (Galli et al., 2012; Steen-Olsen et al., 2012; Van den Bergh and Verbruggen, 1999). It is better to rename the rest of the ecological footprint as "land footprint" which accounts for the actual land and ocean exploited by humanity (Steen-Olsen et al., 2012; Weinzettel et al., 2013). This can be justified by two arguments. First, the carbon uptake land is hypothetical land that does not exist and thus conflicts with the actual appropriated land within the aggregate ecological footprint account (Hubacek and Giljum, 2003; Van den Bergh and Verbruggen, 1999). Second, the carbon uptake land is tightly tied to energy-related carbon emissions and sequestration which are already covered by the energy and carbon footprints (Fang et al., 2013; Steen-Olsen et al., 2012). Nevertheless, there is still a need to get rid of the overlap between the energy and carbon footprints. A potential approach for making the energy footprint completely independent of the carbon footprint will be presented in Section 2.4.2.1.

2.4.1.3. Methodological consistency

Of particular interest in evaluating the footprint family is to what extent the consistency between different footprint indicators has been maintained. From a methodological perspective, this is largely driven by two factors: standardization and harmonization. With respect to the progress of methodological standardization, several accounting standards underlying individual footprints already exist in two internationally standardized formats (Giljum et al., 2011): the normative format (e.g., ISO, PAS) and the descriptive format (e.g., GFN, WFN). The normative one like the ISO never aims to standardize accounting methods in detail, and there is not even common agreement on how to interpret some of the ISO requirements, so diverging approaches have occurred spontaneously (Guinee et al., 2010). The descriptive one contains numerous detailed regulations in which it seems much easier to build consensus concerning methodology, transparency, and communications but is constrained by weak enforcement. In addition, the methodology for the energy footprint is not yet as standardized and scientifically robust as the cases of the ecological, carbon, and water footprints (Galli et al., 2012); therefore, there is much room for improvement in future studies of the energy footprint.

Harmonized accounting of the footprint family on the national level is most likely achieved through a unified MRIO-based framework as a series of MRIO models have been successfully applied to the measurements of the land footprint (Weinzettel et al., 2013), the energy footprint (Wiedmann, 2009a), the carbon footprint (Hertwich and Peters, 2009), and the water footprint (Yu et al., 2010), as well as to the measurements of different sets of footprints (Galli et al., 2013; Lenzen et al., 2012; Moran et al., 2013; Steen-Olsen et al., 2012). The life cycle, consumption-based perspective of the MRIO models make it easy to operationalize the footprint family concept without distracting from the issue of responsibility allocation which is too complicated to be thoroughly settled (Finnveden et al., 2009; Heijungs and Suh, 2002; Lenzen et al., 2007; Wiedmann, 2009a). We, therefore, consider that MRIO models would become a consensus approach for the national-level footprint family studies. More importantly, the ongoing development of MRIO models has been accompanied with the recognition that an updated version should be prepared in such a way that it allows more conceivable footprints to be incorporated (Galli et al., 2013).

In contrast, the methodological harmonization of the micro- or meso-scale footprint family is far from satisfactory. Apart from the low data availability, this can be partially attributed to the fact that consumer products are, in many cases, not compatible with the top-down IOA but rather appear to be compatible with the bottom-up LCA (Peters, 2010). Therefore, the proposal for a full LCA-based footprint family (Ridoutt and Pfister, 2013) would be feasible, though this probably pertains to the product level. An integration of certain existing methods such as hybrid approaches which take advantage of the alignment of both LCA and IOA could be a prospective solution to formulating the organizational-level footprint family. Yet, the likelihood of such a development is not available as no footprint other than the carbon footprint has been tested with a hybrid approach (Ewing et al., 2012).

2.4.1.4. Policy relevance

A remarkable advantage of the footprint family is that it offers policy makers an overall vision of the combined effects of various human pressure which then enables a deeper understanding of environmental complexity. In a policy context, the analysis of any single footprint should not be uncoupled from others as impact categories are so tightly linked that changing one may profoundly affect others in ways that people do not expect. The carbon footprint, for instance, has been found to be a poor representative of the environmental burden of products because problems will shift to other impact categories (like photochemical ozone or human toxicity) if reducing the carbon footprint is overemphasized to manufacturers (Laurent et al., 2012). That is, perhaps, why policy makers are insistent on a set of indicators rather than just a single one (Kitzes et al., 2009). Only the completeness of those indicators can allow one to examine whether an improvement in one category would lead to undesirable situations in other categories. Problem shifting, to some extent, could thus be avoided by using the footprint family.

Meanwhile, the discrepancy between the ecological footprint and the other three footprints becomes apparent when looking at their underlying hypotheses. From an ecological footprint perspective, any emissions of GHGs or use of water resources would be compulsively labeled with "unsustainability" (Fiala, 2008) as a clear recognition of sustainability limits relative to the energy, carbon, and water footprints is lacking in scientific advice for policy makers. In other words, the energy, carbon, and water footprints suffer from the critique that no distinction is made between sustainable and unsustainable activities, while the ecological footprint is criticized for its arbitrariness of assuming zero environmental capacity for carbon accumulation in the atmosphere and for pollutant concentration in the hydrosphere. This illustrates a very important point: without the reference to sustainability limits, the footprint family cannot be used to determine whether or not natural capital is being consumed in a sustainable way; it is only appropriate to measure the environmental impacts derived from natural capital appropriation by consumption of biological resources and water, GHG emissions, and discharge of the resulting waste (Fang et al., 2013). A further extension of the footprint family to relating to planetary boundaries will be delineated in Section 2.4.2.2.

2.4.2. Needs for further development

2.4.2.1. Monitoring stock depletion

The environmental implications of energy use not only affect the atmosphere where anthropogenic warming impacts have been sufficiently denoted using the carbon footprint or the alternatives—the GHG footprint (Hertwich and Peters, 2009) or the climate footprint (Huijbregts et al., 2010), but also affect the lithosphere where fossil fuel is an essential natural resource for humanity and contributes most to the stocks of natural capital maintained throughout the world. Unlike the renewable natural resources such as cropland or grassland, the consumption of fossil energy will undoubtedly result in a diminished stock of natural capital. From a strong sustainability perspective, all renewable flows of resources and ecosystem services could be sparingly consumed, but finite stocks should remain constant (e.g., Costanza and Daly, 1992; Niccolucci et al., 2009; Wackernagel and Rees, 1997). If stock depletion continues to grow a tipping point beyond which a tremendously huge accumulation of debt will never be paid back by natural capital flows, the Earth-system would ultimately collapse with disastrous consequences for human beings (Rockström et al., 2009; Wackernagel and Rees, 1997). Given this concern, it is desirable to find a way of reshaping the energy footprint to monitor the depletion of stocks associated with energy consumption within a given year. We argue that the footprint family will benefit from such a remodeling because it allows for simultaneous tracking of human pressure on four life-supporting compartments of the Earth: the biosphere, lithosphere, atmosphere, and hydrosphere. The major challenges are to calculate the cumulative energy demand for non-renewable stocks and to determine how much stock can be consumed until the global ecosystem breaks down.

2.4.2.2. Setting sustainability limits

The measurement of the biocapacity introduced by the ecological footprint practitioners is unique and important (Ewing et al., 2012); it allows the footprints to be comparable with the relative limits, representing the degree to which biophysical limits have been approached or exceeded (Costanza, 2000). With the aim to complement the footprint family with an integrated sustainability core, it is of great importance to identify and quantify sustainability limits for some other footprints as well. The reality is that seven out of nine Earth-system processes identified by Rockström et al. (2009) including land use change (land footprint), climate change (carbon footprint), freshwater use (water footprint), nitrogen cycle (nitrogen footprint), phosphorus cycle (phosphorus footprint), biodiversity loss (biodiversity footprint), and chemical pollution (chemical footprint), have been addressed through a variety of footprint indicators. In this sense, the combination of those footprints as an extended footprint family which would have sufficient spatial coverage to encompass the majority of Earth-system processes may serve as a starting point for demarcating planetary boundaries. Given the gaps in current knowledge about the complexity of Earth-system, determining critical values for each of the planetary boundaries has to involve normative or even subjective judgments (Lewis, 2012; Rockström et al., 2009). Thus the major constraint on setting sustainability limits for a footprint family is that it is almost unlikely to be able to present reasonable and accurate estimates that can be consistently validated by the scientific community.

2.5. Conclusions

This chapter provides an overview of a new footprint family which is designed in such a way that some significant environmental impacts associated with human activities can be measured through a set of selected footprint indicators. The ecological, energy, carbon, and water footprints involved in the proposed footprint family can be regarded as complementary to each other as each of them focuses on different aspects of environmental issues. The footprint family is found to be able to capture a broad spectrum of sustainability issues in relation to natural resource use and waste discharge, and to provide policy makers with a more complete picture of environmental complexity than single footprints. This study shows that data for the footprint family is already available on the national level in which harmonized accounting is likely to be achieved through a unified MRIO-based framework. Nevertheless, the footprint family still suffers from limitations. Neither the data availability nor the methodological consistency has been satisfactory for products or organizations. Another weakness is due to the significant double counting that exists in terms of energy-related carbon emissions and sequestration.

Our discussion comprises a sequence of suggestions. First, the carbon uptake land is supposed to be disaggregated from the ecological footprint, and it is better to rename the remaining as "land footprint" which addresses the actual land appropriation by humanity. Second, data availability is a limiting factor for the micro- and meso-scale footprint family studies and thus deserves attention in efforts to improve data accessibility and reliability. Third, a full LCA could be appropriate for the footprint family at the product level while the hybrid approach might be a prospective solution to the organizational-level studies. Fourth, the footprint family can currently be used for assessing some relevant environmental impacts but not for evaluating environmental sustainability. To achieve a more rigorous footprint family, two priorities for further development are provided. One is to reshape the energy footprint to monitor the depletion of energy stocks. The other is to identify sustainability limits for a variety of footprints in order to complement the footprint family with a more comprehensive integrated sustainability core.

References

Allan, J. A., 1998. Virtual water: a strategic resource, global solutions to regional deficits. Groundwater 36, 545–546.

Bastianoni, S., Galli, A., Pulselli, R. M., Niccolucci, V., 2007. Environmental and economic evaluation of natural capital appropriation through building construction: practical case study in the Italian context. AMBIO 36, 559–565.

Bastianoni, S., Niccolucci, V., Pulselli, R. M., Marchettini, N., 2012. Indicator and indicandum: "Sustainable way" vs "prevailing conditions" in the Ecological Footprint. Ecological Indicators 16, 47–50.

Berners-Lee, M., Howard, D. C., Moss, J., Kaivanto, K., Scott, W. A., 2011. Greenhouse gas footprinting for small businesses—The use of input–output data. Science of the Total Environment 409, 883–891.

Borucke, M., Moore, D., Cranston, G., Gracey, K., Iha, K., Larson, J., Lazarus, E., Morales, J. C., Wackernagel, M., Galli, A., 2013. Accounting for demand and supply of the biosphere's regenerative capacity: the National Footprint Accounts' underlying methodology and framework. Ecological Indicators 24, 518–533.

Browne, D., O'Regan, B., Moles, R., 2009. Use of ecological footprinting to explore alternative domestic energy and electricity policy scenarios in an Irish city-region. Energy Policy 37, 2205–2213.

BSI (British Standards Institution), 2008. Guide to PAS 2050. How to Assess the Carbon Footprint of Goods and Services. British Standards, London. http://aggie-horticulture.tamu.edu/faculty/hall/publications/PAS2050_Guide.pdf.

Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecological Indicators 21, 17–29.

Carballo-Penela, A., Doménech, J. L., 2010. Managing the carbon footprint of products: the contribution of the method composed of financial statements (MC3). The International Journal of Life Cycle Assessment 15, 962–969.

Castellani, V., Sala, S., 2012. Ecological Footprint and Life Cycle Assessment in the sustainability assessment of tourism activities. Ecological Indicators 16, 135–147.

Chakraborty, D., Roy, J., 2013. Energy and carbon footprint: numbers matter in low energy and low carbon choices. Current Opinion in Environmental Sustainability 5, 237–243.

Chambers, N., Simmons, C., Wackernagel, M., 2000. Sharing Nature's Interest: Ecological Footprints as an Indicator of Sustainability. Earthscan, London.

Chapagain, A. K., Hoekstra, A. Y., 2008. The global component of freshwater demand and supply: an assessment of virtual water flows between nations as a result of trade in agricultural and industrial products. Water International 33, 19–32.

Chapagain, A. K., Hoekstra, A. Y., Savenije, H. H. G., 2006a. Watersaving through international trade of agricultural products. Hydrology and Earth System Sciences 10, 455–468.

Chapagain, A. K., Hoekstra, A. Y., Savenije, H. H. G., Gautam, R., 2006b. The water footprint of cotton consumption: An assessment of the impact of worldwide consumption of cotton products on the water resources in the cotton producing countries. Ecological Economics 60, 186–203.

Chen, B., Chen, G. Q., 2007. Modified ecological footprint accounting and analysis based on embodied exergy—a case study of the Chinese society 1981–2001. Ecological Economics 61, 355–376.

Chen, C. Z., Lin, Z. S., 2008. Multiple timescale analysis and factor analysis of energy ecological footprint growth in China 1953–2006. Energy Policy 36, 1666–1678.

Costanza, R., 2000. The dynamics of the ecological footprint concept. Ecological Economics 32, 341–345.

Costanza, R., Daly, H. E., 1992. Natural capital and sustainable development. Conservation Biology 6, 37–46.

Cranston, G. R., Hammond, G. P., 2012. Carbon footprints in a bipolar, climate-constrained world. Ecological Indicators 16, 91–99.

Čuček, L., Klemeš, J. J., Kravanja, Z., 2012a. A review of Footprint analysis tools for monitoring impacts on sustainability. Journal of Cleaner Production 34, 9–20.

Čuček, L., Klemeš, J. J., Kravanja, Z. 2012b. Carbon and nitrogen trade-offs in biomass energy production. Clean Technologies and Environmental Policy 14, 389–397.

Curry, R., Maguire, C., 2011. The use of Ecological and Carbon Footprint Analysis in regional policy making: application and insights using the REAP model. Local Environment: the International Journal of Justice and Sustainability 16, 917–936.

De Benedetto, L., Klemeš, J., 2009. The Environmental Performance Strategy Map: an integrated LCA approach to support the strategic decision making process. Journal of Cleaner Production 17, 900–906.

Del Borghi, A., Strazza, C., Gallo, M., Messineo, S., Naso, M., 2013. Water supply and sustainability: life cycle assessment of water collection, treatment and distribution service. The International Journal of Life Cycle Assessment 18, 1158–1168.

Dietz, S., Neumayer, E., 2007. Weak and strong sustainability in the SEEA: Concepts and measurement. Ecological Economics 61, 617–626.

Ehrlich, P. R., 1982. Human carrying capacity, extinction, and nature reserves. Bioscience 32, 331–333.

EUREAPA, 2011. Scenario Modelling and Policy Assessment Tool. https://www.eureapa.net/.

Ewing, B., Reed, A., Galli, A, Kitzes, J., Wackernagel, M., 2010. Calculation Methodology for the National Footprint Accounts, 2010 Edition. Global Footprint Network, Oakland. http://www.footprintnetwork.org/images/uploads/National_Footprint_Accounts_Method_Paper_2010.pdf.

Ewing, B. R., Hawkins, T. R., Wiedmann, T. O., Galli, A., Ertug Ercin, A., Weinzettel, J., Steen-Olsen, K., 2012. Integrating ecological and water footprint accounting in a multi-regional input–output framework. Ecological Indicators 23, 1–8.

Fang, K., Heijungs, R., De Snoo, G., 2013. The footprint family: comparison and interaction of the ecological, energy, carbon and water footprints. Revue de Métallurgie 110, 77–86.

Feng, K., Chapagain, A., Suh, S., Pfister, S., Hubacek, K., 2011. Comparison of bottom-up and top-down approaches to calculating the water footprints of nations. Economic Systems Research 23, 371–385.

Ferng, J. J., 2002. Toward a scenario analysis framework for energy footprints. Ecological Economics 40, 53–69.

Fiala, N., 2008. Measuring sustainability: Why the ecological footprint is bad economics and bad environmental science. Ecological Economics 67, 519–525.

Finnveden, G., Hauschild, M. Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in life cycle assessment. Journal of Environmental Management 91, 1–21.

Galli, A., Kitzes, J., Wermer, P., Wackernagel, M., Niccolucci, V., Tiezzi, E., 2007. An exploration of the mathematics behind the ecological footprint. International Journal of Ecodynamics 2, 250–257.

Galli, A., Weinzettel, J., Cranston, G., Ercin, E., 2013. A Footprint Family extended MRIO model to support Europe's transition to a One Planet Economy. Science of Total Environment 461–462, 813–818.

Galli, A., Wiedmann, T., Ercin, E., Knoblauch, D., Ewing, E, Giljum, S., 2012. Integrating ecological, carbon and water footprint into a "Footprint Family" of indicators: Definition and role in tracking human pressure on the planet. Ecological Indicators 16, 100–112.

GFN (Global Footprint Network). Ecological Footprint Standards 2009. Global Footprint Network, Oakland. http://www.footprintnetwork.org/images/uploads/Ecological_Footprint_Standards_2009.pdf.

Giljum, S., Burger, E., Hinterberger, F., Lutter, S., Bruckner, M., 2011. A comprehensive set of resource use indicators from the micro to the macro level. Resources, Conservation and Recycling 55, 300–308.

Giljum, S., Hinterberger, F., Lutter, S., 2008. Measuring natural resource use: context, indicators and EU policy processes. Sustainable Europe Research Institute (SERI) Background Paper 14. SERI, Vienna.

Guinee, J. B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonamici, R., Ekvall, T., Rydberg, T., 2010. Life cycle assessment: Past, present, and future. Environmental Science & Technology 45, 90–96.

Hanafiah, M., Hendriks, J., Huijbregts, M., 2012. Comparing the ecological footprint with the biodiversity footprint of products. Journal of Cleaner Production 37, 107–114.

Hanafiah, M. M., Huijbregts, M. A. J., Hendriks, A. J., 2010. The influence of nutrients and non-CO_2 greenhouse gas emissions on the ecological footprint of products. Sustainability 2, 963–979.

Heijungs, R., Suh, S., 2002. The Computational Structure of Life Cycle Assessment. Kluwer Academic Publisher, Dordrecht, The Netherlands.

Hertwich, E. G., Peters, G. P., 2009. Carbon footprint of nations: A global, trade-linked analysis. Environmental Science & Technology 43, 6414–6420.

Herva, M., Franco, A., Carrasco, E. F., Roca, E., 2011. Review of corporate environmental indicators. Journal of Cleaner Production 19, 1687–1699.

Herva, M., García-Diéguez, C., Franco-Uría, A., Roca, E., 2012. New insights on ecological footprinting as environmental indicator for production processes. Ecological Indicators 16, 84–90.

Hoekstra, A. Y., 2009. Human appropriation of natural capital: a comparison of ecological footprint and water footprint analysis. Ecological Economics 68, 1963–1974.

Hoekstra, A. Y., Chapagain, A. K., Aldaya, M. M. Mekonnen, M. M., 2011. The Water Footprint Assessment Manual: Setting the Global Standard. Water Footprint Network (WFN), Earthscan, London.

Hoekstra, A. Y., Hung, P. Q., 2002. Virtual Water Trade: A Quantification of Virtual Water Flows between Nations in Relation to International Crop Trade. Value of Water Research Report Series (No. 11), UNESCO-IHE Institute for Water Education, Delft. http://www.waterfootprint.org/Reports/Report11.pdf.

Hoekstra, A. Y., Mekonnen, M. M., 2012. The water footprint of humanity. Proceedings of the National Academy of Sciences 109, 3232–3237.

Hoekstra, A. Y., Wiedmann, T. O., 2014. Humanity's unsustainable environmental footprint. Science 344, 1114–1117.

Huang, Y. A., Lenzen, M., Weber, C. L., Murray, J., Matthews, H. S., 2009. The role of input–output analysis for the screening of corporate carbon footprints. Economic Systems Research 21, 217–242.

Hubacek, K., Giljum, S., 2003. Applying physical input-output analysis to estimate land appropriation (ecological footprints) of international trade activities. Ecological Economics 44, 137–151.

Hubacek, K., Guan, D., Barrett, J., Wiedmann, T., 2009. Environmental implications of urbanization and lifestyle change in China: Ecological and water footprints. Journal of Cleaner Production 17, 1241–1248.

Huijbregts, M. A. J., Hellweg, S., Frischknecht, R., Hendriks, H. W., Hungerbühler, K., Hendriks, A. J., 2010. Cumulative energy demand as predictor for the environmental burden of commodity production. Environmental Science & Technology 44, 2189–2196.

Huijbregts, M. A. J, Hellweg, S., Frischknecht, R., Hungerbühler, K., Hendriks, A. J., 2008. Ecological footprint accounting in the life cycle assessment of products. Ecological Economics 64, 798–807.

IPCC (Intergovernmental Panel on Climate Change), 2001. Climate Change 2001: The Scientific Basis. Cambridge University Press, Cambridge, UK.

IPCC (Intergovernmental Panel on Climate Change), 2007. Climate Change 2007: Synthesis Report. Cambridge University Press, New York, USA.

Jess, A., 2010. What might be the energy demand and energy mix to reconcile the world's pursuit of welfare and happiness with the necessity to preserve the integrity of the biosphere? Energy Policy 38, 4663–4678.

Kates, R. W., Clark, W. C., Corell, R., Hall, J. M., Jaeger, C. C., Lowe, I., McCarthy, J. J., Schellnhuber,

H. J., Bolin, B., Dickson, N. M., Faucheux, S., Gallopin, G. C., Grübler, A., Huntley, B., Jäger, J., Jodha, N. S., Kasperson, R. E., Mabogunje, A., Matson, P., Mooney, H., Moore, B. III., O'Riordan, T., Svedin, U., 2001. Environment and development: sustainability science. Science 292, 641–642.

Kitzes, J., Galli, A., Bagliani, M., Barrett, J., Dige, G., Ede, S., Erb, K., Giljum, S., Haberl, H., Hails, C., Jolia-Ferrier, L., Jungwirth, S., Lenzen, M., Lewis, K., Loh, J., Marchettini, N., Messinger, H., Milne, K., Moles, R., Monfreda, C., Moran, D., Nakano, K., Pyhälä, A., Rees, W., Simmons, C., Wackernagel, M., Wada, Y., Walsh, C., Wiedmann, T., 2009. A research agenda for improving national ecological footprint accounts. Ecological Economics 68, 1991–2007.

Kitzes, J., Wackernagel, M., 2009. Answers to common questions in ecological footprint accounting. Ecological Indicators 9, 812–817.

Laurent, A., Olsen, S. I., Hauschild, M. Z., 2012. Limitations of carbon footprint as indicator of environmental sustainability. Environmental Science & Technology 46, 4100–4108.

Leach, A. M., Galloway, J. N., Bleeker, A., Erisman, J. W., Kohn, R., Kitzes, J., 2012. A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. Environmental Development 1, 40–66.

Lenzen, M., Kanemoto, K., Moran, D., Geschke, A., 2012. Mapping the structure of the world economy. Environmental Science & Technology 46, 8374–8381.

Lenzen, M., Murray, J., Sack, F., Wiedmann., T., 2007. Shared producer and consumer responsibility—theory and practice. Ecological Economics 61, 27–42.

Lenzen, M., Murray, S. A., 2001. A modified ecological footprint method and its application to Australia. Ecological Economics 37, 229–255.

Lewis, S. L., 2012. We must set planetary boundaries wisely. Nature 485, 417.

Li, H., Zhang, P. D., He, C. Y., Wang, G., 2007. Evaluating the effects of embodied energy in international trade on ecological footprint in China. Ecological Economics 62, 136–148.

Mamouni Limnios, E. A., Ghadouani, A., Schilizzi, S. G., Mazzarol, T., 2009. Giving the consumer the choice: A methodology for Product Ecological Footprint calculation. Ecological Economics 68, 2525–2534.

Mason, M., Zeitoun, M., 2013. Questioning environmental security. The Geographical Journal 179, 294–297.

Monfreda, C., Wackernagel, M., Deumling, D., 2004. Establishing national natural capital accounts based on detailed Ecological Footprint and biological capacity assessments. Land Use Policy 21, 231–246.

Moore, D., Cranston, G., Reed, A., Galli, A., 2012, Projecting future human demand on the Earth's regenerative capacity. Ecological Indicators 16, 3–10.

Moran, D. D., Lenzen, M., Kanemoto, K., Geschke, A., 2013. Does ecologically unequal exchange occur? Ecological Economics 89, 177–186.

Niccolucci, V., Bastianoni, S., Tiezzi, E. B. P., Wackernagel, M., Marchettini, N., 2009. How deep is the footprint? A 3D representation. Ecological Modelling 220, 2819–2823.

Niccolucci, V., Rugani, B., Botto, S., Gaggi, C, 2010. An integrated footprint based approach for environmental labelling of products: the case of drinking bottled water. International Journal of Design and Nature and Ecodynamics 5, 68–75.

Oosthoek, J., Gills, B. K., 2005. Humanity at the crossroads: The globalization of environmental crisis. Globalizations 2, 283–291.

Page, G., Ridoutt, B., Bellotti, B., 2012. Carbon and water footprint tradeoffs in fresh tomato production. Journal of Cleaner Production 32, 219–226.

Palmer, A. R. P., 1998. Evaluating ecological footprints. Electronic Green Journal 1, 1–11.

Panko, J., Hitchcock, K., 2011. Chemical Footprint Ensuring Product Sustainability. Air and Waste Management Association, pp. 12–15. http://chemrisknano.com/~chemrisk/images/stories/Chemical_Footprint_Ensuring_Product_Sustainability.pdf.

Peters, G. P., 2010. Carbon footprints and embodied carbon at multiple scales. Current Opinion in Environmental Sustainability 2, 245–250.

Rees, W. E., 1992. Ecological footprint and appropriated carrying capacity: what urban economics leaves out. Environment and Urbanization 4, 121–130.

Rees, W. E., 1996. Revisiting carrying capacity: area-based indicators of sustainability. Population and Environment 17, 195–215.

Rees, W. E., 2002. Footprint: our impact on Earth is getting heavier. Nature 420, 267–268.

Ridoutt, B. G., Huang, J., 2012. Environmental relevance — the key to understanding water footprints. Proceedings of the National Academy of Sciences of the United States of America 109, E1424.

Ridoutt, B. G., Pfister, S., 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environmental Change 20, 113–120.

Ridoutt, B. G., Pfister, S., 2013. Towards an integrated family of footprint indicators. Journal of Industrial Ecology 17, 337–339.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F. S., Lambin, E. F., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., De Wit, C. A., Hughes, T., Van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P. K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R. W., Fabry, V. J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J. A., 2009. A safe operating space for humanity. Nature 461, 472–475.

Šantek, B., Gwehenberger, G., Šantek, M. I., Narodoslawsky, M., Horvat, P., 2010. Evaluation of energy demand and the sustainability of different bioethanol production processes from sugar beet. Resources, Conservation and Recycling 54, 872–877.

Sanyé, E., Oliver-Solà, J., Gasol, C. M., Farreny, R., Rieradevall, J., Gabarrell, X., 2012. Life cycle assessment of energy flow and packaging use in food purchasing. Journal of Cleaner Production 25, 51–59.

Simmons, C., Lewis, K., Barrett, J., 2000. Two feet-two approaches: a component-based model of ecological footprinting. Ecological Economics 32, 375–380.

Steen-Olsen, K., Weinzettel, J., Cranston, G., Ercin, A. E., Hertwich, E. G., 2012. Carbon, land, and water footprint accounts for the European Union: Consumption, production, and displacements through international trade. Environmental Science & Technology 46, 10883−10891.

Stöglehner, G., 2003. Ecological footprint — a tool for assessing sustainable energy supplies. Journal of Cleaner Production 11, 267–277.

Stoeglehner, G., Narodoslawsky, M., 2008. Implementing ecological footprinting in decision-making processes. Land Use Policy 25, 421–431.

Tukker, A., Bulavskaya, T., Giljum, S., De Koning, A., Lutter, S., Simas, M., Stadler, K., Wood, R. The Global Resource Footprint of Nations: Carbon, Water, Land and Materials Embodied in Trade and Final Consumption. http://exiobase.eu/index.php/publications/creea-booklet/73-creea-booklet-web-resolution/file.

Tukker, A., Poliakov, E., Heijungs, R., Hawkins, T., Neuwahl, F., Rueda-Cantuche, J. M., Giljum, S., Moll, S., Oosterhaven, J., Bouwmeester, M., 2009. Towards a global multi-regional environmentally extended input–output database. Ecological Economics 68, 1928−1937.

Van den Bergh, J. C. J. M., Verbruggen, H., 1999. Spatial sustainability, trade and indicators: an evaluation of the 'ecological footprint'. Ecological Economics 29, 61–72.

Van den Bergh, J., Grazi, F., 2010. On the policy relevance of ecological footprints. Environmental Science & Technology 44, 4843–4844.

Van Oel, P. R., Mekonnen, M. M., Hoekstra, A. Y., 2009. The external water footprint of the Netherlands: Geographically-explicit quantification and impact assessment. Ecological Economics 69, 82–92.

Venetoulis, J., Talberth, J., 2008. Refining the ecological footprint. Environment Development and Sustainability 10, 441–469.

Virtanen, Y., Kurppa, S., Saarinen, M., Katajajuuri, J. M., Usva, K., Mäenpää, I., Mäkelä, J., Grönroos, J., Nissinen, A., 2011. Carbon footprint of food–approaches from national input–output statistics and a LCA of a food portion. Journal of Cleaner Production 19, 1849–1856.

Vogelsang, K. M., 2002. Footprint: ignoring the facts that don't fit the theory. Nature 420, 267.

Wackernagel, M., Rees, W. E., 1996. Our Ecological Footprint: Reducing Human Impact on the Earth. New Society, Gabriola Island, British Columbia.

Wackernagel, M., Rees, W. E., 1997. Perceptual and structural barriers to investing in natural capital: Economics from an ecological footprint perspective. Ecological Economics 20, 3–24.

Wackernagel, M., Lewan, L., Hansson, C.B., 1999a. Evaluating the use of natural capital with the ecological footprint. Ambio 28, 604–612.

Wackernagel, M., Onisto, L., Bello, P., Callejas Linares, A., Susana López Falfán, I., Méndez García, J., Isabel Suárez Guerrero, A., Guadalupe Suárez Guerrero, M., 1999b. National natural capital accounting with the ecological footprint concept. Ecological Economics 29, 375–390.

Walsh, C., Moles, R., O'Regan, B., 2010. Application of an expanded sequestration estimate to the domestic energy footprint of the Republic of Ireland. Sustainability 2, 2555–2572.

Wang, F., Sims, J. T., Ma, L., Ma, W., Dou, Z., Zhang, F., 2011. The phosphorus footprint of China's food chain: implications for food security, natural resource management, and environmental quality. Journal of Environmental Quality 40, 1081–1089.

Weidema, B.P., Thrane, M., Christensen, P., Schmidt, J., Løkke, S., 2008. Carbon footprint: A catalyst for life cycle assessment. Journal of Industrial Ecology 12, 3–6.

Weinzettel, J., Hertwich, E. G., Peters, G. P., Steen-Olsen, K., Galli, A., 2013. Affluence drives the global displacement of land use. Global Environmental Change 23, 433–438.

Wiedmann, T., 2009a. A first empirical comparison of energy Footprints embodied in trade — MRIO versus PLUM. Ecological Economics 68, 1975–1990.

Wiedmann, T., 2009b. Editorial: carbon footprint and input–output analysis–an introduction. Economic Systems Research 21, 175–186.

Wiedmann, T., Barrett, J., 2010. A review of the ecological footprint indicator—perceptions and methods. Sustainability 2, 1645–1693.

Wiedmann, T., Minx, J., 2008. A definition of 'carbon footprint'. In: Pertsova, C. C. (Ed.), Ecological Economics Research Trends. Nova Science Publishers, Hauppauge, NY, USA, pp. 1–11 (Chapter 1).

Wright, L. A., Coello, J., Kemp, S., Williams, I., 2011. Carbon footprinting for climate change management in cities. Carbon Management 2, 49–60.

WWF (World Wildlife Fund), ZSL (Zoological Society of London), GFN (Global Footprint Network), 2012. Living Planet Report 2012. http://awsassets.panda.org/downloads/1_lpr_2012_online_full_size_single_pages_final_120516.pdf.

Xue, X. B., Landis, A. E., 2010. Eutrophication potential of food consumption patterns. Environmental Science & Technology 44, 6450–6456.

Yaap, B., Struebig, M. J., Paoli, G., Pin Koh, L., 2010. Mitigating the biodiversity impacts of oil palm development. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources 5, No. 19. http://www.daemeter.org/wp-content/files/Yaap_etal_2010_CABReviews_5.pdf.

Yu, Y., Hubacek, K., Feng, K., Guan, D., 2010. Assessing regional and global water footprints for the UK. Ecological Economics 69, 1140–1147.

Zhao, S., Li, Z., Li, W., 2005. A modified method of ecological footprint calculation and its application. Ecological Modelling 185, 65–75.

Zhao, S., Song, K., Gui, F., Cai, H., Jin, W., Wu, C., 2013. The emergy ecological footprint for small fish farm in China. Ecological Indicators 29, 62–67.

Zhao, X., Chen, B., Yang, Z. F., 2009. National water footprint in an input–output framework—a case study of China 2002. Ecological Modelling 220, 245–253.

Chapter 3

Investigating the inventory and characterization aspects of footprinting methods

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Abstract

Inventory and characterization schemes play different roles in shaping a variety of footprint indicators. This chapter performs a systematic and critical investigation into the hidden inventory aspect and characterization aspect of selected environmental footprints with implications for classification and integration of those footprints. It shows that all of the carbon, water, land and material footprints have two fundamentally distinct versions, addressing the environmental exchange of substances in terms of emissions and/or extractions either at the inventory level or at the impact assessment level. We therefore differentiate two broad categories of environmental footprints, namely, the inventory-oriented footprints (IVOFs) and the impact-oriented footprints (IPOFs). The former allow for a physical interpretation of human pressure by inventorying emissions and extractions and aggregating them with value-based weighting factors, whereas the latter assess and aggregate the inventory results according to their potential contributions to a specific environmental impact using science-based characterization factors, with the recognition that these contributing substances are too different to be compared by mass, volume or area. While both categories have individual strengths and weaknesses, the IPOFs have a better performance than the IVOFs on the integration of footprints into a single-score metric in support of policy making. Resembling the general procedure for life cycle impact assessment, we formulate a three-step framework for characterization, normalization and weighting of a set of IPOFs to yield a composite footprint index, which would allow policy makers to better assess the overall environmental impacts of entities at multiple scales ranging from single products, organizations, nations, even to the whole economy. The main value added of this chapter is the establishment of a unified framework for structuring, categorizing and integrating different footprints. It may serve as a starting point for clearing the footprint jungle and for facilitating the ongoing discourse on a truly integrated footprint family.

3.1. Introduction

Over the past years, a rapid expansion of footprint-style indicators has been introduced by companies, governmental bodies and non-governmental organizations, particularly in the field of environmental and sustainability sciences, with the goal of providing a series of pictures of what types of burden are imposed on the planet's environment, and to what extent. Nowadays footprints have reached worldwide popularity, and the environmental issues they are addressing become increasingly diverse, such as climate change (carbon footprint), freshwater use (water footprint), land use (land footprint), material use (material footprint), and so on.

Despite the prevalence of footprint indicators, most studies are narrowed down to one or a few footprints; this, however, brings the risk of problem shifting, as decline in one footprint is often accompanied by undesirable increase in others. For instance, although climate change in many cases dominates the total environmental footprints of a product (Finkbeiner et al., 2014; Page et al., 2012), reducing the carbon footprint is found to lead to a remarkable increase in other footprints (Laurent et al., 2012). Similarly, De Meester et al. (2011) report that 27% of a bioproduct's carbon footprint is cut at the expense of 93% extra land, water and material footprints.

Since environmental issues are getting more and more complex arising from an ever-expanding number of stressors and their interactions (Chapman and Maher, 2014), a shift of focus from issues in isolation to simultaneous assessment in an overall view is needed. Consequently, the concept of "footprint family" was born, with the aim of informing policy makers about the overall environmental burden under a single framework without losing the complexity of the big picture (Galli et al., 2012). It was originated from the combination of the classical carbon, water and land footprints, but gradually extended to accommodate more emerging footprints (Fang et al., 2014; Ridoutt and Pfister, 2013b).

The footprint family concept implies the importance of finding ways to trade-off between different footprints and to minimize the total environmental footprints from a system perspective, rather than emphasizing "net zero" solutions to individual footprints. This gives rise to concern for weighting, as trade-offs among footprints normally cannot be undertaken without any form of weighting (Finnveden et al., 2009; Ridoutt and Pfister, 2013b). The weighting sets have always been a highly controversial subject throughout integrated environmental assessment (Ahlroth et al., 2011). The difficulty of taking such a practice lies in the choice of weighting methods and in the way to deal with uncertainty (Finnveden et al., 2009). This is why weighting practices are basically lacking in present footprint family studies.

Nevertheless, when looking back at how different environmental footprints are structured, we notice that some employ an inventory analysis merely, whereas some others perform an inventory analysis but also an impact characterization. In many cases, unfortunately, the underlying structure has been executed implicitly and remains unexamined by footprint users. It is our conviction that lessons which can be learned from the hidden elements in single footprints will enormously facilitate the ongoing scientific discussions on footprint indicators, including the classification (Čuček et al., 2012), the complementary use and combination in a footprint family (Fang et al., 2014; Galli et al., 2012), and even a single weighted footprint metric (Ridoutt and Pfister, 2013b). This study may also be well connected to the policy domain, with potential to inform and support the development of existing environmental policy frameworks and projects, such as Product Environmental Footprint (PEF) (EC, 2015), Environmental Footprint Analysis (EPA, 2014), PAS 2050 (BSI, 2011), and the related ISO standards (e.g., ISO, 2006).

This chapter aims to propose a general conceptual and mathematical structure that underlies most, if not all, environmental footprints that are en vogue at present, to achieve a harmonization of structure, terminology and notation, to distinguish the inventory aspect and characterization aspect of different footprints, and to provide clarity on some theoretical issues underlying footprint methods. To that end, the remainder of this chapter is structured as follows: Section 3.2 critically examines the inventory analysis and impact characterization in each of the selected footprints; Section 3.3 offers insights on the implications of our findings for the classification and integration of footprints; discussion and conclusions are presented in Sections 3.4 and 3.5, respectively.

3.2. Investigation into the inventory and characterization aspects of selected environmental footprints

3.2.1. Overall terminology and structure of the analysis

In theory, inventory analysis and impact characterization are two successive steps for quantitatively modeling the consequences of man's exploitation of the nature. The fundamentals of the two elements are briefly stated as follows (Finkbeiner et al., 2014; Finnveden et al., 2009; Hauschild et al., 2013; Heijungs and Suh, 2002; Hellweg and Milà i Canals, 2014; Udo de Haes and Heijungs, 2009):

- **Inventory analysis**: a step aimed at tabulating and compiling the exchange of substances (i.e., emission of wastes to and extraction of resources from the environment) within the boundary of an investigated system (e.g., product, organization, nation). In the framework of life cycle assessment (LCA), this corresponds to life cycle inventory (LCI), a compilation of the inputs (resources) and the outputs (emissions) within the system boundaries of the study across its life cycle. The input and output substances are called elementary flows according to the ISO

(2006). In the framework of substance flow analysis (SFA), it corresponds to the system definition and quantification (Van der Voet et al., 1995). In some analytical tools, this activity has no specific name, but it is recognizable as such; see, for instance, Eurostat (2014) for economy-wide material flow accounts (EW-MFA) and Miller and Blair (2009) for input–output analysis (IOA).

- **Impact characterization**: a subsequent step aimed at assessing the inventory results according to their relative contributions to a specific environmental impact or a set of environmental impacts. In LCA, the contributing elementary flows are quantified using characterization factors and translated to common impact units to make them comparable and ready for aggregation into impact indicators. This step is known as life cycle impact assessment (LCIA), where characterization factors are derived from science-based models reflecting the environmental mechanism underlying the impact category under assessment. In MFA—an analytical tool to quantify material flows in well-defined systems, such steps are part of the interpretation of results (Van der Voet et al, 1995). Again, in EW-MFA and IOA, this activity is often present, although without an explicit name. Characterization factors are part of the LCIA-specific jargon, but such factors are used by many other studies as well (e.g., Fuglestvedt et al., 2008; Skeie et al., 2009).

A general mathematical framework for the two steps is as follows. Let M_i be the quantified emission or extraction of substance i (e.g., kg, kg/yr, m^3/yr). Inventory analysis proceeds according to:

$$M_i = \sum_k M_{ik} \tag{3.1}$$

where subscript k denotes all activities that emit or extract substance i within the system boundaries. The resulting inventories of the investigated system can be characterized with substance-specific characterization factors for a chosen impact category (e.g., climate change, resource scarcity) at midpoint or endpoint level:

$$I_j = \sum_i M_i \times cf_{ij} \tag{3.2}$$

where I_j is the indicator result for impact j (e.g., kg-eq., kg-eq./yr); and cf_{ij} is the characterization factor for substance i in relation to impact j (e.g., kg-eq./kg, m^3-eq./kg).

Alternatively, the resulting inventories of an investigated system can be weighted with weighting factors at the option of the users, particularly in cases where well-grounded characterization factors are not sufficiently available. We consider this as part of the inventory analysis, since no impact assessment has been done in the weighting step:

$$I'_j = \sum_i M_i \times iwf_{ij} \quad (3.3)$$

where I'_j is the indicator result j (e.g., kg, m^3, ha); and iwf_{ij} is the inventory weighting factor for substance i in relation to impact j (mostly dimensionless).

Note that, in this chapter, "inventory data" refers to the inputs of the inventory calculations (e.g., unit process data), "inventory results" in contrast refers to the outputs of the inventory calculations (e.g., system-wide emissions and extractions), and the weighted inventory results is out of the scope of these terms. Furthermore, weighting substances within a footprint, namely, inventory aggregation, should be distinguished from weighting footprints, which will be discussed in the following sections. For convenience, we name the former as "(inventory) weighting" and the latter as "(footprint) weighting". By contrast, in the field of LCA, the term "weighting" is restricted to what here we call footprint weighting (ISO, 2006). Inventory weighting is not recognized in the LCA standards, but in other tools for environmental assessment it may occur from time to time (e.g., Hoekstra and Hung, 2002).

In keeping with the framework described above, criteria are developed for selection prior to the evaluation of footprint indicators: (1) documentation of competing versions; (2) transparency in each of the methodologies; and (3) applicability to classification. For these concerns, a systematic and critical investigation into the inventory analysis and/or impact characterization will be presented specifically for four footprints: the carbon footprint, the water footprint, the land footprint, and the material footprint. For each of them, we identify two basic versions. It is worth mentioning that contrary to common practices, the ecological footprint in this chapter is presented in a disaggregate form. It means that we choose to separately investigate its abiotic and biotic components, as will be illustrated below.

3.2.2. Carbon footprint

3.2.2.1. Climate-related carbon footprint (CF_{clim})

The widely used climate-related carbon footprint (CF_{clim}) compiles the inventory data of greenhouse gas (GHG) emissions (e.g., CO_2, CH_4, N_2O) throughout the investigated system boundaries, and characterizes and aggregates them into a mass equivalent metric (e.g., kg CO_2-eq.) on the basis of their respective global warming potentials (GWPs)—a global-specific characterization factor representing the integrated radiative forcing over a specified time horizon, with a reference to CO_2 (IPCC, 2014; Wiedmann and Minx, 2008). While being arbitrarily configured by default to report a time horizon of 100-yr in practice, the GWP is deemed one of the most established and consensus-based characterization factors in LCIA (Hauschild et al., 2013; Laurent et al., 2012), and is also used much outside LCA. International consensus has thus been reached on the characterization modeling used for the CF_{clim} (Hellweg and Milà i Canals, 2014; Ridoutt and Pfister, 2013b). For the inventory part, however, there is much less agreement. In LCA, for instance, the issue of allocation of multi-functional processes is a nagging topic, and there is also disagreement on the accounting of biogenic carbon. In IOA, an example of the controversy is the choice between commodity-by-commodity tables and industry-by-industry tables. Once such inventory choices have been made, the characterization aspect runs smoothly.

3.2.2.2. Classical carbon footprint (CF_{class})

Although the carbon footprint is generally referred to as the CF_{clim}, the classical version of the carbon footprint (CF_{class}) was introduced about two decades ago by Wackernagel and Rees (1996), who convert carbon emissions from fossil energy into the emissions of CO_2 and then into the area of forest required to achieve carbon neutrality. This is why people also call it "energy footprint" (Fang et al., 2014). The CF_{class} premises that all carbon emissions are ideally emitted in the form of CO_2. Carbon contents (CCs) and carbon sequestration rate (CSR) are employed as parameters to make the two phases of conversion come true. Since the CSR is practically assigned with a fixed value of 970 kg/(ha·yr) for all cases (Blomqvist et al., 2013), the aggregation of multiple carbonaceous substances can be seen as a rough inventory analysis, in which the ratio of energy-specific CCs to the CSR serves as weighting factors for different types of fossil energy. It is not surprising that the scientific reliability and accuracy of its estimate have been debated (Blomqvist et al., 2013; Kitzes et al., 2009), thus excluding the CF_{class} from mainstream acceptance.

3.2.3. Water footprint

3.2.3.1. Classical water footprint (WF_{class})

The classical water footprint (WF_{class}) has received great popularity in past years, and its applications have been extended to a wide range of fields, in particular to the arena of basin water resources management (Hoekstra et al., 2009; Hoekstra and Hung, 2002). It measures and sums up the volumetric consumption of green, blue and gray water within an investigated system at the inventory level. The former two water components refer to the cubic meters of freshwater evaporated from the soil and ground surface, respectively, and the latter is expressed as the cubic meters of freshwater needed to dilute pollution below an acceptable standard (Chapagain and Hoekstra, 2008). The WF_{class} in general can be interpreted as an inventory analysis with regard to H_2O. However, its gray component is, to some extent, committed to an impact assessment because of the large similarity between the dilution approach used and certain characterization approaches (Kounina et al., 2013). Finally, the green, blue and gray components are aggregated without a clear characterization principle, signaling an inventory weighting aspect (even though the weighting factors are 1, i.e., equal weighting).

3.2.3.2. Scarcity-related water footprint (WF_{scarc})

In contrast to carbon emissions, which mix globally, water is a highly heterogeneous resource with varying impacts from region to region, which means that the same volumetric water consumption in places could trigger different degrees of water scarcity (Berger and Finkbeiner, 2013). Consequently, it is probably desirable to take local water scarcity into account. The scarcity-related water footprint (WF_{scarc}) complies with the inventory analysis of consumptive water use, and brings the water inventories into an impact characterization model where blue and/or green water are characterized with location-specific withdrawal-to-availability (WTA) ratios or analogous indices (Lenzen et al., 2013; Ridoutt and Pfister, 2010). It proceeds with the recognition that scarcity is the key to understanding the environmental relevance of local water consumption (Ridoutt and Huang, 2012). The final result of the WF_{scarc} is expressed in a H_2O-equivalent volumetric unit (e.g., m^3 H_2O-eq.) (Zonderland-Thomassen et al., 2014).

3.2.4. Land footprint

3.2.4.1. Classical land footprint (LF_{class})

Within the footprint family, the first quantification exercise is the case of the classical land footprint (LF_{class}) (Wackernagel and Rees, 1996). It converts hundreds of primary and secondary bioproducts into the area of cropland, grassland, woodland and fishing ground expressed in world-average bioproductivity. This conversion essentially corresponds to an inventory analysis, irrespective of the approximation of such a process. Bioproductivity, defined as an estimate of the biological production of a specific land type that can renewably support for human consumption (Kitzes et al., 2009; Lenzen et al., 2007), should not be viewed as a characterization factor due to the ambiguity of the impact assessed and the lack of confirmed environmental mechanism. According to the latest National Footprint Accounts (NFA) for the LF_{class}, the inventory weighting of different land use types is fulfilled through equivalence factors (EQFs)—a type of expert knowledge-based weighting assuming that the most suitable land type will be planted to cropland, and that woodland, grassland and fishing ground would be the second, third, and last choice (Borucke et al., 2013).

3.2.4.2. Disturbance-related land footprint ($LF_{disturb}$)

Having realized that the intensity of human-induced changes to land use considerably varies independently of bioproductivity, Lenzen and Murray (2001) come up with a revised version of the land footprint describing the degree of land disturbance ($LF_{disturb}$). All land types are reclassified and expressed in disturbed hectares, by multiplying the land inventories resulting directly from land cover survey or remote sensing with relative land disturbance factors (LDFs) (Kitzes et al., 2009; Lenzen et al., 2007). Contrary to the practitioners' belief, we consider LDF to be a characterization factor more than a weighting factor because disturbance has been identified as one of land use impacts (Udo de Haes, 2006). One may argue that the characterization model implemented in the $LF_{disturb}$ has not been formally validated by the global community. This is true, but land use in LCA is confronted with the same challenge (Klinglmair et al., 2014). For instance, while the ReCiPe method (Goedkoop et al., 2009) is the best among the existing characterization models for recommendation, the databases are nevertheless far from satisfactory and thus leave much room for improvement (Hauschild et al., 2013).

3.2.5. Material footprint

3.2.5.1. Classical material footprint (MF_{class})

The classical material footprint (MF_{class}) is a measure of abiotic and biotic resource use by adding up the quantity of a wide range of raw material consumption (e.g., metals, minerals, fossil energy) (Schoer et al., 2012; Wiedmann et al., 2015). In analogy to the LF_{class}, the MF_{class} commits to not only an inventory calculation, but also an aggregation of the inventory results. By means of the MF_{class}, one can be easily aware of the total resource needs of an investigated system. Interestingly, there is a tradition of summing up a great variety of materials at the kilogram level, as exemplified by the program of "the weight of nations" (Matthews et al., 2000). It should be noted that inventorying in this way is subject to equal weighting, implying that all material categories are regarded as equally important. It is likely to come to the conclusion that the MF_{class} is well suited to facilitating the dematerialization of production and consumption processes, rather than to reducing associated environmental impacts, as the ranking order of materials based on their environmental impacts can significantly deviate from that based on mass (Van der Voet et al., 2004).

3.2.5.2. Scarcity-related material footprint (MF_{scarc})

One obvious impact arising from material extraction is resource scarcity. Although different material categories are quite distinct in nature, they simultaneously contribute to the depletion of natural capital. A scarcity-related material footprint (MF_{scarc}) has been proposed in this context (Fang and Heijungs, 2014a). Given that water and land—two special but elemental resource categories have already been addressed by the water and land footprint accounts, and that most biotic resources can be reproduced by a production process and that this would hardly contribute to scarcity (Guinée et al., 1993; Guinée and Heijungs, 1995), the MF_{scarc} presently restricts its inventory analysis and impact assessment to the abiotic aspect. The MF_{scarc} is distinguished from the MF_{class} in that it substitutes abiotic depletion potentials (ADPs) for equal weighting factors and in that the outcome is expressed in equivalent units, with antimony (Sb) as a reference substance (e.g., kg Sb-eq.). It allows to characterize abiotic resources with respect to relative scarcity and to translate the overall risk of abiotic depletion into an understandable measure of kilograms (Fang and Heijungs, 2014a).

3.3. Lessons from the investigation for the classification and integration of environmental footprints

3.3.1. General lessons

On the basis of the investigation carried out, Figure 3.1 compares some key issues concerning the inventory aspect (including inventory weighting) and the characterization aspect across the footprints aforementioned. Major findings from the investigation are drawn as follows:

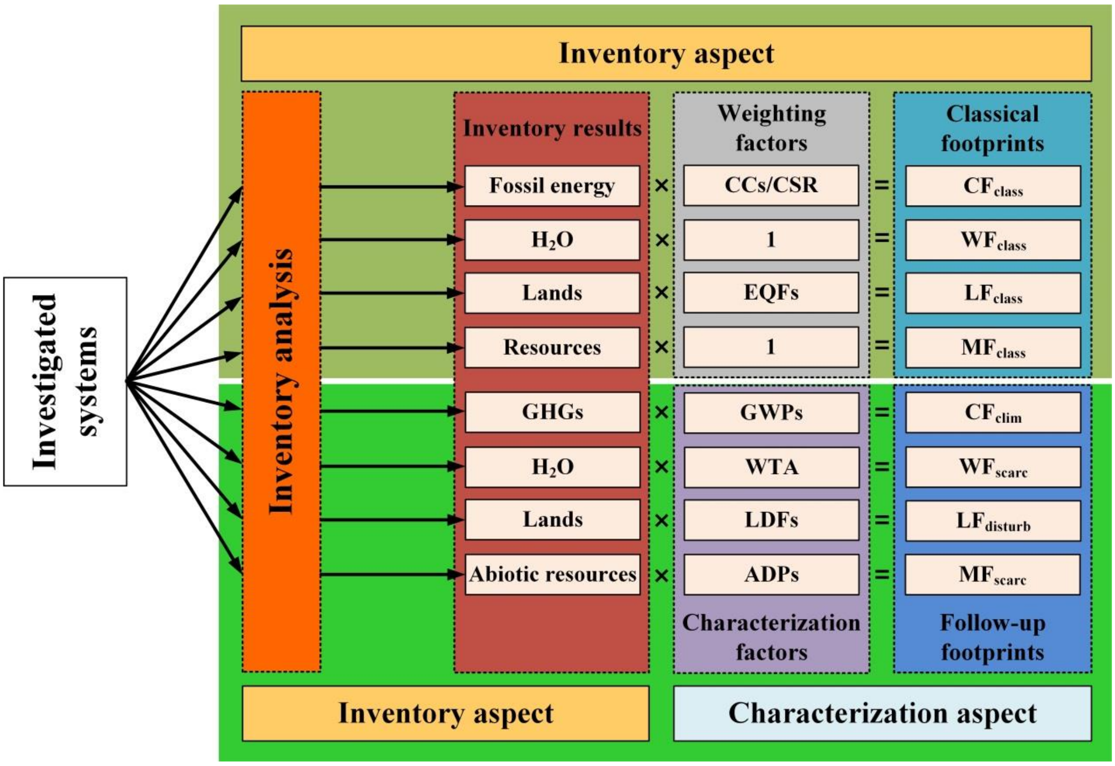

Figure 3.1. Schematic representation of the inventory and characterization aspects of selected footprints. While the inventory results are by and large aligned (some subtle points for CF_{class} and LF_{class} ignored here), two categories of footprints can be distinguished: those that weight inventory results, and those that characterize inventory results.

a) Each of the investigated footprints performs an inventory analysis, whereby data of wastes emitted to and/or resources extracted from the environment are tabulated and compiled specifically for defined system boundaries. The inventory can be LCA-based, and can be non-LCA-based as well. Alternatives to LCA include (parts of) IOA, MFA, NFA, and so on.

b) Some of the footprints further characterize the inventory results with characterization factors to yield single-score impact indicators expressed in equivalent units (e.g., kg CO_2-eq., m^3 H_2O-eq.), which is common to all contributing substances within each single impact category. This practice builds on the premise that environmental mechanism underlying the characterization modeling of the impact exists and can be modeled in a linear approximation as simple multiplication and addition.

c) In other cases, the inventory results are aggregated with weighting factors reflecting the relative importance based on expert-based judgments that contain social, political or ethical value choices in the study. It can be equal weighting in case that all factors are assigned with the same score, as is the case for the WF_{class} and for the MF_{class}. Unequal weighting is observed in the cases of the CF_{class} and LF_{class}. Whichever the weighting scheme, we consider this as part of inventory analysis, as no impact assessment has been done.

d) While the operational formulas are similar in style (both categories of footprints do a multiplication and addition), inventory weighting and impact characterization play different roles in bringing together various inventory results into a footprint. Weighting is value-based, typically using expert knowledge to express stated or revealed preferences and judgments. In contrast, characterization models are developed on the basis of recent scientific evidence, even though subjective choices cannot be completely avoided. The CF_{clim}, for instance, has substantially benefitted from the use of GWPs and had a much broader appeal than many other footprints, despite the arbitrary choice of time horizon.

e) Although the dichotomous framework for contrasting inventory weighting and impact characterization reveals valuable insights into the similarities and differences between these footprints, we admit that there is no sharp boundary between value-based weighting and science-based characterization. One example is the confusion surrounding the nature of the LDF used in the $LF_{disturb}$. This can be primarily attributed to the fact that a recognized consensus on land use characterization modeling is lacking (Hauschild et al., 2013). In view of the difficulties in assessing land use impacts, the $LF_{disturb}$ shows a relatively good performance on scientific robustness by tracing the causal linkages between biotic resource extraction, soil degradation, and biodiversity decline.

3.3.2. Lessons for the classification of footprints: a two-category framework

Simply put, environmental footprints are defined as indicators that measure anthropogenic pressure or associated impacts placed on the environment by human actions, irrespective of their precise units and dimensions. Rather than carrying on the never-ending debate over the definition of footprints, we propose shifting the focus to the classification of footprints which, in our view, is the key to making sense of the footprint concept. In accordance to the investigation conducted above, two broad categories of footprints are identified, namely, the IVOFs and IPOFs. This is a rough dichotomy with respect to the ways in which a growing number of footprints are shaped. In the following we elaborate on the two categories of footprints individually, together with a comparison of their inherent properties.

3.3.2.1. Inventory-oriented footprints (IVOFs)

The IVOFs are the footprints that compile and weight the inputs and outputs of environmental exchange (i.e., hazardous emissions and resource extractions) at the inventory level. It builds on the belief that physical units (e.g., ha, m^3, kg) provide an intuitive and intrinsic understanding of the anthropogenic pressure due to emissions and extractions, which is probably the original intent of what a footprint is supposed to convey to the public. This valuable information, however, is found to be lost when translating into an impact score through characterization modeling (Hoekstra et al., 2009). Apparently, all the four classical footprints, namely the CF_{class}, WF_{class}, LF_{class} and MF_{class}, fall under the scope of the IVOFs.

3.3.2.2. Impact-oriented footprints (IPOFs)

The IPOFs are those which not only comply with an inventory analysis but also characterize the inventory results through their respective contributions to a given impact category at the impact assessment level. The rationale behind the IPOFs is that contributing substances differ in their effects on the same impact category (sometimes by orders of magnitude), and therefore cannot be compared or aggregated without the use of characterization factors reflecting the up-to-date scientific knowledge about the effects of the impact category assessed on the environment and human health. We classify the CF_{clim}, WF_{scarc}, $LF_{disturb}$ and MF_{scarc} as members of the IPOFs. Nevertheless, one should be aware that while some IPOFs use physical-look-alikes units (e.g., the CF_{clim} uses kg CO_2-eq.), such units differ markedly from the real measurable footprints in the IVOF category.

3.3.2.3. Comparison of the two footprint categories

An overall comparison between the IVOFs and IPOFs in terms of some inherent properties is presented in Table 3.1. As shown, the two categories of footprints differ mostly in the ways they address the inventory results. The IVOFs apply value-based weighting to the aggregation of inventory results, whereas the IPOFs are more concerned with the consequences for environmental quality by processing the inventory results with science-based characterization factors.

To further delineate the difference of interpretation between the two footprint categories, we briefly present an illustrative comparison of the WF_{class} and WF_{scarc} for a product system. From an LCA perspective, the WF_{class} deals with the inventory aspect of water use over the product's life cycle, which could serve as a preparatory step to advanced impact assessment. In other words, a subsequent step is to assess the associated environmental impacts using one or more characterization models. Water scarcity that the WF_{scarc} describes is just one of the many environmental consequences resulting from consumptive and degradative water use. Other impacts at midpoint level may include eutrophication (Zonderland-Thomassen et al., 2014) and human toxicity (Boulay et al., 2011), for instance. This is in contrast to endpoint characterization where impacts refer to damage to several of the areas of protection, like human health (Ridoutt and Pfister, 2013a) and ecosystem quality (Hanafiah et al., 2011). All this goes beyond water quantity and thereby constitutes an expanding list of impact/damage-oriented water footprints.

Table 3.1. Comparison of the inventory-oriented and impact-oriented footprints.

Item	Inventory-oriented footprints (IVOFs)	Impact-oriented footprints (IPOFs)
Examples	CF_{class}, WF_{class}, LF_{class}, MF_{class}	CF_{clim}, WF_{scarc}, $LF_{disturb}$, MF_{scarc}
Factors for inventory aggregation	Weighting factors: reflecting the relative importance based on value-based judgments. Easily obtainable but subjective and not reflecting the latest scientific knowledge.	Characterization factors: representing the relative contributions of substances to a specific impact category. Primarily science-based although including some subjective choices.
Rationale	Inventory analysis of contributing substances enables a physical interpretation of human pressure and this may disappear if translating into an impact result through characterization modeling.	Different substances per physical unit vary so notably within any single impact category that they cannot be directly compared and aggregated at the inventory level.
Strengths	Compiling different emissions and extractions with understandable physical units and laying an inventory basis for communicational purposes.	Linking quantitative inventory results to a specific environmental impact and assessing the impact on a scientific characterization basis.
Weaknesses	Not revealing the environmental relevance of different emissions and extractions with respect to their relative contributions to environmental impacts.	Not revealing the physical meaning of emissions and extractions exerted by human activities.
Applicability	Typically oriented towards the macro-level of economy (global, nation-wide, etc.).	Typically oriented towards the micro- or meso-level of individual products or processes.
Policy relevance	Instrumental for policies aimed at measuring the pressure of resource use on the environment and at identifying the driving forces that are causing environmental problems.	Instrumental for policies aimed at assessing potential environmental impacts of emissions and extractions and at finding alternatives for the reduction of impacts.

3.3.3. Lessons for the integration of footprints: a three-step framework

The variation in potential environmental impacts associated with a single stressor (e.g., water use) brings our attention back to the weighting between footprints, referred to as footprint weighting in this chapter. Though the two categories of footprints discussed above both have pros and cons, we argue below that only the IPOFs have the capacity to be integrated into a single composite metric. After that we propose a three-step framework for the integration of IPOFs.

3.3.3.1. The rationale of choosing the IPOFs for integration

There are two main reasons for choosing the IPOFs rather than choosing the IVOFs when it comes to integrating different footprints into one single-score metric:

a) Each IPOF characterized with science-based characterization factor is pinpointing a well-defined impact category such as climate change and water scarcity. Collectively, they are particularly suited to forming an overall picture of the integrated environmental impacts of a product, organization, or nation, without obvious overlapping. This is different from the IVOFs, for which double counting may occur. The WF_{class}, for instance, juxtaposes water use, depletion, and pollution. The pollution part may easily overlap with a chemical or toxic footprint, and it appears difficult to attach a weight to such a heterogeneous footprint.

b) Footprint weighting in the integration of footprints is unavoidable, whereas inventory weighting in shaping a single footprint can be avoided by making use of characterization factors that bring together different inventory results into an IPOF footprint with the best available scientific knowledge on natural resources, human health and environmental impacts. Compared with the IVOFs, the IPOFs have a better performance in the sense that they are able to achieve this integration without performing a double weighting that would certainly compromise the validity of the final estimate.

To sum, double counting and double weighting can be avoided by using the IPOFs rather than using the IVOFs.

3.3.3.2. An initial framework for characterization, normalization and weighting of the IPOFs

Now it seems to be clear that the integration of environmental footprints only makes sense if all involved are members of the IPOF category, to which knowledge of LCIA would be of great benefit. In the LCA framework, LCIA takes place as a subsequent step to LCI, where a number of elementary flows are translated into single impact indicators and, if desired, then weighted and integrated into a comprehensive metric. Essentially, LCIA consists of three major steps: characterization, normalization, and weighting. While according to ISO (2006) the weighting is optional and not recommended for comparative assertions communicated to the public because of the subjectivity and uncertainty, weighting between impact categories greatly facilitates policy making (Hellweg and Milà i Canals, 2014). This is particularly true for analysis aimed at informing policy makers who are in favor of weighted indicators that make alternatives easily comparable (Ahlroth et al, 2011). Therefore, footprint weighting in our view can be considered as part of the established practice although it should be interpreted with caution (Fang et al., 2015). This is moreover a logical extension to the idea of the footprint family that is mainly limited to a conceptual discussion.

To cite an example, we assume that, along the life cycle of a product system, all of the required data are available. In that case, many conceivable IPOFs can be imitated by multiplying the inventory results with corresponding characterization factors such as ozone depletion potential (ODP), terrestrial acidification potential (TAP), and aquatic ecotoxicity potential (AEP). Returning to the mathematical formulation in Section 3.2.1, we calculate a set of *IPOFj* through:

$$IPOF_j = \sum_i M_i \times cf_{ij} \tag{3.4}$$

The outcome is impact indicators expressed in equivalent units (e.g., kg CFC-11-eq., kg SO_2-eq., kg 1,4-DCB-eq.), which are common to all contributing substances within the impact category, but not across impact categories due to incomparable units. This is why normalization is needed after characterization. The calculation principle of normalization is transparent (Heijungs et al., 2007): each normalized footprint $nIPOF_j$ is equal to the ratio of the product footprint to the reference system's footprint for that category $IPOF_{ref,j}$:

$$nIPOF_j = \frac{IPOF_j}{IPOF_{ref,j}} \tag{3.5}$$

The globe is in many cases the most suitable reference system, because a single product may also have a global coverage of waste emission and resource extraction in today's globalized economy (Guinée et al., 2002). However, for some regional-specific impacts, the reference system can specify by a continent scale like Europe or North America (Hauschild et al., 2013; Laurent et al., 2012). The footprints for the reference are usually calculated through inventorying the emissions and extractions of the reference system and subjecting it to the same footprint calculations:

$$IPOF_{ref,j} = \sum_i M_{ref,i} \times cf_{ij} \tag{3.6}$$

By translating abstract impact results into relative contributions of the product to a reference situation, the normalized footprint results (e.g., yr) allow for a direct comparison between impact categories at a broader context. Nevertheless, the sum of normalized footprint results could still be environmentally irrelevant so long as it is not placed in an adequate context (Sleeswijk et al., 2008). For this reason, normalization is often followed by a weighting step in which the footprints of multiple impact categories are integrated into a single composite metric. We name this metric as composite footprint index *CFI*, which is equal to multiplying the normalized results with footprint weighting factor fwf_j for footprint j:

$$CFI = \sum_j nIPOF_j \times fwf_j \tag{3.7}$$

Unlike inventory weighting, footprint weighting in most cases cannot be replaced by a characterization approach given the difficulty of encompassing the full characteristics of divergent environmental impacts without the assistance of any subjective trade-off; with few exceptions, however, such as exergy-based characterization (Huysman et al., 2015). It means that in general value-based weighting is unavoidable when integrating a set of IPOFs into the CFI. This supports, to some extent, that a science communication intended to serve decisions must involve both facts and values (Dietz, 2013). The procedure for deriving weighting factors is beyond the scope of this chapter; see Ahlroth et al. (2011) for an overview.

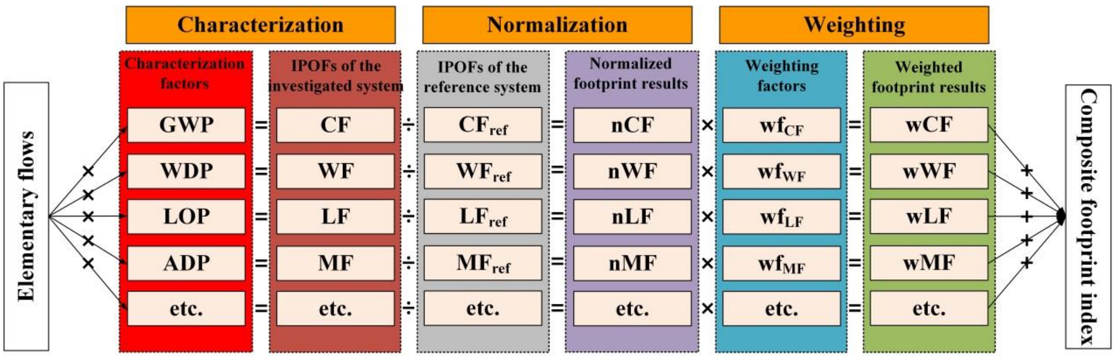

Figure 3.2. Schematic representation of the calculation and integration of a set of impact-oriented footprints under the characterization-normalization-weighting framework. It builds on the premise that all of the necessary data for inventory are sufficiently available. WDP: water depletion potential; LOP: land occupation potential.

Finally, as visualized in Figure 3.2, our proposal for the three-step framework makes a novel contribution to the area and is inherently different from the one that aims to bring together all footprints into LCA, because this framework can be operationalized without a life cycle approach, like for instance what is needed for an organization environmental footprint (OEF) (Fang and Heijungs, 2014b) and, in reverse, doing an LCA does not necessarily follow the inventory-characterization-normalization-weighting (ICNW) logic, like in LCA for EcoDesign (Karlsson and Luttropp, 2006).

3.4. Discussion

The investigation presented in this chapter offers new insights into the inventory aspect and characterization aspect of various footprinting methods, an important subject but being neglected by the majority of footprint practitioners and users. By dividing the selected footprints into two broad categories, namely the IVOFs and IPOFs, this chapter goes to lengths to show what distinguishes a footprint from another that has the same name but a different logic. There is admittedly no sharp boundary between a value-based inventory analysis and a science-based impact characterization due to gaps in scientific knowledge, and neither is there a sharp boundary between an inventory weighting and a footprint weighting. But like there is no such a boundary between dark and light, making the difference is still useful.

One reason for such ambiguity is the fact that subjective choices cannot be completely avoided in shaping an IPOF. The time horizon determined for GWP on which the CF_{clim} strongly depends is a typical example. Another reason is the lack of internationally agreed characterization modeling in some of the identified impact categories. For instance, it is found that even for the same abiotic resource the ADP can differ up to several orders of magnitude depending on the model chosen (Klinglmair et al., 2014). Moreover, resource depletion (including water depletion) is not always seen as a true environmental impact (Finkbeiner et al., 2014), nor the inherent characteristic of abiotic and biotic resources (Hauschild et al., 2013). Then the question comes up: what are the inherent characteristics that should be modeled for resource characterization? The reality is that in many cases resource flows do not have clear input and output characteristics (Udo de Haes, 2006).

In general, there is much more agreement on how to characterize emissions (e.g., GWP, ODP) than on how to characterize resource extractions. This is perhaps the foremost reason why the IPOFs are more advanced for emissions than for other forms of resources (e.g., land, water, material), for which the IVOFs have a considerably wide range of applications. The IVOF category of footprints provides a more practical way of measuring the pressure of resource use that is causing environmental impacts, and this is probably the initial purpose of footprint analysis. Although sometimes there are some simple conversions in the inventory phase of IVOP calculations, valuable physical information is maintained in relation to pressure exerted by the emission of wastes and extraction of resources. These merits, however, are obtained at the expense of undermining environmental relevance because assigned weighting factors are subject to artificial sequences that are almost unwarranted.

According to ISO (2006), footprint weighting is suggested as an option for further evaluation in the context of product LCA. One may easily extend this message to organizations and nations, in order to meet the non-professional demands (e.g., policy making, public communication) for a single-score, stand-alone metric for purposes of easy interpretation and trade-off analysis. In an attempt to standardize the integration of environmental footprints, our study has ascertained that integrating either the IVOFs or a mixture of the IVOFs and IPOFs is technically feasible but essentially meaningless. That is to say, it makes sense only if a set of carefully selected IPOFs are under consideration. Resembling the general procedure for LCIA, a three-step framework has been proposed for characterization, normalization and weighting of the IPOFs. Admittedly this is not much new for LCA experts, but we believe that it provides non-LCA experts with new insights into the fundamental nature of footprints and, more importantly, that it can serve as a unifying framework for discussions in communities that are quite disparate at the present moment, such as those of LCA, the water footprint, and the ecological footprint.

3.5. Conclusions

We herein come up with some conclusions that are supposed to fuel the ongoing discourse on a truly integrated footprint family. First, value-based weighting is hardly avoidable when integrating different footprint indicators into a single composite metric, because a thorough characterization model able to capture the full characteristics of related environmental impacts is virtually non-existent. Second, double counting and double weighting can nevertheless be avoided by the substitution of characterization factors for inventory weighting prior to footprint weighting. Third, the two categories of footprints identified offer two competing paradigms for the development of footprint indicators. Last, the ICNW logic could contribute to the standardization of footprint accounting—a more general framework that is inspired by the results of two decades of intense debate in the LCA community, but that can also be fruitful in life cycle-less contexts where there is no clear life cycle or even without an LCA. As a whole, the main value added of this chapter is the establishment of a unified framework for structuring, categorizing and integrating different footprints.

Accordingly, it should be emphasized that our proposal for restricting the integration practices to the IPOFs does not challenge the validity of the IVOFs, as the latter is appropriate for use in identifying the driving forces of environmental issues using understandable physical units, with the advantages of stimulating more forward-looking policy strategies and of reducing reliance on environmental governance at the end of treatment. Since both of the footprint categories have pros and cons, there is a need for more clarity on the applicability and limitations of each of them, as well as for an exploration of their potential synergies. Gaps in current knowledge, such as the unavailability of data and methods for inventory analysis and characterization modeling, and the uncertainty and subjectivity of normalization and weighting schemes, pose barriers to achieving that goal. In addition to the difficulties associated with footprinting integration, the classification issue is far from being settled as well. For instance, things get more complicated when it comes to the gray component of the WF_{class}, of which the quasi-characterization nature moves a bit away from our two-category framework. Responding to all these challenges that have to be confronted in proceeding with the development of footprint methodologies, we argue for multidisciplinary and interdisciplinary collaboration between footprint users and non-footprint users.

References

Ahlroth, S., Nilsson, M., Finnveden, G., Hjelm, O., Hochschorner, E., 2011. Weighting and valuation in selected environmental systems analysis tools – suggestions for further developments. Journal of Cleaner Production 19, 145–156.

Berger, M., Finkbeiner, M., 2013. Methodological challenges in volumetric and impact-oriented water footprints. Journal of Industrial Ecology 17, 79–89.

Blomqvist, L., Brook, B. W., Ellis, E. C., Kareiva, P. M., Nordhaus, T., Shellenberger, M., 2013. Does the Shoe Fit? Does the shoe fit? Real versus imagined ecological footprints. PLoS Biology 11, e1001700.

Borucke, M., Moore, D., Cranston, G., Gracey, K., Iha, K., Larson, J., Lazarus, E., Morales, J. C., Wackernagel, M., Galli, A., 2013. Accounting for demand and supply of the biosphere's regenerative capacity: the National Footprint Accounts' underlying methodology and framework. Ecological Indicators 24, 518–533.

Boulay, A. M., Bulle, C., Bayart, J. B., Deschênes, L., Margni, M., 2011. Regional characterization of freshwater use in LCA: Modeling direct impacts on human health. Environmental Science & Technology 45, 8948–8957.

BSI (British Standards Institution), 2011. PAS 2050: 2011 Specification for the Assessment of the Life Cycle Greenhouse Gas Emissions of Goods and Services. British Standards Institution, London, UK.

Chapagain, A. K., Hoekstra, A. Y., 2008. The global component of freshwater demand and supply: an assessment of virtual water flows between nations as a result of trade in agricultural and industrial products. Water International 33, 19–32.

Chapman, P. M., Maher, B., 2014. The need for truly integrated environmental assessments. Integrated Environmental Assessment and Management 10, 151–151.

Čuček, L., Klemeš, J. J., Kravanja, Z., 2012. A review of Footprint analysis tools for monitoring impacts on sustainability. Journal of Cleaner Production 34, 9–20.

De Meester, S., Callewaert, C., De Mol, E., Van Langenhove, H., Dewulf, J., 2011. The resource footprint of biobased products: a key issue in the sustainable development of biorefineries. Biofuels, Bioproducts & Biorefining 5, 570–580.

Dietz, T., 2013. Bringing values and deliberation to science communication. Proceedings of the National Academy of Sciences of the United States of America 110, 14081–14087.

EC (European Commission), 2015. Product Environmental Footprint (PEF). http://ec.europa.eu/environment/eussd/smgp/dev_pef.htm.

EPA (United Sates Environmental Protection Agency), 2014. Environmental Footprint Analysis. http://www.epa.gov/sustainability/analytics/environmental-footprint.htm.

Eurostat, 2014. Material Flow Accounts. http://ec.europa.eu/eurostat/statistics-explained/extensions/EurostatPDFGenerator/getfile.php?file=145.118.86.166_1419264195_10.pdf.

Fang, K., Heijungs, R., 2014a. Moving from the material footprint to a resource depletion footprint. Integrated Environmental Assessment and Management 10, 596–598.

Fang, K., Heijungs, R., 2014b. There is still room for a footprint family without a life cycle approach—comment on "Towards an integrated family of footprint indicators". Journal of Industrial Ecology 18, 71–72.

Fang, K., Heijungs, R., De Snoo, G. R., 2014. Theoretical exploration for the combination of the ecological, energy, carbon, and water footprints: Overview of a footprint family. Ecological Indicators 36, 508–518.

Fang, K., Heijungs, R., De Snoo, G. R., 2015. Understanding the complementary linkages between environmental footprints and planetary boundaries in a footprint–boundary environmental sustainability assessment framework. Ecological Economics 114, 218–226.

Finkbeiner, M., Ackermann, R., Bach, V., Berger, M., Brankatschk, G., Chang, Y.-J., Grinberg, M., Lehmann, A., Martínez-Blanco, J., Minkov, N., Neugebauer, S., Scheumann, R., Schneider, L., Wolf, K., 2014. Challenges in life cycle assessment: An overview of current gaps and research needs. In: Klöpffer, W. (Ed.), Background and Future Prospects in Life Cycle Assessment, LCA Compendium – The Complete World of Life Cycle Assessment. Springer Science+Business Media, Dordrecht, The Netherlands, pp. 207–258.

Finnveden, G., Hauschild, M. Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in life cycle assessment. Journal of Environmental Management 91, 1–21.

Fuglestvedt, J., Berntsen, T., Myhre, G., Rypdal, K., Skeie, R. B., 2008. Climate forcing from the transport sectors. Proceedings of the National Academy of Sciences of the United States of America 105, 454–458.

Galli, A., Wiedmann, T., Ercin, E., Knoblauch, D., Ewing, E, Giljum, S., 2012. Integrating ecological, carbon and water footprint into a "Footprint Family" of indicators: Definition and role in tracking human pressure on the planet. Ecological Indicators 16, 100–112.

Goedkoop, M. J., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., Van Zelm, R., 2009. ReCiPe 2008: A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level (First Edition). Report I: Characterisation. http://www.leidenuniv.nl/cml/ssp/publications/recipe_characterisation.pdf.

Guinée, J. B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., De Koning, A., Van Oers, L., Sleeswijk, A.W., Suh, S., Udo de Haes, H. A., De Bruijn, H., Van Duin, R., Huijbregts, M. A. J., Lindeijer, E., Roorda, A. A. H., Van der Ven, B. L., 2002. Handbook on Life Cycle Assessment. An Operational Guide to the ISO Standards. Kluer Academic Publishers, Dordrecht, The Netherlands.

Guinée, J. B., Heijungs, R., 1995. A proposal for the definition of resource equivalency factors for use in product life-cycle assessment. Environmental Toxicology and Chemistry 14, 917–925.

Guinée, J. B., Heijungs, R., Udo de Haes, H. A., Huppes, G., 1993. Quantitative life cycle assessment of products: 2. Classification, valuation and improvement analysis. Journal of Cleaner Production 1, 81–91.

Hanafiah, M. M., Xenopoulos, M. A., Pfister, S., Leuven, R. S., Huijbregts, M. A., 2011. Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction. Environmental Science & Technology 45, 5272–5278.

Hauschild, M. Z., Goedkoop, M., Guinée, J., Heijungs, R., Huijbregts, M., Jolliet, O., Margni, M., De Schryver, A., Humbert, S., Laurent, A., Sala, S., Pant, R., 2013. Identifying best existing practice for characterization modeling in life cycle impact assessment. The International Journal of Life Cycle Assessment 18, 683–697.

Heijungs, R., Guinée, J., Kleijn, R., Rovers, V., 2007. Bias in normalization: Causes, consequences, detection and remedies. The International Journal of Life Cycle Assessment 12, 211–216.

Heijungs, R., Suh, S., 2002. The Computational Structure of Life Cycle Assessment. Kluwer Academic Publisher, Dordrecht, The Netherlands.

Hellweg, S., Milà i Canals, L., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science 344, 1109–1113.

Hoekstra, A. Y., Gerbens-Leenes, W., Van der Meer, T. H., 2009. Reply to Pfister and Hellweg: Water footprint accounting, impact assessment, and life-cycle assessment. Proceedings of the National Academy of Sciences of the United States of America 106, E114–E114.

Hoekstra, A. Y., Hung, P. Q., 2002. Virtual Water Trade: A Quantification of Virtual Water Flows between Nations in Relation to International Crop Trade. Value of Water Research Report Series (No. 11), UNESCO-IHE Institute for Water Education, Delft, The Netherlands.

Huysman, S., Sala, S., Mancini, L., Ardente, F., Alvarenga, R. A., De Meester, S., Mathieux, F., Dewulf, J., 2015. Toward a systematized framework for resource efficiency indicators. Resources, Conservation and Recycling 95, 68–76.

IPCC (Intergovernmental Panel on Climate Change), 2014. Climate Change 2014 Synthesis Report. http://www.ipcc.ch/pdf/assessment-report/ar5/syr/SYR_AR5_LONG ERREPORT.pdf.

ISO (International Organisation for Standardisation), 2006. ISO 14044 International Standard. In: Environmental Management – Life Cycle Assessment – Requirements and Guidelines. International Organisation for Standardisation, Geneva, Switzerland.

Karlsson, R., Luttropp, C., 2006. EcoDesign: what's happening? An overview of the subject area of EcoDesign and of the papers in this special issue. Journal of Cleaner Production 14, 1291–1298.

Kitzes, J., Galli, A., Bagliani, M., Barrett, J., Dige, G., Ede, S., Erb, K., Giljum, S., Haberl, H., Hails, C., Jolia-Ferrier, L., Jungwirth, S., Lenzen, M., Lewis, K., Loh, J., Marchettini, N., Messinger, H., Milne, K., Moles, R., Monfreda, C., Moran, D., Nakano, K., Pyhälä, A., Rees, W., Simmons, C., Wackernagel, M., Wada, Y., Walsh, C., Wiedmann, T., 2009. A research agenda for improving national ecological footprint accounts. Ecological Economics 68, 1991–2007.

Klinglmair, M., Sala, S., Brandão, M., 2014. Assessing resource depletion in LCA: a review of methods and methodological issues. The International Journal of Life Cycle Assessment 19, 580–592.

Kounina, A., Margni, M., Bayart, J. B., Boulay, A. M., Berger, M., Bulle, C., Frischknecht, R., Koehler, A., Milà i Canals, L., Motoshita, M., Núñez, M., Peters, G., Pfister, S., Ridoutt, B., Van Zelm, R., Verones, F., Humbert, S., 2013. Review of methods addressing freshwater use in life cycle inventory and impact assessment. The International Journal of Life Cycle Assessment 18, 707–721.

Laurent, A., Olsen, S. I., Hauschild, M. Z., 2012. Limitations of carbon footprint as indicator of environmental sustainability. Environmental Science & Technology 46, 4100–4108.

Lenzen, M., Borgstrom Hansson, C., Bond, S., 2007. On the bioproductivity and land-disturbance metrics of the Ecological Footprint. Ecological Economics 61, 6–10.

Lenzen, M., Moran, D., Bhaduri, A., Kanemoto, K., Bekchanov, M., Geschke, A., Foran, B., 2013. International trade of scarce water. Ecological Economics 94, 78–85.

Lenzen, M., Murray, S. A., 2001. A modified ecological footprint method and its application to Australia. Ecological Economics 37, 229–255.

Matthews, E., Amann, C., Bringezu, S., Fischer-kowalski, M., Hüttler, W., Kleijn, R., Moriguchi, Y., Ottke, C., Rodenburg, E., Rogich, D., Schandl, H., Schütz, H., Van der Voet, E., Weisz, H., 2000. The Weight of Nations: Material Outflows from Industrial Economies. World Resources Institute. http://citeseerx.ist.psu.edu/viewdoc/download;jsessionid=884F4E9CF9267DC7C1BAF57CE7AB3E58?doi=10.1.1.27.8642&rep=rep1&type=pdf.

Miller, R. E., Blair, P. D., 2009. Input-Output Analysis: Foundations and Extensions (2nd Edition). Cambridge University Press, New York, USA.

Page, G., Ridoutt, B., Bellotti, B., 2012. Carbon and water footprint tradeoffs in fresh tomato production. Journal of Cleaner Production 32, 219–226.

Ridoutt, B. G., Huang, J., 2012. Environmental relevance—the key to understanding water footprints. Proceedings of the National Academy of Sciences of the United States of America 109, E1424–E1424.

Ridoutt, B. G., Pfister, S., 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environmental Change 20, 113–120.

Ridoutt, B. G., Pfister, S., 2013a. A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. The International Journal of Life Cycle Assessment 18, 204–207.

Ridoutt, B. G., Pfister, S., 2013b. Towards an integrated family of footprint indicators. Journal of Industrial Ecology 17, 337–339.

Schoer, K., Weinzettel, J., Kovanda, J., Giegrich, J., Lauwigi, C., 2012. Raw material consumption of the European Union – Concept, calculation method, and results. Environmental Science & Technology 46, 8903–8909.

Skeie, R. B., Fuglestvedt, J. S., Berntsen, T., Lund, M. T., Myhre, G., Rypdal, K., 2009. Global temperature change from the transport sectors: Historical development and future scenarios. Atmospheric Environment 43, 6260–6270.

Sleeswijk, A. W., Van Oers, L. F. C. M., Guinée, J. B., Struijs, J., Huijbregts, M. A. J., 2008. Normalisation in product life cycle assessment: An LCA of the global and European economic systems in the year 2000. Science of the Total Environment 390, 227–240.

Udo de Haes, H. A., 2006. How to approach land use in LCIA or, how to avoid the Cinderella effect? Comments on 'Key elements in a framework for land use impact assessment within LCA'. The International Journal of Life Cycle Assessment 11, 219–221.

Udo de Haes, H. A., Heijungs, R., 2009. Analysis of Physical Interactions between the Economy and the Environment. In: Boersema, J. J., Reijnders, L. (Eds.), Principles of Environmental Sciences. Springer, Dordrecht, The Netherlands, pp. 207–237.

Van der Voet, E., Heijungs, R., Mulder, P., Huele, R., Kleijn, R., Van Oers, L., 1995. Substance flows through the economy and environment of a region. Environmental Science and Pollution Research 2, 137–144.

Van der Voet, E., Van Oers, L., Nikolic, I., 2004. Dematerialization: Not just a matter of weight. Journal of Industrial Ecology 8,121–137.

Wackernagel, M., Rees, W. E., 1996. Our Ecological Footprint: Reducing Human Impact on the Earth. New Society, Gabriola Island, Canada.

Wiedmann, T., Minx, J., 2008. A definition of 'carbon footprint'. In: Pertsova, C. C. (Ed.), Ecological Economics Research Trends (Chapter 1). Nova Science Publishers, New York, USA, pp. 1–11.

Wiedmann, T. O., Schandl, H., Lenzen, M., Moran, D., Suh, S., West, J., Kanemoto, K. 2015. The material footprint of nations. Proceedings of the National Academy of Sciences of the United States of America 112, 6271–6276.

Zonderland-Thomassen, M. A., Lieffering, M., Ledgard, S. F., 2014. Water footprint of beef cattle and sheep produced in New Zealand: water scarcity and eutrophication impacts. Journal of Cleaner Production 73, 253–262.

Chapter 4

Life cycle assessment: Nice to have or essential for environmental footprints?

Based on a synthesis of four short publications:

Fang, K., Heijungs, R., 2015. The role of impact characterization in carbon footprinting. Frontiers in Ecology and the Environment 13, 130–131. DOI: 10.1890/15.WB.005.

Fang, K., Heijungs, R., 2014. Moving from the material footprint to a resource depletion footprint. Integrated Environmental Assessment and Management 10, 596–598. DOI: 10.1002/ieam.1564.

Fang, K., Heijungs, R., 2014. There is still room for a footprint family without a life cycle approach: Comment on "towards an integrated family of footprint indicators". Journal of Industrial Ecology 18, 71–72. DOI: 10.1111/jiec.12067.

Fang, K., Heijungs, R., 2015. Rethinking the relationship between footprints and LCA. Environmental Science & Technology 49, 10–11. DOI: 10.1021/es5057775.

Abstract

The role of life cycle assessment (LCA) in footprinting has been a popular subject of discussion in the literature. The satisfactory performance of LCA on environmental impact assessment (EIA) could allow many footprint topics to be addressed under an LCA framework, in particular those that can be measured in relation to a functional unit. The carbon and abiotic resource footprints are presented as two examples of such LCA-based footprints, in which a variety of inventory flows associated with human disturbance are compiled and translated into impact scores on the basis of science-based characterization modeling. On the other hand, however, narrowing environmental footprints down to an LCA context is found to create blind spots, where exhaustive inventory data for compiling or consensus models for characterization of impact pathways are unavailable. Besides, there are certain important types of questions for which a footprint-type representation is desirable but for which a life cycle perspective is not or only partially appropriate. The organization environmental footprint (OEF) is an obvious example of this. As a result, we argue that footprints are not to be interpreted as a new name for the impact category indicators defined in LCA and, more importantly, that LCA does not substitute but complements footprint analysis. Further investigation into the relationship between environmental footprints and LCA would be critical to the development and refinement of both tools.

4.1. Introduction

The communities of environmental footprints and life cycle assessment (LCA) have lived separately for a long time, but with the advent of the carbon footprint as an LCA tool, the two worlds seem to assimilate. However, there is a tremendous amount of ambiguity, confusion, and controversy surrounding the relationship between environmental footprints and LCA. Is an impact category indicator in LCA the same as a footprint? Is a life cycle perspective indispensable for every footprint accounting? In this chapter, we will discuss some of the points where environmental footprints and LCA agree, but also some of the points where they disagree.

4.2. On the strengths of LCA for environmental footprints

In this section, the strengths of LCA to support the measurement of specific impact categories accounted in environmental footprints are illustrated with two examples: the carbon footprint and the material footprint.

4.2.1. The role of impact characterization in the carbon footprint

The ever-accelerating growth in atmospheric carbon emissions poses the greatest anthropogenic disturbance to the Earth's climate system. As a result, mitigating global warming through reduction of carbon emissions receives top priority among climate-engineering strategies (Cusack et al., 2014). In the pursuit of transitioning to lower carbon output, the concept of the "carbon footprint" was born, with the intention to raise consumer and stakeholder awareness by attributing the responsibility for carbon emissions to products, individuals, organizations, industries, or nations. Although the carbon footprint is internationally recognized as a measure of anthropogenic climate impacts, particularly in the field of LCA (Hellweg and Milà i Canals, 2014), confusion surrounding its meaning still exists.

Of particular concern is what the carbon footprint actually measures and how it deviates from the "ecological footprint" (Borucke et al., 2013). The carbon footprint follows the logic of the LCA framework, in which activities are first translated into the inventory of emissions (resource extractions can be treated on the same level of life cycle inventory) and further processed in a subsequent characterization step, in which the inventory results (emissions or extractions) are modeled quantitatively and expressed as impact scores according to their relative contributions to a specific impact category. By contrast, the ecological footprint translates a given activity into emissions and extractions that are then aggregated into land area required for absorption and regeneration, by a simple conversion that does not involve any characterization modeling. Having recognized probably the most important difference between the carbon and ecological footprints, Hammond (2007) unexpectedly argued that the term "footprint" implies a form of area-based indicator, and that the carbon footprint should thus be renamed "carbon weight" due to its mass unit. This may appear to be an argument over semantics, but underneath is a deeper issue that may indicate a misinterpretation of the term "footprint" and of the rationale behind the carbon footprint calculation. We discuss these issues below.

The carbon footprint assesses not only carbon emissions but also non-carbon greenhouse-gas (GHG) emissions by using substance-specific factors: namely, global warming potentials (GWPs) that account for the relative global warming effects of a mass unit of each GHG. The GWP is determined by sophisticated atmospheric models and set by the equivalence principle, representing the integrated radiative forcing over a specific time horizon (e.g., 100-yr) with a reference to carbon dioxide (CO_2). Subsequently, these comparable results of GHGs are aggregated into a single impact indicator expressed in a CO_2-equivalent mass unit (e.g., kg CO_2-eq.). As such, likening carbon footprint to a mass or weight would be akin to equating blood pressure with a distance because of its unit of measure (mm Hg). This comparison overlooks the distinction between the physical unit of a phenomenon (in this case, infrared radiative forcing) and the accounting unit in which we happen to express something (in this case, the mass of CO_2 that would have to be released to cause an equivalent impact).

Therefore, the success of the carbon footprint concept should not be attributed only to the fashionable term "footprint", borrowed from the ecological footprint community. Rather, a far more fundamental reason is the scientific underpinning of the characterization models, which allows the GWP to be one of the most established, consensus-based characterization factors. By contrast, the way that the ecological footprint deals with carbon emissions is much less rigorous. So-called carbon hectares, which dominate the overall value of the ecological footprint in many studies, are calculated by adding all energy-related carbon compounds in kilograms, and dividing the sum by a constant carbon sequestration rate (CSR). This procedure disregards the difference of impact strength between different carbon emissions. Carbon hectares are thus proportional to the total carbon weight of the energy carriers (e.g., coal). Furthermore, non-carbon GHGs fall outside the scope of the ecological footprint, even though N_2O, for instance, is one of the most important GHGs that contribute to climate change.

In summary, impact characterization is the key to understanding the carbon footprint concept. There is no need to translate the ecological footprint into land area, as proven by the ecological footprint, which attempts to do so but fails to substantiate the conversion convincingly. Hammond's (2007) uneasiness about mistaking weight for impact is indeed reasonable, although not for the carbon footprint, but rather for the ecological footprint.

4.2.2. Moving from the material footprint to a resource depletion footprint

In view of the success of the ecological, water, and carbon footprints, it is not surprising that an expanding list of indicators with "footprint" in their names will be continuously introduced to the public. The recent appearance of the material footprint is an example (e.g., Schoer et al., 2012; Wiedmann et al., 2015). It is defined as the total mass of materials used for economic processes. By using this mass-based material footprint indicator expressed in absolute terms, one can be clearly aware of the total resource needs of an economy.

However, we argue that computing the material footprint in this way is misleading from a life cycle perspective, because in the goal and scope definition of an LCA, material is treated as an upstream process before the manufacture of a product. This means that the material footprint is an analogous but different concept from product environmental footprint (PEF)—an ongoing European Commission policy initiative (EC, 2015a) assessing a broad set of impact categories to provide a comprehensive picture of the life cycle environmental performance of products, for the sake of product labeling. The material footprint, therefore, is expected to encompass a variety of environmental impacts associated with material extraction through the processing, distribution, storage, use, and disposal or recycling stages.

Rather than furthering the discussion on approaches to a veritable material footprint that has not come up, we call for a shift in focus to scarcity—a critical issue which, in our view, the material footprint practitioners were intended to address. The failure to address scarcity is due to summing up the mass of raw materials with equal weights. To cite an example, we assume that Economy A and B both have a material footprint of 100 kg. This, however, does not mean anything except the total mass, because the truth might be that Economy A consumed 1 kg of Au and 99 kg of sands, and Economy B conversely consumed 99 kg of Au and 1 kg of sands! In that case, misleading decisions can be made as a consequence of neglecting the varying importance of different resources in terms of scarcity.

There are several ways to quantify the scarcity of resources, such as exergy (available energy), surplus energy, and market price approaches. In life cycle impact assessment (LCIA), the impact of resource scarcity is evaluated by so-called resource depletion potential (RDP), a form of characterization factor derived from characterization models reflecting the environmental mechanism of depletion in natural capital stocks (Hauschild et al., 2013). In theory, there are two branches of RDP, namely, abiotic depletion potential (ADP) and biotic depletion potential (BDP) (Guinée and Heijungs, 1995). However, the BDP is normally excluded from LCIA as most biotic resources can be reproduced by a production process. This is why deforestation, for example, would not be regarded as a depletion problem but a production process with its particular environmental impacts such as soil erosion, land degradation, and global warming.

We herein propose a resource depletion footprint (RDF) aimed at addressing abiotic resource depletion. The rationale is that abiotic resources, such as minerals and fossil fuels, are a dominant contributor to the depletion of natural stocks. The RDF is calculated by multiplying the ADP by the extraction of resources, where ADP is specified as the ratio between two estimates, indicating how fast the remaining stocks of resources would be exhausted in comparison to a reference resource (such as Sb), both at the current rate of use. Figure 4.1 compares resource categories for which characterization factor ADPs are derived from Van Oers et al. (2002), which serves as an updated version of the baseline method proposed by Guinée and Heijungs (1995).

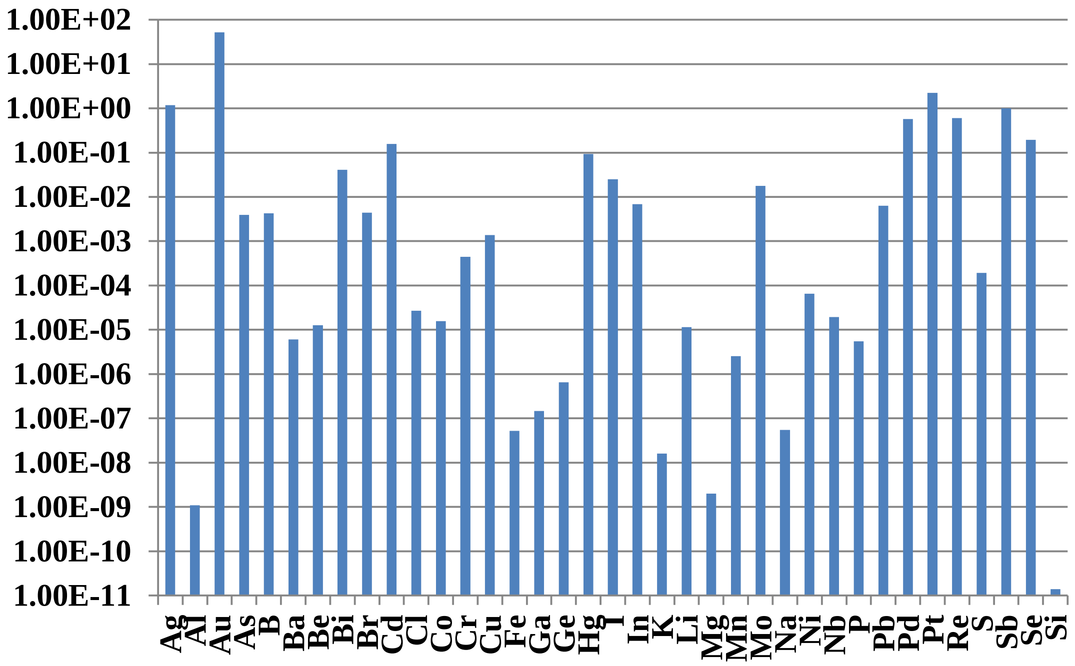

Figure 4.1. ADPs for characterizing abiotic resources against antimony (Sb), based on the estimation of planetary-scale ultimate stocks and the extraction rate for the year 1999. Data derived from Van Oers et al. (2002).

The RDF is distinguished from the material footprint as it uses a set of scientific-based characterization factors as a substitute for arbitrarily equal weights, and the outcome is expressed in relative rather than absolute terms. As a result, it allows one to prioritize abiotic resources with respect to their relative scarcity and to translate the overall risk of abiotic resource depletion into a more understandable measure of kilograms. Moreover, the RDF implies a critical recognition, which has been neglected in many environmental footprints, that human demand for natural capital should be kept within the planetary boundaries of deaccumulation or regeneration, beyond which our planet will be no longer sustainable.

Our proposal enables a harmonization of the RDF and carbon footprint, in the sense that they both aggregate different components based on scientific characterization instead of subjective weighting. The carbon footprint has a broader base of acceptance than other existing environmental footprints, because it is based on GWP—the most complete and accurate characterization factor quantifying the contributions of an emission to climate change. To make transparent the role of characterization in footprinting, we provide a comparison among the carbon footprint, RDF, and an imitating water depletion footprint (WDF) (Table 4.1).

Table 4.1. A proposal for RDF in comparison to the existing carbon footprint and an imitating WDF.

Footprint	Environmental concern	Input/output flow	Impact characterization factor
Carbon footprint	Climate change	Greenhouse emission	GWP
RDF	Resource scarcity	Abiotic extraction	ADP
WDF	Water scarcity	Freshwater extraction	WDP

The WDF has much in common with the RDF. Following this idea, one can easily formulate a suite of environmental footprints which characterize the extractions or emissions through their respective contributions to specific impact categories in a consistent manner. However, although the LCA community has taken an important step in characterizing multiple impact categories, the discrepancy between different characterization models for the same substance and impact category is still large. To fill in this gap, a lot of work needs to be done in the future.

4.2.3. Summary

As such, we underline the wide application and good performance of LCA in footprint studies and, conversely, of the usefulness of footprint principles in LCA studies. It is widely accepted that the product carbon footprint (PCF) or a broader PEF should be based on LCA (Wiedmann and Minx, 2008). LCA is typically an integrative approach in two ways: it covers many types of impact, and it covers the full life cycle. The full life cycle is analyzed to make sure there is no problem shifting from the use phase to the production phase or the disposal phase. The same is true for the broad spectrum of impacts: it is needed to ensure a more complete picture and prevent sub-optimization. Restricting an LCA to a PCF potentially creates blind spots where problem shifting can occur. Extending a PCF with more impact categories to obtain a full PCF is therefore a natural step.

4.3. On the limitations of LCA for environmental footprints: a case study of OEF

In their recent article, Ridoutt and Pfister (2013) call for an integrated family of footprint indicators, for which they highlighted the imperative of a "universal" footprint definition that is entirely based on LCA. That is to say, footprints which are not consistent with a comprehensive LCA (including the description of the goal and scope, inventory analysis, impact assessment, and the interpretation of the inventory and impact assessment results) should be disqualified from the footprint family. We believe that Ridoutt and Pfister overestimated the necessity of LCA to support the establishment of footprints in general or of a footprint family. There are more footprints than product footprints and national footprints. An important and upcoming type is the footprint of an organization (such as a company or an enterprise). In a policy context, the OEF has been described as based on a life cycle (Chomkhamsri and Pelletier, 2011; European Commission (EC, 2015b), without any motivation. We are concerned that such unfounded claims are reinforced by Ridoutt and Pfister, again without a proper justification. It is our conviction that a life cycle-based OEF is likely subject to overcounting. We illustrate this with a brief example. Suppose we calculate the life cycle-based OEF of a copper wire manufacturer. It will be based on cradle-to-gate impacts of its copper wire. Just a few street blocks further, a manufacturer of electrical equipment is using copper wire from the first company. The life cycle-based OEF of the second company will include all cradle-to-gate impacts of its materials, so also of the copper wire. This means that the OEFs of the two plants add to a too big number, because we are double counting the impacts of copper wire. The sum of the parts is bigger than the total; that is a truly holistic LCA! A similar argument was made in the context of product LCAs by Cullen and Allwood (2009).

If one includes the upstream impacts in an organization's footprint, a retailer's footprint will be very high. Likewise, a company that just transports or sells energy (such as a transmission network company or a gas station) would have an excessively large footprint. On the other hand, a flexible permission given by the EC (2013b) either to include or exclude downstream activities could perhaps avoid double counting, but also add uncertain and arbitrary results. There is a need to understand the risks to the environment and investors while recognizing that multiple stakeholders have different needs (Marland et al., 2013). This illustrates a very important point, where LCA can be used only for the final consumers in an economy (Lenzen et al., 2007), rather than for those which serve as both upstream consumers and downstream producers. The solution is to look at the added footprint instead of the life cycle footprint, much as an economist looks at the added value. To find the added footprint of an organization, we must subtract the cradle-to-gate impacts of the inputs from the cradle-to-gate impacts of the outputs, just like a business economist calculates the value added by subtracting the cost of the inputs from that of the outputs. Thus, we can elegantly formulate our framework as:

$$OEF = \sum PEF_{out} - \sum PEF_{in} \qquad (4.1)$$

Notice that this formula contains two LCA-based expressions to calculate the OEF. In this sense therefore, the OEF could be argued to be based on LCA or even doubly so. But notice well that the OEF is defined as a difference between two PEFs, so all overlapping parts of the life cycle are effectively removed.

In conclusion, there is still room for a footprint family without a life cycle approach. Obviously, the definition of the indicators, in terms of how resources and/or emissions are combined into footprints, needs to be aligned between life cycle-based and non-life cycle-based footprints, so between the PEF and the OEF. It would be strange to have a different global warming potential list when doing a PEF and an OEF. In fact, by defining the OEF in terms of a difference between life cycle-based PEFs, a natural harmonization, in terms of method and scope, is achieved.

4.4. Rethinking the relationship between environmental footprints and LCA: a concluding discussion

Over the past two decades, a rapid expansion of footprint-style indicators has been observed by academics, companies, governmental bodies, and nongovernmental organizations, particularly in the arena of environmental and sustainability discourses. Although nowadays footprints have reached worldwide popularity, a dedicated footprint research community is far from being established. The ambiguous relationship with LCA, for which there is such a community, poses a substantial obstacle to achieving that goal.

There has been a growing interest in discussing the relationship between footprints and LCA. Many researchers have stressed the unique contributions of LCA to the identification and quantification of footprints, with the intention of legitimizing footprint indicators from a life cycle perspective. The strengths of LCA in assessing environmental impacts could allow many footprint topics (e.g., climate change, water use, biodiversity) to be addressed under an LCA framework, in particular those that can be measured in relation to a functional unit. Examples include the carbon footprint for climate change and the RDF for abiotic resource depletion (Section 4.1).

Nevertheless, footprint practitioners tend to stand alone in some way. One example is the ecological footprint—the ancestor of the footprint family. From an LCA perspective, the classical ecological footprint analysis (EFA), namely, the National Footprint Accounts (NFA), corresponds to a more rough type of inventory analysis in which hundreds of primary bio-products are simply tabulated and converted into the land use elementary flow. The lack of transparency in defining system boundaries and the exclusion to characterize inventory results make the NFA an unsuccessful LCA, at least in the eyes of LCA experts.

Probably the most important thing that LCA users have learned from the EFA is the name—a good name sometimes means everything. Therefore, the advent of the carbon footprint is not surprising. It begins with the task of competing for public and corporate concerns on global warming—an issue that the ecological footprint attempts to address as well but fails to receive due attention. The prosperity of the carbon footprint moves LCA back to central stage, even though Hammond (2007) suggests calling the carbon footprint "carbon weight" with the belief that footprints should be area-based indicators in line with the ecological footprint.

Meanwhile, environmental input–output analysis (EIOA) has proved useful in accounting for the carbon footprint at national and international scales (Hertwich and Peters, 2009). Increasingly, IO methods have also been found suitable for computing the ecological footprint and many other footprints for nations, such as the water, material and biodiversity footprints. These, however, do not diminish the dominant role of LCA in contemporary footprint analysis, especially in the domain of product footprints where a great amount of theoretical and practical work has been done by the LCA community on various environmental issues associated with production and consumption.

In spite of this, saying that footprints must be LCA-based is, to some extent, analogues to saying that footprints must be area-based—both are due to a lack of mutual understanding between different scientific communities in the field. The reality is that non-area-based footprints are now ubiquitous, and that LCA is not the only way to implement an inventory analysis, in addition to which a footprint is not necessarily committed to an impact assessment. Moreover, there are certain important types of questions for which footprints are desirable but for which a life cycle perspective is not or only partially appropriate. Such a methodological limitation has been demonstrated in Section 4.2, with the case of OEF.

The footprint family has been envisaged in such a way that it can be easily extended to capture a broader scope of sustainability issues. Some emerging footprints, such as the celestial, employment and inequality footprints, open the door for footprint developers to establish and measure human well-being in terms of happiness and equality, which remains the ultimate goal of sustainable development. These social and economic dimension–LCA, however, suffer from difficulties in data availability, societal impact assessment, and result interpretation.

One thing that LCA can learn from environmental footprints is the comparison of a footprint and indicator of carrying capacity. The ecological footprint has a tradition of benchmarking man's land occupation with available planetary area and thereby determining whether the situation is sustainable or not. So do the blue water footprint and chemical footprint—the two can be readily compared with blue water availability and chemical boundary, respectively. The convergence of footprints and planetary boundaries makes sense in that it allows for the evolution of environmental impact assessment to environmental sustainability assessment, which is more informative for policy purposes but lacking or at least inconspicuous in current LCA frameworks.

Admittedly, facilitating the calculation of footprints with mature methodological frameworks is preferred, and because of this, many footprint users have learned and borrowed much from LCA, IOA, or a hybrid of both. Even so, narrowing footprints down to an LCA context potentially creates blind spots, where exhaustive inventory data for compiling and/or consensus models for characterization of impact pathways are not available—and vice versa—some typical impact categories (e.g., ozone depletion, ionizing radiation) are out of the scope of the footprint family in its current form.

To sum up, footprints are not to be interpreted as a new name for the good old impact category indicators defined in LCA and, more importantly, that LCA does not substitute but complements environmental footprints. The nuanced ways that footprints and LCA deal with anthropogenic stressors should not be viewed as merely a source of controversy but rather as an opportunity for complementary use, and for development and refinement of these tools. For instance, an initiative has been launched to investigate the possible synergies between classical water footprint and water-use LCA (Boulay et al., 2013). More investigations are needed into the relationship of individual footprints and LCA scopes. Examples include the ecological footprint and land use, chemical footprint and toxicity, as well as nitrogen footprint and eutrophication. This relies on the collaboration between the footprint community—which is ever-expanding but fragmented and the LCA community—which is sophisticated but more fossilized because its members stick to standards by ISO, EPA, EC, and more.

References

Borucke, M., Moore, D., Cranston, G., Gracey, K., Iha, K., Larson, J., Lazarus, E., Morales, J. C., Wackernagel, M., Galli, A., 2013. Accounting for demand and supply of the biosphere's regenerative capacity: the National Footprint Accounts' underlying methodology and framework. Ecological Indicators 24, 518–533.

Boulay, A. M., Hoekstra, A. Y., Vionnet, S., 2013. Complementarities of water-focused life cycle assessment and water footprint assessment. Environmental Science & Technology 47, 11926–11927.

Chomkhamsri, K., Pelletier, N., 2011. Analysis of existing environmental footprint methodologies for products and organizations: Recommendations, rationale, and alignment. European Commission, Joint Research Centre and Institute for Environment and Sustainability, Ispra, Italy. http://ec.europa.eu/environment/eussd/pdf/Deliverable.pdf.

Cullen, J. M., Allwood. J. M., 2009. The role of washing machines in life cycle assessment studies. Journal of Industrial Ecology 13, 27–37.

Cusack, D. F., Axsen, J., Shwom, R., Hartzell-Nichols, L., White, S., Mackey, K. R. M., 2014. An interdisciplinary assessment of climate engineering strategies. Frontiers in Ecology and the Environment 12, 280–287.

EC (European Commission), 2015a. Product Environmental Footprint (PEF). http://ec.europa.eu/environment/eussd/smgp/product_footprint.htm

EC (European Commission), 2015b. Organization Environmental Footprint (PEF). http://ec.europa.eu/environment/eussd/smgp/organisation_footprint.htm

Fang, K., Heijungs, R., 2014. There is still room for a footprint family without a life cycle approach—comment on "Towards an integrated family of footprint indicators". Journal of Industrial Ecology 18, 71–72.

Guinée, J. B., Heijungs, R., 1995. A proposal for the definition of resource equivalency factors for use in product life-cycle assessment. Environmental Toxicology and Chemistry 14, 917–925.

Hammond, G., 2007. Time to give due weight to the "carbon footprint" issue. Nature 445, 256–256.

Hauschild, M. Z., Goedkoop, M., Guinée, J., Heijungs, R., Huijbregts, M., Jolliet, O., Margni, M., De Schryver, A., Humbert, S., Laurent, A., Sala, S., Pant, R., 2013. Identifying best existing practice for characterization modeling in life cycle impact assessment. The International Journal of Life Cycle Assessment 18, 683–697.

Hellweg, S., Milà i Canals, L., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science 344, 1109–1113.

Lenzen, M., Murray, J., Sack, F., Wiedmann, T., 2007. Shared producer and consumer responsibility—theory and practice. Ecological Economics 61, 27–42.

Marland, G., Buchholz, T., Kowalczyk. T., 2013. Accounting for carbon dioxide emissions: The context and stakeholders matter. Journal of Industrial Ecology 17, 340–342.

Ridoutt, B. G., Pfister, S., 2013. Towards an integrated family of footprint indicators. Journal of Industrial Ecology 17, 337–339.

Schoer, K., Weinzettel, J., Kovanda, J., Giegrich, J., Lauwigi, C., 2012. Raw material consumption of the European Union – Concept, calculation method, and results. Environmental Science & Technology 46, 8903–8909.

Van Oers, L., De Koning, A., Guinée, J. B., Huppes, G., 2002. Abiotic resource depletion in LCA. Road and Hydraulic Engineering Institute, Ministry of Transport and Water, Amsterdam, The Netherlands.

Wiedmann, T. O., Schandl, H., Lenzen, M., Moran, D., Suh, S., West, J., Kanemoto, K., 2015. The material footprint of nations. Proceedings of the National Academy of Sciences of the United States of America 112, 6271–6276.

Chapter 5

Understanding the complementary linkages between environmental footprints and planetary boundaries

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Abstract

While in recent years both environmental footprints and planetary boundaries have gained tremendous popularity throughout the ecological and environmental sciences, their relationship remains largely unexplored. By investigating the roots and developments of environmental footprints and planetary boundaries, this chapter challenges the isolation of the two research fields and provides novel insights into the complementary use of them. Our analysis demonstrates that knowledge of planetary boundaries improves the policy relevance of environmental footprints by providing a set of consensus-based estimates of the regenerative and absorptive capacity at the global scale and, in reverse, that the planetary boundaries framework (PBF) benefits from well-grounded footprint models which allow for more accurate and reliable estimates of human pressure or impact on the planet's environment. A framework for integration of environmental footprints and planetary boundaries is thus proposed. The so-called footprint–boundary environmental sustainability assessment (F–B ESA) framework lays the foundation for evolving environmental impact assessment to environmental sustainability assessment aimed at measuring the sustainability gap between current magnitudes of human activities and associated capacity thresholds. As a first attempt to take advantage of environmental footprints and planetary boundaries in a complementary way, there remain many gaps in our knowledge. We have therefore formulated a research agenda for further scientific discussions, mainly including the development of measurable boundaries in relation to footprints at multiple scales and their trade-offs, and the harmonization of the footprint and boundary metrics in terms of environmental coverage and methodological choices. All these points raised, in our view, will play an important role in setting practical and tangible policy targets for adaptation and mitigation of worldwide environmental unsustainability.

5.1. Introduction

A central challenge for sustainability is how to meet human needs while preserving our planet as a pleasant place for living and as a source of welfare (Kates et al., 2001; Kratena, 2004). A necessary, though not sufficient, step in achieving this goal is the identification and measurement of carrying capacity—the maximum persistently supportable load that the environment can offer without impairing the functional integrity of ecosystems (Catton, 1986; Rees, 1996). Attempts have been made to define human carrying capacity, from a demographic perspective, as the maximum human population which can be raised by the Earth in a way that would ensure the interests of future generations (Daily and Ehrlich, 1992; Ehrlich, 1982). This definition is, however, seemingly somewhat pedantic and meaningless, because the growth in global population remains virtually unchanged and of course cannot be diminished by force even though Ehrlich (1982) already warned of the overshoot of human carrying capacity.

In response to the then-current debates surrounding carrying capacity, the ecological footprint was conceived to represent the spatial appropriation ideally required to support a given population (Rees, 1992; Wackernagel and Rees, 1996). It can be regarded as a complement to carrying capacity. Leaving out many key aspects of sustainability by design (Goldfinger et al., 2014), the ecological footprint practically equates human demand for nature with that for biotic resource provision and energy-related carbon sequestration. Subsequently, an array of footprint-style indicators has been spawned as complements to the communication of pressure or impact that humanity places on the planet's environment. This array includes the water footprint (Hoekstra and Hung, 2002), chemical footprint (Guttikunda et al., 2005), carbon footprint (Wiedmann and Minx, 2008), phosphorus footprint (Wang et al., 2011), nitrogen footprint (Leach et al., 2012), biodiversity footprint (Lenzen et al., 2012), material footprint (Wiedmann et al., 2015), and so on.

At the same time, revisiting sustainability limits has never stopped since the publication of Limits to Growth (Meadows et al., 1972), a remarkable book which for the first time alarmed the public with environmental constraints on population expansion. In 2009, as conceptually similar to carrying capacity, a framework of planetary boundaries was launched by Rockström et al. (2009a, 2009b). By its definition, capacity thresholds for a broad range of environmental issues at the global scale are explicitly identified, including climate change, rate of biodiversity loss, interference with the nitrogen and phosphorus cycles, stratospheric ozone depletion, ocean acidification, global freshwater use, change in land use, chemical pollution, and atmospheric aerosol loading. Because of the initiative of providing quantitative and measurable preconditions for human development, the planetary boundaries concept has grown in interest over recent years, with particular focus on its implications for Earth system governance (Biermann, 2012), biospheric monitoring and forecasting (Barnosky et al., 2012), green economy (Kosoy et al., 2012), food security (De Vries et al., 2013), and environmental equity (Steffen and Stafford Smith, 2013).

There have been a considerable number of studies that deal with either environmental footprints or planetary boundaries, and only very few that discuss both topics within one study. Moreover, the chapters that address environmental footprints together with planetary boundaries employ different principles, frameworks, and terminologies. This chapter aims to highlight the promise of connecting environmental footprints and planetary boundaries by exploring their relationships and synergies, by providing a harmonized framework and terminology, and by offering novel insights into their complementary use.

To that end, the remainder of this chapter is structured as follows: Section 5.2 provides evidence on the importance of the planetary boundaries concept for making environmental footprints policy-relevant; Section 5.3, on the contrary, investigates the role of environmental footprints in improving the scientific robustness of the planetary boundaries framework (PBF); Section 5.4 demonstrates the benefits of jointly defining environmental sustainability; Section 5.5 proceeds with a detailed discussion of the challenges of synthesizing the footprint and boundary metrics and how these inform a research agenda.

5.2. Why knowledge of planetary boundaries is important for making environmental footprints policy-relevant?

Many environmental footprints have proven useful in measuring the pressure or impact exerted by human activities (Galli et al., 2012; Leach et al., 2012). Meanwhile, it has been widely acknowledged that focusing exclusively on a single footprint runs the risk of shifting the environmental burden to other impact categories (Fang et al., 2014). Shrinking the product carbon footprint, for instance, could induce a remarkable increase in other environmental footprints (Laurent et al., 2012). Likewise, reductions in water footprint by inter-basin water or food transfer are found at the expense of increasing energy footprint (Gerbens-Leenes et al., 2009). Considerable evidence from the literature calls for a policy transformation from assessing single footprints in isolation to tackling diverse footprints, i.e., a footprint family (Fang et al., 2014; Galli et al., 2012), from an integrated perspective.

However, this is not enough. Man should not merely minimize his environmental footprints, which many footprint users concentrate on, but make sure these footprints stay within the planetary boundaries, which is a critical prerequisite for sustainable development (Fang and Heijungs, 2015; Heijungs et al., 2014). As pointed out by Lancker and Nijkamp (2000), an indicator does not provide any information on sustainability unless a reference value is given to it. A simultaneous assessment of environmental footprints and related capacity thresholds is therefore of vital importance, representing the evolution of backtracking towards a prognostic and preventive measure that helps prevent human activities from triggering undesirable environmental changes.

The ecological footprint was designed in such a way that it can be readily compared to available bio-productive area of the Earth, which is referred to as "biocapacity" (Rees, 1992; Wackernagel and Rees, 1997). The difference between the ecological footprint and biocapacity reflects a form of sustainability gap, explaining why our world is operating in a state of overshoot with respect to biotic resource extractions and energy-related carbon emissions (Niccolucci et al., 2009; Wackernagel and Rees, 1997). The inclusion of biocapacity is unique and important, making the ecological footprint outstand from many other footprint indicators (Ewing et al., 2012; Hoekstra, 2009).

In a similar case to that of the ecological footprint, the blue and gray water footprints were envisaged as a way of comparing with the blue and gray water boundaries, respectively, where the results are expressed in the form of a quotient (Hoekstra et al., 2012; Liu et al., 2012). The footprint-to-boundary ratios depict the relative severity of water scarcity and pollution as a consequence of the mismatch between water withdrawal and renewable supply. The ecological and water footprints are, in this sense, able to inform policy makers on to which degree the biophysical limits of the biosphere and hydrosphere are being approached or exceeded, respectively (Costanza, 2000; Galli et al., 2012).

Table 5.1 summarizes existing practices that aims at incorporating the boundary concept into footprint analysis. As seen, so far not all of the footprints include a comparison to quantified capacity thresholds. In fact, many do not, although this is being perceived as increasingly useful. Even for those which have been linked to a threshold value already, there remain limitations that have been a notable source of controversy in footprint analysis; thus, we believe that recent developments regarding planetary boundaries will inspire and facilitate the ongoing process of benchmarking environment footprints against capacity thresholds.

Table 5.1. Summary of existing practices for relating environmental footprints to planetary boundaries.

Footprint category	Key elements of relating a footprint to a boundary	Advantages over the sole use of the footprint	Limitations
Blue water footprint (Hoekstra et al., 2012)	• Blue water footprint: a measure of the volume of surface and groundwater consumed and then evaporated or incorporated into a product, sector, or the whole economy. • Blue water availability: a measure of the total natural runoff minus presumed flow requirements for ecological health. • Blue water scarcity: equal to dividing blue water footprint by blue water availability.	By screening the monthly water scarcity in 405 major river basins throughout the world in 1996-2005, a large number of people living under sever water stress. The world-average ratio of blue water footprint to blue water availability is found to be about 94%, which suggests that the globe has experienced a low water scarcity.	The accounting of blue water availability does not properly deal with the perturbation of seasonal runoff patterns by dams' flow regulation, and similarly for the blue water footprint that does not include evaporation from artificial reservoirs. The water scarcity indicator without regard to green water component is incomplete.
Carbon footprint (Hoekstra and Wiedmann, 2014)	• Carbon footprint: a measure of the total amount of greenhouse gas emissions that are directly and indirectly caused by an activity or are accumulated over the life cycle of a product. • Carbon boundary: a measure of the maximum sustainable carbon footprint level at the global scale. • Carbon deficit: equal to subtracting carbon boundary from carbon footprint.	The carbon footprint has been put in the context of a planetary carbon boundary, which is estimated to be 18-25 Gt CO_2-eq./yr. It means that the global carbon footprint should be reduced by 60% in 2010-2050 in order to achieve the global warming target of maximum 2 ℃.	There is not yet a consensus on the most appropriate way of allocating the responsibility for carbon reduction to national and sub-national scales, i.e., a fair share for different stakeholders given their historical performance, capacity and other considerations is lacking.
Chemical	• Chemical footprint: a measure of the	In addition to computing chemical footprint, two	The methodology proposed faces the challenges of

footprint (Zijp et al, 2014)	expected cumulative impacts of chemical mixtures on aquatic ecosystems for a region. • Chemical boundary: a measure of the sustainability level or policy target expressing which chemical impact is acceptable. • Chemical pollution index: equal to dividing chemical footprint by chemical boundary.	approaches to define a chemical boundary are introduced from the realms of chemical management practice (policy boundary) and of research into ecosystem vulnerability (natural boundary), so that one can account for the water volume needed to dilute chemical pollution due to human activities to a level below a specified boundary condition.	finding ways to reduce the uncertainty of weighting that aggregates the impacts on different scales and compartments, and of the complex natural systems that would hamper the distribution of spatially variable and ecosystem specific chemical boundaries. The resulting chemical footprint is hypothetical and thus, comparing the footprint with the boundary should be conducted with care.
Ecological footprint (Borucke et al., 2013)	• Ecological footprint: a measure of the land and water area required to support a given population with biotic resource extractions and energy-related carbon emissions. • Biocapacity: a measure of the biosphere's regenerative capacity in terms of the Earth's terrestrial and aquatic surface that is biologically productive to provide the basic ecosystem services—food, fiber and timber products that humanity consumes. • Ecological deficit/surplus: a measure of the overshoot/reserve of biocapacity relative to its ecological footprint.	The comparison to biocapacity supports the existence of global overshoot which first occurred in the mid-1970s. In 2008, mankind's ecological footprint exceeded at least 50% of the biocapacity, consuming ecosystem services that require about 1.5 planets to regenerate and to assimilate.	The carbon component in many cases contributes almost 100% or even more of the ecological deficit due to the omission of the absorptive capacity in current ecological footprint accounting. Present global overshoot would be replaced by a surplus of 0.6 planets without considering the carbon component.
Gray water footprint (Liu	• Gray water footprint: a measure of the volume of freshwater required to	The calculated water pollution levels of different river basins show a large variation among	The water pollution level of a basin below 1 does not necessarily reflect an avoidance of

et al., 2012)	assimilate the loading of pollutants given natural background concentrations and existing ambient water quality standards. • Pollution assimilation capacity: a measure of the environmental water needs by subtracting the presumed flow requirement for ecological health from the total runoff. • Water pollution level: equal to dividing gray water footprint by pollution assimilative capacity.	different periods, generally increasing in 1970-2000. In 2000, about two-thirds of the basins have their pollution assimilative capacity fully consumed for anthropogenic nitrogen or phosphorus.	eutrophication at the sub-basin level. Defining the overall water pollution level as the largest calculated one among all different nutrient forms of nitrogen or phosphorus is questionable, as this may overly simplify the cumulative effects of multiple aquatic pollutants.

5.3. Why environmental footprints are important for making the PBF scientifically robust?

Contrary to popular belief, Rockström et al.'s framework is not only about planetary boundaries, but also about current state estimates—a neglected field of research into the planetary boundary issues. In other words, its ultimate goal is not to quantify a boundary, but to quantify the transgression or reserve of a boundary, determined by the comparison of planetary boundaries and current human pressure. As a whole, Rockström et al.'s estimates are reliant on literature review reflecting expert knowledge that inevitably contains uncertainty, subjectivity and arbitrariness (De Vries et al., 2013; Lewis, 2012). Nevertheless, currently this is perhaps the best way to quantify planetary boundaries in view of the difficulties of prediction. Furthermore, by using the best available knowledge and the precautionary principle, planetary boundaries are claimed to be more science-based than a common policy framework (Nykvist et al., 2013).

However, the problem is that Rockström et al. do so to measure the current status of investigated environmental issues, which could have been more rigorous and robust if appropriate environmental models are used instead. As environmental footprints are derived from a great number of quantitative models, of which the majority have a broad base of acceptance with respect to documentation, transparency and reproducibility (Fang et al., 2014; Hoekstra and Wiedmann, 2014), it is natural to expect that the methodological maturity of footprints would be able to enhance the expression and quantification of current estimates involved in the PBF.

We illustrate this with two brief examples. On the climate change, for instance, atmospheric concentration of carbon dioxide (CO_2) and radiative forcing have been chosen as two control variables for setting climate boundary, but also for measuring current climate state (Rockström et al., 2009a). The concurrent use of the two variables represents an unnecessary dual-objective trade-off and thus may compromise the usefulness of setting carbon boundary. By using carbon footprint—a consensus impact indicator of climate change (Hellweg and I Canals, 2014; Minx et al., 2013), a convergence of these two independent variables is harmoniously achieved. According to Hoekstra and Wiedmann (2014), global carbon footprint is amounted to 46-55 Gt CO_2-eq./yr for 2011.

In the case of freshwater use, Rockström et al. pose that at present the annual global water consumption is approximately 2600 Gm^3/yr. Apart from the uncertainty of this approximation, the value only accounts for the evaporation and transpiration from surface and ground water—a small fraction of total freshwater usage (Molden, 2009), ignoring green water that is estimated to be 6700 Gm^3/yr (Hoekstra and Mekonnen, 2012). The serious underestimate of human freshwater consumption should have been overcome by aggregating the blue and green water footprints using existing water footprint models with high degrees of scientific certainty.

The two cases as referred to demonstrate the necessity of standardized and reproducible footprint models to support the assessment of actual human-induced environmental pressure or impact. One may extrapolate that the scientific foundation of the PBF will be consolidated by the substitution of well-grounded footprint models for rough current estimates. However, this does not justify the incorporation of capacity thresholds into footprint indicators within the existing footprint discussions. Ambiguity and confusion may occur, as proven by the ecological footprint which sometimes refers to the footprint itself, and at other times refers to both the footprint indicator and biocapacity. As a result, the purpose of the remainder of this chapter is not to consider boundaries as a part of footprints, nor to consider footprints as a part of boundaries. Instead, we keep the footprint metric and boundary metric separate, while taking the two as complements in assessing environmental sustainability.

5.4. Complementary use of environmental footprints and planetary boundaries for environmental sustainability assessment

5.4.1. The root of the environmental sustainability concept

Responding to the increasing challenge of finding ways to maintain the carrying capacity of the global ecosystem, the significance of the boundary concept in making sense of environmental sustainability had already been underlined in the late 20th century. For example, Daly (1990) presented an operational principle of sustainable development; that is, the regenerative and absorptive capacity must be treated as natural capital, of which the failure of maintenance leads to unsustainability. Goodland and Daly (1996) legitimized environmental sustainability by three input–output rules: (1) harvest within the regenerative capacity of renewable resources; (2) waste within the absorptive capacity of natural systems; and (3) depletion of non-renewable resources at a rate less than that of renewable substitutes.

Despite the high transparency, completeness and acceptability that Goodland and Daly's definition provides, a fundamental obstacle to environmental sustainability assessment (ESA) is the difficulty in predicting how long a life-supporting system is to be sustainable, rather than in discriminating sustainability and unsustainability after the fact (Costanza and Patten, 1995). This results from a lack of methods for quantifying the regenerative and absorptive capacity. As a breakthrough to fill in this gap, the PBF gives, for the first time, numerical results for capacity thresholds at the global scale. Meanwhile, the footprint metric serves as a counterpart to the boundary metric by offering background values for environmental issues and thereby helping to better understand the concept of environmental sustainability.

5.4.2. A footprint–boundary ESA (F–B ESA) framework

To preserve the planet's environment from facing unexpected or irreversible changes, a first step would be the development of ways of ascertaining whether human activities are kept within permissible limits. Due to their relative emphases and challenges noted above, neither environmental footprints nor planetary boundaries can adequately address this complicated issue solely; therefore, they should rather be used complementarily to make sense of the ESA. In deriving a footprint–boundary representation of environmental sustainability, clarity on definitions of both environmental footprints and planetary boundaries is required. Although there are already many attempts for making the two concepts transparent, we contend that any definitions work satisfactorily only if placed in an appropriate context, i.e., none is able to fit for all purposes. For this reason, environmental footprints and planetary boundaries will be specified as follows:

- **Environmental footprints**: a measure of human pressure or impact on the planet's environment in relation to resource extractions and hazardous emissions. In a mathematical context, we indicate the footprint of pressure i (e.g., carbon emission, water use, land use) as $U_{footprint,i}$.
- **Planetary boundaries**: a measure of the regenerative and absorptive capacity of the Earth's life-supporting systems, beyond which unacceptable environmental changes for humanity may occur. Accordingly we denote the planetary boundary of pressure i as $U_{boundary,i}$.

Mathematically, we do two steps:

- **Step 1** converts an environmental footprint and/or planetary boundary into a common metric. For example, $U_{footprint,CO_2}$, $U_{footprint,CH_4}$, $U_{footprint,N_2O}$ (in Gt/yr) are collectively converted into $Z_{footprint,climate}$ (in Gt CO_2-eq./yr) using the global warming potential (GWP) values. Likewise, $U_{boundary,temperature}$ (in °C) is converted into $Z_{fboundary,climate}$ (in Gt CO_2-eq./yr).
- **Step 2** creates sustainability indicators by looking at: (1) the difference between a footprint and a boundary ($Z_{footprint,i} - Z_{boundary,i}$); or (2) the ratio of a footprint to a boundary ($Z_{footprint,i}/Z_{boundary,i}$).

On the basis of the two steps, a schematic representation of the F–B ESA framework is provided in Figure 5.1. The main function of the F–B ESA framework is to inform policy makers on the distance or ratio between the actual performance and the estimated thresholds, visualizing if the maximum sustainable level has already been breached. By use of the F–B ESA framework, the distinction between environmental sustainability and unsustainability can be explicitly interpreted as follows:

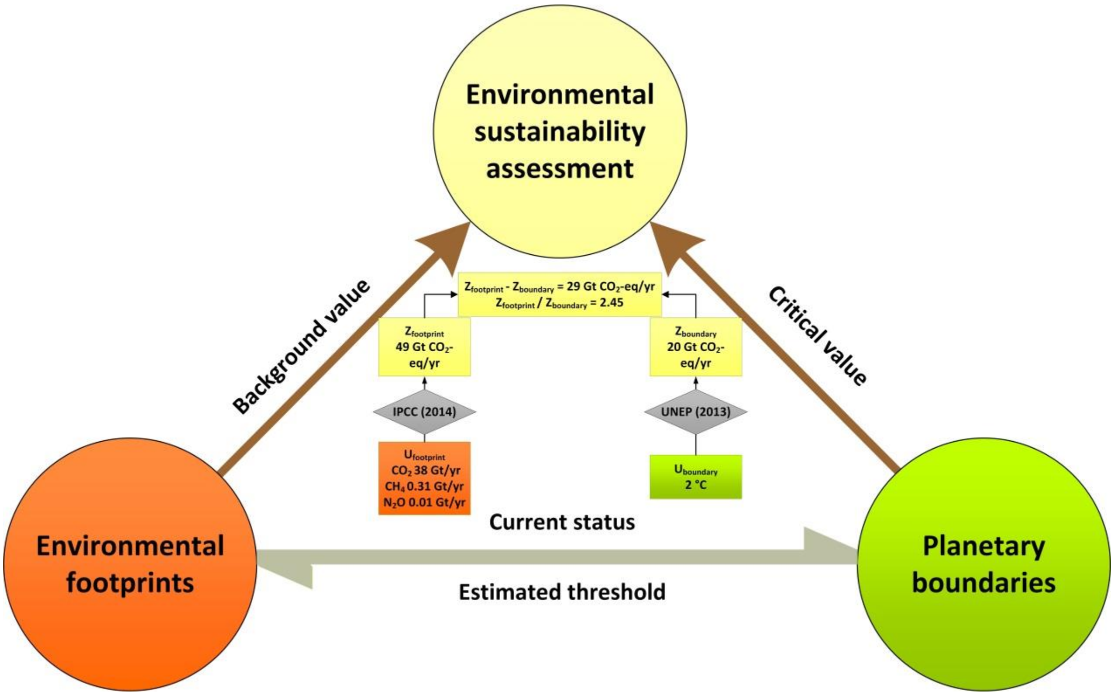

Figure 5.1. The footprint–boundary environmental sustainability framework, along with a procedure for converting and comparing the footprint and boundary metrics, exemplified by measuring the sustainability gap of climate change on the basis of the global carbon budget. While current climate boundary is emerged from the consensus that anthropogenic warming should be limited to below 2 °C (Rogelj et al., 2013), it represents an unnecessary distraction from the "2 °C target" (Allen, 2009). Operational challenges may arise as more than one reduction target should be met simultaneously. For this concern, a proposal for converting the "2 °C target" directly into a mass equivalent metric is given, which is in line with the conversion of three principal greenhouse gases, namely, CO_2, CH_4, and N_2O, to the carbon footprint. IPCC: Intergovernmental Panel on Climate Change; UNEP: United Nations Environment Programme.

- **Environmental sustainability**: the converted footprint of human activities is kept within the relevant converted boundary, ensuring that the planet's environment retains a safe state in which human well-being and prosperity are satisfied ($Z_{footprint,i} - Z_{boundary,i} \leq 0$, or $Z_{footprint,i}/Z_{boundary,i} \leq 1$).
- **Environmental unsustainability**: the converted footprint of human activities already exceeds the relevant converted boundaries, with consequences that would move the planet's environment to an unsafe state in which the stability and resilience of Earth system functioning are being undermined ($Z_{footprint,i} - Z_{boundary,i} > 0$, or $Z_{footprint,i}/Z_{boundary,i} > 1$).

5.4.3. Benefits of the F–B ESA framework

The joint implementation of environmental footprints and planetary boundaries opens the way for a novel and straightforward representation of environmental sustainability. While footprints have been found particularly suited to support decisions in environmental impact assessment (EIA), many of which are limited in visualizing the gaps between what is actually being done and what ought to be done from a sustainability perspective. Examples include the nitrogen, phosphorus and biodiversity footprints. One may argue, for instance, that it is not difficult to imagine the development of nitrogen threshold within the nitrogen footprint framework; this, however, suggests a position that in our view is undesirable because of rejecting the use of existing knowledge on planetary boundaries which has gained considerable interest and support from a broad range of the scientific community.

We believe that ESA represents a step ahead from EIA that is based on descriptive indicators (e.g., environmental footprints) that measure what is happening to the environment (Smeets and Weterings, 1999), as from a consumption-based angle it makes more sense to give consumers the opportunity to take into account their environmental responsibility for closing the sustainability gap. In this regard, a prominent advantage of implementing the F–B ESA framework is that it delivers valuable information on whether or not human activities give rise to a sustainability gap, and to what extent. To meet the public and corporate needs of downscaling planetary boundaries for the allocation of responsibility, developing measurable environmental boundaries at sub-global scales is needed. We classify and exposit the scaling effects of planetary boundaries in depth via Sections 5.5.1 and 5.5.2, together with a discussion of how to harmonize the footprint metric and boundary metric via Section 5.5.3 and of the potential trade-offs of sustainability gaps between various environmental issues via Section 5.5.4.

A further strength of the F–B ESA framework lies in its completeness of capturing key environmental challenges to global sustainability, rather than a single footprint nor a footprint family that covers. As the distinction between a policy target and a natural threshold boundary has been brought to attention (Zijp et al, 2014), there is an ever greater need to understand how the sustainability gap and the policy gap differ. The F–B ESA framework is appropriate for use in distinguishing these two types of gaps. Conceived in simple terms, the sustainability gap is the distance between current status and threshold values anticipated for scientific purposes, though revealed preferences and judgments cannot be completely avoided, and the policy gap is the one between policy targets set with political legitimacy and threshold values. A sustainability gap minus a policy gap represents a measure of implementation gap required to be covered in order to fulfill a commitment that has been enforced by regulation or legislation. A shift from an emphasis on monitoring environmental impacts to an emphasis on measuring sustainability gaps and evaluating the actual effectiveness of policy implementation is therefore realized through the comparison of policy targets and the converted footprints and boundaries (Figure 5.2).

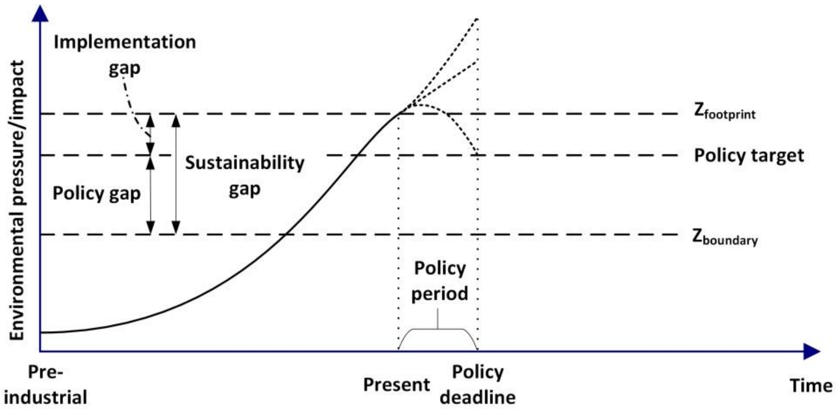

Figure 5.2. A schematic of the sustainability gap, policy gap, and implementation gap. Adapted from Fischer et al. (2007) and Nykvist et al. (2013).

5.5. Research agenda for strengthening the footprint–boundary environmental sustainability framework in future work

5.5.1. Development of measurable aggregated boundaries at multiple scales

It is unclear how regime shifts propagate across scales, and whether local and regional unsustainability necessarily gives rise to global transitions that imply an irreversible collapse worldwide (Hughes et al., 2013). Rockström et al. highlight that for some aggregated issues, such as water use, land use and aerosol loading, it matters where stressors exert negative effects; therefore, the associated environmental consequences are spatially varying and primarily limited to local area. The aerosol loading in East China, for instance, may have led to severe environmental and human health risks, but this hardly contributes to the aerosol loading in New Zealand. In this case, affecting in one region has no unambiguous direct relation to other regions.

As such, aggregated issues are unlikely to show strong evidence of global threshold behaviors (Rockström et al., 2009b); unless their heterogeneous impacts on local environment have been extensively replicated and ultimately spread worldwide (Lewis, 2012). Consequently, setting planetary boundaries merely and waiting until we approach them are dangerous and may significantly obscure the seriousness of environmental degradation at a regional or local scale. For example, while the planetary boundary for phosphorus has not yet been transgressed (Rockström et al., 2009a, 2009b), a striking fact which should not be neglected is that the absorptive capacity for phosphorus in two-thirds of the world's river basins has already been fully consumed (Hoekstra and Wiedmann, 2014).

This justifies the importance of developing measurable aggregated boundaries regionally or locally, which however remains largely unexplored in current PBF. As discussed, the biocapacity and water availability provide paradigms for the measurement of local and regional environmental boundaries. We argue that aggregating local or regional boundaries to the national level would allow for a reasonably refined estimate of national boundaries for those aggregated issues. This could be achieved by means of a bottom-up approach. The sensitivity to place of aggregated issues, however, constitutes a major constraint on applications of such an upscaling (Nykvist et al., 2013). Another challenge concerns the allocation problem (Heijungs and Frischknecht, 1998), as the real geographical scale of anthropogenic perturbation and associated impacts, which for instance can be a river basin, is very likely to go beyond national borders.

5.5.2. Partitioning of systemic planetary boundaries for sub-global assessments

On the other hand, it is also unclear how long the boundary exceedance actually takes to cause catastrophic environmental effects, even though a global threshold effect for systemic issues (climate change, ozone depletion, and ocean acidification) arguably exists regardless of where stressors are imposed (Nykvist et al., 2013; Rockström et al., 2009b). Such a transition is elusive and unpredictable, typically lagged by centuries, millennia, or even millions of years (Hughes et al., 2013), as evidenced in the observation that over the past two billion years global transitions rarely took place, with an estimate of five times merely (Barnosky et al., 2012). This is why the practitioners recently concede that exceeding one or few planetary boundaries is unlikely to give rise to disastrous consequences as immediate as previously thought (Hughes et al., 2013; Lenton and Williams, 2013).

Misinterpretation may occur if one misunderstands the precautionary principle that the planetary boundaries concept relies on and equates a surpassed boundary with a critical transition that corresponds to regime shifts—a sharp and persistent reorganization of the state of an ecosystem, which can hardly be anticipated or reversed by man (Brook et al., 2013). Besides, the global nature of systemic issues does not verify that all entities should take an equal responsibility for emission reduction—a politically salient issue that tends to trigger international instability. To start downscaling systemic boundaries for the allocation of responsibility, establishing internationally agreed criteria is a preparatory step towards a politically acceptable way, just like the distribution of CO_2 emission quotas (Germain and Van Steenberghe, 2003).

Any environmental agreement on the partitioning of systemic planetary boundaries requires negotiation and compromise among different levels of stakeholders such as governments, non-governmental organizations, corporations, and individual citizens. In the absence of reliable databases for local conditions, the partitioning of the "cake" can be first implemented nation by nation. Admittedly, there is more than one way to operationalize such a top-down process, such as on a population size base, a Gross Domestic Product (GDP) base, a territorial area base, or on other bases. Extending the national-scale boundary to sub-national scales will only be realized by a large number of dedicated personnel with sufficient technical and financial support. A schematic representation of the suggested bottom-up and top-down approaches for boundary scaling is given in Figure 5.3.

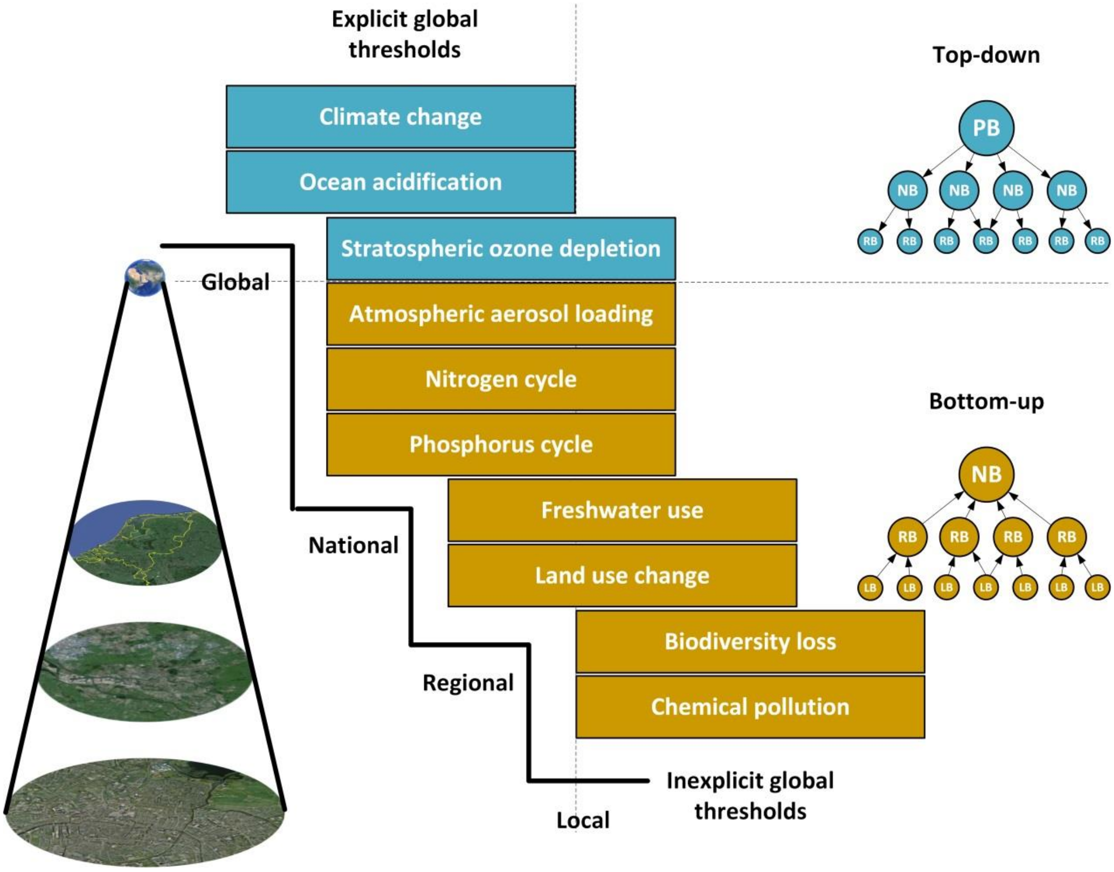

Figure 5.3. A schematic of the bottom-up and top-down approaches to boundary scaling. Adapted from Rockström et al. (2009b). The environmental issues in blue rectangles represent systemic issues, and those in brown rectangles represent aggregated issues. NB: national boundary; RB: regional boundary; LB: local boundary.

5.5.3. Harmonization of the footprint metric and the boundary metric

In concretizing the F–B ESA framework presented in the chapter, maintaining the harmonization between the footprint and boundary metrics deserves priority. We elaborate on this in two aspects: the harmonization of coverage and the harmonization of methodologies.

According to the coverage of major environmental concerns (Figure 5.4), one may easily come to the conclusion that as a whole there is substantial similarity between environmental footprints and planetary boundaries, because they both cover similar and wide-enough spectrums of environmental issues. Nevertheless, converting a footprint and a boundary into a common metric is probably incomplete, as in many cases multiple footprints are related to multiple boundaries. Although the inconsistency of the indicator coverage presented here does not invalidate the foundations and logics of the F–B ESA framework, the system boundaries of each pair of the indicators are better redefined without apparent overlap and inconsistency if the aim is to establish a one-to-one correspondence between the footprint and boundary metrics.

With respect to the obstacles to methodological harmonization, formally recognized approaches to the calculation of sub-global boundaries are still lacking. In contrast, despite the diversity of the footprint family, there has been an empirical principle for selection of appropriate methodologies at sub-global scales: the micro-scale (e.g., product, material) footprints are often subject to bottom-up life cycle assessment (LCA), the macro-scale (e.g., nation, continent) footprints generally commit to top-down input–output analysis (IOA), and the meso-scale (e.g., organization, community) footprints can be addressed by hybrid approaches that combine the strengths of both LCA and IOA (Fang et al., 2014; Peters, 2010). As a result, the ongoing efforts to quantify the boundary metric at sub-global scales should be undertaken in the usual impact systems that matches well with the relevant footprint accounting.

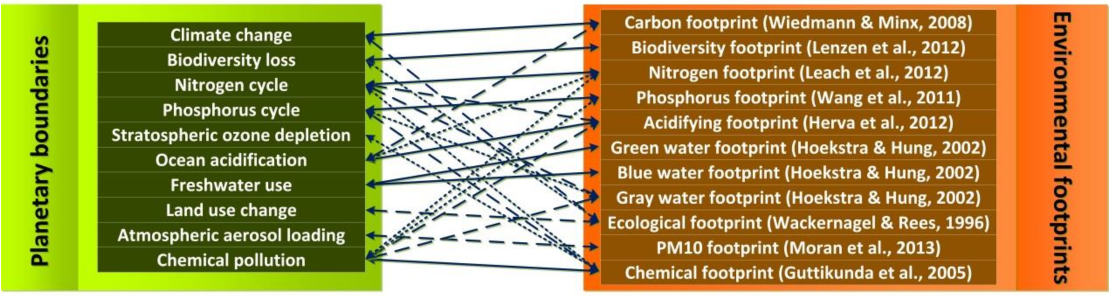

Figure 5.4. Thematic matching of environmental footprints and planetary boundaries. The solid line, long dashed line, and short dashed line represent the degree of matching from high to medium to low.

5.5.4. Trade-offs between different sustainability gaps

Even though in some cases the geographical link between two distant regions is weak, there is no doubt that environmental issues within the Earth's life-supporting systems are essentially interlinked and interactive, and hence cannot be uncoupled from each other (Biermann, 2012). The sustainability of a given region depends, directly or indirectly, on the sustainability of many other regions (Kissinger et al., 2011). In the context of globalization, transgressing one boundary may exert profound intraregional or interregional effects on other boundaries in ways that people do not expect. Thus, maintaining the safe operating space for a single issue without looking at the whole picture seems impractical and no longer a wise policy option. This finding suggests a great need not only for simultaneous assessment of the environmental sustainability of individual issues at different scales, but also for trade-offs between the many sustainability gaps where anthropogenic perturbation ought to be treated with a systematic view.

Such trade-offs can be made thoroughly by ascertaining all categories of environmental footprints and boundaries involved. One difficulty is due to the displacement and leakage effects (Erb et al., 2012), as improvements in some overstepped boundaries are often obtained at the expense of deteriorating other boundaries. In attempts to meet the policy demand for an overall picture of different environmental issues, weighting has been brought to attention with the argument that trade-offs, in many cases, cannot be tackled in a manageable and comprehensive way without some form of weighting (Ahlroth, 2014; Ridoutt and Pfister, 2013).

In contrast to the existing footprint family discourses where conceivable footprints are aggregated to yield a stand-alone, single-score footprint metric (Hadian and Madani, 2015; Ridoutt and Pfister, 2013), we argue for a composite sustainability index that results from weighting and aggregating diverse sustainability gaps into one equation from an integrated perspective. This is accompanied by a recognition that weighting footprints and boundaries simultaneously would offer a more meaningful evaluation of trade-offs than weighting either footprints or boundaries on their own, in particular when policy makers do need a direct comparison of options or entities in terms of their overall performance on environmental sustainability. Approaches to weighting typically include panel methods, distance-to-target, willingness to pay, and more (Ahlroth, 2014).

Weighting sustainability gaps could be considered as a practical possibility, whereas the weighted results should be interpreted with caution. Improper interpretation may violate the intention of the planetary boundaries concept, which is committed to a comprehensive set of non-weighted variables in order to capture diverse global environmental challenges instead of a single-value metric (Nykvist et al., 2013). More importantly, the weighting implicitly legitimates the substitutability of boundaries among environmental issues. While it seems likely to substitute some forms of environmental boundaries, basic life-supporting systems are almost impossible to substitute (Barbier et al., 1994; Moldan et al., 2012).

That is to say, ESA is also a substitution problem in addition to its scaling dimension. A precedent for this is the combination of ecological footprint and biocapacity into a single score, where the weighting scheme has always been steeped in controversy (Kitzes and Wackernagel, 2009; Lenzen and Murray, 2001). The critiques mainly arise from the misinterpretation it may create that the scarcity of carrying capacity for one land's footprint is always allowed to be counteracted by the unconsumed carrying capacity within the boundaries of other lands. Moreover, the lack of differentiation between aggregated issues (e.g., land use) and systemic issues (e.g., carbon emissions) represents another defect in the ecological footprint analysis (EFA), as well as in typical footprint family studies. A partial solution might be to present the results both at the aggregate and disaggregate levels in keeping with the divergent requirements of policy makers and scientists.

5.6. Conclusions

This chapter investigates the complementary linkages between environmental footprints and planetary boundaries originating from two leading communities in the fields of ecological and environmental sciences and doing something quite similar but with their own strengths and weaknesses and surprisingly lacking communication and mutual understanding. The environmental footprints are broadly accepted as a representative of pressure or impact in relation to resource extractions and hazardous emissions, and human knowledge of planetary boundaries provides a set of consensus-based estimates of the regenerative and absorptive capacity of the Earth's life-supporting systems. Although the conceptual roots, calculation methods and policy relevance of different footprints vary widely, in aggregate they show significant similarity to planetary boundaries in terms of environmental coverage.

Both tools are found to be limited in their ability to handle sustainability issues. On the one hand, footprint studies focus typically on measuring and minimizing environmental impacts without seeing if such operationalization is truly in a sustainable way, with some exceptions including the ecological, carbon, water, and chemical footprints, where global and regional threshold measurements are nevertheless far from satisfactory. Misleading conclusions and wrong decisions could be made in the case that the distinction between sustainable and unsustainable activities is vague, ambiguous, or missing at all. On the other hand, while a rough estimate of current human activities is provided in the existing planetary boundaries research besides threshold estimates, it is criticized for the sole use of expert knowledge and lack of quantitative consensus models, which likely lead to unreliable or even false results.

In view of the differing emphases and challenges of footprints and boundaries, it is our conviction that the two metrics should not be viewed as alternatives but rather as complements. The synthesizing of environmental footprints and planetary boundaries makes it possible to benchmark man's contemporary footprints against maximum sustainable footprints (Hoekstra and Wiedmann, 2014), and thereby to indicate the degree to which the functioning of Earth's life-supporting systems has been maintained or crossed. The suggested F–B ESA framework, in this sense, opens the way for a straightforward assessment of environmental sustainability—a non-negotiable prerequisite for the economic and social pillars of sustainable development (Goodland and Daly, 1996).

The growing sustainability gaps between numerous types of environmental burden and the Earth's finite carrying capacity call for a shift from EIA—which may not be informative for policy makers as well as consumers—to ESA, but also from focusing issues in isolation to addressing them simultaneously from an integrated perspective. Today's environmental unsustainability worldwide underpins the need for setting more practical and tangible policy targets for adaptation and mitigation, rather than for unrealistically preventing the overshoot of the Earth's capacity to regenerate resources and assimilate wastes. An example of this is the renegotiation of the "2 °C target" (Parry et al., 2009). Due to the complexity and uncertainty of the environment, a major challenge is how to make trade-offs among various sustainability gaps in support of optimal adaptation strategies in the context of global unsustainability.

We consider this to be a scale problem more than a substitution problem. The quantifications of both boundaries and footprints appear to be strongly scale-dependent (Hughes et al., 2013; Moran et al., 2008; Wiedmann and Lenzen, 2007). This is one reason to take the scale dimension into account when evaluating trade-offs between policy options with consequences for environmental sustainability. Another reason is that the non-transgression of one planetary boundary does not necessarily guarantee a sustainable society, because regional or local boundary exceedance may still give rise to irreversible environmental degradation that is detrimental or even disastrous to the population.

This is particularly true when it comes to aggregated issues that are spatially heterogeneous and local-to-regional in scale. The development of measurable local and regional boundaries is therefore needed. It could serve as a basis for ESA applied to the allocation of environmental responsibility for creating sustainable societies at multiple scales. Lessons can be learned from current methodological choices of environmental footprints. Even for systemic issues, which are believed to have a true global threshold effect, partitioning their planetary boundaries into national or sub-national shares still makes sense. This, however, might be more challenging because of the political attribute of implementing a top-down process that is possibly based on population, GDP, or area, rather than on real regional thresholds.

While the idea of relating descriptive indicators to capacity thresholds is actually not new, this chapter provides concrete discussions of how to bring together the two emerging research fields (environmental footprints and planetary boundaries) as a novel approach for ESA, thus contributing to the ever-developing sustainability discourse. Admittedly, there remain many gaps in our knowledge that may compromise the credibility and applicability of the F–B ESA framework proposed in the thesis. We have therefore gone on at length formulating a research agenda for the global community to continuously improve the performance of the F–B ESA framework on transparency and robustness. This requires a large research effort with contributions from a vast range of fields such as ecology, environmental science, earth science, system science, and social science, and because of this, we call for extensive interdisciplinary communication and collaboration among scientists in each of these disciplines.

References

Ahlroth, S., 2014. The use of valuation and weighting sets in environmental impact assessment. Resources, Conservation and Recycling 85, 34–41.

Allen, M., 2009. Tangible targets are critical. Nature Reports Climate Change 3, 114–115.

Barbier, E. B., Burgess, J. C., Folke, C., 1994. Paradise Lost? — The Ecological Economics of Biodiversity. Earthscan, London, UK.

Barnosky, A. D., Hadly, E. A., Bascompte, J., Berlow, E. L., Brown, J. H., Fortelius, M., Getz, W. M., Harte, J., Hastings, A., Marquet, P. A., Martinez, N. D., Mooers, A., Roopnarine, P., Vermeij, G., Williams, J. W., Gillespie, R., Kitzes, J., Marshall, C., Matzke, N., Mindell, D. P., Revilla, E., Smith, A. B., 2012. Approaching a state shift in Earth's biosphere. Nature 486, 52–58.

Biermann, F., 2012. Planetary boundaries and earth system governance: exploring the links. Ecological Economics 81, 4–9.

Borucke, M., Moore, D., Cranston, G., Gracey, K., Iha, K., Larson, J., Lazarus, E., Morales, J. C., Wackernagel, M., Galli, A., 2013. Accounting for demand and supply of the biosphere's regenerative capacity: the National Footprint Accounts' underlying methodology and framework. Ecological Indicators 24, 518–533.

Brook, B. W., Ellis, E. C., Perring, M. P., Mackay, A. W., Blomqvist, L., 2013. Does the terrestrial biosphere have planetary tipping points? Trends in Ecology & Evolution 28, 396–401.

Catton, W., 1986. Carrying Capacity and the Limits to Freedom. Social Ecology Session, 1, XI World Congress of Sociology. New Delhi, India.

Costanza, R., 2000. The dynamics of the ecological footprint concept. Ecological Economics 32, 341–345.

Costanza, R., Patten, B. C., 1995. Defining and predicting sustainability. Ecological Economics 15, 193–196.

Daily, G. C., Ehrlich, P. R., 1992. Population, sustainability, and Earth's carrying capacity. BioScience 42, 761–771.

Daly, H. E., 1990. Toward some operational principles of sustainable development. Ecological Economics 2, 1–6.

De Vries, W., Kros, J., Kroeze, C., Seitzinger, S. P., 2013. Assessing planetary and regional nitrogen boundaries related to food security and adverse environmental impacts. Current Opinion in Environmental Sustainability 5, 392–402.

Ehrlich, P. R., 1982. Human carrying capacity, extinctions, and nature reserves. BioScience 32, 331–333.

Erb, K.-H., Haberl, H., Defries, R., Ellis, E. C., Krausmann, F., Verburg, P. H., 2012. Pushing the planetary boundaries. Science 338, 1419–1420.

Ewing, B. R., Hawkins, T. R., Wiedmann, T. O., Galli, A., Ertug Ercin, A., Weinzettel, J., Steen-Olsen, K., 2012. Integrating ecological and water footprint accounting in a multi-regional input–output framework. Ecological Indicators 23, 1–8.

Fang, K., Heijungs, R., 2015. Rethinking the relationship between footprints and LCA. Environmental Science & Technology 49, 10–11.

Fang, K., Heijungs, R., De Snoo, G. R., 2014. Theoretical exploration for the combination of the ecological, energy, carbon, and water footprints: overview of a footprint family. Ecological Indicators 36, 508–518.

Fischer, J., Manning, A. D., Steffen, W., Rose, D. B., Daniell, K., Felton, A., Garnett, S, Gilna, B., Heinsohn, R., Lindenmayer, D. B., MacDonald, B., Mills, F., Newell, B., Reid, J., Robin, L., Sherren, K., Wade, A., 2007. Mind the sustainability gap. Trends in Ecology & Evolution 22, 621–624.

Galli, A., Wiedmann, T., Ercin, E., Knoblauch, D., Ewing, E, Giljum, S., 2012. Integrating ecological, carbon and water footprint into a "Footprint Family" of indicators: definition and role in tracking human pressure on the planet. Ecological Indicators 16, 100–112.

Gerbens-Leenes, W., Hoekstra, A. Y., Van der Meer, T. H., 2009. The water footprint of bioenergy. Proceedings of the National Academy of Sciences of the United States of America 106, 10219–10223.

Germain, M., Van Steenberghe, V., 2003. Constraining equitable allocations of tradable CO_2 emission quotas by acceptability. Environmental and Resource Economics 26, 469–492.

Goldfinger, S., Wackernagel, M., Galli, A., Lazarus, E., Lin, D., 2014. Footprint facts and fallacies: a response to Giampietro and Saltelli (2014) "Footprints to Nowhere". Ecological Indicators 46, 622–632.

Goodland, R., Daly, H., 1996. Environmental sustainability: universal and non-negotiable. Ecological Applications 6, 1002–1017.

Guttikunda, S. K., Tang, Y., Carmichael, G. R., Kurata, G., Pan, L., Streets, D. G., Woo, J.-H., Thongboonchoo, N., Fried, A., 2005. Impacts of Asian megacity emissions on regional air quality during spring 2001. Journal of Geophysical Research 110, D20301–D20301.

Hadian, S., Madani, K., 2015. A system of systems approach to energy sustainability assessment: Are all renewables really green? Ecological Indicators 52, 194–206.

Heijungs, R., De Koning, A., Guinée, J. B., 2014. Maximizing affluence within the planetary boundaries. The International Journal of Life Cycle Assessment 19, 1331–1335.

Heijungs, R., Frischknecht, R., 1998. A special view on the nature of the allocation problem. The International Journal of Life Cycle Assessment 3, 321–332.

Hellweg, S., I Canals, L. M., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science 344, 1109–1113.

Hoekstra, A. Y., 2009. Human appropriation of natural capital: a comparison of ecological footprint and water footprint analysis. Ecological Economics 68, 1963–1974.

Hoekstra, A. Y., Hung, P. Q., 2002. Virtual Water Trade: A Quantification of Virtual Water Flows between Nations in Relation to International Crop Trade. Value of Water Research Report Series (No. 11), UNESCO-IHE Institute for Water Education, Delft, The Netherlands.

Hoekstra, A. Y., Mekonnen, M. M., 2012. The water footprint of humanity. Proceedings of the National Academy of Sciences of the United States of America 109, 3232–3237.

Hoekstra, A. Y., Mekonnen, M. M., Chapagain, A. K., Mathews, R. E., Richter, B. D., 2012. Global monthly water scarcity: blue water footprints versus blue water availability. PLoS One 7, e32688–e32688.

Hoekstra, A. Y., Wiedmann, T. O., 2014. Humanity's unsustainable environmental footprint. Science 344, 1114–1117.

Hughes, T. P., Carpenter, S., Rockström, J., Scheffer, M., Walker, B., 2013. Multiscale regime shifts and planetary boundaries. Trends in Ecology & Evolution 28, 389–395.

IPCC, 2014. Climate Change 2014: Mitigation of Climate Change. Working Group III Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, New York, USA.

Kates, R. W., Clark, W. C., Corell, R., Hall, J. M., Jaeger, C. C., Lowe, I., McCarthy, J. J., Schellnhuber, H. J., Bolin, B., Dickson, N. M., Faucheux, S., Gallopin, G. C., Grübler, A., Huntley, B., Jäger, J., Jodha, N. S., Kasperson, R. E., Mabogunje, A., Matson, P., Mooney, H., Moore, B. III., O'Riordan, T., Svedin, U., 2001. Environment and development: sustainability science. Science 292, 641–642.

Kissinger, M., Rees, W. E., Timmer, V., 2011. Interregional sustainability: governance and policy in an ecologically interdependent world. Environmental Science & Policy 14, 965–976.

Kitzes, J., Wackernagel, M., 2009. Answers to common questions in ecological footprint accounting. Ecological Indicators 9, 812–817.

Kosoy, N., Brown, P. G., Bosselmann, K., Duraiappah, A., Mackey, B., Martinez-Alier, J., Rogers, D., Thomson, R., 2012. Pillars for a flourishing Earth: planetary boundaries, economic growth delusion and green economy. Current Opinion in Environmental Sustainability 4, 74–79.

Kratena, K., 2004. 'Ecological value added' in an integrated ecosystem–economy model—an indicator for sustainability. Ecological Economics 48, 189–200.

Lancker, E., Nijkamp, P., 2000. A policy scenario analysis of sustainable agricultural development options: a case study for Nepal. Impact Assessment and Project Appraisal 18, 111–124.

Laurent, A., Olsen, S. I., Hauschild, M. Z., 2012. Limitations of carbon footprint as indicator of environmental sustainability. Environmental Science & Technology 46, 4100–4108.

Leach, A. M., Galloway, J. N., Bleeker, A., Erisman, J. W., Kohn, R., Kitzes, J., 2012. A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. Environmental Development 1, 40–66.

Lenton, T. M., Williams, H. T., 2013. On the origin of planetary-scale tipping points. Trends in Ecology & Evolution 28, 380–382.

Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L., Geschke, A., 2012. International trade drives biodiversity threats in developing nations. Nature 486, 109–112.

Lenzen, M., Murray, S. A., 2001. A modified ecological footprint method and its application to Australia. Ecological Economics 37, 229–255.

Lewis, S. L., 2012. We must set planetary boundaries wisely. Nature 485, 417–417.

Liu, C., Kroeze, C., Hoekstra, A. Y. Gerbens-Leenes, W., 2012. Past and future trends in grey water footprints of anthropogenic nitrogen and phosphorus inputs to major world rivers. Ecological Indicators 18, 42–49.

Meadows, D. H., Meadow, D. L., Randers, J., Behrens, W., 1972. The Limits to Growth. Universe Books, New York. USA.

Minx, J., Baiocchi, G., Wiedmann, T., Barrett, J., Creutzig, F., Feng, K., Förster, M., Pichler, P.-P., Weisz, H., Hubacek, K., 2013. Carbon footprints of cities and other human settlements in the UK. Environmental Research Letters 8, 035039–035039.

Moldan, B., Janoušková, S., Hák, T., 2012. How to understand and measure environmental sustainability: Indicators and targets. Ecological Indicators 17, 4–13.

Molden, D., 2009. The devil is in the detail. Nature Reports Climate Change 3, 116–117.

Moran, D. D., Wackernagel, M., Kitzes, J. A., Goldfinger, S. H., Boutaud, A., 2008. Measuring sustainable development—Nation by nation. Ecological Economics 64, 470–474.

Niccolucci, V., Bastianoni, S., Tiezzi, E. B. P., Wackernagel, M., Marchettini, N., 2009. How deep is the footprint? A 3D representation. Ecological Modelling 220, 2819–2823.

Nykvist B, Persson Å, Moberg F, Persson, L., Cornell, S., Rockström, J., 2013. National Environmental Performance on Planetary Boundaries: a Study for the Swedish Environmental Protection Agency. Stockholm Resilience Center & Stockholm Environment Institute, Stockholm, Sweden.

Parry, M., Lowe, J., Hanson, C., 2009. Overshoot, adapt and recover. Nature 458, 1102–1103.

Peters, G. P., 2010. Carbon footprints and embodied carbon at multiple scales. Current Opinion in Environmental Sustainability 2, 245–250.

Rees, W. E., 1992. Ecological footprint and appropriated carrying capacity: what urban economics leaves out. Environment and Urbanization 4, 121–130.

Rees, W. E., 1996. Revisiting carrying capacity: area-based indicators of sustainability. Population and Environment 17, 195–215.

Ridoutt, B. G., Pfister, S., 2013. Towards an integrated family of footprint indicators. Journal of Industrial Ecology 17, 337–339.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F. S., Lambin, E. F., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., De Wit, C. A., Hughes, T., Van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P. K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R. W., Fabry, V. J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J. A., 2009a. A safe operating space for humanity. Nature 461, 472–475.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F. S., Lambin, E., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., De Wit, C. A., Hughes, T., Van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P. K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R. W., Fabry, V. J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J., 2009b. Planetary boundaries: exploring the safe operating space for humanity. Ecology and Society 14, 32–32.

Rogelj J., McCollum, D., O'Neill, B. C., Riahi K., 2013. 2020 emissions levels required to limit warming to below 2 °C. Nature Climate Change 3, 405–412.

Smeets, E., Weterings, R., 1999. Environmental Indicators: Typology and Overview. Technical Report (No. 25). TNO Centre for Strategy, Technology and Policy, The Netherlands.

Steffen, W., Stafford Smith, M., 2013. Planetary boundaries, equity and global sustainability: why wealthy countries could benefit from more equity. Current Opinion in Environmental Sustainability 5, 403–408.

UNEP, 2013. The Emissions Gap Report 2013: A UNEP Synthesis Report. http://www.unep.org/pdf/UNEPEmissionsGapReport2013.pdf.

Wackernagel, M., Rees, W. E., 1996. Our Ecological Footprint: Reducing Human Impact on the Earth. New Society, Gabriola Island, British Columbia, Canada.

Wackernagel, M., Rees, W. E., 1997. Perceptual and structural barriers to investing in natural capital: Economics from an ecological footprint perspective. Ecological Economics 20, 3–24.

Wang, F., Sims, J. T., Ma, L., Ma, W., Dou, Z., Zhang, F., 2011. The phosphorus footprint of China's food chain: implications for food security, natural resource management, and environmental quality. Journal of Environmental Quality 40, 1081–1089.

Wiedmann, T., Lenzen, M., 2007. On the conversion between local and global hectares in Ecological Footprint analysis. Ecological Economics 60, 673–677,

Wiedmann, T., Minx, J., 2008. A definition of 'carbon footprint'. In: Pertsova, C. C. (Ed.), Ecological Economics Research Trends. Nova Science Publishers, Hauppauge, New York, USA, pp. 1–11.

Wiedmann, T. O., Schandl, H., Lenzen, M., Moran, D., Suh, S., West, J., Kanemoto, K., 2015. The material footprint of nations. Proceedings of the National Academy of Sciences of the United States of America 112, 6271–6276.

Zijp, M. C., Posthuma, L., Van de Meent, D., 2014. Definition and applications of a versatile chemical pollution footprint methodology. Environmental Science & Technology 48, 10588−10597.

Chapter 6

Benchmarking the carbon, water and land footprints against allocated planetary boundaries

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Abstract

Growing scientific evidence for the indispensable role of environmental sustainability in sustainable development calls for appropriate frameworks and indicators for environmental sustainability assessment (ESA). In this chapter, we operationalize and update the footprint–boundary ESA framework, with a particular focus on its methodological and application extensions to the national level. By using the latest datasets available, the planetary boundaries for carbon emissions, water use and land use are allocated to 28 selected countries in comparison to the corresponding environmental footprints. The environmental sustainability ratio (ESR)—an internationally comparable indicator representing the sustainability gap between contemporary anthropogenic interference and critical capacity thresholds—allows one to map the reserve or transgression of the nation-specific environmental boundaries. Although the geographical distribution of the three ESRs varies across nations, in general, the worldwide unsustainability of carbon emissions is largely driven by economic development, while resource endowments play a central role in explaining national performance on water and land use. The main value added of this chapter is to provide concrete evidence of the validity of the proposed framework in allocating overall responsibility for environmental sustainability to sub-global scales and in informing policy makers about the need to prevent the planet's environment from tipping into an undesirable state.

6.1. Introduction

6.1.1. Environmental sustainability assessment (ESA): a brief overview

Humanity has entered a new era of sustainability challenges, the Anthropocene, in which the planet's environment is under significant pressure from social, economic, and demographic forces. In striving to prevent our society and future generations from tipping into disastrous states, sustainable development has remained one of the primary policy goals in the large majority of countries over the world (Griggs et al., 2013). The United Nations is scheduled to announce the Sustainable Development Goals by 2015, an evolving program that is under way to replace the Millennium Development Goals (Costanza et al., 2014). In measuring progress towards sustainable transitions and human well-being, it is necessary to create ways to assess environmental sustainability—a non-negotiable prerequisite for the economic and social pillars of sustainable development (Goodland and Daly, 1996).

There have been many attempts to promote transparency and standardization of ESA. One example is the ecological footprint, which compares human demand for bioproduct provision and carbon sinks with the relevant regenerative and assimilative capacity of the biosphere, thereby explaining why the current economy lives on the depletion of exhaustible stocks rather than on sustainable flows (Wackernagel and Rees, 1997). Apparently, this is by no means the only way of implementing ESA. The Environmental Sustainability Index (Samuel-Johnson and Esty, 2000) and its updated version, the Environmental Performance Index (Esty et al., 2006), for instance, have attracted considerable interest and discussions among science, policy, and in the media. Other influential ESA tools include the Environmental Quality Index (Steinhart et al., 1982), the Index of Environmental Friendliness (Puolamaa et al., 1996), the Environmental Vulnerability Index (Kaly et al., 1999), and the Critical Natural Capital (Sutton and Costanza, 2002).

Despite the continuous efforts made by a large group of researchers, there is no agreement on the most appropriate definition and method for ESA. Nevertheless, on the basis of an in-depth discussion performed in our previous study (Fang et al., 2015), we come up with some key observations on ESA: we come up with some key observations regarding ESA: (1) the essential property of ESA is the comparison of current environmental states and critical capacity thresholds; (2) a descriptive pressure indicator that measures what is currently happening to the environment has no relation to ESA unless it is benchmarked against a critical threshold indicator serving as a reference; (3) the difficulty in prediction of environmental boundaries poses a major challenge to ESA due to uncertainties surrounding the position of the thresholds; and (4) the estimates of ESA are normally expressed either in difference or in ratio, but not in both.

6.1.2. Environmental footprints: descriptive pressure indicators

To represent how much pressure or impact humanity exerts on the Earth's ecosystems, an expanding list of environmental footprints has been introduced to the scientific community over recent years (Čuček et al., 2012; Hoekstra and Wiedmann, 2014). Irrespective of the prevalence of footprint-style indicators, there remains a considerable amount of confusion and controversy regarding the concepts and methodologies (Giampietro and Saltelli, 2014; Kitzes et al., 2009; Van den Bergh and Grazi, 2014). A consequence is that there is, to our knowledge, not yet an explicit and agreed definition of footprints. To bring transparency to this issue, here we propose a general definition as follows: environmental footprints are tools that communicate human-driven pressure on the environment and associated impacts. To our understanding, this is probably the common ground on which most, if not all, footprint indicators depend crucially.

The ecological footprint (Rees, 1992), the water footprint (Hoekstra and Hung, 2002), and the carbon footprint (Wiedmann and Minx, 2008) have been subject to a wide range of scientific scrutiny. A striking overlap has been identified between the ecological and carbon footprints in terms of fossil carbon emissions. To avoid double counting, it is suggested to exclude the energy component from the ecological footprint accounting and to refer to the remainder as "land footprint" which accounts for the pressure associated with actual land use (Fang et al., 2014; O'Brien et al., 2015). Collectively, the carbon, water and land footprints are able to capture the complicated effects of human activities, including carbon emissions (climate change), water use and land use, on the Earth's system processes.

6.1.3. Planetary boundaries: critical threshold indicators

The planetary boundaries framework (PBF) offers a set of quantitative capacity thresholds for vital Earth system processes based on recent scientific evidence (Rockström et al., 2009). Even though planetary boundaries were not designed to downscale to smaller scales (Steffen et al., 2015), there is an increasing demand for allocation to national and regional levels at which numerous environmental policies are formulated and executed with broad participation of stakeholders. This has triggered a growing interest in the development of approaches to detecting sustainability limits and to anticipating critical transitions at multiple scales (Cole et al., 2014; Dearing et al., 2014; Smith et al., 2014). Furthermore, based on recent understanding of the complexity of the Earth system functioning and resilience, Steffen et al. (2015) have implemented updates of planetary boundaries, with a special focus on those that are spatially heterogeneous and show threshold behaviors at sub-global scales.

As one of the most widespread systems of threshold indicators, planetary boundaries present, for the first time, an up-to-date comprehensive estimate of critical values for nine Earth system processes at the global scale. A threshold-behavioral distinction has been made between "systemic processes" with explicit global thresholds, such as climate change, and "aggregated processes" without strong evidence of planetary-scale thresholds, such as water use and land use (Rockström et al., 2009). For systemic processes, in theory their planetary boundaries can be readily allocated to nations and regions through a top-down approach. The reality is, however, that the control variables used for all three systemic planetary boundaries (climate change, ocean acidification and stratospheric ozone depletion) are concentrations or intensity indices (such as 2 °C) that do not allow for yielding remaining budgets and for dividing into country-specific shares. To overcome this issue, all such threshold values must be translated into quantities of flow characters, such as Gt/yr of carbon dioxide (CO_2).

Things get more complicated when it comes to aggregated processes, for which no top-down approach is applicable, because a truly global threshold behavior hardly exists unless boundaries at a local or regional level have been repeatedly transgressed for large parts of the world (Lewis, 2012; Rockström et al., 2009). More specifically, the environmental boundaries for water and land use are essentially dependent on the local availability and scarcity of freshwater and land and thus vary over space, while the planetary carbon boundary potentially can be applied to any specific region, regardless of location. All this points to a key methodological limitation of the present PBF: it is inherently incapable of downscaling environmental boundaries from the planetary level to smaller levels because of the inappropriate control variables and the omission of sub-global dynamics that are critical to both regional and global sustainability.

6.1.4. The need for integrating footprints and boundaries into ESA

In summary, environmental footprints and planetary boundaries fall into the categories of descriptive pressure indicators and critical threshold indicators, respectively. Biophysical elements underlying the two have the potential to constitute an evolving representation of ESA, where a distinction between "sustainable" and "unsustainable" activities is made by showing to what extent a planetary or regional environmental boundary has been approached or exceeded by the corresponding footprint (Fang et al., 2014). This is accompanied by the recognition that planetary boundaries for both systemic and aggregated processes are necessary to allocate to a national or regional level in support of decision-making (Fang et al., 2015).

In this chapter, we first introduce an integrated framework that makes complementary use of environmental footprints and planetary boundaries originating from two isolated research fields. We then apply this framework to a comparative analysis of national-scale environmental sustainability by measuring and integrating three pairs of footprints and boundaries, and finally offer a picture of the overall environmental performance of different countries. The purpose of our analysis is to provide novel insights into the development of a robust methodology for ESA (Section 6.2), and to present available knowledge on the environmental sustainability of 28 nations in a coherent way with respect to carbon emissions, water use and land use (Section 6.3). Section 6.4 comprises discussions on the contribution of selected variables to national performance, the consistency with other estimates, and the policy relevance of a weighted index to trade-offs issues. Conclusions and some final remarks are drawn in Section 6.5.

6.2. Methods

6.2.1. An explanation to the footprint–boundary ESA (F–B ESA) framework

Having recognized the benefits of integrating footprints and boundaries for ESA, Fang et al. (2015) establish a footprint–boundary ESA (F–B ESA) framework for the joint use of a set of footprint and boundary metrics in a systematic way. It comprises the definitions of ecological footprints and planetary boundaries, the development of the concepts of sustainability gap, policy gap and implementation gap, and a technical discussion of the measurement of regional and local environmental footprints and boundaries and of the trade-offs between different sustainability gaps. The logic is to identify some crucial environmental issues, select suitable footprint indicators for each of them, demonstrate at which level the environmental boundaries are most likely to exist, determine the appropriate methods for quantifying relevant boundary indicators, and finally account for the resulting sustainability gaps. To ensure the consistency and reliability of the F–B ESA framework, it is argued that the footprint and boundary metrics should be measured in such a way that they match well with each other.

The F–B ESA framework challenges, for the first time, the isolation of the two research communities and opens the door to collaborative research in defining and assessing environmental sustainability at multiple scales. Its primary purpose is to inform policy makers on the sustainability gap between current environmental states and critical capacity thresholds, indicating whether or not human activities have fallen into an unsustainable state that may result in undesirable environmental changes, with detrimental or even disastrous consequences for the population of a region or the world. By use of the F–B ESA framework, environmental sustainability and unsustainability can be explicitly defined and distinguished from a footprint–boundary perspective:

- **Environmental sustainability**: a safe state in which the footprint of human activities placed on the environment is kept within boundary of capacity.
- **Environmental unsustainability**: an unsafe state in which the footprint of human activities placed on the environment exceeds boundary of capacity.

6.2.2. Selecting key environmental footprints for cross-national analysis

In view of the ever-expanding list of environmental footprints, we have to restrict our analysis to some key footprints that have proved useful in measuring human pressure on the environment. We have therefore made a selection of footprints with a focus on their scientific robustness, documentation, and applicability. Our selection contains the carbon footprint, the water footprint (consisting of the green and blue water footprints) and the land footprint for the following reasons: (1) the three footprints are sufficiently mature to support policy makers and the public in accounting for the appropriation of natural capital in terms of carbon emissions, water use and land use (Fang et al., 2014), in particular for cross-national analysis that will be undertaken in the remainder of this chapter; (2) in the inventory stage, no obvious overlap has been observed among the three footprints, which allows one to avoid counting the same resources or emissions in duplicate; (3) of the aggregated processes, water use and land use are identified as the only two which are likely to show threshold behaviors suited for measurement on a nation-specific basis (Cole et al., 2014); and (4) the reason not to include grey water is that its data presently available are related to nitrogen loads, which mostly contribute to groundwater pollution and therefore ought to be compared with the waste assimilative capacity of groundwater (Schyns and Hoekstra, 2014), for which the data at the national level are not easily obtainable.

6.2.3. Data sources

We conduct an analysis on the latest datasets available. All data relate to nation-specific carbon, water and land footprints. In the absence of better datasets for the same year, we tentatively assemble these data with the assumption that they are sufficiently comparable. Given data availability, 28 countries have been chosen in this study for empirical analysis, namely, Australia, Austria, Belgium, Brazil, Canada, China, Denmark, Finland, France, Germany, India, Indonesia, Ireland, Italy, Japan, Mexico, Netherlands, Norway, Poland, Russia, South Africa, South Korea, Spain, Sweden, Switzerland, Turkey, the UK, and the USA. Data sources are juxtaposed in Table 6.1. Note that, for convenience, we name either the world-average values or the global values as global values.

Table 6.1. Data sources for the estimation of the footprint and boundary metrics used in this chapter.

Footprint metric	Data source	Boundary metric	Data source
Carbon footprint	EUREAPA (2011)	Carbon boundary	IPCC (2014); PRB (2009); UNEP (2014)
Water footprint	Mekonnen and Hoekstra (2011)	Water boundary	FAO (2012)
Land footprint	GFN (2012)	Land boundary	GFN (2012)

6.2.4. Allocating selected planetary boundaries to nations

In accordance with the footprints chosen, three planetary boundaries (carbon, water and land) are supposed to be reconceived in a way that allows for allocating to the national level and for consistency and commensurability with the footprint metric. Clearly the top-down process applies to the downscaling of planetary carbon boundary. For the planetary boundaries for water and land, a different approach is taken due to the geographical heterogeneity of these natural resources. Thus, in the following, we address the two categories of environmental boundaries separately.

6.2.4.1. Downscaling planetary carbon boundary on a per capita basis

It is perceived as a consensus that human-induced carbon emissions contribute to global warming and climate change no matter where emissions take place (IPCC, 2014). The global nature of planetary carbon boundary allows us to allocate quotas of emission permits using a top-down approach. Technically, partitioning the "cake" can be fulfilled in a number of ways, based on the criterion of population size, economic output, territorial area, or historical responsibility. Each of these has relative merits and demerits, and because of this, we consider it as a normative or political issue more than a scientific issue (Fang et al., 2015). Nevertheless, in this chapter, national population is selected as the basis for allocating planetary carbon boundary to country-specific shares, for reasons of simplicity and comparability with the national water boundary and land boundary which will be evaluated on a per capita basis as well but with different methods.

It should be noted that considerations of fairness and equity are not fully accounted for in this case, as the carbon boundary is built upon a scientific and political consensus reached by the global community that from now on every human being ought to be equally responsible for the reduction in carbon emissions in order to limit global warming to 2 °C above the preindustrial level (IPCC, 2014; Rogelj et al., 2013). That is also the premise of the planetary carbon boundary set by Rockström et al.. This corresponds to a maximum of 18–25 Gt CO_2-eq./yr for annual carbon emissions by 2050, where non-carbon greenhouse gas (GHG) emissions have been translated into carbon emissions by using global warming potentials (GWPs) (Hoekstra and Wiedmann, 2014; UNEP, 2014). With the global population of 6.8 billion in 2009 (PRB, 2009), we adopt the mean value of 3.1 t CO_2-eq./yr for both planetary and national carbon boundary per capita.

6.2.4.2. Quantifying national water and land boundaries on a resource availability basis

The distribution of worldwide water and land resources is geographically heterogeneous, with implications for local or regional environment in most cases. This calls for the need to take spatial variations in resource scarcity into account (Aubauer, 2011; Hoekstra, 2009). In theory, nation-specific environmental boundaries for resource use can be quantified either through the aggregation of local or regional scarcity thresholds, if present, or through the overall estimate of resource availability within the national borders. The first approach, which likely leads to more accurate results, is constrained by the lack of data on local resource depletion with evidence of threshold behavior and of knowledge on the cumulative effects of multiple regions (i.e., it is unclear whether the national environmental boundary simply amounts to the sum, maximum or minimum of regional environmental boundaries). By contrast, the second approach under current conditions has been found preferable to measure national water and land boundaries (Cole et al., 2014; Nykvist et al., 2013).

It has been demonstrated that 90% of the green water availability throughout the world is required to maintain the operation of critical ecosystem services irrespective of human actions (Bogardi et al., 2013; Rockström et al., 1999), and that a reference value for the green water availability is not yet available (Hoekstra and Wiedmann, 2014). All this allows the blue water availability to be an approximation to the national water boundary, defined as annually renewable water supply minus environmental flow requirements for ecological health within the border of a country (Hoekstra et al., 2012). Consideration of these environmental flows is necessary to avoid disastrous impacts associated with water scarcity (Gerten et al., 2013). Following Rockström et al. who proceed with planetary water boundary on the explicit understanding that undesirable or even disastrous consequences may trigger if the ratio of water withdrawal to the renewable supply surpasses 40%, this chapter defines the water boundary for a country as 40% of the total renewable water resources—the sum of the internal and external water resources of that country. Data for the renewable water resources of individual countries are obtained from the Food and Agriculture Organization's (FAO, 2012) database AQUASTAT. In theory, the sum of the water boundaries (not per capita) for all nations should be equal to the total PB (not per capita). However, there are some reasons that explain why the two numbers would not be identical, such as the variation in data sources, and the double counting of the external water resources in both the upstream country and downstream country.

In the original version of the PBF, land boundary was assessed by the criterion that a maximum of 15% of Earth surface is allowed to convert to cropland (Rockström et al., 2009). This brings the risk of underestimating the role of crop production in human survival. In practice, converting land for farming would, on the contrary, promote a great deal of welfare and therefore deserving of positive evaluation (Bass, 2009). In the updated version the control variable for land boundary has altered to be the percentage of the forest land area, with a minimum threshold of 75% (Steffen et al., 2015). But this remains the limitation of incompleteness. To overcome this concern, we adopt biocapacity, an aggregate indicator that measures the critical thresholds for biologically productive land use (Wackernagel and Rees, 1997), as a proxy of national land boundaries that are needed to reach self-sufficiency. In contrast to Rockström et al.'s contention, the biocapacity assumes that the most suitable land available will be planted to cropland, and that the second, third, fourth and last choices are forest land, grassland, fishing ground, and built-up land, respectively (Borucke et al., 2013). Data for the revised national land boundary per capita are derived from GFN (2012).

6.2.5. Measuring the sustainability gap: two alternative indicators

As yet, neither the footprint metric nor the boundary metric is able to capture both sides of the sustainability issues; therefore, they should rather be used complementarily to allow for quantitative estimates of the extent to which human pressure or impact on the environment has approached or exceeded the critical capacity thresholds. This leads to the concept of sustainability gap, which aims at providing policy makers with a practical measure of anthropogenic interference with the planet's environment. In principle, a sustainability gap can be measured in two alternative ways, either by subtracting the footprint metric from the respective boundary metric, or by dividing the footprint metric by the respective boundary metric. We elaborate on these in the following subsections. Before that, we define the following symbols:

- **$F_{i,j}$ is the converted footprint for environmental issue *i* for country *j*.**
- **$B_{i,j}$ is the converted boundary for environmental issue *i* for country *j*.**

where the index *i* runs over carbon, water and land, and the index *j* runs over the 28 nations. Because all indicators are measured on an annual basis, the symbols F and B are expressed in the units of kg CO_2-eq./yr for carbon, m^3/yr for water, and ha/yr for land.

6.2.5.1. Environmental sustainability distance (ESD)

The environmental sustainability distance (ESD) refers to the difference between a footprint and the relevant boundary:

$$ESD_i = F_i - B_i \tag{6.1}$$

In cases where $ESD_i>0$, the footprint metric exceeds the corresponding boundary metric and leads to environmental unsustainability, and vice versa.

6.2.5.2. Environmental sustainability ratio (ESR)

The ESR refers to the ratio of a footprint to the relevant boundary:

$$ESR_i = \frac{F_i}{B_i} \tag{6.2}$$

In cases where $ESR_i>1$, the footprint metric exceeds the corresponding boundary metric and leads to environmental unsustainability, and vice versa.

6.2.5.3. *The comparative advantage of ESR over ESD in a cross-national context*

In principle, both ESDs and ESRs that depict how far countries are from their respective environmental boundaries are adequate for representing the magnitude of the (un)sustainability of a single or multiple environmental issue. As a result, there is no need to make use of them at the same time. The difference between ESDs and ESRs is the way of representing sustainability gaps in absolute terms of the former and in relative terms of the latter. An ESR that serves as a dimensionless indicator suffices to provide comparable values for human pressure not only across regions but also across environmental issues, rather than having to convert disparate values into a single unit, as is the case for an ESD. This comparative advantage enables a better performance of ESRs, especially in a cross-national context. We, therefore, choose to utilize ESRs for the following analysis.

6.2.5.4. *Production-based versus consumption-based ESR*

In defining an ESR, the terms in the quotient are made up by the footprint and boundary. For environmental footprints, a distinction has already been made between production-based and consumption-based footprints (Peters, 2008; Wiedmann, 2009). Following this logic, we can contrast production-based ESRs and consumption-based ESRs. Both viewpoints are legitimate and have a policy value. A production-based ESR refers to the degree of environmental unsustainability due to activities within the border of a country. A consumption-based ESR ignores the activities for exported products, but, on the other hand, includes the activities that take place abroad and that are associated with products imported by the region of concern. ESA and the ESRs, therefore, depend on the definition and the perspective. The ESA of nations will be implemented from a production point of view rather than a consumption point of view, irrespective of the inconsistencies with footprint data obtained from the literature.

6.3. Results

To be consistent across our analysis, the same coloring system is employed to graph the ESRs for all the three environmental issues, namely, carbon emissions (climate change), water use, and land use. Further, we choose to display the log(ESR) instead of an ESR, to ensure that a ratio of 2/1 and a ratio of 1/2 are treated equivalently, for instance. We consider this to be superior to data-dependent normalization depending exclusively on what are included in the sample. Countries with environmental sustainability (ESR<1 or log(ESR)<0) are shown in green colors, and countries with environmental unsustainability (ESR>1 or log(ESR)>0) are shown in red colors. ESR = 1 or log(ESR) = 0 is the magical value. However, one cannot claim to be so precise. For this reason, we define the log(ESR) between −0.25 and 0.25 to be the risky intervals, which corresponds to an ESR between 0.56 and 1.78. While an ESR bounded by less than 0.56 might not be the sufficient conditions for environmental sustainability, we believe that it is a prerequisite for that. In the following, we examine the ESRs for the 28 nations in terms of carbon emissions, water use and land use, respectively.

6.3.1. The environmental sustainability of national carbon emissions

In the case of carbon emissions, the global ESR is estimated at 2.42, indicating that the human economy emits nearly 2.5 times the permissible GHGs, beyond which global warming by 2050 is very likely to cross the 2 °C target. This estimate highlights the worldwide unsustainability of the Earth's climate system arising from immoderate carbon emissions. Figure 6.1 presents a "heat map" of national-scale environmental sustainability associated with carbon emissions. India is the only country operating within the carbon boundary, with an ESR of 0.54. The three countries falling in the risky intervals between sustainability and unsustainability are Indonesia, Brazil and China, with an ESR of 0.72, 1.24 and 1.31, respectively. The rest of the countries go beyond the stable environmental state, of which the ESRs vary considerably, ranging from 1.79 for Turkey to 8.62 for the USA. Twenty-two nations have ESRs higher than the global value, and this is perhaps partly due to the fact that many of the countries investigated here are developed countries which tend to rely more heavily on carbon-intensive goods and services.

As a genuine "natural availability" is lacking and is replaced by a per capita share of the planetary carbon boundary, there are several large countries that already exceed this share by far, such as the USA and China. While China has overtaken the USA as the world's top annual emitter of CO_2 (Jones, 2007), our analysis demonstrates that its ESR exceeds 5.5 times that of China and thus remains the primary contributor to the unsustainability of the global climate system. This discrepancy partially results from the difference between production-based accounting and consumption-based accounting—two competing perspectives that have been in conflict to some extent. Moreover, the advances in carbon boundary accounting in this study offers a way to bridge the science-policy interface, as it encompasses scientific and political consensus on tipping points for sustainability under global climate change.

6.3.2. The environmental sustainability of national water use

Unlike carbon emissions, we benchmark the national water footprint against the national water availability within the political boundary, rather than comparing national performance on a global base which is only applicable to systemic processes. As such, the method used is specifically constructed to yield a conservative estimate of the ESRs of some countries. Consequently, another 11 countries are found to fall into the risky intervals, with ESRs ranging from 0.57 for USA to 1.66 for Poland. We attribute the "unsustainable" label to five nations, among which Denmark has the highest ESR of 3.07, followed by South Africa (2.87), South Korea (2.46), Spain (2.23), and Belgium (2.05). The water ESR is 0.48 on average in the world. Lower-than-average ESRs are found in 11 countries, ranging from 0.04 for Norway to 0.43 for Switzerland, as shown in Figure 6.2. They are, in fact, those having ESRs less than the lower limit of the risky intervals as well and therefore can be classified as sustainable nations. The remaining one which is also sustainable is The Netherlands, whose internal water resources account for only 12% and the external account for the remainder of 88%.

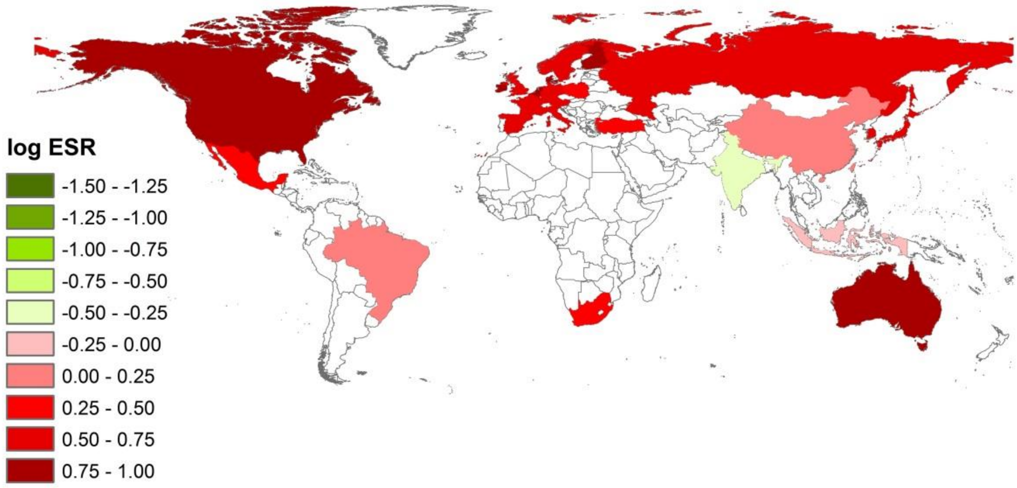


Figure 6.1. The environmental sustainability ratios of carbon emissions for 28 nations.

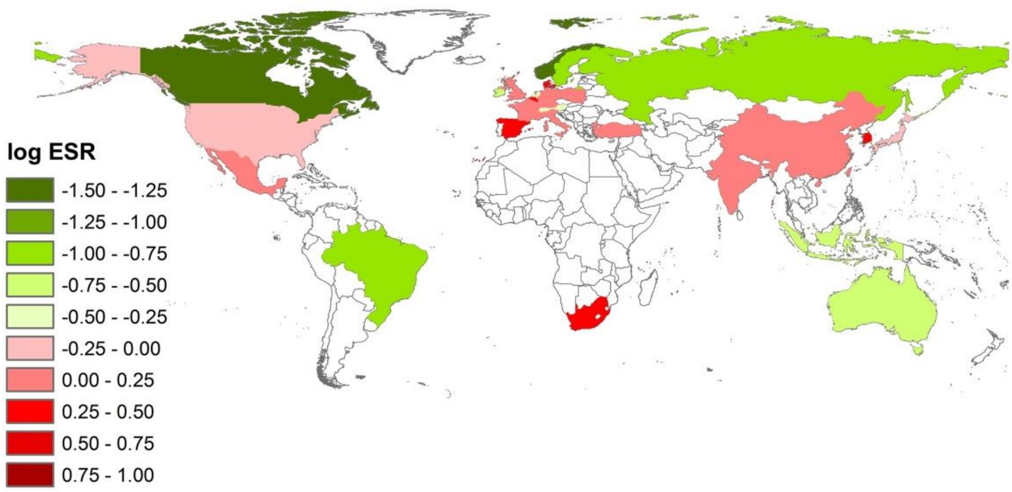


Figure 6.2. The environmental sustainability ratios of water use for 28 nations.

While the distribution of the national water ESRs is in general agreement with monthly water scarcity in the world's river basins monitored by Hoekstra et al. (2012), one should be aware that non-transgression of national water boundary does not necessarily mean environmental sustainability locally and regionally. For instance, though Australia has an ESR of 0.24 merely, large parts of eastern Australia suffer from severe water scarcity due to the disproportionate withdrawal of water resources compared to availability (Hoekstra et al., 2012). However, we are convinced that the ESA of water use at the national scale still makes sense, because, so far, most studies are limited to the basin-level at which little information could be provided on nation-wide water management policy. However, a technical challenge remains in the approximation of the "40% rule", as it fails to capture the spatial variations in the thresholds for water withdrawal within individual nations; this is, nevertheless, a general shortcoming in present methods for defining national water boundaries (Nykvist et al., 2013).

6.3.3. The environmental sustainability of national land use

Figure 6.3 describes which countries approach the individual national land boundaries, and which already overstep. At the global scale, the land ESR is estimated to be 0.67, suggesting that human demand for food, fiber and timber products has been operating in a risky situation in which the regenerative capacity of land supply becomes a limiting factor for sustainable land use. In keeping with this assessment, similar risky situations can be witnessed in 17 countries, accounting for 61% of the countries investigated. Seven countries are able to maintain the maximum capacity within their national land boundaries. Of the seven, Finland and Canada have the lowest ESRs, with a value of 0.15 and 0.18, respectively. The rest of the 28 countries constitute the top-four list, whose land requirements for self-sufficiency under current bioproductivity are much higher than the available resources, with notable ESRs ranging from 2.29 for South Korea to 2.77 for The Netherlands.

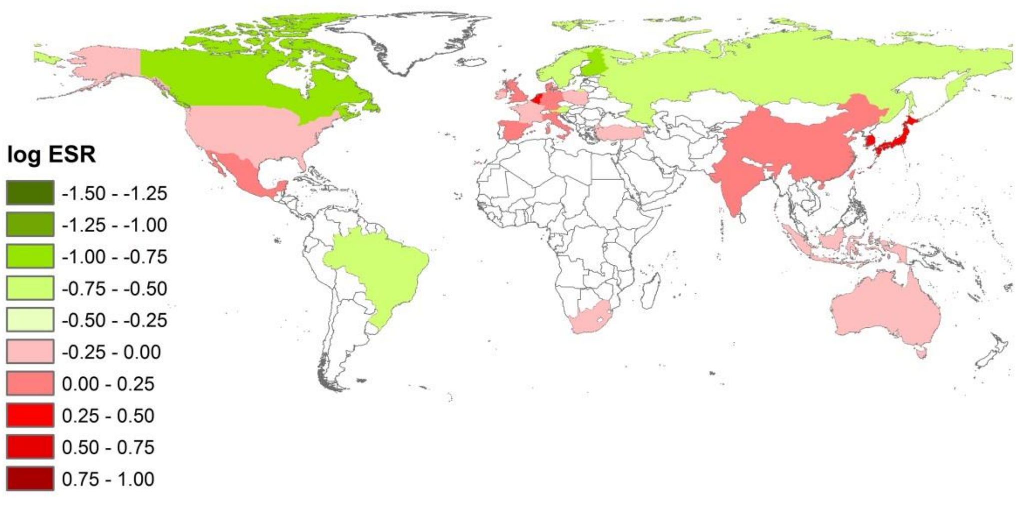


Figure 6.3. The environmental sustainability ratios of land use for 28 nations.

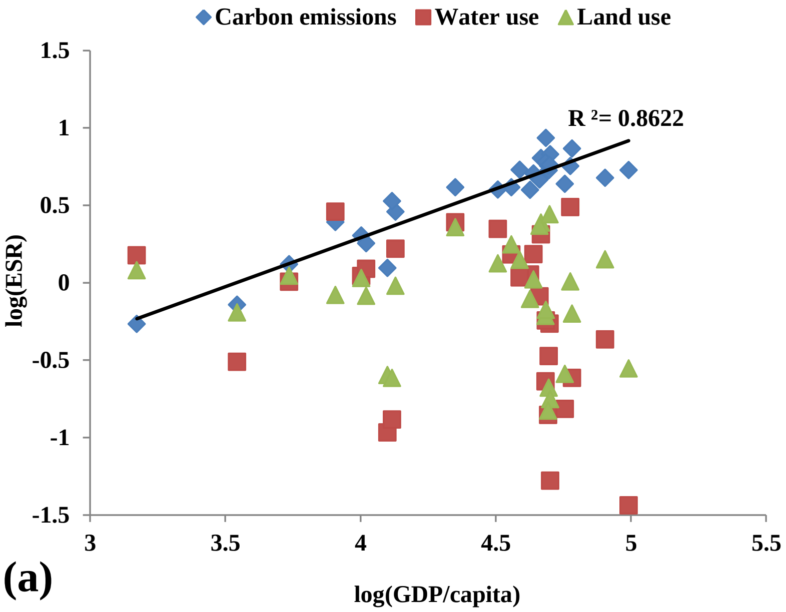
As a whole, the geographical distribution of national ESRs for land use follows a pattern somewhat similar to that for water use. Likewise, the driving forces behind unsustainable land use are complex, including not only the mismatch between region-specific population and land supply, which leads to environmental consequences local-to-regional in scale and which in aggregate may be of national significance, but also the displacement of land use to other countries through international trade (Meyfroidt et al., 2013; Weinzettel et al., 2013). In this sense, the land footprint and boundary measured in this chapter are informative for policy makers who seek to evaluate national performance on land use in a comparative sense. This valuable information is likely to get lost if replaced by the simplistic metrics representing the percentage of land use converted to cropland, as is done by Rockström et al., or the percentage of forest land area, as is done by Steffen et al.

6.4. Discussion

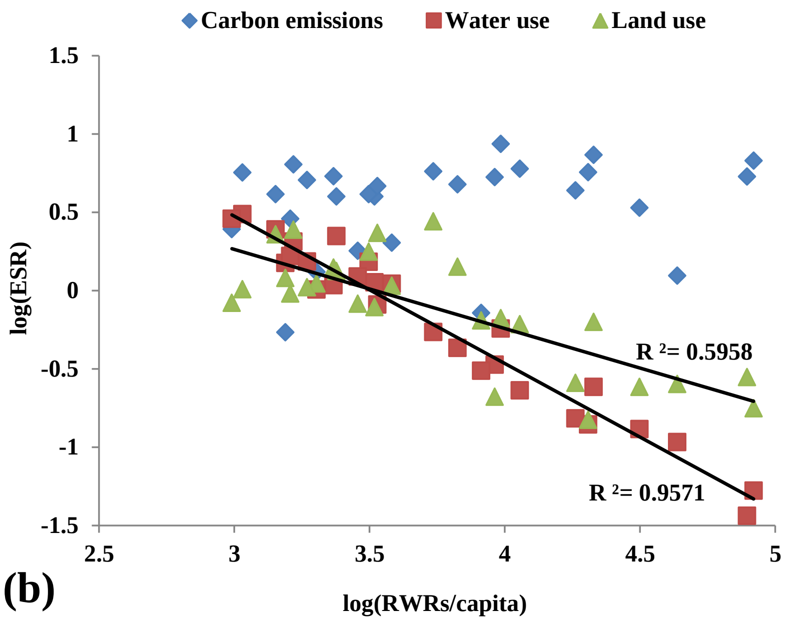
The analysis carried out provides a preliminary integrated assessment of environmental sustainability for 28 countries. Our findings highlight the national-level heterogeneity of both anthropogenic interference and capacity thresholds for carbon emissions, water use and land use. Although the determinants of the sustainability gaps associated with the three environmental issues are definitely complex and may vary across countries, we argue that the complementary distribution of nations between emissions and resource use (water and land use) presented in this study can offer important information to policy makers. To that end, a correlation analysis is undertaken by a regression of log-transformed data to measure the strength of potential linear relationship between the ESR and three variables: (a) Gross Domestic Product (GDP) per capita; (b) renewable water resources (RWRs) per capita; and (c) population density (PD), as will be illustrated below.

As seen in Figure 6.4a, very few countries have already achieved a decoupling of economy from carbon emissions, as evident from the highly significant positive correlation between GDP per capita and the carbon ESR ($R^2 = 0.8622$). By contrast, the ESRs of water and land use have no correlation with GDP per capita since their correlation coefficients do not pass the significance test of the regression with 95-percentile confidence intervals ($p<0.05$). An opposite situation is observed when it comes to Figure 6.4b, in which per capita RWRs have a non-significant correlation coefficient with the carbon ESR, while being closely correlated to that of land use ($R^2 = 0.5958$) and, not surprisingly, significantly to that of water use ($R^2 = 0.8756$). Figure 6.4c exposits positive significant correlations between the PD and ESRs of water use ($R^2 = 0.6310$) and land use ($R^2 = 0.6157$), while the latter is not as strong as commonly thought. One reason for that might be the salient differences across land types and nations in bioproductivity—the key parameter for determining the land footprint and land boundary.

The correlation coefficient between each pair of the explanatory variables indicates that, for the 28 countries on average, approximately 86% of the variation in the log(ESR) of carbon emissions can be explained by log(GDP/capita). This is quite different from the cases of water use and land use, where between 53% and 96% of the variation can be explained by log(RWRs/capita) or log(PD). We further test the three variables together in a multiple regression (Table 6.2), and see that all the three ESRs are explained very well by the three variables. GDP is a major driving factor for the carbon ESR, but not for that of water and land. For water, the RWRs are highly significant, and for land, the interplay turns out to be more complicated and there is no clear driving factor.



(a)



(b)

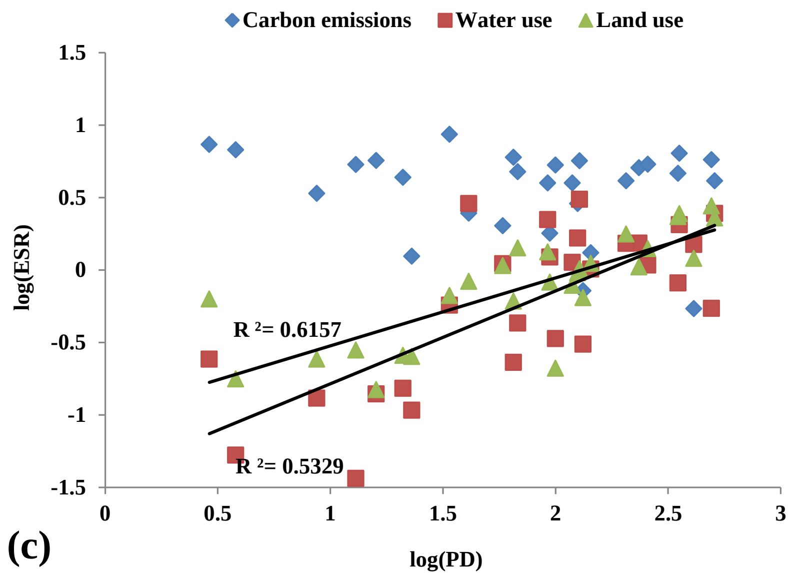


Figure 6.4. Correlations between log(ESR) and (a) log(GDP/capita), (b) log(RWR/capita) (c) log(PD). R^2 is the coefficient of determination (the square of the correlation coefficient). Lines with a non-significant correlation (p>0.05) are not shown.

Table 6.2. Multiple regression of log(ESRs) on three log-transformed variables.

Log (ESR)	log(GDP)			log(RWR)			log(PD)			Model fit		
	b	*SE*	*p*	*b*	*SE*	*p*	*b*	*SE*	*p*	R^2	*F*	*p'*
Carbon	0.658	0.049	0.000	−0.149	0.061	0.023	-0.110	0.055	0.055	0.889	64.226	0.000
Water	0.054	0.050	0.290	−1.037	0.063	0.000	-0.097	0.056	0.097	0.963	209.248	0.000
Land	0.128	0.098	0.204	−0.299	0.123	0.023	0.275	0.109	0.019	0.698	18.524	0.000

b: regression coefficient; *SE*: standard error; *p*: the *p*-value of the *t*-test for β=0; R^2: the coefficient of determination of the regression model; *F*: *F*-statistic; *p*': the significance of the overall model.

In summary, the unsustainability of carbon emissions is largely driven by the stage of economic development on which a nation finds itself, while resource endowments play an important role in the degree of unsustainable water and land use (PD can be seen as the inverse of per capita land resources). These findings confirm the inherent distinction between the systemic and aggregated processes. In addition, Figure 6.5 shows that the global ESR values for the three environmental issues agree well with other estimates of the ratios of pressures to thresholds in the literature, supporting the conclusions of earlier studies on the transgression of planetary carbon boundary and on the reserve of planetary water and land boundaries. Nevertheless, a major divergence is encountered in the case of carbon emissions, where the planetary carbon boundary is measured based either on joint use of two control variables—atmospheric CO_2 concentration and radiative forcing—not allowing to downscale, as is done by Rockström et al., or on an integrated alternative—CO_2-equivalent emissions—a mass-based indicator that is more inclusive and expressed in the same unit of the carbon footprint, as is done by Hoekstra and Wiedmann (2014) and by our study.

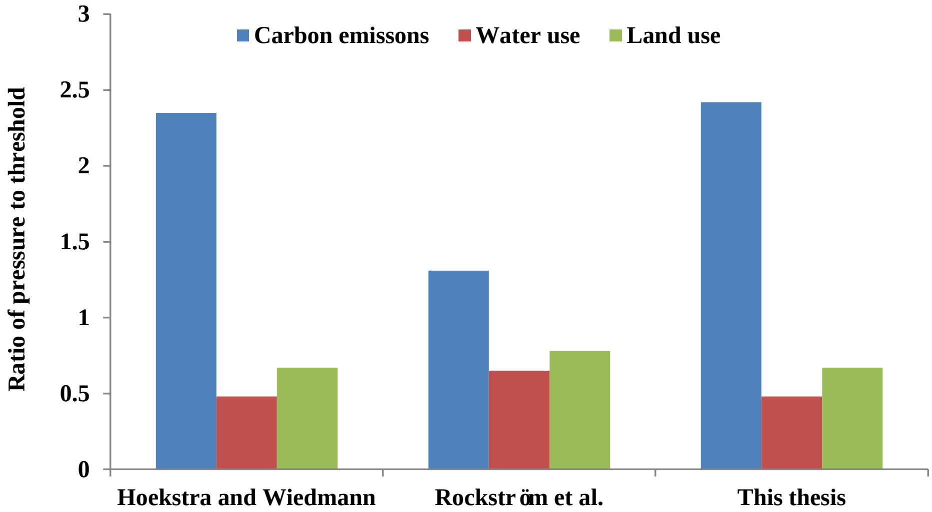


Figure 6.5. Comparison of the ratios of pressures to thresholds between our estimate and the estimates in the literature.

As such, it is our conviction that the F–B ESA framework allows one to improve the performance of the planetary boundaries accounting on integrity, transparency and, more importantly, the harmonization with footprint indicators. Another prominent merit of the F–B ESA framework is its capacity to outline the methodology for assessing trade-off issues, such as one country and another that generates a lower carbon footprint but a higher water footprint. One solution is not only to benchmark each national environmental footprint against the respective environmental boundary in commensurable units but also to aggregate different sustainability gaps between them into a composite index through weighting factors. The rationale behind this is that weighting the footprint and boundary metrics simultaneously enables a more meaningful evaluation of trade-offs than weighting either footprints or boundaries on their own, in particular when policy makers are in great need of a quantitative comparison of the overall performance of nations on environmental sustainability (Fang et al., 2015).

To illustrate the idea of weighting without pretending to give a conclusive answer, for each single country we purposely bring together the three ESRs into a composite index of ESR (ESRI) with an assigned weighting factor of 1/3. All resulting ESRI scores for the 28 countries are presented in ascending order (Figure 6.6). Specifying an ESRI of 1 as the threshold value for national-scale environmental sustainability, we find that only Indonesia and Brazil maneuver within their repective safe operating space, and that the other 26 countries are in the state of environmental unsustainability, among which the disparity of the ESRIs is significant, ranging from 1.13 for China to 4.34 for The Netherlands. By carrying out a multiple regression of log(ESRI) on log(GDP), log(RWR) and log(PD) (Table 6.3), we find that the ESRI is explained quite well by GDP and the renewable water resources, but that the population density is not helpful in explaining the weighted ESRI according to 95-percentile confidence intervals. As a whole, the ranking of countries reflects the overall performance of national-scale environmental sustainability influenced by multi-factors, justifying the use of ESRI in providing solutions to trade-off issues.

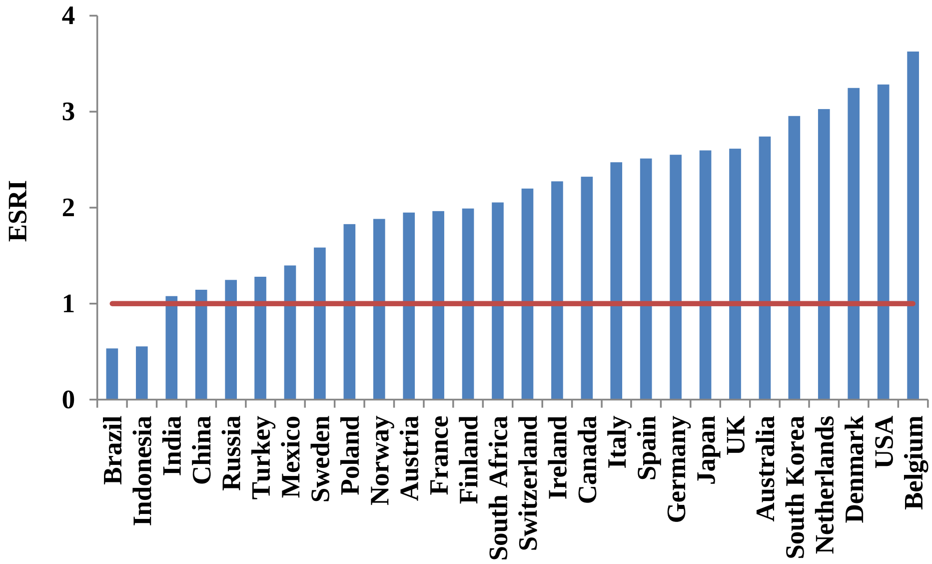


Figure 6.6. The index of environmental sustainability ratio for each of the 28 nations.

Table 6.3. Multiple regression of log(ESRs) on three log-transformed variables.

	log(GDP)			log(RWR)			log(PD)			Model fit		
	b	*SE*	*p*	*b*	*SE*	*p*	*b*	*SE*	*p*	R^2	*F*	*p*
log(ESRI)	0.412	0.049	0.000	−0.255	0.061	0.000	−0.066	0.055	0.240	0.766	26.117	0.000

See Table 6.2 for an explanation of the symbols.

As is the case for all weighting practices, equal weighting that assumes equal importance of the carbon, water and land ESRs in our case has been steeped in controversy (Berger and Finkbeiner, 2013; Bruns and Shefferson, 2004). That is to say, it would be preferred if the final results are presented both at aggregate and disaggregate levels. Other weighting schemes may easily be implemented, if desired, to investigate the sensitivity of final results to weighting factors. For instance, Tuomisto et al. (2012) define weighting factors for the aggregation of diverse local-specific environmental issues in a way similar to the calculation of the global ESRs calculated here, so that one can weigh the seriousness of different issues at the global level. As an aside, from a broader point of view, the ESRs that describe the gap between the footprint and boundary metrics can be understood as a measure of environmental impacts. In this sense, weighting and aggregating ESRs into a single-value ESRI would facilitate the ongoing development of integrated environmental impact assessment—a topic that remains a priority for future endeavors (Chapman and Maher, 2014).

6.5. Conclusions

It has been increasingly acknowledged that environmental sustainability serves as a critical prerequisite for the economic and social pillars of sustainable development. Environmental sustainability analysis, therefore, deserves priority in sustainability sciences (Kates, 2011). This chapter develops ways to allocate planetary boundaries to the national scale and to benchmark against environmental footprints, with the intention of bridging the disciplinary gap and, more importantly, of making use of the synergies for ESA. By means of the F–B ESA framework, we are able to uncover the sustainability gaps of carbon emissions, water use and land use for 28 countries and the whole world. The well accordance with previous studies at the global scale allows our study to be as a whole reliable and reproducible. By examining the correlation between the resulting ESRs and selected explanatory variables, we also discuss certain possible driving factors for unsustainable resource use patterns. Furthermore, seeking to meet the rising policy demand for an overall picture of the environmental sustainability of nations, the ESRI is launched for snapshotting and ranking nations' overall performance on the three environmental issues investigated.

The main value added of the chapter is to provide concrete evidence of how the F–B ESA framework makes it possible to allocate global responsibility for environmental sustainability to individual countries where environmental policy initiatives massively take place. It is not difficult to expect application extensions to sub-national scales, such as regions, cities, and organizations, on which environmental issues are dealt with even more often. A key difficulty concerns the disparate scaling effects of the systemic and aggregated processes. For carbon emissions, for instance, the unambiguous global nature enables the carbon boundary to be one of the few planetary boundaries that can be downscaled to the national level through top-down approaches that seem to be a normative or political issue more than a scientific issue. Conversely, water and land boundaries are in many cases a sub-national problem, for which reliable local assessments could only be fulfilled by the access to high-resolution data for local-scale resource availability. The real challenges are therefore a mix of downscaling and upscaling, as well as a convergence of scientific and political considerations.

Admittedly, we realize that the analysis presented is limited in the capacity to capture the full complexity of sustainability, as proved by the crude data gathering, the exclusion of many environmental issues by design, and the orientation towards macro- or meso-level that hampers the allocation of overall responsibility to a single process or product at the micro-level (Hoekstra, 2015). A further critical point is that, although benchmarking the green and blue water footprints against the blue water availability creates a solution to the overly optimistic estimate of planetary water boundary (Jaramillo and Destouni, 2015; Molden, 2009), this comparison is flawed in the sense that the scopes of the water footprint and boundary metrics are not identical. To improve the comparability of studies that define environmental sustainability from authors' point of view, further work needs to specify the precise realms of application of the production-based and consumption-based ESRs. But regardless of the choice, it is preferable to define the footprint and boundary metrics along the same principle, so that both numerator and denominator are either production-based or consumption-based. In addition, the investigation into the driving forces behind sustainability gaps is far from an exhaustive factorial analysis, even though it manages to provide an interesting basis for discussion on the importance and complexity of economic, natural and demographic factors for understanding national performance on environmental sustainability. Besides, the use of equal weighting is debatable with high uncertainty in the final estimate. This, however, is due to the recognition that trade-offs between sustainability gaps, in many cases, cannot be tackled without any form of weighting. Anyhow, in response to all these challenges that have to be confronted in implementing and developing the F–B ESA framework, we call for multidisciplinary and interdisciplinary collaboration between the fields of environmental footprints and planetary boundaries.

References

Aubauer, H. P., 2011. Development of ecological footprint to an essential economic and political tool. Sustainability 3, 649–665.

Bass, S., 2009. Keep off the grass. Nature Reports Climate Change 3, 113–114.

Berger, M., Finkbeiner, M., 2013. Methodological challenges in volumetric and impact-oriented water footprints. Journal of Industrial Ecology 17, 79–89.

Bogardi, J. J., Fekete, B. M., Vörösmarty, C. J., 2013. Planetary boundaries revisited: A view through the "water lens". Current Opinion in Environmental Sustainability 5, 581–589.

Borucke, M., Moore, D., Cranston, G., Gracey, K., Iha, K., Larson, J., Lazarus, E., Morales, J. C., Wackernagel, M., Galli, A., 2013. Accounting for demand and supply of the biosphere's regenerative capacity: The National Footprint Accounts' underlying methodology and framework. Ecological Indicators 24, 518–533.

Bruns, T. D., Shefferson, R. P., 2004. Evolutionary studies of ectomycorrhizal fungi: recent advances and future directions. Canadian Journal of Botany 82, 1122–1132.

Chapman, P. M., Maher, B., 2014. The need for truly integrated environmental assessments. Integrated Environmental Assessment and Management 10, 151–151.

Cole, M. J., Bailey, R. M., New, M. G., 2014. Tracking sustainable development with a national barometer for South Africa using a downscaled "safe and just space" framework. Proceedings of the National Academy of Sciences of the United States of America 111, E4399–E4408.

Costanza, R., Kubiszewski, I., Giovannini, E., Lovins, H., McGlade, J., Pickett, K. E., Ragnarsdóttir, K. V., Roberts, D., Vogli, R. D., Wilkinson, R., 2014. Time to leave GDP behind. Nature 505, 283–285.

Čuček, L., Klemeš, J. J., Kravanja, Z., 2012. A review of Footprint analysis tools for monitoring impacts on sustainability. Journal of Cleaner Production 34, 9–20.

Dearing, J. A., Wang, R., Zhang, K., Dyke, J. G., Haberl, H., Sarwar Hossain, M., Langdon, P. G., Lenton, T. M., Raworth, R., Brown, S., Carstensen, J., Cole, M. J., Cornell, S. E., Dawson, T. P., Doncaster, C. P., Eigenbrod, F., Flörke, M. , Jeffers, E., Mackay, A. W., Nykvist, B., Poppy, G. M., 2014. Safe and just operating spaces for regional social–ecological systems. Global Environmental Change 28, 227–238.

Esty, D. C., Levy, M. A., Srebotnjak, T., De Sherbinin, A., Kim, C. H., Anderson, B., 2006. Pilot Environmental Performance Index. Yale Center for Environmental Law & Policy, New Haven, USA.

EUREAPA, 2011. Scenario Modelling and Policy Assessment Tool. https://www.eureapa.net/.

Fang, K., Heijungs, R., De Snoo, G. R., 2014. Theoretical exploration for the combination of the ecological, energy, carbon, and water footprints: Overview of a footprint family. Ecological Indicators 36, 508–518.

Fang, K., Heijungs, R., De Snoo, G. R., 2015. Understanding the complementary linkages between environmental footprints and planetary boundaries in a footprint–boundary environmental sustainability assessment framework. Ecological Economics 114, 218–226.

FAO (Food and Agriculture Organization of the United Nations). 2012. AQUASTAT. http://www.fao.org/nr/water/aquastat/data/query/index.html.

Gerten, D., Hoff, H., Rockström, J., Jägermeyr, J., Kummu, M., Pastor, A. V., 2013. Towards a revised planetary boundary for consumptive freshwater use: role of environmental flow requirements. Current Opinion in Environmental Sustainability 5, 551–558.

GFN (Global Footprint Network). 2012. National Footprint Accounts 2012 Edition. http://www.footprintnetwork.org/images/article_uploads/National_Footprint_Accounts_2012_Edition_Report.pdf.

Giampietro, M., Saltelli, A., 2014. Footprints to nowhere. Ecological Indicators 46, 610–621.

Goodland, R., Daly, H., 1996. Environmental sustainability: Universal and non-negotiable. Ecological Applications 6, 1002–1017.

Griggs, D., Stafford-Smith, M., Gaffney, O., Rockström, J., Öhman, M. C., Shyamsundar, P., Steffen, W., Glaser, G., Kanie, N., Noble, I., 2013. Sustainable development goal for people and planet. Nature 495, 305–307.

Hoekstra, A. Y., 2009. Human appropriation of natural capital: A comparison of ecological footprint and water footprint analysis. Ecological Economics 68, 1963–1974.

Hoekstra, A. Y., 2015. The sustainability of a single activity, production process or product. Ecological Indicators 57, 82–84.

Hoekstra, A. Y., Hung, P. Q., 2002. Virtual Water Trade: A Quantification of Virtual Water Flows between Nations in Relation to International Crop Trade. Value of Water Research Report Series (No. 11), UNESCO-IHE Institute for Water Education, Delft, The Netherlands.

Hoekstra, A. Y., Mekonnen, M. M., Chapagain, A. K., Mathews, R. E., Richter, B. D., 2012. Global monthly water scarcity: blue water footprints versus blue water availability. PLoS One 7, e32688.

Hoekstra, A. Y., Wiedmann, T. O., 2014. Humanity's unsustainable environmental footprint. Science 344, 1114–1117.

IPCC (Intergovernmental Panel on Climate Change), 2014. Climate Change 2014 Synthesis Report. http://www.ipcc.ch/pdf/assessment-report/ar5/syr/SYR_AR5_LONG ERREPORT.pdf.

Jaramillo, F., Destouni, G., 2015. Comment on "Planetary boundaries: Guiding human development on a changing planet". Science 348, DOI: 10.1126/science.aaa9629.

Jones, N., 2007. China tops CO_2 emissions. Nature 448, DOI: 10.1038/news070618-9.

Kaly, U., Briguglio, L., McLeod, H., Schmall, S., Pratt, C., Pal, R., 1999. Environmental Vulnerability Index (EVI) to Summarise National Environmental Vulnerability Profiles. SOPAC Technical Report 275. http://ict.sopac.org/VirLib/TR0275.pdf.

Kates, R. W., 2011. What kind of a science is sustainability science? Proceedings of the National Academy of Sciences of the United States of America 108, 19449–19450.

Kitzes, J., Galli, A., Bagliani, M., Barrett, J., Dige, G., Ede, S., Erb, K., Giljum, S., Haberl, H., Hails, C., Jolia-Ferrier, L., Jungwirth, S., Lenzen, M., Lewis, K., Loh, J., Marchettini, N., Messinger, H., Milne, K., Moles, R., Monfreda, C., Moran, D., Nakano, K., Pyhälä, A., Rees, W., Simmons, C., Wackernagel, M., Wada, Y., Walsh, C., Wiedmann, T., 2009. A research agenda for improving national Ecological Footprint accounts. Ecological Economics 68, 1991–2007.

Lewis, S. L., 2012. We must set planetary boundaries wisely. Nature 485, 417–417.

Mekonnen, M. M., Hoekstra, A. Y., 2011. National Water Footprint Accounts: The Green, Blue and Gray Water Footprint of Production and Consumption. Volume 2: Appendices. http://www.waterfootprint.org/Reports/Report50-NationalWaterFootprints-Vol2.pdf.

Meyfroidt, P., Lambin, E. F., Erb, K. H., Hertel, T. W., 2013. Globalization of land use: distant drivers of land change and geographic displacement of land use. Current Opinion in Environmental Sustainability 5, 438–444.

Molden, D., 2009. The devil is in the detail. Nature Reports Climate Change 3, 116–117.

Nykvist, B., Persson, Å., Moberg, F., Persson, L., Cornell, S., Rockström, J., 2013. National Environmental Performance on Planetary Boundaries: a Study for the Swedish Environmental Protection Agency. Stockholm Resilience Center & Stockholm Environment Institute, Stockholm, Sweden.

O'Brien, M., Schütz, H., Bringezu, S., 2015. The land footprint of the EU bioeconomy: Monitoring tools, gaps and needs. Land Use Policy 47, 235–246.

Peters, G. P., 2008. From production-based to consumption-based national emission inventories. Ecological Economics 65, 13–23.

PRB (Population Reference Bureau), 2009. 2009 World Population Data Sheet. http://www.prb.org/pdf09/09wpds_eng.pdf.

Puolamaa, M., Kaplas, M., Reinikainen, T., 1996. Index of Environmental Friendliness: A Methodological Study. Official Statistics of Finland, Helsinki, Finland.

Rees, W. E., 1992. Ecological footprint and appropriated carrying capacity: what urban economics leaves out. Environment and Urbanization 4, 121–130.

Rockström, J., Gordon, L., Folke, C., Falkenmark, M., Engwall, M., 1999. Linkages among water vapor flows, food production, and terrestrial ecosystem services. Conservation Ecology 3 (2): 5.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F. S., Lambin, E. F., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., De Wit, C. A., Hughes, T., Van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P. K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R. W., Fabry, V. J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J. A., 2009. A safe operating space for humanity. Nature 461, 472–475.

Rogelj J., McCollum, D., O'Neill, B. C., Riahi K., 2013. 2020 emissions levels required to limit warming to below 2 °C. Nature Climate Change 3, 405–412.

Samuel-Johnson, K., Esty, D. C., 2000. Pilot Environmental Sustainability Index Report. World Economic Forum: Annual Meeting, Davos, Switzerland.

Schyns, J. F., Hoekstra, A. Y., 2014. The added value of water footprint assessment for national water policy: A case study for Morocco. PloS One 9, e99705.

Smith, N. J., McDonald, G. W., Patterson, M. G., 2014. Is there overshoot of planetary limits? New indicators of human appropriation of the global biogeochemical cycles relative to their regenerative capacity based on 'ecotime' analysis. Ecological Economics 104, 80−92.

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S. R., De Vries, W., De Wit, C. A., Folke, C, Gerten, D., Heinke, J., Mace, G. M., Persson, L. M., Ramanathan, V., Reyers, B., Sörlin, S., 2015. Planetary boundaries: Guiding human development on a changing planet. Science 347, DOI: 10.1126/science.1259855.

Steinhart, C. E., Schierow, L.-J., Sonzogni, W. C., 1982. An environmental quality index for the great lakes. Journal of the American Water Resources Association 18, 1025–1031.

Sutton, P., Costanza, R., 2002. Global estimates of market and non-market values derived from nighttime satellite imagery, land cover, and ecosystem service valuation. Ecological Economics 41, 509–527.

Tuomisto, H. L., Hodge, I. D., Riordan, P., Macdonald, D. W., 2012. Exploring a safe operating approach to weighting in life cycle impact assessment – a case study of organic, conventional and integrated farming systems. Journal of Cleaner Production 37, 147–153.

UNEP (United Nations Environment Programme), 2014. The Emissions Gap Report 2014: A UNEP Synthesis Report. http://www.unep.org/publications/ebooks/emissionsgapreport2014/portals/50268/pdf/EGR2014_LOWRES.pdf.

Van den Bergh, J. C. J. M., Grazi, F., 2014. Ecological footprint policy? Land use as an environmental indicator. Journal of Industrial Ecology 18, 10–19.

Wackernagel, M., Rees, W. E., 1997. Perceptual and structural barriers to investing in natural capital: Economics from an ecological footprint perspective. Ecological Economics 20, 3–24.

Weinzettel, J., Hertwich, E. G., Peters, G. P., Steen-Olsen, K., Galli, A., 2013. Affluence drives the global displacement of land use. Global Environmental Change 23, 433–438.

Wiedmann, T., 2009. A review of recent multi-region input-output models used for consumption-based emissions and resource accounting. Ecological Economics 69, 211–222.

Wiedmann, T., Minx, J., 2008. A definition of 'carbon footprint'. In: Pertsova, C. C. (Ed.), Ecological Economics Research Trends. Nova Science Publishers, Hauppauge, New York, USA, pp. 1–11.

Chapter 7

General discussion

7.1. Answers to research questions

RQ1: Does it make sense to bring together different environmental footprints into a unified framework?

Environmental footprints have now received a lot of attention from academia, the public, organizations and governments; however, none of the existing footprints is able to adequately communicate all aspects of anthropogenic effects on the environment. In addition, focusing on single footprints in isolation runs the risk of shifting the environmental burden to other footprints, as environmental issues within the Earth system are extensively interlinked and interactive and cannot be uncoupled from others. This is particularly true in the context of globalized economy, where a simple product may have a global coverage of resource extractions and hazardous emissions. Without a systemic view of the complexity of human–environment interactions, reducing one type of product environmental footprint (PEF) may induce a remarkable increase in others (see Chapter 2).

Stemming from the firm belief that environmental issues are getting increasingly complex, and that wise environmental policies cannot be formulated without looking at the whole picture, the combination of the ecological, energy, carbon and water footprints takes a fundamental step towards constructing a unified footprint family (see Chapter 2). Although these four footprints differ in more aspects than only in the impacts that are addressed, the footprint family has proved effective in making use of them in a complementary way. The value added of the footprint family lines in its systemic view that allows to provide policy makers with a complete picture of human disturbance associated with the demand for the regenerative and assimilative capacity of the biosphere. By that, one can examine whether or not a reduction in one footprint would lead to undesirable consequences for others. Problem shifting, in this sense, would be avoided to some extent by operationalizing the footprint family concept.

RQ2: How to make use of a selection of environmental footprints to constitute a truly integrated footprint family?

The integration of environmental footprints goes beyond framing a footprint family and requires a deep understanding of the general structure that underlies existing footprint indicators. Defined in simple terms, environmental footprints are indicators that measure anthropogenic effects on the planet's environment by human actions, irrespective of the precise units and dimensions. An investigation into the conceptual and mathematical structure behind different versions of the carbon, water, land and material footprints suggests that there are two broad categories of environmental footprints, namely, the inventory-oriented footprints (IVOFs) and the impact-oriented footprints (IPOFs) (see Chapter 3). The two-category classification captures the inherent distinction between most, if not all, footprint accounts in terms of inventory analysis and impact characterization; that is, the IVOFs present a physical interpretation of the pressure of resource use which is causing environmental impacts at the inventory level, whereas the IPOFs further link inventory flows to a specific environmental impact and assess the impact category on a scientific characterization basis.

The next step to the categorization is selection. While both footprint categories have their own strengths and weaknesses, the integration of environmental footprints only makes sense if all involved are members of the IPOF category. The foremost reason for choosing the IPOFs rather than choosing the IVOFs is that the former can prevent the process of integration from double counting and double weighting that would undoubtedly compromise the validity of the final composite metric. To meet the policy demands for a single-score, stand-alone metric, a framework for characterization, normalization and weighting of conceivable IPOFs is proposed, whereby the results of inventory analysis are first to be translated into single impact category indicators, and subsequently normalized, weighted and integrated into a composite footprint index (CFI) (see Chapter 3). The three-step framework differs from life cycle impact assessment (LCIA) in that it can be fruitful in life cycle-less contexts as well. Besides, it offers experts without life cycle assessment (LCA)-expertise new insights into how to form a truly integrated footprint family—which remains unsolved and steeped in controversy (see Chapter 4).

RQ3: Is life cycle assessment a necessity for accounting for environmental footprints?

There is no doubt that footprint practitioners and users have learned and borrowed much from the LCA community. The strengths of LCA in assessing environmental impacts could allow many footprint topics (e.g., climate change, resource use) to be addressed under an LCA framework, in particular those that can be measured in relation to a functional unit. The carbon and abiotic resource footprints are two obvious examples of LCA-based footprints, where a variety of human disturbance is tabulated and translated into the inventory of greenhouse gas (GHG) emissions and abiotic resource extractions and further modeled quantitatively and expressed as impact scores according to their relative contributions to climate change and resource depletion, respectively (see Chapter 4). Given the satisfactory performance of life cycle approaches on scientific robustness, environmental relevance and reproducibility, taking advantage of LCA has now been a fashion trend followed by a growing group of footprint users.

Regardless of the ubiquity of life cycle approaches to footprinting, like any methodologies, however, LCA has its own limitations and uncertainties. Narrowing environmental footprints down to an LCA context potentially creates blind spots, where exhaustive inventory data for compiling or consensus models for characterization of impact pathways are unavailable. Moreover, some of the environmental footprints such as the classical ecological and water footprints are designed in a way that permits a measure of pressures, not impacts, of anthropogenic activities on the planet's environment, with the belief that this valuable information may get lost if translating into an impact score through characterization modeling. In addition, there are certain important types of questions for which a footprint-type representation would be preferable to a life cycle-type representation, as is the case for organization environmental footprint (OEF). For these reasons, LCA should not be interpreted as a necessity, but rather an option, for defining and computing environmental footprints (see Chapter 4).

RQ4: What are the complementarities of environmental footprints and planetary boundaries?

While focusing on the measurement of planetary boundaries for several environmental issues, the ultimately aim of the planetary boundaries framework (PBF) is to identify the remaining safe operating space for humans by comparing planetary boundaries with current environmental states. The sole use of expert knowledge and lack of quantitative methods, however, compromise the reliability of current estimate—a neglected part of the PBF which could have been more rigorous and accurate if appropriate footprint models are employed instead. On the contrary, the lack of comparison to threshold indicators makes many of the environmental footprints (e.g., nitrogen footprint, biodiversity footprint) policy-irrelevant. Even for those which have already been linked to threshold values, the threshold estimates are far from satisfactory and much work remains to be done. As a result, it becomes clear that recent developments regarding planetary boundaries could facilitate the ongoing process of benchmarking environment footprints against the corresponding critical thresholds.

Substantial similarity exists between environmental footprints and planetary boundaries, because almost all environmental issues that the PBF concerns, such as climate change, water use, and land use, can be found in existing footprint accounts. Interestingly, the two research communities are doing something quite similar but with different strengths and challenges and lacking communication and mutual understanding. As they are found to be limited in their own abilities to implement environmental sustainability assessment (ESA), it makes great sense to take advantage of both in a complementary way. To that end, a footprint–boundary (F–B) ESA framework is proposed as a tool for jointly assessing environmental sustainability (see Chapter 5). It challenges the isolation of the footprint community and the planetary boundary community, thus opening the door to collaborative research into ESA. The primary purpose of the F–B ESA framework is to support policy makers in responding to the widening sustainability gap and in finding new ways to prevent the planet's environment from undesirable transitions.

RQ5: How to allocate planetary boundaries to nations and how does this relate

to nation-specific environmental sustainability assessment?

The environmental issues that the F–B ESA framework deals with come in two broad types, namely, the systemic processes and aggregated processes. Climate change, for instance, fits within a systemic process whose unambiguous global nature enables the climate boundary to be one of the few planetary boundaries which can be downscaled to the national level through top-down approaches that seem to be a normative or political issue more than a scientific issue. Water use, conversely, is an aggregated process for which the environmental boundaries at the national level can be quantified by upscaling local and regional water boundaries that are spatially heterogeneous, depending on a great deal of well-documented survey data for site-specific runoff and water availability. The main challenge to allocate planetary boundaries to nations is therefore a mix of downscaling and upscaling, as well as a balance between scientific and political considerations (see Chapters 5 and 6).

One prominent advantage of the F–B ESA framework is its capability to create solutions to this challenge. Converting environmental boundaries from the planetary scale to the national scale can be fulfilled by a series of steps, including the identification of target environmental issues, the selection of suitable footprint metrics, the determination of the level at which specific environmental boundaries rely on, the choice of appropriate methods for boundary metrics, the harmonization of the footprint and boundary metrics, and the measurement of sustainability gaps (see Chapter 6). In the case of climate change, atmospheric CO_2 concentration and radiative forcing—two independent control variables for defining the climate boundary—are integrated in a way consistent with the calculation of carbon footprint—a consensus-based impact indicator describing the current status of the Earth's climate system. On the national scale, the difference between the footprint and boundary metrics, in either absolute or relative terms, offers a straightforward and practical means of nation-specific ESA, explaining how far countries are from their individual environmental boundaries.

7.2. Further reflections

7.2.1. Advances to the establishment of a footprint family

Chapters 2 and 3 are dedicated to the development of a conceptual and methodological framework of the footprint family, with the aim to call for a policy transformation from assessing single environmental footprints in isolation towards an integrated assessment. Each of the ecological, energy, carbon and water footprints involved in the footprint family focuses on limited aspects of human-induced environmental change to the Earth's ecosystems, but in combination they are able to provide an overall picture of anthropogenic effects on the biosphere, atmosphere and hydrosphere. However, there is still a need to avoid overlapping and conflicting messages of the current footprint family. For this concern, we suggest to: (1) exclude the energy-related component from the ecological footprint accounting and rename the remaining as "land footprint"; (2) reshape the energy footprint to make it relevant for assessing the depletion of non-renewable energy resources in the lithosphere; (3) keep the footprint family systematized and flexible so that emerging environmental footprints could be continuously taken into account; (4) ensure the harmonization of all selected footprint accounts by including them in unified procedures for structuring, categorization and integration; and (5) present the final results both at aggregate and disaggregate levels in keeping with the varying requirements of policy makers and scientists.

7.2.2. Environmental impact assessment *vs.* environmental sustainability assessment

Chapters 5 and 6 are committed to an investigation into the complementary use of environmental footprints and planetary boundaries, for which there are two leading but disparate communities in the fields of environmental and ecological sciences. The simultaneous assessment of the footprint and boundary metrics not only aims at describing what is happening to the environment, as is done in environmental impact assessment (EIA), but also at determining whether this is deviating away from a state of environmental sustainability—a prerequisite for the economic and social pillars of sustainable development. It builds on the recognition that man should not merely minimize his environmental footprints which many footprint users currently concentrate on, but make sure these footprints stay within the planetary boundaries. As pointed out by Lancker and Nijkamp (2000), an indicator does not provide any information on sustainability unless a reference value is given to it. The F–B ESA framework is, to our knowledge, the first attempt to create synergies between environmental footprints and planetary boundaries, representing an evolution of prognostic and preventive ESA—a step ahead from EIA. To obtain a better understanding of the implications for ESA, further improvements need to focus on the following issues: (1) development of measurable aggregated boundaries at multiple scales; (2) partitioning of systemic planetary boundaries for sub-global assessments; (3) harmonized accounting of the footprint and boundary metrics; and (4) trade-offs between different sustainability gaps.

7.2.3. The role of methodological standards

Chapter 4 gives novel insights into the relationship between environmental footprints and LCA. At present, a series of initiatives have been made or are under way to reach agreement on methodological standards for footprint accounting, such as the PAS 2050 (BSI, 2011) for the carbon footprint, the ISO 14046 (ISO, 2014) for the water footprint, and the Environmental Footprint Analysis (EPA, 2014). More recently, guidelines for Product and Organization Environmental Footprints (PEF/OEF) have been launched (EC, 2015a, 2015b), in the hope that the environmental impacts of products and organizations can be assessed within LCA frameworks. While it is an appealing idea to proceed with the development of footprinting standards from an LCA perspective, inappropriate and misleading results may be encountered when equating environmental footprints with impact category indicators defined in LCA. This is why until now there is no consensus on the most appropriate footprinting method, and neither is there a clear recognition of the applicability of every conceivable method nor of the compatibility of the results due to varying methods chosen. All this leads to a question about the role of international standards for the calculation and interpretation of environmental footprints: is it a boost or obstacle?

7.2.4. Data quality and uncertainty

This thesis as a whole is not intended to generate new data but to generate new insights obtained by critical reading and contemplation. The contradicting literature and the non-quantitative nature of the analysis make the research distinguished from others. Nevertheless, on the basis of the findings of the thesis, the following remarks can be drawn: (1) in the absence of exhaustive data for life cycle inventory analysis and impact assessment, footprint accounts may be satisfied by using classical footprinting methods, such as National Footprint Accounts (NFA) and the Water Footprint Network (WFN) approach, which provide feasible alternatives to LCA; (2) recent development in environmentally extended multi-regional input–output (MRIO) models offers an opportunity for footprint users to get access to more comprehensive and detailed datasets for macro- and meso-scale studies than previously available; (3) analysis of uncertainty in both raw data and final results is necessary to add to the procedure for interpretation of environmental footprints; (4) there is a need for high-resolution remote sensing data to map the spatial and temporal variations in anthropogenic extractions and emissions; and (5) data availability for a variety of operational control variables is an important limiting factor for the allocation of planetary boundaries to country-specific shares.

7.2.5. Production-based *vs.* consumption-based perspective

A further point of attention concerns the choice between consumption-based and production-based footprint accounts in relation to nationalized planetary boundaries. For environmental footprints, a distinction has been made between production-based and consumption-based accounting (Peters, 2008; Wiedmann, 2009). Production-based footprints refer to the pressure or impact exerted where production of goods and services takes place. Given that more than ever consumers are driving environmental impacts far beyond the geographical borders of their own locations (Moran et al., 2013), there is a growing focus on the consumption-based footprints which ignore the pressures by activities for exported products, but on the other hand include the activities that take place abroad and that are associated with products imported by the region of concern (Chao et al., 2013). Both viewpoints are legitimate and have a policy value, although the precise realms of application of the two are still under debate. But regardless of the choice for production-based or consumption-based footprints, the environmental boundaries are supposed to be defined along the same principle. In this regard, in Chapter 6 there is a striking inconsistency between data for the footprint metric and data for the boundary metric. That should be resolved in the follow-up work.

7.3. Recommendations for future research

The thesis aims at bringing clarity and transparency to a number of unresolved and important issues regarding environmental footprints; nevertheless, there remain many gaps in our knowledge. In order to proceed with the development of environmental footprints, we come up with a research agenda that includes the following prioritized topics:

- Harmonization of the structure, terminology and notation of environmental footprints.
- Identification of the applicability and limitations of each environmental footprint;
- Evaluation of footprinting methods on scientific quality, policy relevance, and public acceptance.
- Development of approaches for uncertainty and sensitivity analysis of footprint models.
- Development of systematic and dynamic frameworks of the footprint family to integrate other sustainability-related (e.g., social, economic) footprints.
- Exploration of the methodological synergies with LCA, IOA, MFA, energy-based methods (e.g., emergy, exergy), etc.
- Improvement in the consistency and compatibility of the footprint and boundary metrics.

References

BSI (British Standards Institution), 2011. PAS 2050: 2011 Specification for the Assessment of the Life Cycle Greenhouse Gas Emissions of Goods and Services. British Standards Institution, London, UK.

Chao, C.-W. Heijungs, R., Ma, H.-W., 2013. Development and application of dynamic hybrid multi-region inventory analysis for macro-level environmental policy analysis. A case study on climate policy in Taiwan. Environmental Science and Technology 47, 2512–2519.

EC (European Commission), 2015a. Product Environmental Footprint (PEF). http://ec.europa.eu/environment/eussd/smgp/product_footprint.htm

EC (European Commission), 2015b. Organization Environmental Footprint (PEF). http://ec.europa.eu/environment/eussd/smgp/organisation_footprint.htm

EPA (United Sates Environmental Protection Agency), 2014. Environmental Footprint Analysis. http://www.epa.gov/sustainability/analytics/environmental-footprint.htm.

ISO (International Organization for Standardization) 14046. 2014. Environmental Management —

Water Footprint — Principles, Requirements and Guidance. International Organization for Standardization, Geneva, Switzerland.

Lancker, E., Nijkamp, P., 2000. A policy scenario analysis of sustainable agricultural development options: a case study for Nepal. Impact Assessment and Project Appraisal 18, 111–124.

Moran, D. D., Lenzen, M., Kanemoto, K., Geschke, A., 2013. Does ecologically unequal exchange occur? Ecological Economics 89, 177–186.

Peters, G. P., 2008. From production-based to consumption-based national emission inventories. Ecological Economics 65, 13–23.

Wiedmann, T., 2009. A review of recent multi-region input-output models used for consumption-based emissions and resource accounting. Ecological Economics 69, 211–222.

Summary

Introduction

Attention to develop indicators of anthropogenic interference with ecosystems in a footprint context is shared by a wide range of research communities, especially in the fields of ecological economics and industrial ecology. The concept of environmental footprint has now been broadly recognized as a proxy for human-induced pressure or impact on the planet's environment. A substantial number of publications released in recent decades have contributed to establishing the scientific underpinning and communication standards of environmental footprints. Meanwhile, the widespread diffusion of footprint indicators that employ disparate or conflicting hypotheses, principles and methodologies is increasingly facing the challenge of finding ways to achieve harmonization between different footprint studies. Despite the emergence of the footprint family concept, the related knowledge is still scarce and fragmented. Moreover, there remains a lack of exploration of the common ground and individual characteristics of various environmental footprints, which significantly restricts the policy-oriented use of footprint indicators.

Research questions

As a starting point for clearing the footprint jungle, this thesis aims to present, for the first time, a comprehensive investigation into the theoretical and methodological aspects of environmental footprints and the disciplinary relationship with the latest science in defining planetary boundaries for human activities. Research questions have been condensed into five items:

RQ1: Does it make sense to bring together different environmental footprints into a unified framework?

RQ2: How to make use of a selection of environmental footprints to constitute a truly integrated footprint family?

RQ3: Is life cycle assessment a necessity for accounting for environmental footprints?

RQ4: What are the complementarities of environmental footprints and planetary boundaries?

RQ5: How to allocate planetary boundaries to nations and how does this relate to nation-specific environmental sustainability assessment?

Answers to research questions

Answers to RQ1: Chapter 2 brings together the ecological, energy, carbon and water footprints into a footprint family. It shows that the four footprints differ in more aspects than only in the impacts that are addressed. Although there is some overlap and there are some inconsistencies between these footprint accounts, the footprint family established as a whole has proved effective in offering an overall picture of anthropogenic effects on some major compartments of the Earth system. Our study provides a strong conceptual and visionary basis for discussion on the integration of different environmental footprints, which remains one of the most meaningful and challenging academic tasks for the current footprint community. Without this systemic view, problem shifting is likely to occur and trade-offs cannot be implemented in a convincing way.

Answers to RQ2: Chapter 3 develops a stepwise approach to uncover the conceptual and mathematical structure hidden in existing versions of the carbon, water, land and material footprints. The differing elements allow most, if not all, environmental footprints to be classified into two broad categories: the inventory-oriented footprints (IVOFs) and the impact-oriented footprints (IPOFs). While both footprint categories have their own strengths and weaknesses, the integration of environmental footprints is found to be feasible only if all involved are members of the IPOF category. Furthermore, a unified framework is proposed to integrate different IPOFs into a composite footprint index in support of environmental decision-making. Our study touches upon a fundamental issue which is the key to making sense of the footprint concept.

Answers to RQ3: Chapter 4 discusses the pros and cons of applying life cycle assessment (LCA) to environmental footprints. On the one hand, the strengths of LCA in assessing environmental impacts could allow many footprint topics to be addressed under an LCA framework, as exemplified by the carbon and abiotic resource footprints which are subject to life cycle approaches. On the other hand, narrowing environmental footprints down to an LCA context may create blind spots where exhaustive inventory data for compiling or consensus models for impact characterization are unavailable. Moreover, there are certain important types of questions for which a life cycle perspective is problematic, as is the case for organization environmental footprint (OEF). As such, LCA offers an option, not a necessity, to account for environmental footprints.

Answers to RQ4: Chapter 5 departs from the position of challenging the isolation of environmental footprints and planetary boundaries. These two research communities are found to have much in common but with different strengths and weaknesses. Our analysis demonstrates that the latest scientific knowledge of planetary boundaries is able to provide environmental footprints with a set of consensus-based threshold estimates as reference indicators, and in reverse that the planetary boundaries framework (PBF) could benefit from well-grounded footprint models which allow for more accurate and reliable measurement of current human disturbance. For these reasons, we propose a framework for complementary use of the footprint and boundary metrics, where the concept environmental sustainability assessment (ESA) is defined in a novel and explicit way.

Answers to RQ5: Chapter 6 performs a practical implementation of the footprint–boundary ESA (F–B ESA) framework to assess the environmental sustainability of 30 nations primarily from a production point of view. The downscaling of planetary boundaries to nations is fulfilled by a top-down approach based on population, as is the case for carbon emissions, or by a bottom-up approach based on natural endowments, as is the case for water and land use. On the national scale, the sustainability gaps of the above three environmental issues are determined in relative terms, explaining how far countries approach or exceed their respective environmental boundaries for sustainable development. Through this work that provides concrete evidence of the validity of the F–B ESA framework, policy makers can be adequately informed of national performance on environmental sustainability both at the disaggregate and aggregate levels.

Main conclusions

- Environmental footprints are defined as measures of anthropogenic pressure or impact on the planet's environment irrespective of their precise units and dimensions.
- The IVOFs and IPOFs that address environmental exchange (extractions and emissions) at the inventory and impact assessment levels respectively offer two competing paradigms for footprint indicators.
- The framework for integration of the IPOFs provides policy makers with a unified approach to assessing overall environmental impacts and has a broader scope of applicability than LCA.
- While footprint users have learned and borrowed much from LCA, some limitations encountered suggest that LCA cannot be interpreted as a versatile tool for accounting for all possible environmental footprints.
- Latest science in planetary boundaries is found to complement environmental footprints in assessing environmental sustainability that is a critical prerequisite for the economic and social pillars of sustainable development.
- The sustainability gap between the converted footprint and boundary metrics plays a central role in understanding the national performance on individual and collective environmental issues.

Outlooks

- Theoretical fundamentals of environmental footprints should be strengthened in support of the establishment of footprint science.
- It is necessary to improve the performance of individual footprints on conceptual transparency, methodological robustness, data availability and policy relevance.

- There is a need for the development of systematic and dynamic frameworks where a well-defined footprint family is quantified and integrated in a uniform way.
- It may be appropriate to further clarify the pros and cons of LCA-based footprint accounting with more examples.
- Enhancing consistency and compatibility of the footprint and boundary metrics are of great importance to the operationalization of ESA.
- There seems to be room for use of environmental footprints in combination with other analytical tools both from production-based and consumption-based points of view.

Samenvatting

Inleiding

Diverse onderzoeksgroepen houden zich bezig met de ontwikkeling van voetafdruk-achtige indicatoren voor de invloed van de mens op ecosystemen. Dit zijn met name onderzoeksgroepen op het gebied van de ecologische economie en de industriële ecologie. Het achterliggende idee van een ecologische voetafdruk is inmiddels breed aanvaard als een benaderende maat voor de druk op of het effect van de mens op het milieu. Een groot aantal publicaties uit de afgelopen decennia heeft bijgedragen aan de totstandkoming van een wetenschappelijk begrip van en een communicatiestandaard voor de milieugerichte voetafdruk. Tegelijkertijd is er een tendens tot divergentie, waarbij verschillende en conflicterende veronderstellingen, principes en methodes de basis vormen voor de uitdaging om tot harmonisatie te komen. Ondanks de pogingen om een familie van voetafdrukken te creëren, is de kennis op dit gebied beperkt en gefragmenteerd. Bovendien is er een gebrek aan kennis van de gezamenlijke én individuele principes van de verschillende voetafdrukken, wat het gebruik in het milieubeleid in grote mate hindert.

Onderzoeksvragen

Om de verwarring in de voetafdrukwereld weg te nemen beschrijft dit proefschrift, voor het eerst, een omvattend onderzoek naar de theoretische en methodologische aspecten van milieugerichte voetafdrukken en de verbanden met de meest recente inzchten uit de de disciplines betreffende de planetaire grenzen voor door de mens ondernomen activiteiten. Er worden daarbij vijf onderzoeksvragen geformuleerd:

OV1: Is het zinvol om verschillende milieugerichte voetafdrukken in één raamwerk onder te brengen?

OV2: Hoe kan een selectie van de milieugerichte voetafdrukken gebruikt worden om tot een goede geïntegreerde familie van voetafdrukken te komen?

OV3: Is een levenscyclusanalyse nodig om de milieugerichte voetafdrukken te berekenen?

OV4: Wat is de aanvullende betekenis van milieugerichte voetafdrukken en planetaire grenzen?

OV5: Hoe kunnen planetaire grenzen worden toebedeeld aan landen, en wat is het verband met het nationale duurzaamheidsbeleid?

Antwoorden op de onderzoeksvragen

Antwoord op OV1: Hoofdstuk 2 combineert de ecologische voetafdruk, de energie-, koolstof- en watervoetafdruk in een voetafdrukfamilie. Het laat zien dat de vier voetafdrukken in meer verschillen dan alleen in hun onderwerp. Ondanks de overlap van en de inconsistenties tussen de verschillende soorten voetafdrukken is de voetafdrukfamilie in staat gebleken een omvattend beeld van de druk van de mensheid op de diverse componenten van het systeem Aarde te geven. Het onderzoek biedt een goed uitgangspunt voor een nadere discussie over de integratie van de verschillende soorten voetafdruk; een belangrijke taak die nog geheel open lag voor de academische gemeenschap. Zonder deze systeemblik zal probleemafwenteling met een grote mate van waarschijnlijkheid optreden.

Antwoord op OV2: Hoofdstuk 3 ontwikkelt een stapsgewijze aanpak om de onderliggende conceptuele en wiskundige structuur van de bestaande koolstof-, water-, land- en materiaalvoetafdrukken bloot te leggen. De verschillende elementen maken een onderscheid mogelijk tussen de meeste, zo niet alle, voetafdrukken in twee categorieën: de inventarisatiegeorienteerde voetafdrukken (inventory-oriented footprints, IVOFs) and de effectgeorienteerde voetafdrukken (impact-oriented footprints, IPOFs). Hoewel beide typen voetafdrukken sterke en zwakke kanten kennen, is een integratie van voetafdrukken alleen mogelijk voor voetafdrukken van het IPOF-type. Daarnaast wordt een raamwerk ontwikkeld waarmee verschillende voetafdrukken van het IPOF-type verenigd kunnen worden in een samengestelde voetafdrukindex om het milieubeleid te ondersteunen. Dit is een fundamentele voorwaarde om betekenis te geven aan het begrip voetafdruk.

Antwoord op OV3: Hoofdstuk 4 bespreekt de voor- en nadelen van het gebruik van levenscyclusanalyse (LCA) voor milieugerichte voetafdrukken. Aan de ene kant maakt LCA het mogelijk om verschillende effecttypen in een gezamenlijk raamwerk onder te brengen, zoals dat bij de koolstof- en abiotische grondstofvoetafdruk gebeurt. Aan de andere kant is de beperking tot LCA een probleem wanneer er geen goede LCA-gegevens of karakterisatiefeactoren beschikbaar zijn. Daarnaast zijn er belangrijke typen vragen waarvoor een LCA niet geschikt is. Een voorbeeld hiervan is de milieugerichte voetafdruk voor organisaties (organization environmental footprint, OEF). Daarmee is LCA een mogelijkheid en geen noodzaak bij het berekenen van milieugerichte voetafdrukken.

Antwoord op OV4: Hoofdstuk 5 betwijfelt het nut van de isolatie van de milieugerichte voetafdruk en de planetaire grenzen. De twee ondersoeksgemeenschappen blijken veel gemeen te hebben, maar ze hebben ook verschillende sterke en zwakke plekken. Uit de analyse blijkt dat het met de meest recente wetenschappelijke inzichten omtrent de planetaire grenzen mogelijk is om de milieugerichte voetafdrukken van een aantal op consensus-gebaseerde grenswaardes te voorzien. Omgekeerd kan het raamwerk van de planetaire grenzen worden verbeterd met behulp van goed-gefundeerde voetafdrukmodellen die een preciezere en betrouwbaardere meting van de menselijke verstoring van het milieu geven. Om deze redenen wordt een raamwerk voorgesteld, waarbij het begrip milieugerichte duurzaamheidsbeoordeling (environmental sustainability assessment, ESA) op een nieuwe en explicietere manier wordt gedefinieerd.

Antwoord op OV5: In hoofdstuk 6 wordt de voetafdruk-grenzenduurzaamheidsanalyse (footprint–boundary environmental sustainability assessment, F–B ESA) in de praktijk geïmplementeerd, om de milieuduurzaamheid van 30 landen vanuit een productieperspectief te bepalen. De toedeling van de planetaire grenzen aan landen wordt bereikt op basis van bevolkingsomvang voor de koolstofuitstoot, en op basis van het bezit van natuurlijke grondstoffen voor water- en landgebruik. De duurzaamheidskloof kan aldus op nationale schaal worden vastgesteld in relatieve termen, waarbij duidelijk wordt in welke mate landen de milieugrenzen voor duurzame ontwikkeling naderen of overschrijden. Dit stelt beleidsmakers in staat om zich goed te laten informeren over de nationale duurzaamheidsprestaties, zowel op geaggregeerd als op gedesaggregeerd niveau. Verder ondersteunt dit werk de validiteit van het F–B ESA raamwerk.

Hoofdconclusies

- Milieugerichte voetafdrukken zijn gedefinieerd als maten voor de door de mens veroorzaakte druk of effecten op het milieu van de de aarde, onafhankelijk van de precieze eenheden en maatstaven.
- De IVOF's en IPOF's, die beiden op de milieu-ingrepen (onttrekkingen en emissies) werken op respectievelijk inventarisatie- en effectniveau, bieden twee strijdige paradigma's voor de voetafdrukfamilie.
- Het raamwerk voor de integratie van de IPOF's biedt beleidsmakers een eenduidige benadering om milieueffecten te beoordelen, en heeft een groter toepassingsgebied dan LCA.
- Hoewel gebruikers van de voetafdruk veel hebben geleerd en overgenomen van LCA zijn er beperkingen aan het gebruik van LCA voor milieugerichte voetafdrukken.
- Recente bevindingen wat betreft de planetaire grenzen blijken aanvullende waarde op de milieugerichte voetafdrukken te hebben, wat een voorwaarde is voor de economische en sociale dimensies van duurzame ontwikkeling.
- De duurzaamheidskloof tussen de omgerekende voetafdruk en de grensmaat speelt een centrale rol voor het begrijpen van de nationale duurzaamheidsprestaties op zowel individuele als collectieve gebieden.

Vooruitblik

- De theoretische grondslagen van milieugerichte voetafdrukken moeten versterkt worden om de wetenschap van de voetafdrukken beter te ondersteunen.
- Verbetering van de prestaties van de individuele voetafdrukken op het gebied van conceptuele helderheid, methodologische robuustheid, beschikbaarheid van gegevens en beleidsrelevantie, is noodzakelijk.
- Er is behoefte aan de ontwikkeling van een systematisch en dynamisch raamwerk waarin een goed-gedefinieerde voetafdrukfamilie op eenduidige wijze kan worden gekwantificeerd en geïntegreerd.
- Verdere verheldering aan de hand van voorbeelden van de voor- en nadelen van op LCA gebaseerde voetafdrukken kan nuttig zijn.
- Vergroting van de consistentie en compatibiliteit van voetafdrukken en grensmaten zijn van groot belang voor de invulling van ESA.
- Er lijkt behoefte te zijn aan het gebruik van milieugerichte voetafdrukken in combinatie met andere analyse-instrumenten, zowel uit het oogpunt van productie als uit dat van consumptie.

Publications in this thesis

Chapter	Reference
2	Fang, K., Heijungs, R., De Snoo, R. G., 2014. Theoretical exploration for the combination of the ecological, energy, carbon, and water footprints: Overview of a footprint family. *Ecological Indicators* 36, 508–518. DOI: 10.1016/j.ecolind.2013.08.017.
3	Fang, K., Heijungs, R., 2015. Investigating the inventory and characterization schemes in footprinting: lessons for the classification and integration of footprints. *Journal of Cleaner Production* 108, 1028–1036. DOI: 10.1016/j.jclepro.2015.06.086.
4	Fang, K., Heijungs, R., 2015. The role of impact characterization in carbon footprinting. *Frontiers in Ecology and the Environment* 13, 130–131. DOI: 10.1890/15.WB.005.
	Fang, K., Heijungs, R., 2014. Moving from the material footprint to a resource depletion footprint. *Integrated Environmental Assessment and Management* 10, 596–598. DOI: 10.1002/ieam.1564.
	Fang, K., Heijungs, R., 2014. There is still room for a footprint family without a life cycle approach: Comment on "towards an integrated family of footprint indicators". *Journal of Industrial Ecology* 18, 71–72. DOI: 10.1111/jiec.12067.
	Fang, K., Heijungs, R., 2015. Rethinking the relationship between footprints and LCA. *Environmental Science & Technology*, 49, 10–11. DOI: 10.1021/es5057775.
5	Fang, K., Heijungs, R., De Snoo, R. G., 2015. Understanding the complementary linkages between environmental footprints and planetary boundaries in a footprint–boundary environmental sustainability assessment framework. *Ecological Economics* 114, 218–226. DOI: 10.1016/j.ecolecon.2015.04.008.
6	Fang, K., Heijungs, R., Duan, Z., De Snoo, R. G. The environmental sustainability of nations: Benchmarking the water, carbon and land footprints with allocated planetary boundaries. *Sustainability* 7, 11285–11305. DOI: 10.3390/su70811285.

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Curriculum Vitae

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Publications in peer-reviewed journals (in English)

1. Fang, K., Heijungs, R., 2015. Investigating the inventory and characterization schemes in footprinting: lessons for the classification and integration of footprints. *Journal of Cleaner Production* 108. DOI: 10.1016/j.jclepro.2015.06.086.
2. Fang, K., Heijungs, R., De Snoo, R. G., 2015. Understanding the complementary linkages between environmental footprints and planetary boundaries in a footprint–boundary environmental sustainability assessment framework. *Ecological Economics* 114. DOI: 10.1016/j.ecolecon.2015.04.008.
3. Fang, K., Heijungs, R., Duan, Z., De Snoo, R. G., 2015. The environmental sustainability of nations: Benchmarking the water, carbon and land footprints with allocated planetary boundaries. *Sustainability* 7. DOI: 10.3390/su70811285.
4. Fang, K., Heijungs, R., 2015. Rethinking the relationship between footprints and LCA. *Environmental Science & Technology* 49. DOI: 10.1021/es5057775.
5. Fang, K., Heijungs, R., 2015. The role of impact characterization in carbon footprinting. *Frontiers in Ecology and the Environment* 13. DOI: 10.1890/15.WB.005.
6. Fang, K., Heijungs, R., De Snoo, R. G., 2014. Theoretical exploration for the combination of the ecological, energy, carbon, and water footprints: Overview of a footprint family. *Ecological Indicators* 36. DOI: 10.1016/j.ecolind.2013.08.017.
7. Fang, K., Heijungs, R., 2014. There is still room for a footprint family without a life cycle approach: Comment on "towards an integrated family of footprint indicators". *Journal of Industrial Ecology* 18. DOI: 10.1111/jiec.12067.
8. Fang, K., Heijungs, R., 2014. Moving from the material footprint to a resource depletion footprint. *Integrated Environmental Assessment and Management* 10. DOI: 10.1002/ieam.1564.
9. Fang, K., Heijungs, R., De Snoo, G., 2013. The footprint family: Comparison and interaction of the ecological, energy, carbon and water footprints. *Revue De*

Metallurgie, 110. DOI: 10.1051/metal/2013051.

10. Wiloso, E. I., Heijungs, R., Huppes, G., Fang, K. Effect of biogenic carbon inventory on the life cycle assessment of bioenergy: challenges to the neutrality assumption. *Journal of Cleaner Production*, submitted.

Publications in peer-reviewed journals (in Chinese)

11. Fang, K., 2015. Footprint family: current practices, challenges, and future prospects. *Acta Ecologica Sinica*, accepted. DOI: 10.5846/stxb201407031373.
12. Fang, K., Duan, Z., 2015. An integrated assessment of national environmental sustainability by synthesizing carbon, water and land footprints and boundaries. *Journal of Natural Resources* 30. DOI: 10.11849/zrzyxb.2015.04.001.
13. Fang, K., 2015. Assessing the natural capital use of eleven nations: an application of a revised three-dimensional model of ecological footprint. *Acta Ecologica Sinica* 35. DOI: 10.5846/stxb201305211128.
14. Fang, K., 2015. Paradigms for the categorization and integration of environmental footprints. *Ecological Economy* 31. DOI: 10.3969/j.issn.1671-4407.2015.07.005.
15. Fang, K., 2015. Footprint family: Concept, classification, theoretical framework and integrated pattern. *Acta Ecologica Sinica* 35. DOI: 10.5846/stxb201305211128.
16. Fang, K., 2014. Multidimensional assessment of national environmental sustainability based on footprint family and planetary boundaries. *Ecology and Environment Sciences* 23. DOI: 10.3969/j.issn.1674-5906.2014.11.022.
17. Fang, K., 2014. Changes in the spatial distribution of natural capital use among G20 countries from 1999 to 2008. *Resources Science* 36. DOI: zykx201404016
18. Fang, K., Gao, K., Li, H., 2013. International comparison of natural capital use: A three-dimensional model optimization of ecological footprint. *Geographical Research* 32. DOI: 10.11821/dlyj201309008.
19. Fang, K., 2013. Ecological footprint depth and size: New indicators for a 3D model. *Acta Ecologica Sinica* 33. DOI: 10.5846/stxb201111051670.
20. Fang, K., Shen, W., Dong, D., 2013. Impacts of economic growth and technological progress on energy eco-footprint in Jilin Province. *Arid Land Geography* 34. DOI: 10.13826/j.cnki.cn65-1103/x.2013.01.013.
21. Fang, K., Heijungs, R., 2012. A review on three-dimensional ecological footprint model for natural capital accounting. *Progress in Geography* 31. DOI: 10.11820/dlkxjz.2012.12.016.
22. Fang, K., Li, H., 2012. Provincial pattern of China's natural capital use: A modification and application of ecological footprint depth and size. *Journal of Natural Resources* 27. DOI: 10.11849/zrzyxb.2012.12.001.
23. Fang, K., Shen, W., Dong, D., Lin, Z., 2012. Calculation method of energy ecological footprint based on global net primary productivity. *Acta Ecologica Sinica* 32. DOI: 10.5846/stxb201103280399.
24. Fang, K., Shen, W., Dong, D., 2012. Effects of multiple factors on changes of

energy eco-footprint: An empirical study. *Journal of Ecology and Rural Environment* 28. DOI: 10.3969/j.issn.1673-4831.2012.02.004.

25. Fang, K., Dong, D., Shen, W., 2012. Carbon footprint of global electricity and its equivalent calculation. *Chinese Journal of Ecology* 31. DOI: 10.13292/j.1000-4890.2012.0408.
26. Wang, J., Fang, K., 2012. Estimation of China's carbon footprint of electric power based on sources diversity. *Journal of Electric Power* 27. DOI: 10.3969/j.issn.1005-6548.2012.02.018.
27. Fang, K., Shen, W., Dong, D., 2011. Modification and prediction of energy ecological footprint accounting: Case study of Jilin Province. *Geographical Research* 30. DOI: 10.11821/yj2011100010.
28. Fang, K., Shen, W., Zheng, Q., Gao, K., Liu, J., 2011. Study on ecological carrying capacity of fossil energy land based on ecological footprint theory. *Environmental Science & Technology* 34. DOI: 10.3969/j.issn.1003-6504.2011.12.043.
29. Fang, K., Dong, D., Shen, W., 2010. Discussion on shortcomings and improvement of ecological footprint theory in energy consumption assessment. *Journal of Natural Resources* 25. DOI: 10.11849/zrzyxb.2010.06.013.
30. Fang, K., Dong, D., Shen, W., 2010. A new energy ecological footprint model based on net primary productivity and its comparison with traditional model. *Ecology and Environmental Sciences* 19. DOI: 10.3969/j.issn.1674-5906.2010.09.005.
31. Fang, K., Dong, D., Shen, W., 2010. Modified energy ecological footprint method and its application to analysis and evaluation of regional energy utilization benefit. *Scientia Geographica Sinica* 30. DOI: 10.13249/j.cnki.sgs.2010.05.017.
32. Fang, K., Shen, W., Wang, L., 2010. Sustainability analysis in agricultural area based on modified energy ecological footprint model. *Geography and Geo-Information Science* 26. DOI: 10.3969/j.issn.1672-0504.2010.06.017.
33. Fang, K., Wang, L., Li, H., 2010. Impact assessment of land use planning on traditional agricultural area of Southern Yangtze River based on ecosystem services value. *Resources and Environment in the Yangtze Basin* 19. DOI: 10.11870/cjlyzyyhj2010z2017.
34. Fang, K., Shen, W., 2010. Construction and application of calculation model for CO_2 emission carrying capacity. *Ecological Science* 29. DOI: 10.3969/j.issn.1008-8873.2010.06.011.
35. Fang, K. A framework for quantification and integration of environmental footprints: A perspective of life cycle assessment. *Acta Ecologica Sinica*, submitted.

Presentations

16-18 Oct. 2015	A framework for quantification and integration of environmental footprints. The 2015 China Society of Natural Resources (CSNR) Annual Conference, Kunming, China.
2 Sep. 2015	Environmental footprints: Concept, accounting and integration.

	School of Geographic and Oceanographic Sciences, Nanjing University, Nanjing, China.
3-7 May 2015	Investigating the characterization and weighting schemes for footprinting. Society of Environmental Toxicology and Chemistry (SETAC) Europe 25th Annual Meeting, Barcelona, Spain.
23 Apr. 2015	Environmental footprints: Concept, classification, and the relevance to planetary boundaries. Water Footprint Network (WFN), University of Twente, Twente, The Netherlands.
23 Mar. 2015	The applications and implications of environmental footprints for environmental governance. School of Public Affairs, Zhejiang University, Hangzhou, China.
30-31 May 2012	The footprint family: A comparison and discussion of ecological, water and carbon footprints. The 6th Society & Materials (SAM) International Conference, University of Leuven, Leuven, Belgium.
20-24 May 2012	The footprint family: Differentiation, classification, and integration. The 6th SETA World Congress, Berlin, Germany.
12-16 Feb. 2011	Net primary productivity as the basis for assessing energy ecological footprint. Center for Energy and Environmental Sciences, University of Groningen, Groningen, The Netherlands.
22-23 Oct. 2010	Calculation method of energy ecological footprint based on global net primary productivity. The 2010 CSNR Annual Conference, Chongqing, China.