WORLD RESOURCES FORUM

Towards a Resource Efficiency Index of Nations

Discussion Paper
May 2015

Universiteit Leiden

CML Institute of Environmental Sciences

Schweizerische Eidgenossenschaft
Confédération suisse
Confederazione Svizzera
Confederaziun svizra

Swiss Confederation

Federal Office for the Environment FOEN

EMPA Materials Science & Technology

UNEP International Resource Panel

Stadt St.Gallen

Akademien der Wissenschaften Schweiz
Académies suisses des sciences
Accademie svizzere delle scienze
Academias svizras da las scienzas
Swiss Academies of Arts and Sciences

Discussion paper

Towards a resource efficiency index of nations

Authors

Arnold Tukker
Jeroen Guinée
Lauran van Oers
Ester van der Voet

Institute of Environmental Sciences (CML), Leiden University, Netherlands

Under assignment of the World Resources Forum Association (WRF), St. Gallen, Switzerland

Version: 25.05.2015

Printed by
World Resources Forum WRF

Available from
World Resources Forum Secretariat
Lerchenfeldstr. 5
9014 St. Gallen, Switzerland
+41 71 554 0900
info@wrforum.org

Electronic version also available via
http://www.wrforum.org

ISBN 978-3-906177-08-3

This report was prepared under contract to the Federal Office for the Environment (FOEN). The contractor bears sole responsibility for the content, which does not necessarily reflect the views of FOEN.

Table of contents

Glossary

3R	Reduce, Re-use, Recycling
AGWP	Absolute Global Warming Potential
ANS	Adjusted net saving
BEES	Building for Environmental and Economic Sustainability
BLI	Better Life Index
CH4	Methane
CML	Institute of Environmental Sciences
CO2	Carbon dioxide
DMC	Domestic Material Consumption
EE IO	Environmentally extended input output
EEA	European Environment Agency
EMC	Environmentally Weighted Material Consumption
EU	European Union
EW-MFA	Economy-wide Material Flow Accounts
FAO	Food and Agricultural Organisation
GDP	Gross Domestic Product
GHG	Greenhouse Gas
GPI	Genuine Progress Indicator
GWP	Global Warming Potential
HANPP	Human appropriation of net primary production
HLY	Happy life years
IEA	International Energy Agency
ILCD	International Reference Life Cycle Data System
IOT	Input output table
IPPC	International Panel on Climate Change
IRP	International Resources Panel
ISEW	Index of Sustainable Economic Welfare
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
MFA	Material Flow Analysis
MR EE IO	Multi-regional environmentally extended input output
MSA	Mean Species Abundance
N_2O	Nitrous oxide
NOGEPA	Nederlandse Olie en Gas Exploratie en Productie Associatie
OECD	Organisation for Economic Collaboration and Development
PDF	Potential Disappeared Fraction of species
RMC	Raw Material Consumption
RME	Raw Material Equivalents
SUT	Supply and Use Tables
TMC	Total Material Requirement
UN	United Nations
UN SEEA	United Nations System of Economic and Environmental

	Accounts
UNDP	United Nations Development Program
UNEP	United Nations Environmental Program
UNSD	UN Statistical Division
US EPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WEI	Water exploitation index
WRF	World Resources Forum

1 Introduction

The World Resources Forum (WRF) is the science-based platform for sharing knowledge about the economic, political, social and environmental implications of the extraction, use and management of natural resources. Its main activities are the annual flagship World Resources Forum conference, which is held every 2 years in Davos, Switzerland, and every 2 years in another part of the World.

The WRF secretariat is interested in the development of an integrated 'resource use' or 'resource-efficiency' index of nations, as a counterpart of the well-known Gross Domestic Product (GDP) that measures economic activity of nations. The WRF secretariat has granted the Institute of Environmental Sciences (CML) of Leiden University, the Netherlands, a budget for a small pilot project that should result in a discussion paper (and if possible some quantitative examples) of how such an integrated indicator could be formulated.

In the last 25 years, the community of sustainability scientists has done a lot of work on material flow indicators, life cycle impact assessment indicators, and resource footprint indicators for e.g. water, land and resources. As we will show in this discussion paper, many of the discussions about what is the 'best' indicator have been far from concluded. It is hence inevitable that any attempt to produce an integrated 'resource use' or 'resource efficiency' index will be based on value choices. This is nothing new: as will be shown in section 3, the now widely accepted 'Human development index (HDI)' as produced by the United Nations Development Program (UNDP) is also based on a reasoned, but in essence reductionist selection of indicators and one specific weighting method. In this first attempt, the only thing we can do is come to reasoned choices and being transparent about them. We also do not see the suggested indices as final – we rather would offer them as first proposals that can kick-start a discussion over the next years, allowing for refinement of the approach, until – like happened with GDP, HDI and other now well accepted indicators – a consensus will arise in due time. We further do not see a priori that the indicators proposed would have a different applicability across different countries – the indicators proposed measure what they measure, impartially, although as usual interpretation of values by country and comparisons across countries always needs care. A point may be that certain countries may have more capabilities for in-depth data gathering as others, but as will be shown later our feeling is that many of the indicators proposed can be composed using globally available data sets from e.g. the International Energy Agency (IEA) and the Food and Agricultural Organisation (FAO).

This report is build up as follows

a) Chapter 2 gives some policy backgrounds, and options of how to structure resource-indices in general, next to deliberations about data inventory
b) Chapter 3 discusses possible reference indicators, such as GDP and HDI, to assess resource efficiency
c) Chapter 4 discusses possible indicators for individual resource categories (e.g. water, land, materials)
d) Chapter 5 discusses how such indicators for resource categories could be weighted
e) Chapter 6 ends with reflections, key deliberations and relevant choices, conclusions and suggestions.

2 Considerations in building a resource use or resource-efficiency index

2.1 Introduction

Various important economic blocks in the world have already developed some sort of a resource policy. China has embraced the ‘circular economy’ concept. Japan is pursuing a 3R (Reduce, Re-use, Recycling) strategy. And the European Union (EU) has launched two major Communications in this field, one on the Resource-efficiency Roadmap (EC, 2011) another on the Circular economy (EC, 2014). In relation, the scientific community has developed during the last 20-25 years ample of thoughts about how to define indicators for resource use and resource-efficiency, of which the economy-wide Material flow framework and various resource footprinting methods are best known.

In this section we want to discuss

a) Environment versus economy, and the key difference between production based and consumption based resource indicators
b) Options for practical data inventory
c) Some policy and scientific backgrounds, and possible definitions
d) Resources to be considered
e) Options for data inventory with regard to resource indicators

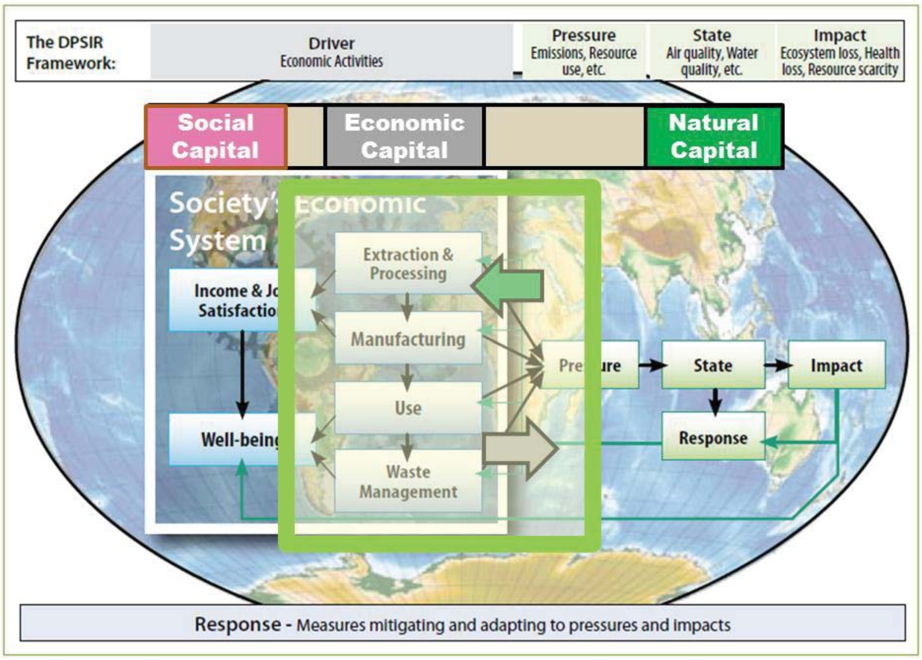
2.2 Environment versus economy and production versus consumption

2.2.1 Environment versus economy

The United Nations System of Economic and Environmental Accounts (UN SEEA), like methods for e.g. product Life cycle assessment, and prominent scientific publications, make a clear distinction between the economy and environment. Figure 2.1 gives an extension of a visualisation provided in a recent UNEP International Resources Panel report (Hertwich et al, 2010). In short, it distinguishes an economic system (divided in life cycles of extraction, manufacturing, use and waste management of products) that provides income and job satisfaction, and ultimately well-being, next to a natural system from which resources are extracted and emissions and waste is expelled. Further, capital stocks related to the social, economic and natural dimension are present. Resource extractions and emissions form pressures on the natural system, impacting it and altering its state. Note further that this conceptualisation frames the economy as being a sub-system (and hence limited by) the Natural system (c.f. Griggs et al, 2013, Daly and Cobb, 1989).

Figure 2.1 allows for conceptual development of indicators at a global level. Usually, however, one wants to understand issues with regard to resource use and relations to other (economic and social) performance indicators at country level. Emissions and resource extractions can be monitored at the level of sectors within a country, usually called the production perspective. This says however little about how emissions and resource use relate to consumption in a country – in our globalising economy trade has become more and more important, and the emissions and resource use to make a specific product can take place in a totally different country as where the product is consumed.

Figure 2.1: The Earth's natural system and economic system, resource extraction and emissions, and capital stocks

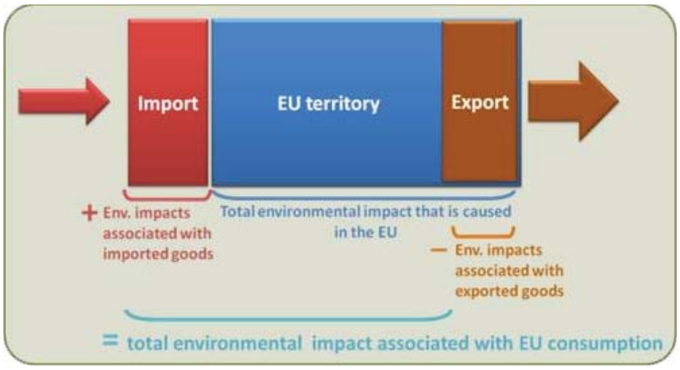


2.2.2 Production versus consumption

We need hence an environmental accounting system that makes such relations visible, and can both cover production-based or territorial emissions and resource use, and consumption-based emissions and resource use.

As a start, it is always needed to make an inventory of emissions and resource extraction in that country. This gives the *production oriented* or *territorial* emissions and resource use. One can then look at imports and export statistics, and try to assess the emissions and resource use in the supply chains of imports, next to trying to calculate the emissions and resource use related to exports. The consumption-based emissions then can be estimated by calculating the territorial/production based emissions/resource use plus those related to imports minus those related to exports (see Figure 2.2).

Figure 2.2: Production versus consumption based accounts, example for the EU[1]



[1] Figure produced by DG JRC IES, European Platform on Life cycle assessment, see: http://eplca.jrc.ec.europa.eu/?page_id=95, accessed 12.02.2015

2.3 Options for practical data inventory

2.3.1 Introduction

There are in essence two approaches to create an inventory of resource extractions related to the production and consumption of a country. The first are so-called Multi-regional environmentally extended input output (MR EE IO) approaches, discussed in section 2.3.2. The second are so-called co-efficient approaches, discussed in section 2.3.3.

Note in this that this data inventory at the level of resource extraction and economic relations is – similar to the life cycle inventory step in Life cycle assessment (LCA) – the most time-consuming issue. It is these basic data that change from year to year and in principle have to be gathered from year to year. Aggregation of e.g. such extraction data for individual resources to e.g. indices for e.g. water or material use as discussed in chapter 4, is in essence an aggregation step for which a method needs to be developed, but in most cases does not need any new data gathering. The clearest example is the aggregation of different energy carriers to MJ energy extracted: using the average energy content per carrier, one can do this aggregation for different years without major mistakes. As discussed further in chapter 5 the further aggregation of resource extraction indices to a single indicator again is mainly a methodological issue, without the need for an annual data gathering exercise. So, as in Life cycle impact assessment, the aggregation step to indicators and indices can be scientifically very challenging, but once solved, is a ‘piece of cake’ provided the underlying economic and resource extraction data are available.

Section 2.3.4 will discuss in more detail some of the relative advantages of the EE IO and co-efficient approaches when it comes down to data inventory and –availability.

2.3.2 MR EE IO approaches and data availability

<u>Introduction</u>
So-called ‘Multi-regional Environmentally Extended Supply and Use/Input-Output Tables’ (MR EE SUT/IOT) are now widely seen as the most promising approach to create such an accounting system (see figure 2.3). In short, a SUT / IOT shows a country’s total economy, with production divided in a few dozen industry sectors, and consumption divided into a few dozen product (and service) groups. The tables show how much each industry sector produces of these specific products (output) – for instance, how many cars, expressed in Euros, the car industry in that country produces. The tables also show for each industry sector how much of other products they need to realise this production (input) – e.g. the amount of steel, glass, plastics, electricity and electronics the car industry in that country needs to produce the output of cars. Furthermore, for each industry one can identify the primary resource use and emissions (‘environmental extensions’) – for instance land use by the agricultural sector, or CO2 emissions by the electricity production sector. At country level, it now becomes possible to analyse how the economy is interconnected. For instance, for the final use of cars by consumers, it becomes possible to analyse how much production value was contributed by the car industry, the glass production industry, the steel industry, and so on. But since we also know for each industry the emissions and primary resource extraction per Euro, we can estimate now the total primary resource extraction and life cycle emissions for the total consumption of cars in that country. This example is for one country only, and we know that imports and exports in the current global economy are substantial. So, one also must understand the emissions and primary resource use involved in imports. For that, one needs to create EE SUT/IOTs for the most important economies of the world, and identify the trade flows between countries. This then gives the aforementioned MR EE SUT, which gives a rather detailed picture of all linkages between production and consumption in the global economy.

Figure 2.3: A multi-regional Supply and Use Table (SUT) with extensions for 3 countries/regions. Each country/region is split up in industry sectors showing emissions and resource extractions per sector as extensions. Each industry sector per country/region has product inputs and outputs from/to other sectors and regions (usually measured in economic units, but it is equally possible to give them in physical units). Since all economic links are present in this figure, the environmental interventions in each industry sector can be allocated to the final use of specific products in each region

Data availability and other considerations

The key advantage of an MR EE IO database is that it provides consistent data for all countries covered in that database including all trade relations, and hence allows calculating resource footprints for all these countries in one go (compare Tukker et al, 2014). Further, the MR EE IO approach has as important advantage that value chains across various countries can be followed, relevant for the highly internationalised supply chains of manufactured products.

A clear drawback is that creating an MR EE IO is laborious. The research community has created about 4-5 major databases, of which 2 (EXIOBASE and EORA) show a high sector and product detail and provide a lot of environmental extensions making them suitable for use in environmental analyses (e.g. Tukker and Dietzenbacher, 2013; Lenzen et al., 2013; Tukker et al., 2013)[2]. Particularly creating the economic Multi-regional input-output core is crucial, since that implies gathering IO tables from different countries in the world, and linking them via trade. Since IO tables of individual countries are not consistent, this inevitably implies adjusting data in country tables and/or adjusting classifications[3].

---

[2] A high product detail in sectors like agriculture, mining and electricity production is not crucial for economic analysis, but very relevant for environmental analyses. Sub-sectors of these areas can have highly different impact intensities; compare e.g. the emissions from electricity generation by coal or electricity generation from hydropower.

[3] Statistical agencies often do not appreciate such adjustments, although the inconsistency at global level is a hard fact and only can be solved if Statistical agencies harmonize their data so that they make up a consistent global picture. The clearest example is the issue of 'trade with aliens'. All IO tables of individual countries from

The resource use data by sector are less crucial, since if there are no data at national level these can be gathered from databases with a good global coverage, and allocation of these resource use or resource extraction data to industry sectors usually is straightforward. These include the IEA database (energy carriers), FAOSTAT and Aquastat (biomass production by crop, wood production, water use, land use) and databases from the US and other Geological surveys (extraction of other materials).

2.3.3 Co-efficient approaches and data availability

Particularly for water and land use often other, so-called co-efficient approaches are used as well. In this approach, a single country is put central which in essence is depicted as a black box (unlike in the MR EE IO approach, that discerns economic transactions in a country). Where MR EE IO tables discern at best 200 product groups, co-efficient approaches can make full use of the detailed international trade databases, discerning some 5000 products and several 100 agricultural products, which dominate land use and water use. Agricultural production and related water use in physical terms is covered in great detail by FAO databases. Imported agricultural products usually are produced in the country of exports, with no further supply chains behind it that entail important levels of water or land use. If one hence then knows, say the physical amount of bananas imported from Costa Rica to Germany, and one knows the average land use and water use per ton bananas grown in Costa Rica, the land and water embodied in this banana trade can be estimated with a reasonable level of detail and accuracy. In essence such co-efficient approaches, particularly for water and land use, hence use similar or even the same resource extraction databases as the MR EE IO approach (i.e. FAOSTAT, Aquastat).

Co-efficient approaches also have been applied for e.g. the materials embodied in trade, but in that case Life cycle inventory databases have been used to estimate the primary resource extractions related to e.g. a ton of imports of a specific product. An advantage of this approach is that the corresponding ‘Raw material equivalents (RME)’ of a ton of imports give a statistical agency a simply way to use their official import and export data to calculate the primary material extraction in imports (cf. Schoer et al, 2012). A disadvantage is that a totally different type of data-sources is introduced, and that the RME is calculated just for the country from which a product is imported – where many components of that product may have been made elsewhere.

2.3.4 Reflections

As indicated primary data mining is one of the main challenges in creating a resource-efficiency index of nations, rather than developing a method for the calculating of the resource-indicators and resource-index itself. The latter is an effort, but once – as discussed in this document – a method has been chosen or further developed, the index can be calculated with a ‘push on the button’, whereas the initial data always need to be inventoried per year.

Some considerations which data gathering approach will be followed are the following:

a) A comparative ‘resource index of nations’ is best based on a single database using consistently gathered data, for all countries and all types of resources the index has to cover.

---

statistical agencies include imports and exports from and to that country. If one however adds up all the imports and all the imports of all countries having them available, covering >95% of the global GDP, one ends up with clear differences in total global imports and global exports that cannot be explained by imports and exports from and to the few remaining countries. But at global level imports and exports obviously must be balanced. This must be solved by changing imports and exports of countries, and hence altering national IO tables.

b) Primary resource use and extraction data can be obtained from national institutes in specific countries, but for countries without official data they can be rather easily gathered from major international databases (e.g. FAOSTAT and Aquastat for land use, biotic material extraction and water extraction; IEA for extraction of energy carriers, and USGS for extraction of other materials).
c) Next to national production-oriented or territorial resource use data consumption-oriented data are needed reflecting the life cycle ‘resource footprint’ of the consumption in a nation.
d) Co-efficient approaches have been used for calculating the ecological and water footprint, usually relying on global resource use databases for land and water like FAOSTAT and Aquastat in combination import and export databases that somehow have been harmonized by third parties (e.g. Hoekstra and Mekonnen (2012); Wackernagel et al,, (2005).
e) For material resources, a co-efficient approach requires calculating ‘Raw material equivalents (RMEs)’ for a large set of products for all countries covered. This is not only a significant job, but will lead to major inconsistences. The Domestic (material) extraction used per country is usually based on data from Geological surveys (e.g. USGS). The materials embodied in consumption are however in part based on the Life cycle assessments performed for calculating the RMEs for imports and exports. By this, it is virtually certain that the total Domestic (material) extraction used for all countries together (i.e. the total global material extraction), will not equal the calculated materials embodied in consumption for all countries together. These however should be by definition the same.
f) The MR EE IO approach in principle provides an elegant approach in which all trade relations and resource extraction data are gathered in a consistent framework at global level. In MR EE IO the extraction and primary use of resources is by definition consistent with the resource use embodied in final consumption.
g) Building global MR EE IOs – particularly their economic cores – is a major job. For environmental assessments, a rather high detail in sectors like agriculture, resource extraction and electricity generation is needed. Most countries have the building blocks for national EE IOs available (IO tables and/or resource extraction data as discussed under b). But building global MR EE IOs by nature exceeds a mandate of an individual statistical office, International organisations (OECD, UN Statistical Division, the UN Commission on Economic and Environmental Accounts) have started discussions on building Global MR EE IOs but official, detailed Global MR EE IOs are unlikely to be available on the short term.

From the above, it seems that the MR EE IO approach is the most promising way forwards for data gathering and organisation. For WRF, it seems however most practical for this moment to avoid practical data gathering but to use one of the global research databases that has for many countries for which the WRF wants to cover the data already available.

2.4 Some policy and scientific backgrounds, and possible definitions

The EU’s Resource efficiency roadmap was faced with the problem that the ‘ideal’ indicator for resource use or resource efficiency does not yet exits. Pragmatically the Roadmap suggests as so-called headline indicator the Domestic Material Consumption (DMC) over Gross domestic product (GDP). The DMC is simply said the weight of national resource extraction plus the weight of imported goods minus the weight of exported goods. This indicator has two clear drawbacks:

a) It does not cover all resources, most notably land and water
b) It allows for the possibility of burden shifting, i.e. that countries stop in part mining primary resources (which lowers DMC) and import finished goods instead (of which

the DMC only counts the weight of the imported product, rather than all primary materials needed to produce that product)

The Resource efficiency roadmap hence suggests to complement DMC/GDP with a dashboard of other key indicators, covering carbon, water, land and resources. From section 2 one can deduct that such resource-related indicators can be formulated from a production perspective and a consumption perspective, next to being related to the resource-extraction itself (pressure indicator) and the impacts of resource-extraction (impact indicator). In a study for the Resource-efficiency roadmap, Mudgal et al. (2012), further expanded in the DESIRE project (Giljum et al, 2013), suggested a classification of indicators as shown in Figure 2.4[4].

Figure 2.4: Basket of resource use relevant indicators (Mudgal et al., 2012)

	Resource use-oriented		Environmental impact-oriented	
	Domestic resource use *(resources directly used for domestic production and consumption)*	*Global resource demand* *(domestic resource use plus resource use embodied in trade)*	*Environmental impacts related to domestic resource use*	*Environmental impacts related to global resource demand*
Material use	Domestic material use Domestic Material Consumption	Global material demand Raw Material Consumption	*Territorial part of* Life-Cycle Resource Indicator *(of Environmentally-weighted Material Consumption)**	Life-Cycle Resource Indicator *(Environmentally-weighted Material Consumption)**
Energy use and climate change	Domestic energy use Gross Inland Energy Consumption	Global energy demand Energy Footprint	Domestic GHG emissions Territorial GHG Emissions	Global GHG emissions Carbon Footprint
Water use	Domestic water use Water consumption *(Water abstraction)**	Global water demand Water Footprint	Domestic water exploit. Water Exploitation Index	Global water exploit. Global Water Consumption Index
Land use	Domestic land use Domestic Land Demand	Global land demand Actual Land Demand (Land Footprint)	Domestic LU intensity Human Appropriation of Net Primary Production	Global LU intensity *eHANPP, LEAC and other indicators on ecosystem quality*

Note further that the indicators in Figure 2.4 are related to resource use itself, and do not say much yet about resource-efficiency. Resource efficiency has two dimensions, being the use of resources on the one hand and the value created on the other hand. The value is usually expressed in economic terms, but includes also technological efficiency. Technological efficiency refers to the minimisation of resource losses along the production chain.

Resource productivity is a related concept and might be seen as the counterpart of resource intensity. It expresses the value that is derived out of resources. Resource productivity can be defined as the value obtained per unit of resource. This value can be expressed in monetary terms, but can also refer to the fulfilment of a specific function or service, or a group of functions. Whereas resource intensity is typically expressed in units like kg/euro, resource productivity has a dimension of euro/kg. Using this framework, we come to Table 2.1. Note that apart from value expressed in economic terms (e.g. Gross Domestic Product (GDP) at national level), as will be further explained in chapter 3 also indicators related to life satisfaction and well-being could be considered[5].

[4] Note that Figure 2.3 can be criticized for the following reasons. It does not only look at resource uses, but also mixes things up with impacts. Some impacts, like territorial GHG emissions are not even fully related to resources, but also due to land use change, agricultural processes, etc.

[5] Other reference values include: Added value at product or sector level (resulting in e.g. euro/kg resource; at this level it is often quite difficult to develop non-economic reference values as suggested in chapter 3); number of people using the resource (e.g. DMC per capita); indicators for specific functions (e.g. kg oil per km transport, MJ delivered energy, m2 dwelling)

Table 2.1: Types of 'resource efficiency' indicators (adapted from Huppes and Ishikawa, 2005)

	Economic (production)	**Environmental (resource saving)**
Euro / resource use	€ / unit resource use Resource productivity	€ / unit avoided resource extraction Costs of resource saving
resource use / euro	Resource use / € Resource intensity	Avoided resource extraction / € cost effectiveness of resource saving

2.5 Deliberations about resource categories to be considered

"Resources" is sometimes used as a very broad concept, including all raw materials and services of the environment. Next to metals or fossil fuels, also water, air and soil, even down to biodiversity and cultural landscape are considered resources. For the purpose of this paper, we propose to take a somewhat narrower view and define resources as raw materials that we extract from the environment in order to use them in the economic processing and final use. These raw materials include metals, non-metallic minerals, fossil fuels, biomass (from both forestry and agriculture), and natural water supplied that we extract and process for use by households, industry and agriculture. It also includes the land we use during agricultural and forestry production. These resources are extracted from nature to supply society with its need for energy, food, cleaning services, transport, shelter etc. In this narrower view, resources do **not** include emissions, environmental quality or impacts on human and ecosystem health from either the extraction, processing, use or waste management of these resources. In this way, we maintain a clear distinction between resource (availability, scarcity) itself and other impacts of economic processes involving resources including resource extraction, processing, use and waste management.

In essence, we face a scoping discussion around the following questions:

- A first question if one just wants to look at resource use *as such*, or also to the *impacts* of resource use. Looking at resource use as such requires mainly some simple accounting exercises (e.g. the amount of water, land and resources used), whereas taking impacts into account complicates things, but also makes analyses more meaningful (e.g. for water use, scarcity in the area where water is extracted is taken into account). In this discussion paper we took the position that impacts at least to some extent need to be taken into account.
- The second question is if one wants to take into account all impacts of resource extraction, transformation, use and waste management (i.e. the life cycle of a resource), or merely those related to resource extraction itself.

One can in fact discern three levels of how one wants to account for resource use and related impacts:

1. Level 1: impacts of resources used are not accounted for. This leads to for instance a simple addition of water extraction in m3, without taking into account that water extracted in river basins with water scarcity is worse as water extractions from river basins with abundant water.
2. Level 2: impacts of resources are included, as far as this has to do with extraction and scarcity. In the example of water above, one would assess water extraction from river basins where water scarcity exists as less desirable.
3. Level 3: impacts of resources are included, including all emissions over the full life cycle of extraction, transformation, use and waste management. An example of this is the Environmentally Weighted Material Consumption (EMC) indicator, that adds up the use of materials as a function of volume and life cycle impact.

We propose to use in this discussion paper for level 2, since level 1 clearly leads to a too simplistic approach, while level 3 makes things extremely complicated. Resources are used in numerous products. The impacts of products can be mapped adopting techniques such as Life cycle assessment (LCA) results in impact scores for products or Input-Output Analysis resulting in impact scores for product groups and nations. However, we are aiming for a resource efficiency index of nations in this paper and not for an environmental footprint of nations. The impacts of products and product groups can only partially be attributed to the resources used in those product (groups). Mapping the impacts of resources only over the full life cycle of the resource would thus lead to all kinds of allocation discussions[6]. To keep things manageable, we here opt for just taking impacts of resource extraction into account.

Box 2.1: Definition of resources
For the purpose of this paper, we define resources as raw materials that we extract from the environment in order to use them in the economic processes and goods. These raw materials include metals, non-metallic minerals, fossil fuels, natural biomass (including biomass produced by forestry and agriculture), and natural water supplies that we extract and process for use by households, industry and agriculture. This definition excludes emissions, environmental quality or impacts on human and ecosystem health from either the extraction, processing, use or waste management of these resources. In this way, we maintain a clear distinction between resource (availability, scarcity) itself and other impacts of economic processes involving resources including resource extraction, processing, use and waste management. Note that in this we further used a definition of resources common in the Material flow analysis community, that includes forestry and agricultural products as resources extracted from nature. An alternative view is seeing such materials being produced via processes in the economic system.

This position creates some tension between the approach proposed in this paper and the discussion in section 2.3 and the EU's Resource Efficiency Roadmap. As indicated, that Roadmap presented a dashboard of indicators including materials, land, water and Greenhouse Gas (GHG) emissions. In the demarcation suggested in the above, GHG emissions do not fit the definition. Carbon as a resource might fit in: this can be linked to fossil fuels and biomass. We propose to include the following rough categories of resources:

- Materials, following the convention in Material Flow Analysis (MFA) discerning:
  - Fossil fuels
  - Metals
  - Other minerals, including construction minerals and industrial minerals
  - Biomass, including biomass harvested from agriculture and forestry, and biomass taken from nature.
- Water, in so far it is used as a resource for drinking, cleaning, agriculture and industry, therefore excluding water as an environmental compartment
- Land, broken down to
  - Built-up land
  - Agricultural land
  - Forestry land
  - Unused land (nature).

[6] In the Environmentally weighted Material Consumption this allocation has been done, but there allocation choices have been made as well: "For every considered material, an estimate is made of its contribution to environmental problems throughout its life cycle. This includes not only the impacts related to the material itself, but also the impacts of auxiliary materials, energy used for its extraction and production, emissions of impurities and pollutants included in the material during use or waste treatment, etcetera. Energy use in the consumption phase is not allocated to the materials chains. We consider this energy use - for example, petrol in cars or electricity for computers - to be related to products rather than materials. It is difficult to allocate the use of energy to the individual materials a product is composed of, and quite often the energy use is hardly related to these materials. Energy use in the consumption phase however is not excluded from the EMC: it is included in the chains of fossil fuels, and any change due to shifts to less energy-intensive products will bevisible in the EMC (van der Voet et al., 2005: 7).

In many resource categories, challenges occur with the supply, either already now or in the foreseeable future. However, most, if not all, of the options for meeting future demand turn out to have consequences for resources other than that which is targeted. These resource "linkages" can severely constrain potential solutions. Because all major resource categories are challenged in this way, a quantitative understanding of the linkages is very important for exploring pathways toward sustainable development (see Box 2.2).

Box 2.2: Linkages between resource uses

There are different types of linkages that may be important:

- Via urbanisation: a larger share of the growing world population lives in cities. There will be an increased need for infrastructure, such as industrialized water supplies and wastewater treatment facilities. Water infrastructure has a high energy demand as well as significant material requirements. Emission reduction technologies attached to waste and water treatment plants, power plants, and other such large-scale facilities are necessary in urban areas to keep up environmental quality. If implemented on a large scale, the quantity of materials involved is large. The use of emission reduction technologies, such as CCS, may reduce CO2 emissions, but it also reduces energy efficiency considerably.
- Via an increased difficulty in accessing resources as a result of an ever and rapidly growing demand. Declining ore grades, for example, have substantial energy and water implications. Biomass supply for a growing population (food and energy) may lead to increased pressure on land as well as on water resources; agriculture already accounts for 70% of the world's total water use. To offset a declining supply and meet future demand, energy-intensive efforts (e.g., purification of polluted resources or desalinization of seawater) may be needed to ensure the water supply.
- Via the envisaged energy transition: the viability and upscaling of alternative energy pathways. To supply the world with a significant share of bioenergy, for example, biomass production will have to rise by an order of magnitude. A shift to bio-based energy, therefore, has dramatic implications for land and water use, and is likely to encounter constraints quite quickly. A large-scale transition to solar energy technology may meet constraints in rare metal resources.
- Via environmental degradation as a results of resource use. A continued use of fossil energy sources, for example, may harm the potential for biomass production via climate change, especially via changes in precipitation patterns. Water quality degradation may increase as a result of the agrochemicals needed to secure the (increased) supply of biomass. Similarly, the increased need for mining can have consequences for water depletion and water quality degradation, especially when mines are located in water-poor regions.

These linkages are very important and ideally should be included in a system of resource efficiency indicators. As there are still many unknowns in this area, this may not be feasible yet. However including them in a forward looking indicator system for resource efficiency would be indicated.

2.6 Reflection and conclusions

This section indicates that it is important to include the classical, sharp distinction between an economic system that extracts resources and expels emissions and waste to the natural system. The indicator system further should distinguish between a production-oriented and a consumption-oriented perspective. We further see that indicator systems often distinguish land, water and materials as resource inputs into the economy, whereas often also energy use and related GHG emissions are included. The latter is controversial, since GHG are emissions, and not inherently related to resource extraction. Finally, we see that to calculated resource-efficiency indicators some reference for economic activity or well-being has to be chosen. We therefore discuss in section 3 GDP and alternative reference indicators, whereas in section 4 we zoom in further on how indicators for land use, water use, materials use and energy use and related GHG could be defined.

This section further indicated that selection and calculation of indicators is probably a minor job compared to gathering primary data to calculate the indicators. For pragmatic reasons, it is suggested that WRF for now will use one of the existing, major so-called MR EE IO research databases in which economic and resource extraction data are stored at global level.

3 Economic reference or ‘outcome’ indicator

3.1 Introduction

Increased well-being of its citizens is generally the ultimate goal of a modern society. Usually, increasing the well-being and higher development of a society requires more resources. As indicated in the introduction, it is therefore useful to relate resource use of a country to indicators for well-being and development, to arrive at a measure for resource-efficiency. The traditional reference value for well-being and development is the Gross Domestic Product (GDP), but in the context of the ‘Beyond GDP’ debate other indicators for well-being have been proposed, for instance the Human Development Index (HDI), ‘Happy life years’ (HLY), Better Life Index (BLI) and others like the Genuine Progress Indicator (GPI), the Index of Sustainable Economic Welfare (ISED), Adjusted Net Savings, etc. .

As indicated in the introduction, for the purpose of this discussion paper we want to focus on ‘outcome’ indicators, i.e. indicators that provide some measure of quality of life of citizens of nations. For this reason we will not consider e.g. the GPI, ISEW or similar indicators as economic reference, since they measure already in some form resource use or resource depletion, which we want to keep separately to measure economic outcomes against. Below we discuss in some more detail GDP, HDI, HLY, BLI whereas summarizing some other ‘Beyond GDP’ indicators in Box 3.1

Box 3.1: Some ‘Beyond GDP’ indicators

Adjusted net saving (ANS) measures the true rate of saving in an economy after taking into account investments in human capital, depletion of natural resources and damages caused by pollution[7].

Index of Sustainable Economic Welfare (ISEW) The ISEW and GPI based on the concept of sustainable income, i.e. the amount a person or an economy can consume at one point in time without decreasing consumption in future (Daly and Cobb, 1989; cf. Hicks, 1946). In its simplest form, the ISEW can be calculated as the sum of private consumption, non-market services generating welfare, increase in capital stock and balance of international trade, minus costs of deterioration of nature and natural resources, crime, defensive expenditures, reduction of leisure time[8].

Genuine progress indicator (GPI). The GPI uses the same conceptual approach as ISEW but is somewhat more comprehensive in its coverage of elements for with the GDP has to be adjusted (see e.g. Costanza et al., 2004).

3.2 Gross domestic product

Gross domestic product (GDP) is defined by Organistion for Economic Collaboration and Development (OECD) as "an aggregate measure of production equal to the sum of the gross values added of all resident institutional units engaged in production (plus any taxes, and minus any subsidies, on products not included in the value of their outputs)”[9] The GDP of a country can be determined in 3 ways, that in principle all should give the same value:

- Expenditure approach: GDP (Y) is the sum of consumption (C), investment (I), government spending (G) and net exports (X – M).
- Production approach: GDP sum of gross value added (value of product and service output minus input of intermediate products and services) by all resident producers in

[7] http://web.worldbank.org/WBSITE/EXTERNAL/TOPICS/ENVIRONMENT/EXTDATASTA/0,,contentMDK:21061847~menuPK:2935543~pagePK:64168445~piPK:64168309~theSitePK:2875751,00.html (accessed 12 October 2014)
[8] http://rprogress.org/sustainability_indicators/genuine_progress_indicator.htm (accessed 12 October 2014)
[9] http://stats.oecd.org/glossary/detail.asp?ID=1163, accessed 12 October 2014

the economy plus any product taxes and minus any subsidies not included in the value of the products.

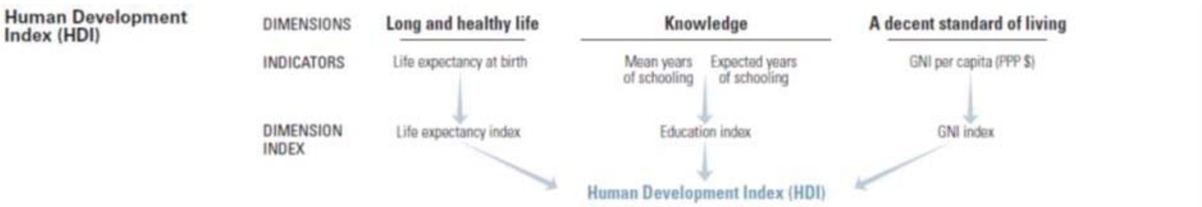
- Income approach: GDP is measured as the sum of all incomes, i.e. GDP = Compensation of employees + Rent + Interest + Proprietor's Income + Corporate Profits + Indirect business taxes + Depreciation + Net foreign factor income.

Statistical offices often use the system of Supply and Use tables/Input Output tables to reconcile these three approaches of calculating GDP. In a SUT/IOT all elements to calculate GDP via the different routes are present, and in a balanced SUT/IOT GDP calculated via any of these approaches should be equal.

3.3 Human development index

The Human Development Index (HDI) was created by the United Nations Development Program (UNDP) in 1990 in recognition that people and their capabilities should be the key criterion for assessment of development of a country, rather than economic growth per se. It was developed by a team lead by the Pakistani economist Mahbub ul Haq inspired a.o. by the capabilities approach developed by the Indian economist Amartya Sen. The HDI measures the achievement of countries in three key dimensions of human development: providing long and healthy lives, education and knowledge, and a decent standard of living (see figure 3.1).

Figure 3.1: steps in calculating HDI (UNDP, 2014)



Minimum and maximum values are used as 'goalposts' to transform the indicators expressed in different units into indices between 0 and 1. For life expectancy, the 'natural zero' is set at 20 years since no country in the 20th century had a lower life expectancy. The aspirational goal for life expectancy has been set at 85 years. For education, societies can subsist without formal education, justifying a minimum of 0 years. The maximum is 18 years, equivalent to achieving an MSc degree in most countries. For income, the minimum is set a Gross National Income of 100$, whereas the maximum is set at 75.000$, a value above which extra income gives virtually no gain in human development (e.g. Kahneman and Deaton, 2010). Using for a country the actual life expectancy at birth, mean years of schooling, expected years of schooling and GNI per capita in purchasing power parity, indices are calculated as follows (UNDP, 2014):

- Health index or Life expectancy index (LEI): (Life expectancy – 20) / 85 – 20

- Education index (EI): (MYSI + EYSI) / 2
  - Mean Years of Schooling index (MYSI) = MYS / 15
  - Expected Years of Schooling index (EYSI) = EYS /18

- Income index (II)= (ln (GNI per capita) – Ln (100)) / (ln (75.000)- ln (100))

UNDP (2014) considered that the translation from income to capabilities is most likely concave, i.e. each additional dollar of income has a smaller effect on expanding capabilities. For this reason, for the income index the natural logarithm of the actual, minimum and maximum value is used.

Finally, the HDI is calculated as by multiplying all three indices and taking the cube root of this multiplication:

- $HDI = (LEI * EI * II)^{1/3}$

3.4 Happy life expectancy / happy life years

The Happy Life Expectancy (HLE) is a concept developed by prof. Ruut Veenhoven of the Erasmus University Rotterdam (EUR). The basic indicator is proposed in the paper “Happy Life-Expectancy, A Comprehensive Measure of Quality-of Life in Nations”, which appeared in the journal Social Indicators Research in 1996 (Veenhoven, 1996). Veenhoven argues that HLE may be a better indicator of quality of life than others as it relies on subjective measures of happiness, as opposed indicators like the HDI that use an arbitrary set of aspects/determinants for quality of life, arbitrary weighting across these aspects, and limited universal value across culture and time. The use of Happiness and the Happy Life Expectancy as indicators was significantly enhanced by the development of the new economics foundation (nef, 2012) of the ‘Happy Planet Index’, that plots the HLE of nations against the Ecological footprint of nations (being a measure of resource pressure caused by nations), and the publication of the World Happiness Report by the Sustainable Development Solutions Network (SDSN; Helliwell et al, 2013). The main difference between the SDSN and nef is that nef does not use happiness on its own, but includes life expectancy in its indicator.

Both the new economics foundation as the SDSN draw data for average levels of well-being by country from responses to the ‘ladder of life question’ in the Gallup World Poll. This poll uses samples of around 1000 individuals aged 15 or older in each of more than 150 countries. The question asked is:

Please imagine a ladder with steps numbered from zero at the bottom to 10 at the top. Suppose we say that the top of the ladder represents the best possible life for you and the bottom of the ladder represents the worst possible life for you. On which step of the ladder would you say you personally feel you stand at this time, assuming that the higher the step the better you feel about your life, and the lower the step the worse you feel about it? Which step comes closest to the way you feel?

The Happy Life Years (HLY) then are calculated by multiplying the ladder of life score by life expectancy for each country.

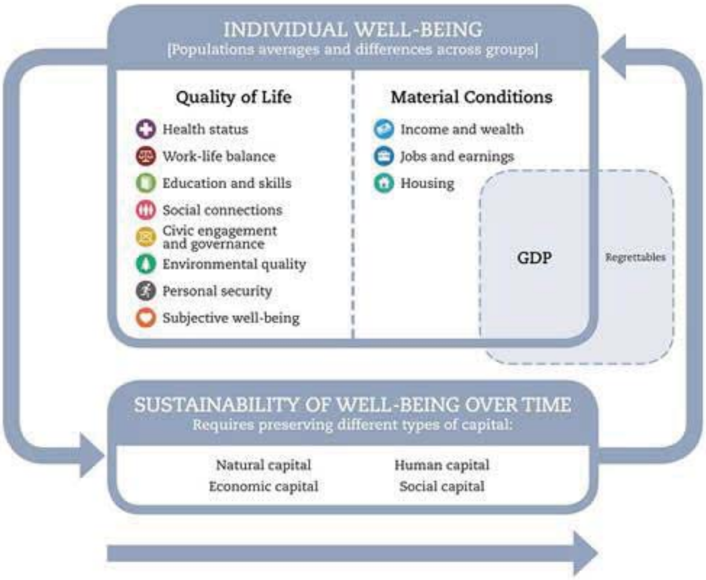
3.5 Better Life Index

The OECD launched the 'Better life index' based on the conceptual approached visualized in Figure 3.2. The OECD measures quality of life using 11 categories encompassing a dashboard of 25 sub-indicators. It concerns:

1. Housing (Dwellings without basic facilities; Housing expenditure; Rooms per person)
2. Income (Household net adjusted disposable income; Household net financial wealth)
3. Jobs (Employment rate; Job security; Long-term unemployment rate; Personal earnings)
4. Community (Quality of support network)
5. Education (Educational attainment; Student skills; Years in education)
6. Environment (Air pollution; Water quality)
7. Civic engagement (Consultation on rule-making; Voter turnout)
8. Health (Life expectancy; Self-reported health)
9. Life Satisfaction (Life satisfaction)
10. Safety (Assault rate; Homicide rate)
11. Work-Life Balance (Employees working very long hours; Time devoted to leisure and personal care)

Most of these (sub)indicators can be derived from official statistics, with the exception of e.g. life satisfaction, and quality of the support network. The OECD was hesitant to construct a synthetic index since this could only be done 'if individual level data as well as country level data are available from the same survey'. OECD therefore presents a dashboard of 25 individual indicators related to the categories above. Via an interactive website, however, OECD allows individuals to create their own composite index by weighting the various dimensions according to what they consider more important for their well-being.

Figure 3.2: OECD framework for measuring well-being and progress (OECD, 2013)



Source: OECD, 2013

3.6 Reflection

From the indicators above, we feel ANS, ISEW and GPI best are not used as an alternative reference indicator. These correct GDP for elements that do not contribute to well-being and capital losses. Particularly dealing with natural, social and economic capital losses is an important issue to consider in a resource-use or resource-efficiency index of nations. But the aforementioned indices do not measure well-being or quality of life per se. We want to measure how well the economy performs in terms of output (providing high quality lives), given a certain resource-input.

Indices such as HDI, Happiness, HLY and BLI seem hence more useful indices for our purpose. The BLI is rather comprehensive, but has as drawback that it consists of a dashboard of 25 sub-indicators without aggregation. Just measuring happiness, as in the World happiness report, seems not as comprehensive as measuring Happy Life Years (HLY). The HDI finally has as advantage that it is published by the UNDP, which gives it a more official character as the HLY which is calculated by a think-tank.

4 Indicators for carbon, water, land and materials

4.1 Introduction

As indicated in section 2, the classical division in e.g. production and consumption ('footprint') related indicators for carbon, water, land and materials has a number of problems. A carbon footprint is an indicator measuring (the impact of) emissions rather than resource use. Water and land are used to produce biotic materials (wood and agricultural crops), which in turn are part of the material footprint. The carbon footprint is related to extraction of fossil fuels or energy materials, which are also included in the material footprint. The carbon footprint and the use of fossil fuels can be reduced by using biomass, leading to higher land and probably higher water use. And ground- and river water use could be reduced by desalination of sea water, with in turn requires significant amounts of energy. The categories are hence not fully independent but 'linked' (e.g. Graedel and van der Voet, 2010), whereas the carbon footprint as an impact oriented indicator may not have a place at all in a resource-use or resource-efficiency index of nations.

Having said this, it is however useful as an intermediate step to a proposal for a resource-use or resource-efficiency index of nations to highlight some methods that create an aggregated indicator within the 'resource categories' carbon, water, land and materials. We discuss them in the next 4 sections. Note that these aggregated indicators are still independent from the question if a territorial/production perspective or a consumption/footprint perspective is chosen – we merely want to address the question how aggregation within a category of resource pressures can be done. The inventory (see chapter 2) determines if this pressure is calculated from a production or consumption perspective.

4.2 Carbon

As indicated above, including an indicator for carbon is something of an oddity since usually indicators for carbon relate to territorial or consumption-based carbon emissions, and it is hence an indicator measuring impacts of emissions rather than resource input.

Usually different greenhouse gases (e.g. CO2, CH4, N2O) are aggregated on the basis of the amount of solar radiation that they can retain within the atmosphere over a period of time. The International Panel on Climate Change (IPCC, 2013) provides for this purpose so-called Absolute Global Warming Potentials (AGWPs) for different greenhouse gases. If expressed relative to the AGWP of CO2 which usually is taken as a reference, it is called the Global Warming Potential (GWP). Usually a time horizon of 100 years is taken as default, although also (A)GWPs for longer time horizons are published. These are generally seen as less robust, but give better insight in the climate impacts of greenhouse gases with long life times in the atmosphere. In Life cycle impact assessment it is commonplace to calculate in this way the GWP caused by a large number of greenhouse gases emitted over the life-cycle of a product as a so-called mid-point indicator. See for further details various authoritative manuals in this field (e.g. Guinée, 2002; Goedkoop et al, 2009; Huijbregts et al., 2014).

An alternative to this emission-related indicator would be to calculate carbon truly as a resource input to the economy. In energy statistics such an indicator would be *Primary production of energy*. Eurostat defines this as "*any extraction of energy products in a useable form from natural sources. This occurs either when natural sources are exploited (for example, in coal mines, crude oil fields, hydro power plants) or in the fabrication of* *biofuels"*[10].Primary energy production usually is expressed in tons of oil equivalent, but could also be expressed in e.g. Joules. Such an indicator would clearly overlap with the domestic material extraction discussed in section 4.5 'Materials'. There however coal, crude oil and gas simply would be added up on a mass basis rather than their energy content, and energy generated by hydro and solar power plants would be excluded (since not depending on material extractions from nature).

4.3 Water

Water use is often divided into so-called 'green', 'blue' and 'grey' water use. Green water comes from natural precipitation. Blue water is water extracted from rivers and aquifers for human (including agricultural) use. Where 'green' and 'blue' water hence represent in principle genuine water extractions, 'grey' water is used as a rather simple measure for water pollution. If an emission of a toxic substance to water occurs, the grey water use is calculated as the amount of water needed to dilute this emission so that the concentration of the toxic substance is below a relevant toxicological standard. This way of calculating 'grey water' use is heavily criticized – it mixes up an indicator for water pollution with an indicator for water extraction, and it neglects in full the 'fate assessment' step that is standard in any toxicological risk assessment or Life cycle impact assessment of toxic releases. For instance, in grey water use calculations all emitted substances are assumed to be persistent to eternity, while in practice many substances decompose more or less quickly.

Authors affiliated to the Water Footprint Network tend to calculate the Green, Blue and Grey water use by economic sector or product (including agriculture) in m^3, and add these volumes simply up to a total (e.g. Gerbens-Leenes et al., 2009; Hoekstra and Mekkonen, 2012). In this simple way we also calculated recently the 'Blue water footprint of nations' (Tukker et al., 2014).

Such simple additive approached to calculating domestic water use (production perspective) or water footprints (consumption perspective) have been severely criticized (e.g. Pfister and Hellweg; 2009; Ridoutt and Huang, 2012; see also Pfister et al, 2009). Critics include:

a) Green water is not to be included. Natural vegetation also will extract rainwater, and there no good reason to see why agricultural vegetation will extract more rainwater or create undesirable by-effects. A counter-argument to this critique obviously is that agricultural vegetation is largely removed from the land where it grows, implying the water part of the plant is lost for other uses[11].
b) Inclusion of grey water is complicated due to the factors mentioned above.
c) Only Blue water is to be seen as a resource-extraction. Yet, such blue water extraction is much more severe in areas of high water stress as in areas where water is abundant[12].

The latter line of reasoning hence calls for weighting water extraction with some scarcity index that is specific per country or river basin. The European Environment Agency defines the water exploitation index (WEI), or withdrawal ratio, in a country as the mean annual total

---

[10] http://epp.eurostat.ec.europa.eu/statistics_explained/index.php/Glossary:Primary_energy_production

[11] This discussion could be further refined by looking how much water (agricultural ore natural) vegetation uses and dissipates via evapotranspiration, versus the water content of harvested plants. Only the latter is water that gets lost by agriculture.

[12] Ridoutt and Huang (2012) give the following example. In China, the water footprint of wheat grown in the highly water-stressed northern Huang basin is 800 m3·tonne (t)−1, compared with 1,031 m3·t−1 in the water-rich southern Chang basin (4). Simply showing the water footprint gives the impression that the efficiency of water use in the Huang basin is positive, whereas in practice this level of water use may be (much) less sustainable as the higher water use per ton in the Chang basin.

abstraction of fresh water divided by the long-term average freshwater resources. The long-term average freshwater resource is derived from the long-term average precipitation minus the long-term average evapotranspiration plus the long-term average inflow from neighbouring countries. A withdrawal of less than 20% is seen as a situation of no stress. Severe water stress can occur for WEI>40%, which indicates strong competition for water (EEA, 2003). Building upon this concept of the Water exploitation index, Pfister et al. (2009) proposed watershed specific Water stress indices, that can be used to weight blue water use per river basin.

This watershed specific WEI (Pfister et al., 2009) is a further geographical refinement of the Ecological Scarcity Method (Frischknecht et al., 2008). The latter is the default characterisation model for water consumption at midpoint level as recommended by the ILCD[13] (EC, 2012). According to ILCD, at the endpoint level, all methods evaluated are too immature to be recommended. However, the ReCiPe (Goedkoop et al., 2009/2013) method may be used as an interim solution.

4.4 Land

Accounting for land use, and particularly if some kind of impact assessment is included, is still relatively immature (e.g. Hauschild et al. 2013; Jolliet et al. 2014). A large number of land use indicators exist, but many of them stay close to (life cycle) inventory data (Pfister et al, 2014). Figure 4.1 and 4.2 allow for explaining some of the complexities involved. Note that figure 4.1 already assumes that impacts on biodiversity (species richness) is to be used as indicator for quality (reduction) caused by land use occupation and transformation. In relation to Figure 4.1 and 4.2, one now can try to classify a number of often used land use inventory and impact assessment methods.

The CML 2001 Life cycle impact assessment method, a very simple indicator for land use (competition) is suggested (Guinée, 2002). It concerns simply the occupied land area multiplied by the time of the occupation. For the type of annual accounting done in e.g. the Global resource footprint of nations this simply boils down to the amount of m2 or km2 land occupied by an economic activity in this year.

A main limitation of this simple approach is obvious: it does not take into account the original 'quality' of the land that is occupied. Next to land use occupation, the ecological footprint hence also takes into account the capacity of different land cover types to produce resources for humans (Wackernagel and Rees, 1996). Occupation of highly productive croplands is hence seen as significantly worse as occupation of lowly productive pastures (e.g. Ewing et al, 2010)[14].

An indicator that also includes land productivity is the human appropriation of net primary production (HANPP). For each geographical area (grid cell) it calculates the NPP left for ecosystems between the current land use and a reference state: HANPP = natural state NPP – reduction in NPP (by changing from the natural state to current land use) – harvested NPP. The HANPP hence can be used to express land use in biomass loss in a specific year.

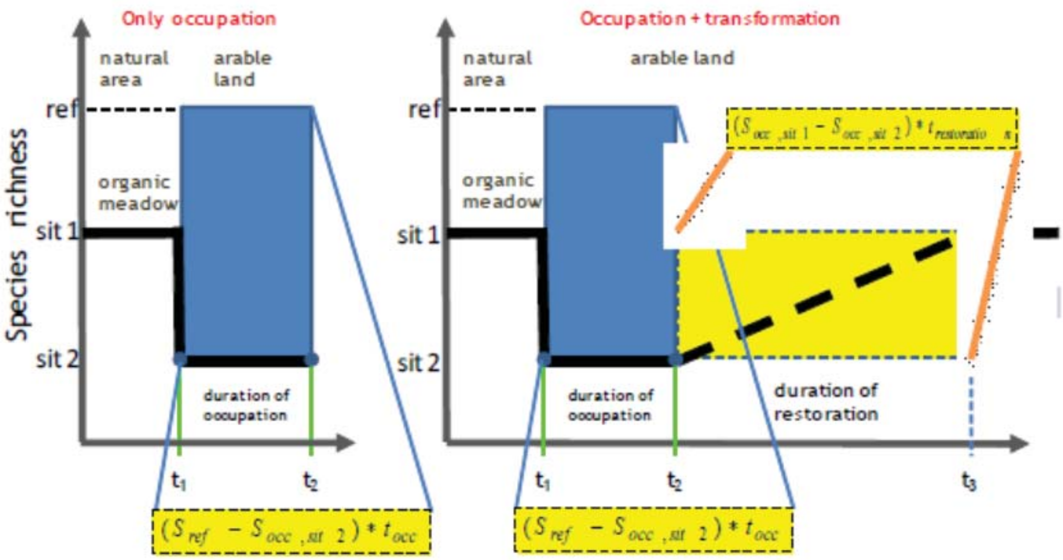
[13] International Reference Life Cycle Data System (ILCD). http://eplca.jrc.ec.europa.eu/?page_id=86

[14] The Ecological footprint method further adjusts for productivity differences between the same land types in different countries. Finally the Ecological footprint methods also accounts for a form of virtual land use, i.e. the amount of land that would have been required to capture CO2 emissions from fossil fuels in the form of biomass or the area to grow biomass that can replace fossil fuel use. The latter aspect is less relevant for this discussion paper

Other indicators use biodiversity loss related to land use occupation or land use change as an end-point (e.g. Croezen et al, 2011; Goedkoop et al, 2009/2013):

a) The Potential Disappeared Fraction of species (PDF). This parameter measures, by geographical area / grid cell, the relative change in biodiversity compared to a reference situation. For land occupation, this reference is the natural, pristine situation that would have existed without any human intervention. For land transformation, the reference is the situation after land use change.
b) The Mean Species Abundance (MSA). This indicator expresses the species richness in a geographical area / grid cell compared to the species richness in a reference situation.

Figure 4.1: Interpretation of Croezen et al. (2013) of the relationships applied in the ReCiPe project for quantifying impact of land use on biodiversity (Goedkoop et al, 2009/2013)



PDF and MSA are closely related, as can be seen from the following formulas:

PDF calculated for occupation:

$$PDF_{nature \rightarrow occ.} = \frac{S_{nature} - S_{occupation}}{S_{nature}} \text{ or } PDF_{nature \rightarrow occ.} = \frac{a_{nature} - a_{occupation}}{a_{nature}}$$

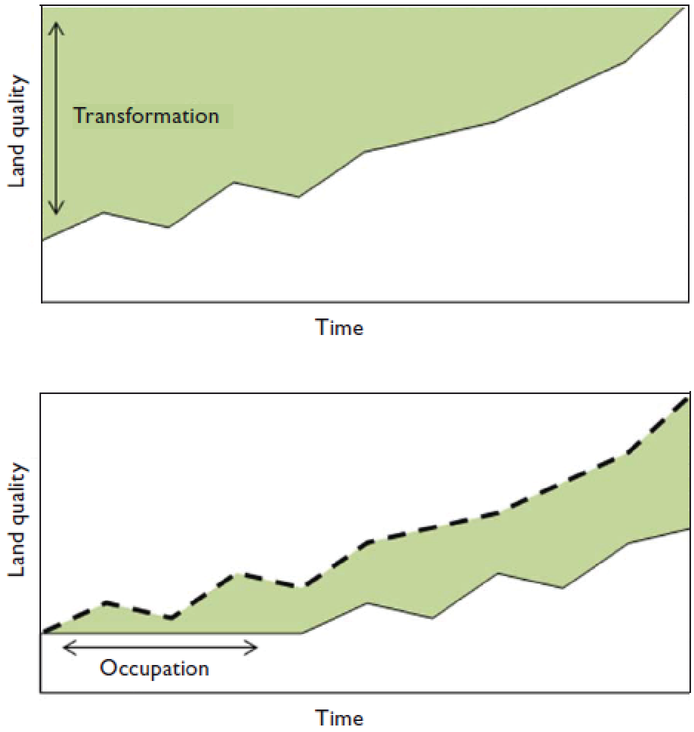
PDF calculated for transformation:

$$PDF_{1 \rightarrow 2.} = \frac{S_1 - S_2}{S_2} \text{ or } PDF_{1 \rightarrow 2} = \frac{a_1 - a_2}{a_2}$$

MSA

$$MSA = \frac{S_{occ}}{S_{nature}} \text{ or } MSA = \frac{a_{occ}}{a_{nature}}$$

Figure 4.2: Interpretation of Matilla et al. (2011) of work of Milá I Canals et al. (2007), suggesting that land transformation can be considered as an occupation impact, related to a certain time required to restore the system to a natural state. Prolonged occupation is then seen as a delay to restoration to this state



Relation between MSA and PDF:

$$PDF_{nature \rightarrow occ.} = \frac{S_{nature} - S_{occupation}}{S_{nature}} = 1 - MSA$$

Further, in the context of Life cycle impact assessment (LCIA) indicators related to emissions and quality changes are suggested (emissions from land use and transformation; changes in organic carbon content of land; nutrient losses; other emissions such as related to biomass burning/clearing of land, erosion, etc.; see EC, 2010).

Apart from these quite different approaches to develop indicators related to land use, there are additional complexities which include:

a) Some methods use a 'natural reference situation' with regard to the amount of biomass produced, or amounts and diversity of species available in a specific geographical area[15]. Obviously, this estimate of the reference brings in uncertainties, while there is also discussion is a static or dynamic reference should be used (i.e. using a 'pristine' or 'current' natural reference (e.g. Koellner and Scholz, 2007), or

[15] For instance, the MSA counts only the original species in a particular area. The rationale behind this is that it avoids counting other species that filled the niches left by the original species. The implication is for instance that logging down pristine forest and replacing it by production forest, or the transformation of original grassland into intensive agriculture results in a major loss of MSA since the original species are replaced.

using some suitable existing land use as reference in the same area where the transformation took place (Milà i Canals et al. 2007)).

b) Usually, current land use is not a matter of one transformation step from the natural reference situation to a new state, e.g. build up land. Intermediate changes may have occurred (e.g. from natural vegetation to production forest, from production forest to agriculture, and from agriculture to build up land). This results in questions about what to take as the reference situation, and if indeed a ‘natural reference situation’ is taken, one has the option to allocate land use occupation and related quality losses to the different land uses over time.
c) Assumptions with regard to the recovery time (and the question if full recovery is possible and how to penalize less than full recovery) are highly uncertain.
d) Methods often apply implicit weighting of deviations of the reference situation across geographical area. For instance, the HANPP simply counts the amount of biomass removed regardless of the geographical area where it happens, and potential quality impact on specific biotopes. The MSA counts the % loss of species per grid cell, and then adds up implying that grid cells with a high species density with some loss are penalized in comparison to grid cells with very low species density (e.g. deserts) with no loss.
e) Inclusion or exclusion of indirect land use change effects. For instance, using palm oil from existing production site for biofuel rather than food production, may indirectly lead to land use change elsewhere to compensate for the lost food production (e.g. Cramer, 2007).

4.5 Materials

4.5.1 Introduction

In section 4.5.2 we will discuss some classical economy-wide Material flow indicators. Although such indicators can be used to express resource use and -efficiency in an aggregate way, they have limitations. As we will show the main limitation is that materials simply are aggregated by weight.

They do not provide information on either resource scarcity or impacts related to resource use. For resource scarcity, it makes more sense to account for specific materials instead of adding it all up. For environmental impacts, a connection must be made with environmental impacts. Such approaches for adding materials in a weighted way are discussed in section 4.5.3 and 4.5.4.

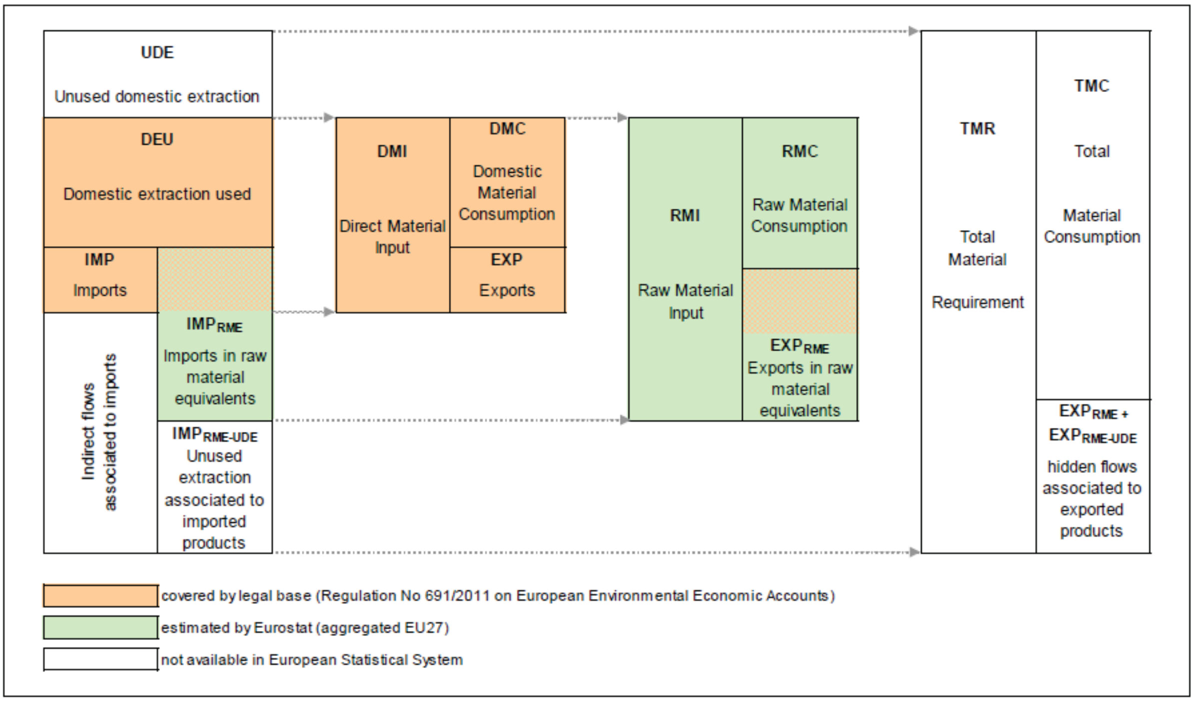
4.5.2 Economy-wide Material Flow Accounts (EW-MFA)

Material indicators, and material efficiency indicators, are often drawn from Material Flow Accounts. Economy-wide material flow accounting is a systematic and comprehensive conceptual framework that allows describing the interaction of a domestic economy with its natural environment and the rest of the world economy in physical terms (Eurostat 2012). It accounts for flows of materials, except for water and air, in a given year. Several aggregate indicators can be derived from this accounting framework, with DMC (Domestic Material Consumption) being the currently most prominent one. Methods for calculating EW-MFAs are highly standardized and implemented in international environmental accounting systems such as the EU (see Eurostat, 2001 and Eurostat, 2013) and the SEEA framework of the UN (UNSD, available at http://unstats.un.org/unsd/envaccounting/seea.asp).

From EW-MFA several aggregate indicators can be derived. Resource efficiency within EW-MFA is usually measured as resource productivity in €/ton. As already indicated in chapter 2, in the Resource Efficiency Roadmap (European Commission 2011) "**Gross Domestic Product GDP/Domestic Material Consumption (DMC)**" is proposed as lead indicator for monitoring the general development of material productivity in Europe. Although called "consumption" this indicator takes (in part) a regional approach. It does not take in full a global consumption perspective and is hence not able to monitor shifting of environmental burden to other countries via international trade. The DMC full includes Domestic Extraction Used, but then uses only the actual mass of imported and exported products to calculate a DMC. The upstream material requirements for imported and exported products hence are neglected.

Methods to calculate upstream material requirements, and thus using a functional system, are currently in development. There are basically two approaches for calculating these upstream material requirements of traded products, similar to the difference between co-efficient approaches and input-output approaches discussed in section 2.2.3 for water accounts before. One approach identifies the imported products using detailed trade statistics, and then uses a Life cycle impact assessment approach to estimate the upstream primary material requirements. Another approach is the Input-output bases approach discussed in section 2.2.3. In a hybrid account, these two approaches can be combined. All these methods lead to the **Raw Material Consumption (RMC)** indicator, reflecting the total global material extraction used related to consumption in a country. If also unused extraction is accounted for (e.g. mine tailings in coal mining), one speaks of **Total Material Consumption (TMC).** Figure 4.3 reviews how the different indicators focusing on materials are related.

Figure 4.3: Scheme for EW-MFA and derived indicators (from: Eurostat[16])



[16] See: http://ec.europa.eu/eurostat/documents/1798247/6191533/RME-project-Introduction.pdf/96bbe739-bfda-47a2-99c7-cc3ec1048832, accessed 11.02.15

Next to these economy-wide indicators for resource use, one can establish accounts for specific materials (see Box 4.1). Since the aim of this project is to define an overall resource use or –efficiency indicator for nations, we do not discuss these in more detail here.

Box 4.1: Accounting for specific materials
At the global level, the USGS statistics for production of metals and minerals provide information on the amounts mined. This information is provided per country as well as worldwide, and therefore could be used as the resource dimension of resource efficiency. It is not straightforward what the economic dimension should be. In the literature we find indicators such as kg iron, aluminium or copper produced per capita, as well as per € of GDP. This could give a picture of a society's iron, aluminium or copper intensity. These indicators are generally based on primary production.

Besides production, also the use of such materials is relevant. An indicator for materials use is the stock-in-use, the total amount of a certain material being present in the technosphere. Stock information is scarce; it exists only for a few materials on a global level. It is however very relevant information for resource efficiency, as these stocks can be considered as potential secondary materials. Only recently has the attention for stocks begun to grow and a few estimates and models are starting to become available. Stocks can be calculated from time series information on flows, which has been attempted for a number of materials, especially metals, for various countries and regions. For a circular economy, information on stocks in society is essential. As this is one of the major ways forward to increase resource efficiency, indicators on stocks in society should be developed and the database should be created to calculate those indicators.

Some indicators exist for specific materials regarding the waste stage. The amount of scrap generated is generally part of USGS statistics, as is secondary production. The recycling rate is known for a only few elements, for a greater number we have crude estimates at the global level. It is clear that recycling rates > 50% can be found only for the very large scale metals such as iron and copper. For most smaller scale metals recycling rates are quite low. This is an issue of concern for resource efficiency: closing loops and secondary production are very important to improve resource efficiency. Whereas "general" MFA based information is available, this is not the case for information on the use and waste stage of specific resources. The lack of such data can be considered a major information gap for a resource policy aiming at a closed loop economy.

4.5.3 Methods using environmental impacts of resource use

For regional systems, especially countries, the total of environmental impacts of a nation could be approached by using the emissions inventory. One could say that the aggregate impacts of resource handling in societies is specified by measuring or estimating a nations emissions. For functional systems, however, this is more complicated. Lately, there are efforts to calculate so-called footprint indicators at the national level. The best known indicator in this area, also included in the Roadmap's dashboard, is the carbon (or CO_2) footprint. Of all specific footprint emissions, this is the only one that is used at the macro-level of national economies. A database exists with a time series of 10 years, specifying CO_2 emissions of consumption, including emissions abroad. It was compiled using trade linked EE-IOTs, again illustrating the versatility of this accounting tool. It is possible to compile footprint emissions for other pollutants as well (Hertwich & Peters, 2009).

Such footprint accounts do use a functional system, but it is very difficult to relate them to the use of specific resources, and therefore, resource efficiency. Another effort at specifying life cycle or footprint impacts, in this instance explicitly related to resource use, is the EMC indicator (van der Voet et al., 2005). This indicator uses MFA information to compile material balances for national economies and adds Life cycle inventory (LCI) information to specify life cycle impacts for each material. This offers information on the contribution of different resources to different environmental impacts. It is specifically designed for the macro-level as an indicator of "double decoupling", and is still the most complete indicator for that purpose. Although scaling down to sector level will be difficult, it can be broken down to separate resources as well as separate impact categories.

4.5.4 Methods based on scarcity and criticality

For resources, several impact categories have been developed for midpoint and endpoint categories (see Annex 1). Impact assessment methods have been particularly developed for abiotic resources. The diversity of methods developed is huge and there is no consensus on one best method because the *definitions of the problem are different* (e.g., decrease of the resource availability itself; decreasing reserves of useful energy/exergy; the contribution of current or future extraction processes). Assuming that we adopt the definition that abiotic depletion is about the decreasing availability of resources, there are still different methods possible (only focusing on reserves, or also taking into account annual total rates of extraction), and different definitions of ‘reserve’ can be adopted (economic reserves, ultimate reserves, or also reserves in the economy (‘urban mining’)).

5 Options for combining indicators for individual resource categories to a single index

5.1 Introduction

In section 2.4 we narrowed the definition of resources down to raw materials that we extract from the environment in order to use them in the economic processing and final use while excluding emissions and related impacts on human and ecosystem health from either the extraction, processing, use or waste management of these resources. However, we still are dealing with a quite divergent number of resource types (metals, non-metallic minerals, fossil fuels, biomass, natural water, and land).. For the purpose of one overall resource efficiency index we need to further aggregate these results. For that, implicit or explicit weighting can be applied. Options for this will briefly be discussed below, followed by a brief reflection on the state-of-the-art in weighting of resource indicator results.

Huppes et al. (2012) drafted a survey of operational weighting methods applicable in LCA (consumption based) and in larger systems like countries (production based) (Huppes et al., 2012; Huppes & van Oers, 2011). The methods inventoried differed in terms of modelling (midpoint or endpoint), with respect to the type of values and preferences involved (collective or individual preferences) in the weighting, and in the basis for measuring them (revealed or stated preferences) The weighting factors are based on values and preferences including monetary values, political standards, or expert panel's judgments (see annex 4).

The types of methods also can be classified as follows:
1) One-issue methods, so no weighting
2) Implicit weighting
3) Explicit weighting.

5.2 One-issue methods, no weighting

As to the methods classified under "single item methods," these technically are no weighting methods because they refer only to one impact category. One might interpret them as weighting methods placing a fully dominant weight at the issue chosen, like the carbon footprint leaving out all non-carbon impacts, the ecological footprint method that transforms all impacts in essence to land use, or the externality school of thought that transforms all impacts to damage costs.

5.3 Implicit weighting

The methods classified under "collective revealed preferences" (see annex 4) are not based on the evaluation of different expected impacts and thus lack an explicit weighting step across impact categories. These methods use distance-to-target for weight factors, or effectively also other measures, like the (prevention) cost to reach a certain reduction target for a specific impact or amount of emissions, to specify an overall score.

These methods touch on weighting but do not make that step explicitly or recognizably. Apart from methodology issues like targets that are not only derived from environmental but also include economic, technical etc. considerations, they still lack the inter-impact factor, which indicates how important some impact is relative to another. All reduction targets for specific impacts or emissions are treated as of equal importance, without any true weighting.

Also in EW-MFA (Economy Wide Material Flow Analysis) implicit weighting has been adopted in e.g. the DMC approach, where all materials are added on a mass basis (kg) and no explicit weights between the different types of materials are defined.

5.4 Explicit weighting

This leaves the methods based on stated preferences (See Annex 2) as explicit weighting method, be they individual utility-based or having some collective preference or value element.

Explicit weighting is known from the field of multi-criteria analysis and has particularly been developed and applied within LCA. Weighting is an optional step of LCIA in which the (normalised) indicator results for each impact category assessed are assigned numerical weighting factors according to their relative importance, multiplied by these factors and possibly aggregated. Before weighting can be performed, the various indicator results must first be converted into the same units. One possible method for this is normalisation.

If a midpoint modelling approach of impact indicators is adopted weighting needs to be separately added to the impact modelling; if an endpoint modelling approach is adopted, and the impact assessment is restricted to the area of protection of 'resources', the aggregation of the mineral use, water use and land use is part of the endpoint modelling and the valuation technically has become a one-issue method.

5.4.1 Midpoint-methods

When adopting midpoint modelling of the resource related impact categories, there are three panel-based weighting factor sets available from 1) US EPA Science Advisory Board (Lippiat, 2007); 2) BEES Stake-holder Panel (Lippiat, 2007); 3) NOGEPA (Huppes et al., 2007). Originally, weighting factors are provided for 9 to 13 different midpoint impact categories, depending on the method (see Annex 2). Weighting factors are thus provided for climate change and acidification, but also for depletion of abiotic resources (fossil fuels and minerals), water and land use. These midpoint impact categories to a large extent comply with the midpoint impact categories as adopted in the ILCD Handbook. There are however two restrictions:

- the NOGEPA panel didn't derive weighting factors for the impact categories 'resource depletion', 'water depletion' and 'land use'.
- the weighting factor of EPA and BEES for resource depletion originally is narrowed down to fossil fuels.

If we simply expand the EPA and BEES weighting factors for fossil fuels to include also minerals, we can extrapolate from the original weighting sets of these two methods two weighting sets for impacts of resources in a narrow sense. A third set is added representing an average of the EPA and BEES sets:

	EPA Science Advisory Board (Lippiat, 2007)	**BEES Stake-holder Panel (Lippiat, 2007)**	**average**
Resource depletion (minerals and fossils)	21	42	31
Resource depletion water	13	33	23
Land use	67	25	46

5.4.2 Endpoint methods

In endpoint methods the weighting step is actually part of the endpoint modelling itself. For example, climate change as midpoint point indicator is modelled with radiative forcing as indicator expressed in the GWP; climate change as endpoint indicator is modelled from the radiative forcing on to GWP on to damage to human health and damage to ecosystem health. A third damage category in endpoint approaches is ‘damage to resource availability’ merging all abiotic, biotic, water and land ‘availability’ or ‘depletion’ impacts. Thus, at an end-point level the mid-point impacts for 'abiotic resource depletion', 'water depletion' and 'land use' are further modelled into effects on the end-point level. These effects are all part of one end-point ‘damage to resource availability’. So, at the end-point level no additional weighting is required anymore.

However, existing endpoint models including EPS2000, ReCiPe, Eco-Indicator 99 and IMPACT2000+ only cover the ‘abiotic depletion’ part (fossil fuels, minerals) and are based on quite new and immature endpoint modelling (EC , 2011). Moreover, there are no default endpoint models for ‘water depletion’ and ‘land use’ (in terms of occupation).

5.5 Reflection

We advise to always apply explicit weighting because implicit weighting basically assumes equal weights for the different impact categories, which is still a weighting. Because weighting has a value-based normative nature, it is of utmost importance to be fully transparent on the process towards weighting factors. This hence suggests that the routes discussed in Section 5.2 and 5.4 should be pursued. Assuming that underlying data and methods for e.g. water, land and material indicators are available, the data needs for the weighting step are relatively modest. The explicit weighting as discussed in section 5.4 needs only some kind of panel procedure, which seems relatively straightforward to organize, or otherwise even can rely on earlier weights suggested by panel procedures done earlier (Huppes and van Oers, 2011). The one-issue weighting methods discussed in section 5.2 in many cases are relatively uncomplicated (e.g. cumulative energy demand, carbon footprint), or can be based on well-elaborated methods (e.g. as is the case for the ecological footprint). A problem may occur however if the method or related data for translation of certain impacts into the ‘one issue’ chosen is not fully available yet. This could for instance be the case if one would chose damage costs as the ‘one issue’: to our knowledge there is still limited availability of the damage costs for resource- and water extraction, particularly if local impacts around the globe have to be considered.

From the explicit weighting methods discussed in section 5.4, the only ones that provide practical sets of weighting factors are the EPA Science Advisory Board and the BEES Stake-holder Panel (Lippiat, 2007). On top of that, an average could be made out of these two sets. However, again realizing that weighting is highly value-based and leads to weighting factors on which there is no general consensus at any societal level, it may sensible to derive a new set of specific weighting factors for 'abiotic resource depletion', 'water depletion' and 'land use' based on expert judgments from the resource experts, for example representatives from the UNEP International Resource Panel.

6 Conclusions, key discussion points and recommendations

From the analysis in this discussion paper we can already see that the conclusion inevitably will be problematic: there are many ways to define resource categories, there are many ways to aggregate resources within categories, and there are many ways to aggregate (or weight) between resource categories. There is hence most probably no value-free or fully science-based approach to create an aggregated resource-use or resource-efficiency index of nations.

Chapter 2 already indicated some of the key decisions that had to be made with regard to the question if one just would like to account for resource use, or also would like to use impacts of resource use as a factor to be taken into account in the index. We there discerned the following levels

1. Level 1: impacts of resources used are not accounted for. This leads to for instance a simple addition of water extraction in m3, without taking into account that water extracted in river basins with water scarcity is worse as water extractions from river basins with abundant water.
2. Level 2: impacts of resources are included, as far as this has to do with extraction and scarcity. In the example of water above, one would assess water extraction from river basins where water scarcity exists as less desirable.
3. Level 3: impacts of resources are included, including all emissions over the full life cycle of extraction, transformation, use and waste management. An example of this is the Environmentally Weighted Material Consumption (EMC) indicator, that adds up the use of materials as a function of volume and life cycle impact.

We chose in this discussion paper for level 2, since level 1 clearly leads to a too simplistic approach, while level 3 makes things extremely complicated. Resources are used in numerous products, that often create impacts which only partially can be attributed to the resources used in that product, leading to all kinds of allocation discussions. Since resources end up in virtually all products used in the global economy, one would end up with the need of some kind of global impact assessment. To keep things manageable, we here opt for just taking impacts of resource extraction into account[17].

Having chosen this position, the former sections do indicate some points on which a reasonable agreement seems to exist, and also identifies some clear points for discussion. When building a resource-use or resource-efficiency index of nations, the following elements play a role:

a) Making an inventory of relevant environmental and economic data (compare chapter 2)
b) Defining an economic reference indicator (compare chapter 3)
c) Defining resource categories to be included, and aggregation methods within resource categories (compare chapter 4)
d) Defining methods to aggregate across resources/weighting (compare chapter 5)

[17] To be self-critical, we would like to make the following remark. The popular booklet 'The global resource footprint of nations' (Tukker et al., 20014) used for water, materials and land indicators that simply aggregated cubic meters of blue water, tons of materials, and km2 of land occupied. It did not take into account local water scarcity, scarcity of materials, or land productivity. This discussion paper suggests taking such factors into account is essential.

We summarize the discussions and choices to be made below. Overall we can conclude the following

a) Data inventory methods: **reasonable to high consensus**. MR EE IO and co-efficient methods are now quite widely used to make inventories of territorial and consumption based accounts for water, materials, land use and carbon. There are significant uncertainties, but it seems that the scientific community is converging on methodologies here, and that it is mainly a matter of getting more accurate and detailed data as that major conceptual discussions remain. Also the data sources used for resource extraction and land use are converging, with FAOSTAT highly relevant for water use, land use and biotic material use, the IEA database highly relevant for fossil fuel extraction, and databases like USGS highly relevant for other resource extractions. Initiatives of a.o. the UNEP International Resources Panel and the Water Footprint Network are helping to consolidate such data sets. But particularly when one wants to calculate consumption-based indicators, the time investment for gathering data on linking extraction to consumption (economic and trade relations) will be very high compared to defining and calculating the indictors as such.. For pragmatic reasons, it is suggested that WRF for now will use one of the existing, major so-called MR EE IO research databases in which economic and resource extraction data are stored at global level.

b) Economic reference indicator: **reasonable to high consensus**. GDP, with all its drawbacks, is the default economic reference indicator to be used. From the Beyound GDP debate fairly well accepted indicators arose like the Human Development Index and the number of Happy Life Years.

c) Aggregation methods within resource categories:

   a. Carbon: **high consensus** on the aggregation method (GWP); but **low consensus** if a carbon footprint should be included in a resource use or resource efficiency index of nations, being an impact indicator (related to level 3 discussed above) rather than a resource use indicator.

   b. Water: **moderate consensus**. The Water extraction index and its elaboration by Pfister et al. (2009), the Water stress index, is a watershed specific measure for the use of water from river and aquifers and seems the best candidate for a resource use index for water (related to level 2 discussed above). An important discussion is however if also for the use of rain water should be accounted for, and if yes, how this then should be operationalized. Simply adding up water use in m3 irrespective of the region where it is used, as done by the Water footprint network, is not ideal since it neglects region-specific scarcity issues..

   c. Land: **moderate consensus**. Accounting for land use occupation in m2 is hardly controversial (level 1 mentioned above), but if one wants to include some form of impacts of land use occupation things get different. First, there is a difference between land use occupation and land use change. Further, there are approaches that take into account the productivity of the land occupied (e.g. the ecological footprint method), but also approaches that assess according to the impact on e.g. biodiversity.

   d. Materials: **low consensus**. Like for land use, adding up materials by weight (level 1 above) as is commonplace in the MFA community is a rather simple exercise. But it has severe drawbacks, since neither scarcity nor the impacts related to materials is taken into account. Indicators that include the impacts of emissions related to resource use (level 3 above) are less desirable in a framework focusing on resource use mainly (compare the discussion on carbon). In the Life cycle impact assessment community different assessment methods for materials based on scarcity indicators (level 2 above) are relatively mature. But since they are based on quite different principles one must conclude there is overall still low consensus.

d) Weighting/aggregating across resource categories: **low consensus**. As discussed in chapter 5 some panel weighting methods have been developed, that seem to be in line reasonably with each other when it comes down to weight water depletion, resource depletion and land use. Yet, this apparent consensus is mainly due to the fact that few studies on weighting have been tried, and that there are hence much more diverse views as apparent from this discussion paper. For instance, in the MFA community currently a lively discussion on a ‘safe operating space’ for resource use has started, that in turn could be used for weighting purposes as well.

Given the state of the art above, a resource use or resource efficiency index for nations hence by definition will be subjective. Indeed, even indicators for individual resources like land, water and materials will be subjective. A pragmatic way forward could be:

a) Simply to get some visual impression how a resource use or resource efficiency indicator of Nations could look like, use the (simple, as called above ‘level 1’) water-, materials, and land indicators per country as calculated by Tukker et al. (2014) as a basis and combine them with the weights discussed in chapter 5.
b) In a more advanced approach, which needs additional resources, use
    a. For water: use the Water stress index of Pfister et al. (2009)
    b. For land: use land occupation corrected for land use productivity (e.g. applying the HANPP method or elements of the ecological footprint method)
    c. For materials: use one of the scarcity based assessment methods, one that has currently a good endorsement in the LCIA community
    d. Weight these 3 indicators then again with the panel-based weights reviewed in chapter 5, or do some additional investment in developing specific weighting factors via a dedicated WRF-related panel.

7 References

Ahlroth, S. and Finnveden, G. (2009): Ecovalue08 – a new valuation method for environmental systems analysis tools. In Ahlroth, S. (2009): Valuation of environmental impacts and its use in environmental systems analysis tools. Doctoral thesis. KTH, Stockholm.

Anonymous (1992a) Life-Cycle Assessment. Proceedings of SETAC-Europe workshop on Environmental Life Cycle Assessment of Products, December 2-3 1991 in Leiden. SETAC-Europe, Brussels

Ayres, R. U., K. Martinàs & L.W. Ayres, 1996. Eco-thermodynamics. Exergy and life cycle analysis. Working Paper (96/04/EPS), INSEAD, Fontainebleau, France.

Ayres, R.U., 1998. Eco-thermodynamics: economics and the second law. Ecol. Econ. 26 (2): 189-209.

Bare JC (2011) TRACI 2.0: the tool for the reduction and assessment of chemical and other environmental impacts 2.0. Clean Techn Environ Policy 13:687–696

Bruyn, S. de, M. Korteland, A. Markowska, M. Davidson, F. de Jong, M. Bles, M. Sevenster (2010) Shadow Prices Handbook. Valuation and weighting of emissions and environmental impacts. Delft: CE http://www.ce.nl/publicatie/handboek_schaduwprijzen_%3A_waardering_en_weging_van_emissies_en_milieueffecten/1027

Blonk, T.J., M.D. Davidson & M.C.C. Lafleur, 1997a. Working document: Feasibility of operationalisation of depletion of abiotic resources in LCA via the key resources energy and land. IVAM Environmental Research, Amsterdam.

Costanza, R., Erickson, J.D., Fligger, K., Adams, A., Adams, C., Altschuler, B., Balter, S., Fisher, B., Hike, J., Kelly, J., Kerr, T., McCauley, M., Montone, K., Rauch, M., Schmiedeskamp, K., Saxton, D., Sparacino, L., Tusinski, W. and L. Williams. 2004. "Estimates of the Genuine Progress Indicator (GPI) for Vermont, Chittenden County, and Burlington, from 1950 to 2000." Ecological Economics 51: 139-155.

Cramer Commission (2007). The greenhouse gas calculation methodology for biomass-based electricity, heat and fuels. Project group Sustainable Biomass. The Netherlands. Report from Working Group. CO2 Methodology. January 2007. Pp. 39.

Croezen, H., G. Bergsma, A. Clemens, M. Sevenster and B. Tulleners (2011). Biodiversity and Land Use. A Search for Suitable Indicators for Policy Use. CE, Delft, Netherlands

Daly,H. and Cobb, J., 1989. For the Common Good. Beacon Press, Boston

EC (2011). Roadmap to a Resource Efficient Europe. Communication of the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. COM / 2011 / 571. EC, Brussels, Belgium

EC (2014). Towards a circular economy: A zero waste programme for Europe. Communication of the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. COM / 2014 / 0398 final. EC, Brussels, Belgium

EEA (2003). Indicator Fact Sheet (WQ01c) Water exploitation index. European Environment Agency, Copenhagen, Denmark, available from: http://www.eea.europa.eu/data-and-maps/indicators/water-exploitation-index (accessed 12 October 2014).

Ekvall, T., T. Rydberg, Ö. Hedenberg, B. Backlund Jacobson, T. Pajula & H. Wessman, 1997. Guidelines on life cycle Impact assessment of pulp and paper. NORDPAP DP2/55, SCAN Forskrapport 688.

European Commision (EC, 2010). ILCD Handbook: General guide for Life Cycle Assessment - Provisions and action steps. 1st ed. European Commission, Joint Research Centre, Institute for Environment and Sustainability. http://lct.jrc.ec.europa.eu/pdf-directory/ILCD-Handbook-General-guide-for-LCA-PROVISIONSonline-12March2010.pdf.

European Commission (EC, 2011). International Reference Life Cycle Data System (ILCD) Handbook- Recommendations for Life Cycle Impact Assessment in the European context. Joint Research Centre - Institute for Environment and Sustainability. First edition November 2011. EUR 24571 EN. Luxemburg. Publications Office of the European Union.

European Commission (EC, 2012). Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment methods. Ispra, Italy: European Commission, Joint Research Centre, Institute for Environment and Sustainability. http://lct.jrc.ec.europa.eu/assessment/assessment/projects

Ewing, B., Moore, D., Goldfinger, S., Oursler, A., Reed, and Wackernagel, M. (2010) Calculation Methodology for the National Footprint Accounts, 2010 Edition, Oakland, CA: Global Footprint Network

Finnveden, G., 1996a. “Resources” and related impact categories. Part III. In: Udo de Haes, H.A. (ed.), 1996.

Finnveden, G., 1999b. Methodological aspects of life cycle assessment of integrated solid waste management systems. Resour. Conserv. Recy. 26 (3-4): 173-187.

Finnveden, G., Eldh, P., Johansson, J., 2006. Weighting in LCA based on ecotaxes – development of a mid-point method and experiences from case studies. Int. J. LCA 11 (1), 81–88 (special issue).

Frischknecht R, Steiner R, Jungbluth N (2009)The Ecological Scarcity Method – Eco-Factors 2006. A method for impact assessment in LCA. Environmental studies no. 0906. Federal Office for the Environment, Bern. Available from: www.environment-switzerland.ch/uw-0906-e. Accessed 19 Dec 2012

Frischknecht, R., Steiner, R., Jungbluth, N. (2008). The Ecological Scarcity Method – Eco-Factors 2006. A method for impact assessment in LCA. Environmental studies no. 0906. Federal Office for the Environment (FOEN), Bern: 188 pp.

Gerbens-Leenes W, Hoekstra AY, Van der Meer TH (2009) The water footprint of bioenergy. Proc Natl Acad Sci USA 106:10219–10223.

Giljum Stefan Giljum, Leisa Burrell, Katharina Giesecke, Stephan Lutter, Sebastiaan Deetman, René Kleijn, Nina Eisenmenger, Michaela Theurl, Sylvia Gierlinger (2013). Deliverable 3.2 Policy Analysis. Deliverable 3.2 from the EU FP7 project Development of a System of Indicators for a Resource Efficient Europe (DESIRE), available from: http://fp7desire.eu/index.php/documents2. Accessed 5 December 2014.

Goedkoop M, Heijungs R, Huijbregts M, De Schryver A, Struijs J, van Zelm R (2012) ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level . First edition (revised). Report I: Characterisation. 6 January 2009, http://www.lcia-recipe.net/. Accessed July 2012

Goedkoop M, Spriensma R (2000) The Eco-indicator 99: a damage oriented method for life cycle assessment, Methodology Report, 2nd edn. Pré Consultants, Amersfoort (NL), Netherlands

Goedkoop, M., 1995. The Eco-indicator 95. NOH report 9523. PRé Consultants, Amersfoort.

Goedkoop, M., R. Heijungs, M. Huijbregts, A. de Schryver, J. Struijs, R. van Zelm (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. Ministry of Housing, Spatial Planning and Environment, den Haag, Netherlands

Goedkoop. M, R Heijungs, M. Huijbregts, A. de Schryver, J. Struijs, R. van Zelm (2009/2013). ReCiPe 2008 A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Ministry of Public Housing, Spatial Planning and Environment, den Haag, Netherlands (2nd edition, 2013)

Graedel, T. and E. van der Voet (eds., 2010). Linkages of sustainability. The MIT Press. Cambridge, Massachusetts, US and London, England

Griggs, David, Mark Stafford-Smith, Owen Gaffney, Johan Rockström, Marcus C. Öhman Priya Shyamsundar, Will Steffen, Gisbert Glaser, Norichika Kanie, Ian Noble (2013). Sustainable development goals for people and planet. Nature 495, 305–307 doi:10.1038/495305a

Guinée JB (1995) Development of a methodology for the environmental life-cycle assessment of products; with a case study on margarines. Thesis, Leiden University

Guinée JB (2014). Selection of impact categories and classification of LCI results to impact categories. In: LCA Compendium - The Complete World of Life Cycle Assessment, series eds Klöpffer W, Curran MA. Life Cycle Impact Assessment, eds Hauschild M, Huijbregts M., Springer Dordrecht.

Guinée JB (2014). Selection of impact categories and classification of LCI results to impact categories. In: LCA Compendium - The Complete World of Life Cycle Assessment, series eds Klöpffer W, Curran MA. Life Cycle Impact Assessment, eds Hauschild M, Huijbregts M., Springer Dordrecht.

Guinée JB (Ed), Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, van Oers L, Wegener Sleeswijk A, Suh S, Udo de Haes HA, de Bruijn JA, van Duin R, Huijbregts MAJ (2002) Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Series: Eco-Efficiency in Industry and Science, Vol. 7. Springer, Dordrecht

Guinée, J. & R. Heijungs, 1995. A proposal for the definition of resource equivalency factors for use in product Life-Cycle Assessment. Environ. Toxicol. Chem. 14 (5): 917-925.

Guinée, J.B.; Gorrée, M.; Heijungs, R.; Huppes, G.; Kleijn, R.; Koning, A. de; Oers, L. van; Wegener Sleeswijk, A.; Suh, S.; Udo de Haes, H.A.; Bruijn, H. de; Duin, R. van; Huijbregts, M.A.J. (2002). Handbook on life cycle assessment. Operational guide to the ISO standards. I: LCA in perspective. IIa: Guide. IIb: Operational annex. III: Scientific background. Kluwer Academic Publishers, ISBN 1-4020-0228-9, Dordrecht, 692 pp.

Hauschild M, Potting J (2005) Spatial differentiation in life cycle impact assessment – the EDIP2003 methodology. Environmental News no. 80. The Danish Ministry of the Environment, Environmental Protection Agency, Copenhagen

Hauschild M, Wenzel H (1998) Environmental assessment of products - Volume 2: Scientific background. Chapman & Hall, United Kingdom, 1997, Kluwer Academic Publishers, Hingham, MA. USA. ISBN 0-412-80810-2

Hauschild MZ, Goedkoop M, Guinée J, Heijungs R, Huijbregts M, Jolliet O, Margni M, De Schryver A, Hum-bert S, Laurent A, Sala S, Pant R (2013) Identifying best existing practice for characterization modelling in life cycle impact assessment. Int J Life Cycle Assess 18:683-697

Heijungs R, Guinée JB, Huppes G, Lankreijer RM, Udo de Haes HA, Wegener Sleeswijk A, Ansems AAM, Eggels PG, van DuinR, de Goede HP(1992) Environmental life cycle assessment of products. Guide & Backgrounds – October 1992. Centre of Environmental Science, Leiden University, Leiden

Heijungs, R. , 2008. The weighting step in life cycle impact assessment. Three explorations at the midpoint and endpoint level. Weighting with damage costs. CML, Leiden University, Netherlands.

Helliwell, John, Richard Layard and Jeffrey Sachs (2013). World Happiness Report 2013. Sustainable Development Solutions Network, Paris/New York/New Delhi, see unsdsn.org

Hicks, J.R. (1939, 2nd ed. 1946). Value and Capital: An Inquiry into Some Fundamental Principles of Economic Theory. Oxford: Clarendon Press.

Hoekstra, A.Y. and M.M. Mekonnen (2012). The water footprint of humanity. PNAS, vol. 109, no. 93, p. 232–3237

Huijbregts, M.A.J., Rombouts, L.J.A., Hellweg, S., Frischknecht, R., Dik van de Meent, J.H., Ragas, A.M.J., Reijnders, L., Struijs, J., 2006. Is cumulative fossil energy demand a useful indicator for the environmental performance of products? Environ. Sci. Technol. 40, 641–648.

Huijbregts M.A.J., Steinmann Z.J.N., Stam G., Van Zelm R. (2014). Update of emission-related impact categories in ReCiPe2008. Department of Environmental Science, Radboud University Nijmegen.

Huppes, G. & L. van Oers (2011) Background Review of Existing Weighting Approaches in Life Cycle Impact Assessment (LCIA). JRC Scientific and Technical Reports, Luxembourg: Publications Office of the European Union, ISBN 978-92-79-21751-7, ISSN 1831-9424. | online available at JRC |

Huppes, G. & L. van Oers (2011) Evaluation of Weighting Methods for Measuring the EU-27 Overall Environmental Impact. JRC Scientific and Technical Reports, Luxembourg: Publications Office of the European Union. ISBN 978-92-79-21643-5, ISSN 1831-9424. | online available at JRC |

Huppes, G., L. van Oers, U. Pretato & D.W. Pennington (2012) Weighting environmental effects: Analytic survey with operational evaluation methods and a meta-method. The International Journal of Life Cycle Assessment17(7):876-891.

Huppes, G., M.D. Davidson, J. Kuyper, L. van Oers, H.A. Udo de Haes and G. Warringa. 2007. Eco-efficient environmental policy in oil and gas production in The Netherlands. Ecological Economics 61(1): 43-51.

Huppes, Gjalt and Ishikawa, Masanobu (2005), A Framework for Quantified Eco-Efficiency Analysis. Journal of Industrial Ecology, Vol. 9, Issue 4, pp. 43-46

IPCC, 2013: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 1535 pp.

Itsubo, N., M., Sakagami, T. Washida, K. Kokubu & A. Inaba, 2004. Weighting Across Safeguard Subjects for LCIA through the Application of Conjoint Analysis. Int J LCA 9 (3) 196-205.

Itsubo N, Inaba A (2012) LIME2, Life-cycle Impact assessment Method based on Endpoint modelling: summary. Available from: http://lca-forum.org/english/pdf/No12_Summary.pdf. Accessed 20 Dec 2012

Jolliet O, Frischknecht R, Bare J, Boulay AM, Bulle C, Fantke P, Gheewala S, Hauschild M, Itsubo N, Margni M, McKone TE, Mila i Canals L, Postuma L, Prado-Lopez V, Ridoutt B, Sonnemann G, Rosenbaum RK, Seager T, Struijs J, van Zelm R, Vigon B, Weisbrod A (2014) Global guidance on environmental life cycle impact assessment indicators: Findings of the scoping phase. Int J Life Cycle Assess 19:962-967

Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G, Rosenbaum R (2003b) IMPACT 2002+: A New Life Cycle Impact Assessment Methodology. Int J Life Cycle Assess 8:324- 330

Kahneman, D., and A. Deaton. 2014. “High Income Improves Evaluation of Life But Not Emotional well-being. Psychological and Cognitive Sciences.” Proceedings of National Academy of Sciences 107(38) 16489–16493.

Koellner, T., Scholz, R.W., 2007. Assessment of land use impacts on the natural environment. Part I: An analytical framework for pure land occupation and land use change. Int.J.LCA 12 (1) 16-23.

Lenzen, M., Moran, D., Kanemoto, K., Geschke, A. (2013) Building Eora: A Global Multi-regional Input-Output Database at High Country and Sector Reso lution, Economic Systems Research, 25:1, 20-49, DOI:10.1080/09535314.2013.769 938

Lindfors, L.G, K. Christiansen, L. Hoffman, Y. Virtanen, V. Juntilla, O.J. Hansen, A. Rønning , T. Ekvall & G. Finnveden, 1995c. LCA-NORDIC Technical Reports No 10. Nordic Council of Ministers, Copenhagen.

Lindfors, L.G., 1996. A Desk study on characterisation methods applicable in EU Ecolabeling programmes. In: Udo de Haes, H.A., R. Clift, R. Griesshammer, L. Grisel & A.A. Jensen, 1996. Practical guidelines for Life Cycle assessment for the EU ecolabelling programme. Leiden.

Lippiatt, B.C., 2007. BEES 4.0. Building for Environmental and Economic Sustainability Technical Manual and User Guide. National Institute of Standards and Technology (NIST), Gaithersburg, USA. http://www.bfrl.nist.gov/oae/software/bees/

Mattila T, Helin T, Antikainen R (2012) Land use indicators in life cycle assessment: A case study on beer pro-duction. Int J Life Cycle Assess 17:277-286

Mattila, T. T. Helin, R. Antikainen, S. Soimakallio, K. Pingoud and H. Wessman (2011). Land use in life cycle assessment. The Finnish Environment 24, Finnish Environment Institute, Helsinki, Finland

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007a. Key elements in a framework for land use impact assessment within LCA. Int J LCA 12 (1): 5-15.

Mudgal, S., A. Tan, S. Lockwood, N. Eisenmenger, M. Fischer-Kowalski, S. Giljum and M. Bruckner (2012). Assessment of resource efficiency indicators and targets. BIO Intelligence Service (BIO), Institute for Social Ecology (SEC) and Sustainable Europe Research Institute (SERI) for EU DG Environment, Brussels, Belgium

Müller-Wenk, R., 1998. Depletion of abiotic resources weighted on base of “virtual” impacts of lower grade deposits used in future. IWÖ- Diskussionsbeitrag nr. 57, St Gallen.

Nef (2012). The Happy Planet Index: 2012 Report. A global index of sustainable well-being. New economics foundation, London, UK

OECD (2013). How’s Life? 2013. Measuring Well-being. OECD, Paris, France

Oers, L. van, A. de Koning, J.B. Guinée & G. Huppes, 2002. Abiotic resource depletion in LCA - Improving characterisation factors for abiotic resource depletion as recommended in the new Dutch LCA Handbook. DWW report, Delft; see http://www.cml.leiden.edu/research/industrialecology/researchprojects/finished/abiotic

Oers, L. van, A. de Koning, J.B. Guinée & G. Huppes, 2002. Abiotic resource depletion in LCA - Improving characterisation factors for abiotic resource depletion as recommended in the new Dutch LCA Handbook. DWW report, Delft.

Pedersen, B. 1991. Hvad er et bæredygtigt ressourceforbrug? Tværfagligt center, Danmarks Tekniske Højskole, Lyngby.

Pfister S, Hellweg S (2009) The water ‘‘shoesize’’ vs. footprint of bioenergy. Proc Natl Acad Sci USA 106:E93-E94.

Pfister, S., M. Motoshita, B. G. Riddoutt (2014). Progress toward an LCA impact assessment model linking land use and malnutrition-related DALYs. Paper for the LCA Foods 2014 conference, October 2014, San Francisco (see: http://lcafood2014.org/)

Preiss P., and V. Klotz (2008) NEEDS New Energy Externalities Developments for Sustainability. Technical Paper no. 7.4 – RS 1b. “Description of updated and extended draft tools for the detailed site-dependent assessment of external costs”. Universität Stuttgart, Germany. http://www.needs-project.org/

Ridoutt, B.G and J. Huang (2012). Environmental relevance—the key to understanding water footprints PNAS, vol. 109 no. 22, E1423-1424

Schneider L, Berger M, Finkbeiner M (2011) The anthropogenic stock extended abiotic depletion potential (AADP) as a new parameterisation to model the depletion of abiotic resources. Int J Life Cycle Assess 16 (9):929-936

Schneider L, Berger M, Schüler-Hainsch E, Knöfel S, Ruhland K, Mosig J, Bach V, Finkbeiner M (2013) The economic resource scarcity potential (ESP) for evaluating resource use based on life cycle assessment. International Journal of Life Cycle Assessment 19 (3):601-610

Schoer, Karl, Jürgen Giegrich, Jan Kovanda, Christoph Lauwigi, Axel Liebich, Sarka Buyny, Josefine Matthias (2012). Conversions of European Product Flows into Raw Material Equivalents. Final report of the project: Assistance in the development and maintenance of Raw Material Equivalents conversion factors and calculation of RMC time series Ifeu, Heidelberg; Sustainable Solutions Germany – Consultants, Wiesbaden; Charles University Prague

Steen, B (1999a) A Systematic approach to environmental priority strategies in product development (EPS). Version 2000-general system characteristics; CPM report 1999:4, Chalmers University of Technology, Gothenburg

Steen, B (1999b). A Systematic approach to environmental priority strategies in product development (EPS). Version 2000-Models and data of the default method; CPM report 1999:5, Chalmers University of Technology, Gothenburg, Sweden

Steen, B., 1995. Valuation of environmental impacts from depletion of metal an fossil mineral reserves and from emission of CO2. AFR-Report, AFR (Swedish Waste Research Council), Stockholm.

Stern, N. (2007) The Economics of Climate Change. The Stern Review. Cambridge University Press, Cambridge

Toffoletto L, Bulle C, Godin J, Reid C, Deschênes L (2007). LUCAS – A New LCIA Method Used for a CAnadian-Specific Context.Int J Life Cycle Assess 12: 93-102

Tol, Richard SJ (2008) The Social Cost of Carbon: Trends, Outliers and Catastrophes. Economics: The Open-Access, Open-Assessment E-Journal, Vol. 2, 2008-25. http://www.economics-ejournal.org/economics/journalarticles/2008-25

Tukker, Arnold, Arjan de Koning , Richard Wood , Troy Hawkins , Stephan

Lutter , Jose Acosta , Jose M. Rueda Cantuche , Maaike Bouwmeester , Jan Oosterhaven , Thomas Drosdowski & Jeroen Kuenen (2013): EXIOPOL – Development and illustrative analysis of a detailed global MR EE SUT/IOT, Economic Systems Research, 25:1, 50-70

Tukker, Arnold, Tatyana Bulavskaya, Stefan Giljum, Arjan de Koning, Stephan Lutter, Moana Simas, Konstantin Stadler, Richard Wood (2014). The global resource footprint of nations. Carbon, water, land and materials embodied in trade and final consumption. Available from: http://creea.eu/index.php/7-project/8-creea-booklet (accessed 5 December 2014)

Tukker, Arnold & Erik Dietzenbacher, 2013. "Global Multiregional Input-Output Frameworks: An Introduction And Outlook," Economic Systems Research, Taylor & Francis Journals, vol. 25(1), pages 1-19

UNDP (2014). Human Development Report 2014. Sustaining Human Progress: Reducing Vulnerabilities and Building Resilience. United Nations Development Program, New York, US

United States Department of the Interior - Bureau of Mines, 1993. Mineral commodity summaries 1993. U.S. Government Printing Office, Washington DC

Voet, Ester van der, Lauran van Oers, Stephan Moll, Helmut Schütz, Stefan Bringezu, Sander de Bruyn, Maartje Sevenster, Geert Warringa (2005). Policy Review on Decoupling: Development of indicators to assess decoupling of economic development and environmental pressure in the EU-25 and AC-3 countries. CML report 166, CML, Leiden University, Leiden, Netherlands

Vogtländer, Joost , Bianca Baetens, Arianne Bijma, Eduard Brandjes, Erwin Lindeijer, Merel Segers, Flip Witte, J.C. Brezet, Ch.F. Hendriks+ LCA-based assessment of sustainability The Eco-costs/Value Ratio: EVR Delft: 2009 / xiv + 219 pp

Wackernagel, M. and W. Rees. 1996. Our Ecological Footprint: Reducing Human Impact on the Earth. Gabriola Island, BC: New Society Publishers. ISBN 0-86571-312-X.

Wackernagel, M. C. Monfreda, D. Moran, P. Wermer, S. Goldfinger, D. Deumling, M. Murray. 2005. National Footprint and Biocapacity Accounts 2005: The underlying calculation method. Global Footprint Network, Oakland, California, USA

Weidema, B.P., 2009. Using the budget constraint to monetarise impact assessment results. Ecol. Economics 68, 1591–1598.

Weitzman, Martin L., 1999. Pricing the limits to growth from minerals depletion. Quarterly Journal of Economics, MIT Press, vol. 114(2), pages 691-706

Wenzel, H, M.Z. Hauschild & L. Alting, 1997 Environmental assessment of products. Vol.1. 544 pp. Chapman & Hall, United Kingdom, Kluwer Academic Publishers, Hingham, MA. USA. ISBN 0412808005.

Annex 1: Review of midpoint and end-point impact categories

In this Annex we review the impact categories (both mid-point as end-point) as usually applied in Life cycle impact assessment. With regard to mid-point categories (Table A1.1) we can see that the diversity and overlap in impact categories among different methods is huge. Sometimes it is merely a matter of semantics and the differences are likely to be minor, sometimes differences are much more fundamental. For example, the way that EPS2000 (Steen 1999a,b) defines midpoint impact categories is fundamentally different from the way that Guinée et al. (2002), Hauschild and Potting (2005) and Bare (2011) have done. EPS2000 basically is an endpoint-approach and derives midpoint impact categories from endpoints. Guinée et al. (2002), Hauschild and Potting (2005) and Bare (2011) define midpoint impact categories basically from problem-oriented cross-media approach adopting environmental themes as defined in environmental policy (Anonymous 1992a).

Table A1.2 shows that the diversity in damage or endpoint impact categories among different methods is significant but not as huge as for midpoint impact categories. The differences are actually dominated by the approach taken in LIME. LIME choose to use the same names for midpoint and endpoint impact categories elaborating different indicators for each of them, which results in two similar lists of LIME impact categories in Table A1.1 and Table A1.2.

As part of Life Cycle Impact Assessment (LCIA) methods have been developed for assessing the depletion of abiotic resources over the past decades. These methods predominantly focus on the depletion of abiotic resources, i.e. natural resources (including energy resources) regarded as non-living, e.g. zinc ore, crude oil, wind energy (Guinée et al. 2002). This focus is thus different from the focus on materials above. .Abiotic resource depletion is one of the most frequently discussed impact categories and there is consequently a wide variety of methods available for characterising contributions to this category. To a large extent these different methodologies reflect differences in problem definition. For this reason, there is also no generally accepted method to assess the depletion of abiotic resources.

Reviews of existing Impact assessment methods for the depletion of abiotic resources are provided in several publications. Broadly speaking, the available methods fall into six groups, reviewed very briefly below (Guinée et al. 2002).

1. No assessment or aggregation (e.g. Lindfors, 1996)
2. Aggregation of natural resource extractions on a mass basis (e.g. Lindfors *et al.*, 1995c).
3. Aggregation and assessment based either on a) ‘ultimate reserves’, i.e. the quantity of resource (as a chemical element or compound) that is ultimately available, estimated by multiplying the average natural concentration of the resource in the primary extraction media (e.g. the earth’s crust) by the mass or volume of these media (e.g. the mass of the crust) (Guinée, 1995); or b) on ‘economic reserves’, i.e. that part of the reserve base[18] which can be economically extracted at the time of determination (United States Department of the Interior - Bureau of Mines, 1993) and/or current extraction rate. The method of Heijungs *et al.* (1992) is an example of assessment based on reserves. Alternative assessment methods proceed from resource extraction rates relative to reserves (see Guinée & Heijungs, 1995 and Ekvall *et al.*, 1997), from extraction rates only (Goedkoop, 1995) or from per capita reserves (see Hauschild & Wenzel, 1998); or C) on AADP (e.g. work of L. Schneider).

[18] The reserve base is that part of an identified resource that meets specified minimum physical and chemical criteria related to current mining practice (United States Department of the Interior - Bureau of Mines, 1993).

Table A1.1. Lists of midpoint impact categories (Guinée et al., 2014)

Midpoint categories	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**
1. Abiotic resource depletion					X			X			
2. Depletion of abiotic resources				X							X
3. Depletion of biotic resources				X							
4. Depletion of element reserves (element)						X					
5. Depletion of fossil reserves (Coal)						X					
6. Depletion of fossil reserves (Gas)						X					
7. Depletion of fossil reserves (Oil)						X					
8. Depletion of mineral reserves (ore)						X					
9. Mineral extraction	X										
10. Mineral resources consumption							X				
11. Gravel									X		
12. Energy resources									X		
13. Non-renewable energy	X										
14. Fossil fuel consumption							X				
15. Fossil fuel depletion										X	
16. Resource depletion, water											X
17. Resources									X		
18. Fish and meat production capacity						X					
19. Crop production capacity						X					
20. Wood production capacity						X					
21. Forest resources consumption							X				
22. Freshwater									X		
23. Land competition				X							
24. Land occupation	X										
25. Agricultural land occupation		X									
26. Natural land transformation		X									
27. Rural land occupation		X									
28. Land use							X	X	X		X
29. Impacts of land use				X							
30. Share of species extinction [NEX]						X					
31. Loss of biodiversity				X							
32. Loss of life support functions				X							

1=IMPACT2002+ (Jolliet et al. 2003b); 2= ReCiPe (Goedkoop et al. 2012); 3= EI99 (Goedkoop and Spriensma1999) ; 4= CML 2002 (Guinée et al. 2002); 5= EDIP 2003 (Hauschild and Potting 2005); 6= EPS 2000 (Steen 1999a,b); 7=LIME2 (Itsubo and Inaba 2012); 8=LUCAS (Toffoletto et al. 2007); 9=Swiss Ecoscarcity 2006 (Frischknecht et al. 2009); 10=TRACI 2.0 (Bare 2011); 11=ILCD (EC-JRC 2011)

Table A2.12: Default lists of damage or endpoint impact categories (Source: Guinée 2014).

Damage or endpoint categories	1	3	6	7
1. Resources	X	X	X	
2. Biodiversity			X	
3. Land use				X
4. Mineral resources consumption				X
5. Fossil fuel consumption				X
6. Forest resources consumption				X

1=IMPACT2002+ (Jolliet et al. 2003b); 3= EI99 (Goedkoop and Spriensma 1999); 6= EPS 2000 (Steen 1999a,b); 7=LIME2 (Itsubo and Inaba 2012)

4. Aggregation and assessment based on the cost of ‘restoring’ the resource to its original, natural state, or on the costs associated with substituting current extraction processes by presumed ‘sustainable’ processes. Pedersen (1991) and Steen (1995) describe such methods.
5. Aggregation and assessment based on energy content or exergy content or consumption (e.g. Finnveden, 1996b; see also Ayres *et al.*, 1996 and Ayres, 1998). Exergy is the amount of energy that can be obtained when matter is brought reversibly into equilibrium with its surroundings. It is the fraction of the energy content that can be used for work (= available energy). The exergy consumed is the exergy of the resources extracted as input minus the exergy of the outputs (Finnveden, 1996b). Finnveden suggests that the potential exergy of an ore might be used as a measure for depletion of abiotic resources in LCA.
6. Aggregation and assessment based on the change in the anticipated environmental impact of the resource extraction process due to lower-grade deposits having to be mined in the future. This method is described by Blonk *et al.* (1997a) and Müller-Wenk (1998) among others. It has been operationalised for metal ores and energy resources. In the case of metal ores, the virtual additional energy that will be required for future extraction processes is estimated. This additional energy, and the energy content of energy resources, are converted into the weighting indices of the Eco-indicator 95 approach by performing an LCA for the transformation of 1 kg ‘heavy-grade’ oil into thermal energy. Within the Eco-indicator 99 methodology a method has been developed based on that of Müller-Wenk (1998) (Goedkoop & Spriensma, 1999).

There is as yet no consensus about what constitutes the best category indicator for ‘abiotic depletion’, the choice depending crucially on the definition of the problem. Although in most publications the problem of abiotic depletion is not precisely defined, four groups of definitions can be broadly distinguished, based on what is seen as the key problem:

A. the decrease of the resource itself – method groups 2 and 3
B. the decreasing reserves of useful energy/exergy in the world – method groups 5 and 7
C. the contribution of current extraction processes, or possible ‘restoration’ of the resource, to other impact categories (in the first case, this means that resource depletion is not in fact regarded as a separate environmental problem at all, as the environmental impacts of extraction processes are already included in LCAs (*cf.* Finnveden, 1996a) – method groups 1 and 4
D. the change in the environmental impact of extraction processes at some point in the future (e.g. as a result of having to extract lower-grade ores or recover materials from scrap) – method groups 4 and 6.

We will not further determine here what would be the preferred method as that is not the scope of this discussion paper; for this we refer to for example Guinée et al. (2002) or ILCD (2010). Within each option there are more choices to make. If the decrease of the resource itself is taken as the key problem (option A), and assessment based on reserves and/or current extraction rates is preferred (method group 3), there is still room for debate on several points such as on which of the many types of reserves do we consider: economic reserves, ultimate reserves in primary media (ore, fossil fuels) or also reserves in the economy (in products, scrap)?

For the assessment of abiotic resource depletion on the midpoint level the CML method (Oers et al., 2002) is recommended in the ILCD framework (EC, 2012). The method captures scarcity by including extraction as well as reserves of a given resource. Characterization factors are given for metals, fossil fuels and, in the case of reserve base and economic reserves, mineral compounds. In addition, the method covers most of the substances/materials identified as critical by the European Commission's Ad-hoc Working Group on defining critical raw materials (European Commission 2010).

Data on reserves and production are taken from the US Geological Survey (http://minerals.usgs.gov/minerals/pubs/mcs/).

Oers et al. (2002) give characterization factors for economic reserves, reserve base, and ultimate reserves. The characterization factors given for the *reserve base* are recommended, as this reflects a longer time horizon and the possibility of improvement in mining technology, making feasible the exploitation of previously sub-economic deposits. The reserve base includes deposits which meet certain minimal chemical and physical requirements to potentially become economically exploitable within planning horizons.

In an updated version of Oers et al. (2002) the impact category "abiotic resource depletion" is split up into two for "elements" and "fossil energy". Fossil fuels are assumed to be full substitutes.

At the endpoint level, all methods evaluated are too immature to be recommended.
However, the ReCiPe method (Goedkoop et al., 2009/2013) may be used as an interim solution.

Annex 2: Weighting factors of three panel panel-based weighting sets

ILCD draft recommended midpoint impact categories	Impact category on midpoint level	Panel weighting set		
		EPA Science Advisory Board (Lippiat, 2007)	BEES Stake-holder Panel (Lippiat, 2007)	NOGEPA (Huppes et al., 2007)
		%	%	%
Climate change	Climate change	16	29	32
Ozone depletion	Ozone depletion	5	2	5
Acidification	Acidification	5	3	6
	Eutrophication	5	6	13
Eutrophication, terrestrial				
Eutrophication, fresh water				
Eutrophication, marine				
Photochemical ozone formation	Photochemical ozone formation	6	4	8
	Human health ((non)cancerous)	11		16
Human toxicity cancerous	Human health cancerous		8	
Human toxicity non-cancerous	Human health non-cancerous		5	
Particulate matter/Respiratory inorganics	Human health criteria pollutants	6	9	
	Ecotoxicity	11	7	
Fresh water ecotoxicity	Fresh water ecotoxicity			6
	Marine ecotoxicity			8
	Terrestrial ecotoxicity			5
Ionizing radiation, human health				
Ionizing radiation, ecosystems				
Resource depletion (minerals fossils and renewables)				
	resource depletion (fossil fuel)	5	10	
Resource depletion water	Water intake	3	8	
	Indoor air quality	11	3	
Land use	Habitat alteration	16	6	
	Total	100	100	100

Annex 3: Taxonomy and classification of weighting methods

Figure A3.1: Taxonomy of weighting methods

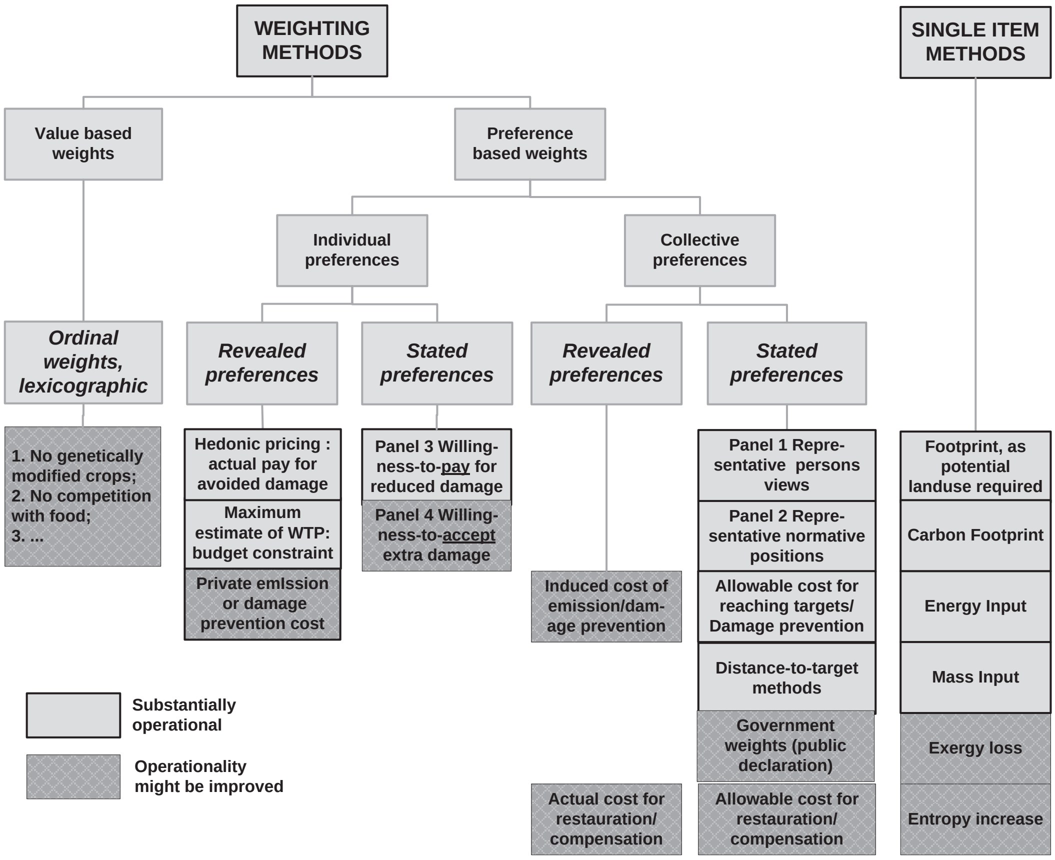


Table A3.1: Classification of quantified operational weighting methods

Modelling level	**Weighting type**				
	Collective stated preferences	**Collective revealed preferences**		**Individual stated preferences**	**Single item methods**
	Non-monetised	**Non-monetised**	**Monetised**	**Monetised**	
	Panel 1	**DTT**	**Damage prevention cost**	**WTP Panel 3**	
Inter-ventions	Ecotax (98/02/06) (Finnveden et al, 2006)	Ecopoints (Frischknecht et al, 2008)			Ecological Footprint (Wackernagel et al, 2005)
Midpoint	BEES and EPA (Lippiatt, 2007)	EDIP (Wenzel et al., 1997, Hauschild and Wenzel, 1998)		Ecovalue08 (Ahlroth & Finnveden 2009)	Cumulative. Fossil Energy Demand, CED (Huijbregts et al, 2005)
	Nogepa (Huppes et al., 1995/2007)				
Endpoint			ReCiPe-shadow prices (de Bruyn et al., 2010)	ReCiPe-damage cost (Heijungs, 2008).	
	Ecoindicator 99 (Goedkoop & Spriensma, 2000)		Eco-costs/Value Ratio (Vogtländer et al, 2009)	LIME (Itsubo et al., 2004).	
				EPS (Steen, 1999)	
				Budget constrained WTP (Weidema, 2009)	
Integrated modelling and weighting				NEEDS (Preiss & Klotz, 2008)	Damage cost GHG emissions (Tol (2008) (Stern, 2007)
					Damage cost resource extractions (Weitzman, 1999)

Note: in the scope of the Resource Efficiency Index the methods mentioned as 'integrated modelling and weighting' are not relevant. The methods of Preiss et al (2008), Tol (2008) and Stern (2008) are restricted to the evaluation of emissions and do not include extraction of resources. Weitzman (1999) focuses on valuation of resources but the method is restricted to (a limited number of) mineral extractions and land use and water use are not included.

ISBN **978-3-906177-08-3**

Contact information
WRF Secretariat
Lerchenfeldstrasse 5, CH-9014 St. Gallen, Switzerland
Phone + 41 71 554 09 00
info@wrforum.org
www.wrforum.org
Visit us on Facebook, Twitter and Linkedin

WORLD RESOURCES FORUM