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Background Review of Existing Weighting Approaches in Life Cycle Impact Assessment (LCIA)

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Introduction

This report presents the first part of the work carried out towards the development of a scheme for weighting indicators across the impact categories (climate change, acidification, resource depletion, human cancer effects, and others) that are commonly considered in life cycle assessment. Weighting is essential to derive a single indicator of the overall environmental impact of the EU-27 and to build the resources indicators as set out in the Thematic Strategy.

Citation taken from the Administrative Arrangement

Task A2

A ***quantitative weighting scheme*** *across different impacts of emissions on the natural environment and human health will be developed in close coordination with DG Environment as this will include a value-based weighting step, i.e. should draw on the societal and economic perspectives. The weighting approach will be prepared by JRC with expert input from the scientific domain.* ***Various approaches will be compared*** *towards the proposal.*

According to ISO 14040 and 14044, weighting is an optional element in Life Cycle Impact Assessment (LCIA). This converts indicator results of different impact categories into a common indicator by using numerical factors based on value-choices.

Weighting has always been a controversial topic in Life Cycle Assessment (LCA), partly because this element requires the incorporation of social, political and ethical values. Despite that, weighting is frequently used in LCA practice and several weighting methods have been developed over the last ten years.

Methods for weighting can be classified in different categories, namely:

- Panel methods, where a group of experts representing different stakeholders are asked to provide their weighting factors;
- Monetisation methods, where the weighting factors are expressed in monetary costs according to the estimated economic damage incurred in an impact category or to what is necessary to prevent the damage itself;
- Distance-to-target methods, where the weighting factors are calculated as a function of some type of target values, which are often based on political decisions.

A further distinction can be made between midpoint methods and endpoint methods, depending on the level of impact indicators to which the weighting step is applied. A combination of the two may also be considered in order to reduce the number of indicators and/or to provide complementary information during the weighting.

This report further details the above classification scheme and analyses a number of relevant weighting approaches. Each of the methods considered has been characterized in terms of methodological foundations, geographical representativeness, procedure for values definition, communication impact and major applications in the LCA practice.

Building on this review, the 2nd part of the project will define a weighting approach and procedure to support the construction of the overall EU eco-efficiency indicator. The weighting will be applied to the impact assessment categories recommended in the context of the ILCD - International Reference Life Cycle Data System[1], leading to the final indicator of the overall environmental impact.

[1] http://lct.jrc.ec.europa.eu/assessment/projects

1 Policy context and aims of weighting

The aim of the weighting procedure - as is to be established - is to combine different environmental effect indicators based on their relative importance. This allows for an easier survey of otherwise complex indicators. For example, if we want to determine the extent to which society is achieving decoupling, one may survey all possible decoupling indicators among the hundreds existing. However, then the overall question still remains: did we decouple overall?

The overall score does not replace more detailed scores. One base requirement on such an overall scheme is that it is responsive to relevant underlying mechanisms contributing to the overall score. More extraction from nature of some scarce resource should lead to a higher depletion score and to a higher overall score. The same holds true for any emission of a substance exerting negative effects on the environment. The trade-off between different emissions, for example how many tons of additional CO_2 emissions are allowed for one ton reduction of SO_2, is established through the weighting procedure. It is not the emissions which are weighted but the expected effects of these emissions.

With such trade-offs, established for all relevant environmental interventions, countries can be scored according to their overall environmental performance. Next, the development in time and the comparison with other countries can be made, allowing statements on decoupling. The applications in this project are exemplary. The prime aim is to establish an adequate weighting procedure and a set of weighting factors. In this paper, we start with a survey on available weighting methods and on operational sets of weighting factors.

1.1 Policy context

This project falls in the context of the EU Thematic Strategy on the Sustainable Use of Natural Resources (COM(2005)670)[2]. The Institute for Environment and Resources (IES) of the Joint Research Centre (JRC) has developed three sets of decoupling indicators[3]. Such indicators require the definition of a measure of the EU-27 overall environmental impact, through a weighting procedure across the whole range of impact categories.

1.2 Scope

The project is in the context of the development process of decoupling indicators as set out in the above mentioned Thematic Strategy on Natural Resources, contributing to the development of the overall eco-efficiency indicator for total consumption in the

[2] http://ec.europa.eu/environment/natres/index.htm
[3] Continuously updated information is provided at: http://lct.jrc.ec.europa.eu/assessment/projects

EU and for the eco-efficiency of consumption in the individual Member States. The eco-efficiency indicator specifies the relation between the level of Overall Economic Activity and Overall Environmental Impact.

1.3 Objectives of weighting

The prime objective of the weighting procedure in this project is to support benchmarking of the Overall Environmental Impact of the EU-27 Member States. Combined with their economic performance, this gives the eco-efficiency score. Analysis of time series then can establish the decoupling performance and allows for comparison of performance among Member States.

Other applications, like in technology assessment, might require different methods. The weighting result will be established in a fully independent way from different options for describing and aggregating economic activities in their economic aspects, like in private preferences for private goods and services or integrated social accounting methods.

1.4 Goals of environmental impact assessment

The goal of environmental impact assessment in LCA is to provide indicators related to the effects of environmental interventions. Environmental indicators may be specified at midpoint, as with Global Warming Potential (GWP) establishing the amount of climate forcing relative to carbon dioxide, or at endpoint, stating the effects in terms of areas of protection, like human health and biodiversity.

1.5 Goals of weighting

The goal of weighting in LCA is to facilitate the establishment of an overall indicator of environmental impact. Results of such overall weighing methods may be expressed in monetary terms or in dimensionless weighting factors, or possibly in some other unit.

1.6 What is weighted

The environmental effects of economic activities are to be weighted. At first, these are mostly recorded in terms of environmental interventions, such as emissions or resources extracted. Next, midpoint indicators as specified in Life Cycle Impact Assessment (LCIA) can be specified, in terms of e.g. radiative forcing relative to an emission of carbon dioxide (Global Warming Potentials - GWP) or other impact categories such as acidification. Relevant consequences - related to human health, ecosystem health and life support functions as well as damages to relevant products - are observed further down in the effect chain. Models specifying these endpoints can be incomplete and sometimes highly uncertain.

In this project, the ILCD (International Life Cycle Data System) recommended impact methods and factors are used as the basis for the indicators for impact categories, covering emissions, land use and resource extractions.

The two-step procedure (from environmental interventions to midpoints and endpoints) as established in life cycle assessment is not used in other domains. Willingness-to-pay methods often cover the full effect chains towards the to-be-evaluated effects in a single model, as in climate models. These do not use Global Warming Potentials, but use the underlying dynamic climate model with effect models included. They do not have a fixed time horizon as in GWP_{20} or GWP_{100}, but use a discounting method, diminishing the contribution to an overall effect gradually.

2 Weighting approaches analysed

2.1 Modelling and evaluation: principles for establishing weights

For answering the question of ***'how serious are effects?'*** we first need a link between the activities involved and the effects to be evaluated. The first step in linking is to specify the environmental interventions created by the activities, e.g. emissions, land use and resources extraction. As the further effects in the causal chains involved are not visible or measurable, they are established through some form of modelling. The models to be used take the environmental interventions as a starting point and then specify effects through the mechanisms as deemed relevant and modelled. The effects to be evaluated are whatever is deemed important. The more stable, quantified models specify the first steps in effect mechanisms. Models may, for example, specify climate forcing and climate change. The effects of climate change are highly dependant on the measures being taken. With a two meter sea level rise, low lying lands may be diked to prevent flooding. Whether or not that is an option, is difficult to model. However, it is not the climate change but the flooding, and the effects thereof, which constitutes the effect to be evaluated. As models are becoming increasingly uncertain when adding effect mechanisms and elongating the time horizon, they often stop at some midpoint level, allowing a more subjective estimation of further effects and the always subjective step of evaluating them. Prior to the weighting step, we therefore need models linking environmental interventions to effects covering several causal mechanisms. Then the resulting effects, as modelled quantitatively and more subjectively estimated, are to be evaluated.

So the next question, ***'which effects, how modelled and estimated?'*** is on the table. There is a huge variety of models which, for very practical reasons, will have to be reduced to a limited number. The life cycle impact modelling as developed in LCA is the guiding principle and is followed, but the very extensive modelling of other domains should be used as well. With the modelled and subjectively estimated effects established, we can start to answer the core question: '***how serious, how important are they?***'.

Ultimately, the importance of the environmental effects considered is based on a judgement. Whose judgement is not yet the point here. It is the fact that it is a judgement. The judgement may relate to what we feel as fair or pleasant, or alternatively it may relate to what we do not like or do not judge as proper to happen. As citizens, we do not like to be ill and we like to live in a healthy, stable and rich ecosystem as part of our surroundings. From a public point of view, we may also have more indirect effects as focus point for the judgement. For example, biodiversity is not always considered as having inherent value. General ideas, on how biodiversity relates to ecosystem health and stability as well as on potential value of species in an economic sense, together lead to the evaluation of biodiversity as a valued good. Concerns on ecosystem stability are of a public or collective nature, as they cannot easily be linked to specific effects on individuals. Such public concerns are common in the social domain, as when judging on the skewedness of income distribution, given a total income. These public concerns are dominant in Brundtland's report sustainability values of intragenerational and intergenerational and equity. Equity values also refer to how seriously we judge environmental impacts – like production losses due to climate change, differentiating the same effect depending on income: for poor people the loss of one euro may count higher than for a rich person.

Specifying and ordering such considerations to allow a comparative view on importance of effects constitutes the core of a weighting procedure.

2.2 A taxonomy of weighting approaches

The taxonomy of weighting approaches specifies the main types of methods used to arrive to a single score. Some methods are not really weighting methods, like single item methods which pick out "the" main effect score. This is the first main distinction presented in figure 2-1, narrowing down the field of weighting methods. Next, there are value based and preference based methods. Value based judgements focus on one aspect corresponding to one value. Each value constitutes a reason why one option is preferred to another. As these considerations are of a qualitative nature, orderings are possible only based on a lexicographic basis: one value being more important than other values, or slightly more modest, one value to be met before other values can usefully be considered. For example, for many people products based on child labour are ***always*** inferior, other considerations together being less important. Value based reasoning is not easily linked to complex situations, with many relevant aspects varying in different directions.

The solution then is to analyse preferences over options directly. These preferences reflect, in an unspecified way, the values behind a choice. Assuming some degree of rationality, one may infer a set of values from a broad set of overall judgements in terms of preferences, as von Neumann & Morgenstern (1947) proved. However, there is broad literature indicating that reasons given by decision makers for their preferences are not consistent or stable in time. A good survey in this conceptually difficult domain is provided by Keeney & Raiffa (1993). For the current problem of this study, there is no broad set of preference judgements to form the basis for deriving the values behind. We still have to arrive at a broad set of preferences.

Methods based on preferences, with quantified trade offs being specified, form the core for the analysis here. These may be further distinguished as to the kind of preferences involved. These may refer to what concerns an individual as the judge of effects, i.e. how do ***you***, as a private person, value effects of climate change in ***your*** life? These are individual preference methods. There are, however, reasons why other considerations may play a role which is not covered by individual preferences. One main example relates to societal risk aversion. Objections to low chance/high impact effects of nuclear installations or climate change may be based on justice considerations as well as on the fear of unforeseeable social and cultural disruption. Such reasoning in evaluation goes beyond the expected value of effects based on private preferences. It is value based, not easily allowing for the establishment of trade offs, or is reflected in a not clearly specified way in collective preferences. Next, the question is “whose preferences count” – which is a value judgement by itself. First, we may choose individuals as the only relevant subjects for judgement, as natural persons. This is the position taken by most economists. How to derive an overall judgement from such individual preferences is a subject of major concern, with Bentham stating the core principle of ‘the greatest pleasure for the greatest numbers’. The Benthamian welfare function for society depends solely on the welfare (or utility) of individuals. Next, we may assume some collective decision maker with preferences, like a representative body or a relevant person. Using a collective decision maker makes it easier to accommodate considerations regarding society at a more aggregated level, as when referring to the low-chance high impact effects and when considering distributional effects (e.g., the distribution between current income groups and intergenerational distributions).

The final distinction in figure 2-1 pertains to measurement of preferences, either through explicit statements, as stated preferences, or derived from decisions actually taken, as revealed preferences. Direct statements on preferences are what we want to arrive at here: an expression on what is to be preferred over what, i.e. which set of environmental effects over another. How such statements are collected is very open. A much used method is asking persons or public bodies to rank alternatives. This method is flexible but prone to manipulation and inconsistency. The other option is to see how decisions are actually made, and then infer the preferences from the decisions. In this revealed preference method the main problem is getting clear choice situations. A basic example is the situation where prices of housing, where houses and surroundings are equal, and where the difference in price between options depends only on air pollution of some kind. The price difference for houses then indicates the value of clean air. The assumption is that the buyers of houses can adequately judge the empirical consequences of differences in air pollution, including threshold effects and time delays. The applicability in the domain of individual preferences is limited. Application in the domain of collective preferences is more straightforward. If governments implement policies of a more general nature, the cost induced can be used for deriving the trade off factors behind decisions – especially when abatement costs for specific substances and effects are involved. When collective preferences are derived in this way, they reflect choices of the past. Therefore, they cannot guide new policies for example based on new insights in the seriousness of the effects of climate change.

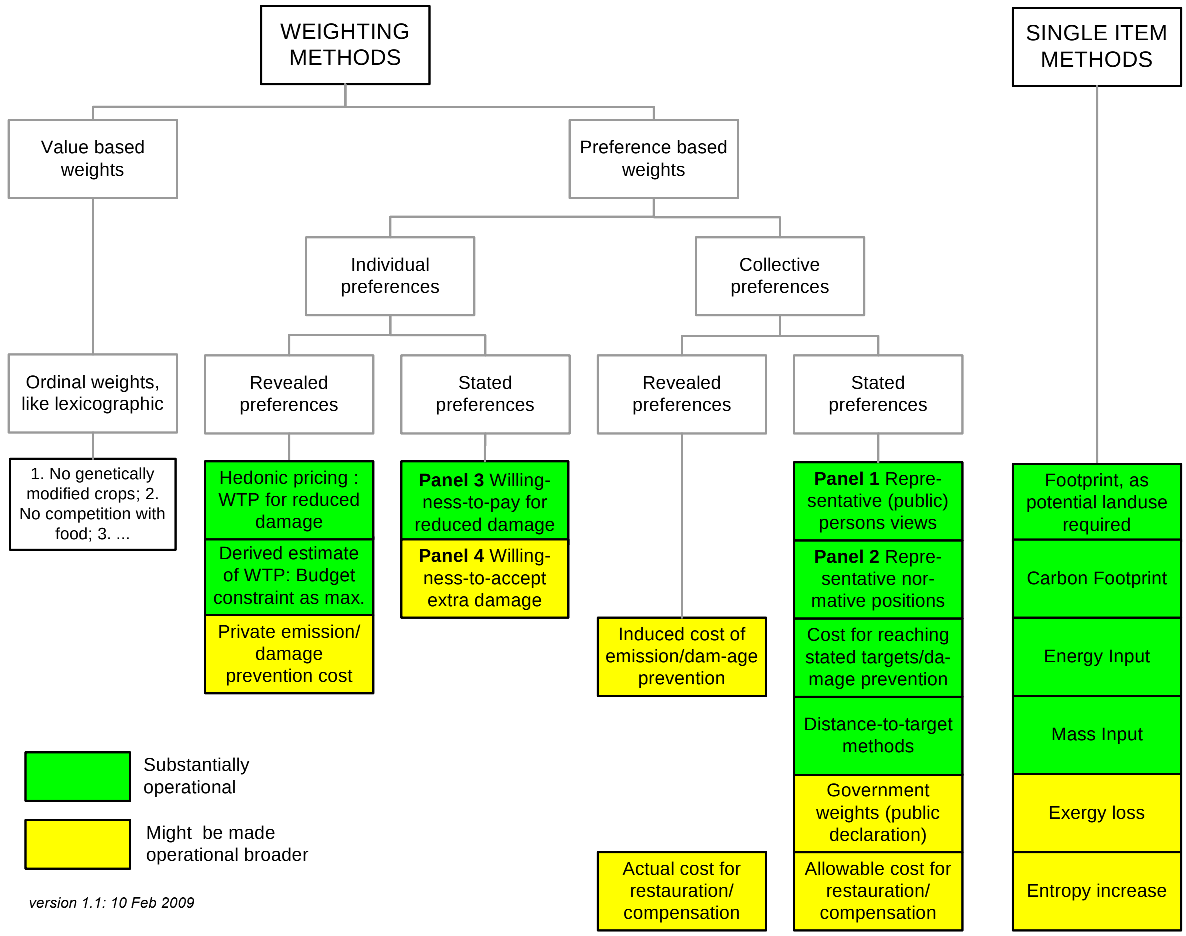
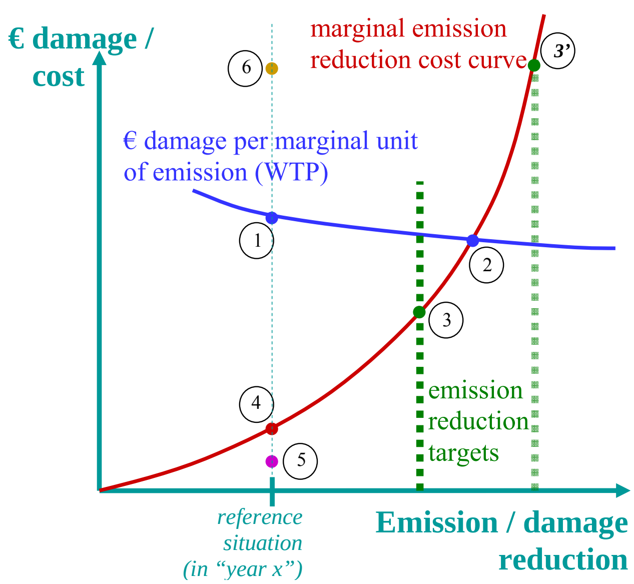


Figure 2-1 Taxonomy of weighting approaches

Some main types of methods may be combined in a consistent way, like hedonic pricing and Panel 3, establishing willingness-to-pay for reduced environmental damages. Other methods seem to be mutually incompatible. For example, the willingness-to-pay for avoiding damages cannot be combined with the expected cost of damage reduction to reach a stated target, even though expressed in monetary units like euro. Complementary application leads to non-interpretable results here, as different methods with differing outcomes are applied partially. When stating the allowable level of an emission (or more generally an impact type or damage type), emission reduction options can be specified, indicating the cost level required to reach the reduced level. This cost figure does not coincide with current actual cost of emission reduction, nor does it fit with the cost of damages as specified through panel methods. This damage cost method can establish the allowable emission level, while the cost of allowable emission level method obviously cannot be used for that purpose. The actual cost of emission reduction cannot be used for either purpose.

Each of the main type of method may be filled-in in different ways. Methods to establish stated collective preferences may be expressed as a dimensionless weighted score, like representative panel methods mostly do, or in terms of monetary units, e.g. the cost of reaching a stated target.

Especially the different types of monetising methods may erroneously be combined or confused, as they are all expressed in a monetary unit (e.g. euro or dollar). Their scores may well be orders of magnitude apart, differences between 5 and 3 being especially large, but all other options for monetisation can frequently differ by a factor 2 or more. Each option for monetisation has a large variation in outcomes as well.



Legend:

1. Willingness-to-pay, marginal, for emission/damage reduction or prevention in current (or reference) emission situation.
2. Willingness-to-pay, marginal, for emission/damage reduction in hypothetical optimum situation.
3. Marginal cost to reach stated emission (reduction) target. Idem for higher reduction target ***3'***.
4. Currently induced highest marginal cost of emission reduction.
5. Current private marginal expenditure for emission/damage prevention.
6. Budget limit for marginal willingness-to-pay for damage reduction, current income.

Figure 2-2 Possible levels of marginal damages or cost, for selected monetizing methods

Figure 2-2 presents several monetizing methods and their possible levels of damages or cost. The private preference based method as willingness-to-pay is most fitting in the economic valuation approach. The marginal valuation, in principle, is relative to a reference situation. With lower emission levels, the value to the damages resulting will be lower in principle than the curve showing a demand for emission reduction. Such curves are not available. It is one point on the hypothetical curve which can be specified.

The core point to note here is that all other methods may deviate substantially from this damage evaluation method. Current policy induced emission reduction cost may be an order of magnitude lower, while cost currently incurred for private considerations may be an order of magnitude lower again.

The cost of reaching targets, often used as a proxy for missing data on willingness-to-pay, may be substantially lower, or substantially higher than willingness-to-pay. For higher targets, the cost usually surge to orders of magnitude above the blue line. Mixing methods, therefore, is unallowable if interpretable results are to be produced.

The choice of relevant methods starts with a survey of potentially relevant methods. One basic requirement for all methods is that in their empirical part “in the economy” they are to link to environmental interventions: resource extractions, land use and emissions of substances and energy. There are monetising methods (which refers to aspects of activities, like input of fertiliser) which model effects to valued damages without specifying emissions at the boundary economic activity – environment. These cannot be used here.

In the taxonomy of weighting methods, the focus is on preference based weights, not on value based weights or on single issue methods, which therefore receive less attention.

2.3 Application to country level data

The environmental indicator, and therefore the weighting method across environmental impact categories, should be applicable to the EU and its Member States. This can be done in terms of the direct environmental effects of the total of economic activities. The final intention, however, is to use the weighting method at the level of countries in the sense of total consumption of countries and EU. This requires the specification of the environmental effects of imports, upstream, and the subtraction of activities and imports required for exports. The latter corrections, necessary to arrive to the overall impact of consumption, will not be possible in this project. In the third stage of this project, we will restrict the exemplary applications to the total of interventions by all economic sectors in the EU27.

2.4 Weighting methods reviewed

Weighting level	**Non-monetised**		**Monetised**	
	Panel 1	**DTT**	**Damage prevention cost**	**WTP Panel 3**
activity				EXIOPOL (partially)
interventions		Ecopoints		
midpoint	BEES (2x)	EDIP		
	ReCiPe-Nogepa			
endpoint	ReCiPe-Pré		ReCiPe-CE	ReCiPe-CML
	Ecoindicator99			ExternE/NEEDS/ EXIOPOL
				LIME
				EPS
				Weidema (indirect)
other	Ecological Footprint			Top,
				Stern,
				Weitzman

Table 2-1 Operational quantified weighting methods considered

3 Characteristics of approaches: survey

3.1 Introduction

The taxonomy can be used for classifying relevant approaches, starting from value versus preference approaches; individual versus collective preference approaches; and revealed versus stated preference approaches.

At a more practical level several additional criteria apply, such as:

- weighting procedure: distance-to-target (DTT), panel;
- modelling of cause-effect chain;
- environmental interventions covered;
- impacts covered, connection to generally accepted impact indicators and in particular to those of the ILCD;
- scientific quality and acceptance of the method;
- societal acceptance of the content;
- reproducibility;
- range of applicability;
- treatment of uncertainty;
- geographic and temporal representativeness;
- degree of being operational.

These criteria form a basis for reasoned choice, but mostly they are not direct evaluation criteria. In making choices, the operationality requirement is the most important in this project. The models and methods that are better but do not provide results so far are irrelevant in a practical sense. However, they can indicate shortcomings of current methods.

3.2 Panel judgements versus assumptions and models

Panel judgements are judgements and, hence, cover the core of the evaluation step. Whatever one may think of the precise questions being asked and of the methods for the processing of answers into scores, the evaluation element is there, explicitly. All other methods are based on assumptions and models, to extract the evaluation indirectly. For example, the cost for reaching emission reduction targets is based upon cost curves on the one hand and a stated goal in some future point of time. Such goals may be set for different time horizons, like for climate change emissions in 2030 and in 2050. It is not easy to finish the reasoning for such cost goals with ambient air quality goals.

3.3 Modelling of cause-effect chains

Weighting is applied across different impact category indicators. The impact category indicators can be classified according to the position in the cause effect chain. Weighting approaches differ in:

1) the position in the cause effect chain of the impact category indicators on which the weighting is applied:

- activity,
- intervention,
- midpoint,
- endpoint.

2) the environmental mechanisms taken into account between interventions and environmental impact indicators.

Models will differ in the type and number of impact categories at midpoint level that are taken into account in the impact assessment, either explicitly or implicitly. Furthermore, there will be differences in the type and number of damages that are estimated. Note that the number of indicators that are taken into account within a specific endpoint or area of protection may already imply a form of weighting: what is not covered receives weight zero.

It should be noted that for the evaluation of decoupling a link to interventions is required. Interventions are the link between economic activities and environmental impact categories, midpoint and endpoint, and from there to evaluation. Therefore, calculated results on midpoint or endpoint impact categories based on monitored interventions are suitable for measuring decoupling. In contrast, monitored states of the environment and damages do not have a direct link to economic activities in a region. These monitored impacts might be the result of interventions in the past and/or from other regions. Alternatively, they might be the result of other developments, like zoning laws and nature conservation activities, and of autonomous development. Thus, monitored impacts are not suitable for measuring decoupling.

3.4 Environmental interventions to be covered

There are different sets of environmental interventions that may be covered in different steps of modelling and evaluation:

A) Environmental interventions profile of a country, e.g. total of emissions, extractions and land use by a country or group of countries like the EU.

B) Environmental interventions covered in the weighting scheme.

C) Environmental interventions for normalisation[4] as required for panel weighting, not for DTT and most monetising methods.

Ideally, all relevant interventions are covered in the systems to be evaluated, like a country's activities or country's consumption. Next, these are covered in the evaluation methods and, where required, in the normalisation step. Different incomplete sets are available, both about data on countries and on method. We survey:

Ad A) choice of emission profiles to be assessed:

1) Extensive environmental extensions: e.g. EIPRO, to be disaggregated and specified to country level
2) Country data: Wegener Sleeswijk et al. 2008
3) Less extensive, more related to official sources: EXIOPOL
4) NAMEAs: very limited and incomplete

Ad B) The number of interventions that are covered in the impact assessment and weighting scheme varies between the different weighting approaches. Some approaches have a limited coverage, like EPS, Ecopoints, EDIP (200-500 interventions), while others have a much larger domain of application, e.g. ReCiPe (3000 interventions).

Ad C) The reference for normalisation is an external normalisation and will be at the global level[5] (Wegener Sleeswijk *et al.,* 2008). This allows for expressing EU (or country scores) as a dimensionless fraction of that global total, be they measured as national production and consumption activities or as national consumption, including upstream foreign activities through imports and subtracting production chains for export.

Economic valuation methods don't usually have a normalisation step. It is possible, however, to express global totals of monetised environmental interventions in a single weighted score. The national scores then can be expressed in exactly the same way as it is done in life cycle analysis creating dimensionless weighting fractions. Of course, the normalisation in life cycle approaches is done at the midpoint level, before weighting.

[4] The scores for the different impact categories are expressed in different units. To facilitate the interpretation in LCA often a normalization procedure is used. If the environmental impact assessment includes a weighting across impact categories into one overall environmental impact score the normalization is a necessary step in case panel weighting is used.
So after normalization for each impact category the normalized score of the case study is given. This normalized score expresses the relative contribution of the case study to the impact score based on the interventions of the total world economy.

[5] Global reference is required for effects with global mechanisms, and global reference is required for global normative positions, like on ***sustainability***.

3.5 Impacts covered

The weighting scheme should connect to generally accepted damage types and in particular to the impact indicators of Life Cycle Impact Assessment methods as recommended in the ILCD Handbook – Framework and requirements for Life Cycle Impact Assessment models and indicators[6]. So, for each weighting approach the impact indicators on which weighting can be applied will be described.

Many recommended methods still need adaptation, i.e. factors are not yet available. The development of impact assessment (IA) factors is outside the scope of this project. So a practical choice might be to use the IA factors that are available in ReCiPe. ReCiPe defines IA factors on two levels: midpoint and endpoint. Also the relation between midpoint and endpoint is well described. These ReCiPe categories can be adapted to the ILCD Handbook requirements, and adjusted for the slight differences which may come up. Input related interventions and impact categories, as related to resource extraction, water use and land use, have a more limited acceptance and applicability. For example, land use at a country level makes sense only if the relevant land use categories are distinguished. These are relative to specific impact assessment methods.

Most monetising methods do not specify midpoints, and often are quite secretive about the endpoints covered – meta-study like Tol (2008) covers around 200 different studies. It gives dollars per emission, discounted, but what exactly is discounted remains unspecified, and will be different in the studies surveyed.

Methods like developed and used in ExternE and NEEDS project are more detailed in the health effects covered than for example ReCiPe. In the majority of broader methods it is acknowledged that a high degree of incompleteness makes them less useful and therefore "methods transfer" is applied to enlarge the damage categories covered.

[6]http://lct.jrc.ec.europa.eu/assessment/projects

Table 2-2 Recommended impact categories, LCIA methods and indicators – ILCD Handbook[7]

	Recommendation at midpoint		Recommendation from midpoint to endpoint	
Impact category	***Recommended default LCIA method***	***Indicator***	***Recommended default LCIA method***	***Indicator***
Climate change	Baseline model of 100 years of the IPCC (2007)	Radiative forcing as Global Warming Potential (GWP100)	No methods recommended	/
Ozone depletion	Steady-state ODPs 1999 as in WMO 1999 assessment	Ozone Depletion Potential (ODP)	No methods recommended	/
Human toxicity, cancer effects	USEtox model (Rosenbaum et al, 2008)	Comparative Toxic Unit for humans (CTU_h)	DALY calculation applied to USEtox midpoint (Adapted from Huijbregts et al., 2005)	Disability Adjusted Life Years (DALY)
Human toxicity, non- cancer effects	USEtox model (Rosenbaum et al, 2008)	Comparative Toxic Unit for humans (CTU_h)	No methods recommended	/
Particulate matter/Respiratory inorganics	RiskPoll model (Rabl and Spadaro, 2004)	Intake fraction for fine particles (kg PM2.5-eq/kg)	Adapted DALY calculation applied to midpoint (Adapted from Van Zelm et al, 2008, Pope et al, 2002)	Disability Adjusted Life Years (DALY)
Ionising radiation, human health	Human health effect model as developed by Dreicer et al. 1995 (Frischknecht et al, 2000)	Human exposure efficiency relative to U^{235}	No methods recommended	/
Ionising radiation, ecosystems	No methods recommended	/	No methods recommended	/
Photochemical ozone formation	LOTOS-EUROS (Van Zelm et al, 2008) as applied in ReCiPe	Tropospheric ozone concentration increase		Disability Adjusted Life Years (DALY)
Acidification	Accumulated Exceedance (Seppälä et al., 2006, Posch et al, 2008)	Accumulated Exceedance (AE)	No methods recommended	/
Eutrophication, terrestrial	Accumulated Exceedance (Seppälä et al., 2006, Posch et al, 2008)	Accumulated Exceedance (AE)	No methods recommended	/

[7] Version 3.1 and 3.2, d.d. 01 February 2010 of the ILCD characterisation set is not complete yet; http://lct.jrc.ec.europa.eu/assessment/projects

Table 2-2 Recommended impact categories, LCIA methods and indicators – ILCD Handbook[7]

	Recommendation at midpoint		Recommendation from midpoint to endpoint	
Impact category	***Recommended default LCIA method***	***Indicator***	***Recommended default LCIA method***	***Indicator***
Eutrophication, aquatic	EUTREND model (Struijs et al, 2009) as implemented in ReCiPe	Fraction of nutrients reaching freshwater end compartment (P) or marine end compartment (N)	No methods recommended	/
Ecotoxicity – fresh water[8]	USEtox model, (Rosenbaum et al, 2008)	Comparative Toxic Unit for ecosystems (CTU_e)	No methods recommended	/
Land use	Model based on Soil Organic Matter (SOM) (Milà i Canals et al,, 2007)	Soil Organic Matter (SOM)	No methods recommended	/
Resource depletion, water	Model for water consumption as in the Swiss Ecoscarcity (Frischknecht et al, 2008)	Water use related to local scarcity of water	No methods recommended	/
Resource depletion (mineral, fossil and renewable)	EDIP97 update 2004 (Hauschild and Wenzel,1998-update 2004) and CML 2002 (Guinée et al., 2002)	Scarcity	No methods recommended	/

[8] This refers only to freshwater ecotoxicity. Marine and terrestrial haven't, at the moment, methods recommended

3.6 Geographic and temporal representation

Spatially differentiated environmental effect chains are modelled and used by economists in evaluating the consequences of emissions. A coal-fired power station to the west of Paris clearly causes more health damages than one to the east, given the typically western winds prevailing. Weighting values may thus be site specific, versus site independent as is usual in LCA oriented methods, where technology choices and not location choices are to be supported. Thus, methods and values resulting may be representative for different geographical scales, from the local level to world level. Willingness-to-pay methods as set up by economists, like the impact Pathway approach[9], hence tend to differentiate below the country level as to source and effect mechanisms, and to some extent also in weighting. We will not follow that approach to spatial differentiation.

This creates a number of problems in specifying the willingness-to-pay scores, as the spatial detail has to be aggregated to the EU27 as well as to the country level, at least for a number of countries.

3.7 Spatially and temporarily differentiated weighting

Even if models do not differentiate in these respects, the persons making the judgement can differ in their preferences. There are several reasons for differentiating the evaluation of effects which take place in different locations and time:

- There are cultural differences in willingness-to-pay, also between regions.
- There are income based differences in willingness-to-pay between income groups and countries with varying income per head. This may result in attitudes like “what is wrong with us is OK with them”.
- Willingness-to-pay may be based on expected future earnings.
- Future preferences may lead to adapted willingness-to-pay for other reasons, like cultural developments regarding the environment and changed attitudes towards income and growth.
- Time integration and discounting issues, versus zero discounting as with toxicity effects on nature based on ‘fate modelling towards infinity’.

[9] E.g. the monetised values using the impact pathway approach (NEEDS; EXIOPOL; …) are location specific.

3.8 Discounting

There is a substantial discussion on principles for discounting. World Resources Institute (WRI) has produced a survey on the subject (Portney and Weyant 1999). The subject is broader than just discounting, it is about intergenerational equity. There first is a subtle subject related to modelling. There is some agreement that technological uncertainty is a reason for discounting. Maybe, the climate can be geo-engineered and we need not bother so much about long lasting climate changing emissions, like CO_2. Such technological options may be treated differently, by including them in technology scenarios, keeping them out of the procedure of discounting. If not, a subjective technology uncertainty estimate should be included in the discount option being used. With this subject solved, one way or another, the remaining part of the discussion is about reasoning on how important later effects are compared to the same ones earlier. Most people will agree that in the long run we are all dead as Keynes remarked, but also that in thousands of years of time our next generations will be disturbed by other intervening mechanisms will have become of overriding importance by then.

In a practical sense, the importance of discount rate can be indicated by the corresponding half-time reduction of importance. With 4%, a future effect is reduced by a factor 2 in 17.5 years, with 2% in 35 years; with 1% in 70 years and with 0.1% in 700 years. Clearly, this choice matters, differently for different types of environmental effects of concern. Climate change and species extinction have an extremely long effect horizon, while the toxic effects of fine dust will become negligible within years. The choice therefore influences the relative importance of different midpoint type environmental problems. For models going to infinite time without discounting, that is discount rate zero, the outcome is infinite. In practice all models reduce effects in time, either through cut off or through an explicit or implicit limitation of effects in time or through an implicit or explicit discounting procedure.

One of the conceptual difficulties in discounting relates to the subjects having the time preference. In purely classical utility theory, utility functions are independent of each other. Nobody cares about other persons, now or in the future. Suppose that the overall welfare function combines the utilities of all concerned now. Then our concerns stop with old age, and next generations have no voice. Contrary to independence of utility notions, we do have concerns about the future, also the far away future. Discounting with a discount rate relevant within generations leads to fully insignificant contributions of the far away future, contrary to what people express as being relevant.

One solution has been brought up by Weitzman and similarly Gollier, who propose to differentiate the discount rate, with a decreasing discount rate towards the more far away future, also stressing the special nature of discounting long term low chance high impact environmental effects. This approach is attracting many supporters at the moment, including main line neo-classical economists like Tol. Without such a correction, current choices would erroneously, that is contrary to general views, leave out their long term effects.

Another solution has been proposed by Kopp & Portney (1999). They propose to treat time preference in exactly the same way as in eliciting the value of effects, by asking the respondents in the panel. Respondents typically come up with Weitzman type of discounting, without the same reasoning however. Moreover, respondents vary as to their time preference profiles. A simple average or mean may seem the best solution. However, it is not the discount rate which should be weighted but the net present value. In computing the net present value, based on the views the public panel holds, persons with the lower discount rates will have a higher influence on the net present value. This constitutes one reason to go for a smaller discount rate than the one based on averaging time preferences as measured in discount rates. Adding the personal discounted net present values would take everybody's opinion equally; averaging their preferred discount rates would not.

The discussion on discount rates comes back when treating specific damage monetisation methods.

Choice:
If possible we will use the Weitzman method of discounting as developed in the EXIOPOL project, and as used in climate effect modelling as surveyed by Tol (2008). If this cannot be done consistently, a middle value discount rate will be used, that is two to three percent.

3.9 Normalisation: reasons and level

There are two main reasons to apply a normalisation step. One is related to inherent elements in weighting steps and one to comparison between systems.

In weighting, different types of effects, like health effects and species loss, are to be compared. The comparison requires that such different effects are expressed in the same unit, like monetary units, or are dimensionless. In both cases the original unit, like DALY (Disability Adjusted Life Years) for human health impacts and PAF (Potentially Affected Fraction) for ecosystem health impacts, are to be removed by division by a reference value. This is a customary step in all multi-criteria evaluation procedures, formalised or deliberative. The weights set can be applied only to the standardised scores with equal units. In weighting at midpoints, the same reasoning holds. We cannot apply a weighting set to variables in different units; they need to be standardised. Climate forcing in CO_2-equivalents cannot be compared to acidification in terms of SO_2-equivalents. In midpoint weighting, the reference by which the case score is divided usually is a geographic region, the region for which the environmental problem resulting from the midpoint score is envisaged. All midpoints are weighted as to the seriousness of their region score. The system analysed receives the share it would have in the overall score per midpoint, by division of its score by the region score. The region weights then can be applied to this standardised score per midpoint. If the system score is in equivalents per year, as with country scores, and the region reference is per year as well, the resulting ration is dimensionless. For non-time defined systems, like product systems, all midpoint scores have the unit 'year', before and after weighting. Which reference to take? Let us first check requirements from the second level.

The comparative evaluation of systems - between different systems, e.g. countries or between the same system at different moments in time - requires a reference, in order not to be dependent on shifting and arbitrary references. For general applicability, a most general reference is to be used, independent from the case of application at hand. For the regional type of normalisation, the global level then is most appropriate. An extensive data set for global normalisation is available.

Several available weighting methods have one or another region as a reference. These are to be transformed from regional to global level.

With midpoint panel weighting, the need for normalisation inside the weighting step is clear. For willingness-to-pay methods this is less so. They can be set up independent of normalisation. The computation of the damage is based on a monetary value per unit of damage, available for each damage type. The resulting score has the monetary unit only (and possibly the unit year). Willingness-to-pay scores can then be added over all types of damage.

When analysing time series, the reference year should be fixed. If we would adapt the normalisation reference to the year of analysis, the outcome would always be 1 for each year. As a consequence an increase or decrease of the value would not be visible. When taking a fixed year of reference, the decoupling score can be formulated as an index, with the normalisation year as the base year, and a score of 100 in the base year (see figure 2.3). Choosing the same base year for all weighting methods allows them to be expressed in the same index.

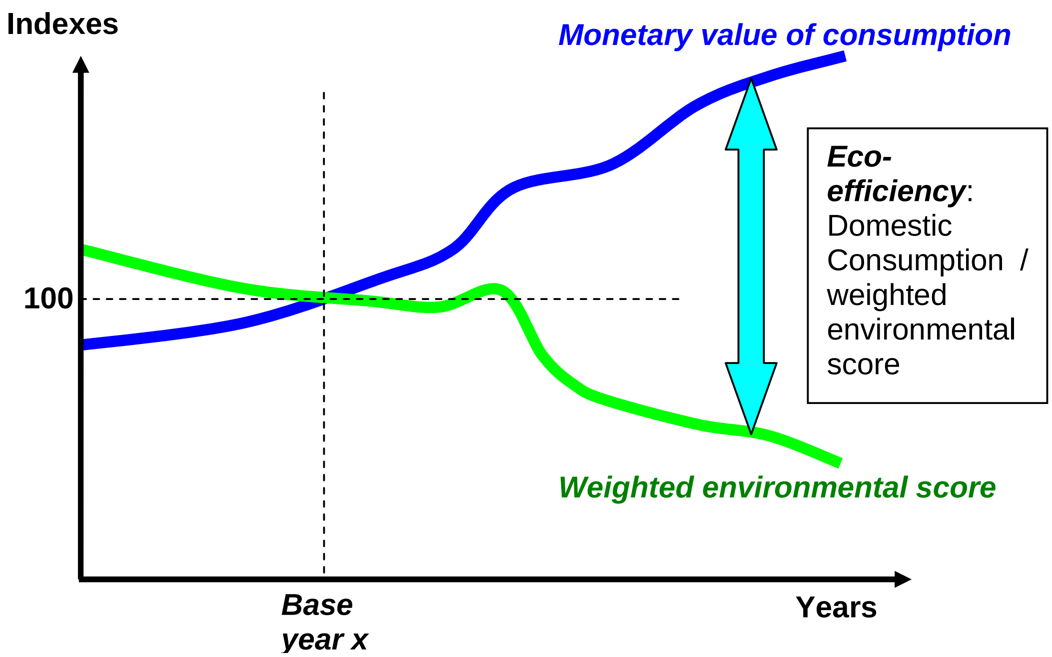


Figure 2-3 Normalisation data for the base year in the decoupling analysis.

It is possible to transform midpoint oriented methods with a dimensionless weighted score into a monetary score by giving a reference unit, for example $CO_{2\text{-equiv}}$, a monetary value and thus express all scores in monetary units. The converse is possible as well, expressing all scores in dimensionless units, also those based on monetary values. It should be noted that also for some monetisation methods, like the conjoint procedure used in LIME, information is used on the present state of the environment based on the same reference emissions as used for the normalisation.

Choice:

All reasoning leads to the use of a global normalization level where normalization is required and use of the same reference for constructing indexes when developments in time are to be measured, as in decoupling analysis of countries.

3.10 Degree of being operational

Impact assessment methods without operational weighting sets cannot be used now and are excluded from further analysis. However, impact assessment and weighting methods often are partial in effect chains and limited in interventions covered, practically or reasoned. For neo-classical economists the depletion of resources caused by their extraction is covered in their prices already, so there are not externalities involved, and no weights can be established. There may be political, technical or economic disruption in supply, but that is not an environmental problem.

As most methods are partial or incomplete in one respect or another, a combination of different methods into a broader applicable one is envisaged in stage 2 of the project.

Choice:
All methods covering weighting over different types of environmental effects will be included in the analysis.

3.11 Scientific quality and acceptance of the method

The following criteria on quality and acceptance are considered:

- Scientific soundness of methodology and data
- Indicative value: is the meaning of the indicator clear and is it relevant
- Peer reviews available
- Simplicity and transparency of the calculation, procedure for values definition
- Treatment of uncertainty
- Major applications in LCA practice
- Scientific Institutes supporting the method
- Clarity and ease of understanding

3.12 Reproducibility

Many methods build on layers of transformation, with questionnaires often forming the starting point. However, the transformation from answers given on an ordinal scale to weights are mostly hidden in partial original studies.

3.13 Comparative completeness in applicability

Impact assessment and weighting methods have been developed for application to product systems and broader technology systems. Applicability at country level, the subject here, primarily is a matter of data availability. The issue of data availability for country level analysis is crucial but is not a point of concern in this part of the project, which will focus on application to EU27, and exemplary country level. Trade linked country systems are being developed and will come available in increasing quality, as in the EXIOPOL project by the end of 2009. There are impact assessment and weighting methods, especially monetising ones, which cover relatively few environmental interventions only, and also cover limited effect chains. This leads to problems in comparative applicability. For comparing evaluation outcomes we can either leave out relevant interventions, like N_2O (highly relevant in bio-energy and agriculture), or those requiring an addition missing interventions (and mechanisms and effects) to the methods so as to make them equal the one covering the broader set of interventions.

Neither option is really satisfactory, nor is it allowable for such undue differences to remain.

Choice:
For the moment, the practical solution is chosen to estimate weighting factors for missing substance-emissions. The estimate will be based on the proportional contribution as expressed in weighting methods that do encompass the substance-emission.

3.14 Consequential vs attributional

However the modelling of economic activities has been done, this modelling results in a set of environmental interventions, possibly specified at a point in time or as a time series. The impact assessment methods cannot be specified in an attributional way, as most effects are consequential by definition.

No choice required

3.15 Marginal vs total

Ideally, effects of environmental interventions are modelled and evaluated per unit, as marginal effects, against some background scenario. For example, Global Warming Potentials are the time integrated climate forcing scores of one additional unit emitted, GWP_{100} over one hundred years, against a scenario background. This marginal score is used in most life cycle impact models. Totals then can be constructed by multiplying the marginal value with the total score. There is a parallel to specifying total market value by multiplying the marginal price with the total volume, disregarding the different values for the intra-marginal units.

In practice, many models are rough and don’t distinguish between average and marginal effects. Only climate models have generally been developed to this sophisticated level. For simple models, there is no difference between marginal and average, they are linear models through the origin, with each unit having the same effect independent of the volume of emissions. Acidifying emissions in midpoint models are an example.

The marginal weights may refer to endpoint or to midpoints, with implicit endpoints in the panel procedure. Both types of weighted scores can be translated back to the environmental interventions being covered. Then one unit of intervention is the marginal unit, like a kg of emission of a certain substance.

4 Survey of operational weighting methods

4.1 Introduction

Weighting approaches will be described according to the criteria developed in chapter 2 and chapter 3. We place them in three categories, based on the modelling characteristics and the method of weighting, all giving results in terms of a judgement on the seriousness of expected effects involved. There is a fourth category for methods which do specify an overall score but which are not based on the evaluation of expected effects but on some other measure, like prevention cost or distance-to-target.

The three main categories of weighting methods are the following.

1. Integrated modelling and evaluation

Number one is based on full integrated modelling of the causal chains from environmental interventions to damages, with values attached to damages based on the willingness-to-pay for their reduction. These values are based on panel procedures, with substantial additional processing, as through discounting procedures. They have been developed especially in the domain of climate modelling and for health effects of atmospheric pollution. Their broader application requires substantial “methods transfer”, using other methods for filling gaps in interventions, effect mechanisms and damages.

2. Midpoint modelling and evaluation

The second modelling group is based on midpoint modelling as developed in the realm of LCA. The modelling steps from midpoints to relevant endpoints are done subjectively, in the panel weighting procedure. This implicit subjective modelling step can be supported by specialist knowledge supplied in the panel procedure, partially quantified, but remains subjective and implicit in how it influenced the weights coming out of the panel procedure. The weights given constitute a combined view of what is to be expected, empirically, and the importance given to these effects.

3. Midpoint-endpoint 2-step modelling and evaluation

The third group formalises the modelling from midpoint to endpoint damages, and then applies monetised weights to damages. In a way this looks like integrated modelling. However, the modelling steps are fully separate for the two-step procedure and there is no dynamic mechanism involved. Both midpoint and endpoint effects are specified as totals, regardless their occurrence in time and hence making discounting impossible. Cut-offs, as in Global Warming Potentials, make a split at midpoint between effects reckoned with, fully, and not reckoned with, all later effects.

4. Other methods for reaching an aggregated score

There is a heterogeneous group of methods resulting in a single score, but not based on an evaluation of relevant effects. Distance-to-target methods are one example, stating how far an emission or midpoint score is failing to reach a target. Another one is what the marginal cost would be to reach a target. Such methods can be integrated into an overall score, but they are not weighting methods in the sense of evaluating what is important. The marginal cost of reaching the target for CO_2 emission reduction in 2020 (or $CO_{2\text{-equiv}}$ emission reduction, which is totally different) may be substantially lower per kg than the cost of reaching the target for SO_2 reduction. It seems that such methods still lack an inter-effect factor, which is a weighting factor, though they can be used to specify an overall score.

4.2 Integrated modelling and monetary evaluation

4.2.1 Introduction

The evaluation of effects caused ideally has the form of a model specifying the relevant changes induced, combined with their evaluation. Integrated models express the evaluation in monetary terms, as willingness-to-pay by the subjects concerned. Economists, mainly of the neo-classical type, play a core role in this development. The underlying panel studies might however express importance of effects in dimensionless weights as well. The changes induced ideally refer to changes in economic activities and from there to changes induced in the environment. These changes in the environment, in turn, will have consequences for economic functioning, as in reduced crop harvests per ha and reduced availability of mineral resources, and hence higher prices. In the integrated modelling approaches surveyed here, there is a first cut, in separating modelling of economic activities and of environmental consequences. Next, there is a cut in terms of the economic consequences of environmental changes. Only direct effects at a physical-technical level are taken into account, sometimes, as in reduced crops, damages to buildings due to acidification and in the cost of hospitalisation due to illness. Criteria for inclusion are not well developed.

Models as developed mostly were partial models. By far the largest efforts have been put into the modelling of effects of climate changing emissions. A recent meta-analysis (Tol, 2008) covers 47 studies on the economic impact of climate changing emissions. The second subject with extensive modelling concerns the health effects of toxic emissions, and within that subject a prime focus on emissions to air, as in ExternE. A survey on monetisation of broader effects is in Turner et al (2004). This excellent study for DEFRA shows how basic studies in terms of panel surveys link to secondary and tertiary studies, like in the ExternE project. In the process of recombining and re-interpreting basic studies, there is a convergence of these widely diverging base studies to damage levels *deemed acceptable*, in the group producing these studies. The NEEDS project and the ongoing EXIOPOL project extend the applicability to broader domains of environmental mechanisms and effects.

Though partial, focused on climate change only, we start with evaluation of climate changing emissions, using the survey by Tol, in terms of willingness-to-pay for emission reduction. Next the line of EU studies is covered, from ExternE to NEEDS and EXIOPOL. The climate studies as surveyed by Tol are important to include as NEEDS and hence EXIOPOL do not specify willingness-to-pay but reduction cost, for climate effects.

4.2.2 Climate models and abiotic resource depletion

Climate models

Climate effects cover all complexity in modelling and evaluation, relating to uncertainty, conditionality, low chance high impact effects and to difficulties in judging effects and compressing them in a single score based on some aggregation of preferences. The focus of many studies is on the last subject: how to value and aggregate effects. These studies are not very precise in describing the underlying environmental modelling. Origins for that part of modelling, however, are relatively stable in reports for the IPCC. One major study using that type of modelling and evaluation is the Stern report. In a review of that report, Weitzman (2007) sees the discount rate used as too low, but sees reasons not treated by Stern for attaching higher damage values to low chance effects. They constitute an insurance premium to catastrophic effects which cannot be compensated for by reduced consumption and increased saving.

How do we get to operational numbers? The review study by Tol analyses the results of 211 estimates in 47 studies[10]. Studies which have been externally reviewed and published in scientific papers tend to converge in a relatively low value for emission reduction of CO_2. There is however a substantial number of studies which have a substantially higher score, up to an order of magnitude. When wanting to avoid the lower chance on higher damage values, with 1% uncertainty, the value per ton of CO_2 rises from 58EUR to 226EUR, see table 4-1. The results strongly depend on the choice of discount rate. A rate of 3% seems the most favoured by Tol. This discount rate does not reflect the outcomes of the more gray literature, opening up options for higher damage estimates. The choice of damage value now is for 20EUR per ton, with reasoned adaptation in D2, also based on experts' comments.

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[10] Tol 2008, p4: "The 211 estimates are classified as follows. Most estimates use the Ramsey discount rule—$\delta = \rho + \eta g$—but some estimates use a constant consumption discount rate rather than a constant utility discount rate. A few recent studies use a *declining* discount rate (inspired by Gollier 2002, and Weitzman 2001), a few studies fail to report what discount rate was used, and a few studies include the discount rate in the uncertainty analysis. Some studies use equity weighting (Fankhauser et al. 1997) with the global average income as normalisation (Anthoff et al., forthcoming), but most studies simply add the regional dollar values (for which normalisation is irrelevant; cf. Fankhauser et al. 1998)."

	Dollar per ton C	Dollar Per ton CO_2	Euro per ton CO_2
Modus	20	73	58
1% uncertainty	78	286	226

Table 4-1 Damage value estimates per ton of carbon: modus and 1% exceedance value.

The use of these willingness-to-pay based damage estimates for climate changing emissions may replace the conceptually less adequate measures in NEEDS and EXIOPOL based on some cost measure like reduction cost to reach a target.

Resource depletion

The economic measures for depletion of abiotic resources are viewed upon differently by different economists. Ideally, markets reckon with increasing scarcity, current prices already reflecting future scarcity. General reasoning has led Hotelling to formulate a rule on pricing of exhaustible resources: the price should rise so much that the value of the remaining resource remains equal. This Hotelling rule has not led to operational measures. In practice, long term price trends for abiotic resources show a downward linear trend, combined with S-shaped or exponential growth of primary production. The fundamental reason for this form of curves is that technological progress reduces cost of production, allowing for expanded production and for increases in economically attractive resources. The levelling off may well relate to the investment nature of many flows. Iron is required in large amounts during take off of economic growth, for building up infrastructure and buildings. At a certain stage such investments level off and recycling becomes more dominant, reducing primary production. Weitzman (1999) has formulated a method to quantify depletion of exhaustible resources, like fossil fuels. He compares the hypothetical situation of not depletion, made operational where global extraction of minerals is allowed to remain forever constant at today's flow rates and extraction costs, with the actual development of volumes and prices. He concludes that world loses the equivalent of about 1 percent of final consumption per year from finiteness of the earth's resources, compared with the counterfactual hypothetical trajectory.

A recent survey on economic analysis of resource depletion is by Halvorsen (2009). It is conceptually interesting but does not give operational figures. The Weitzman (1999) study hence seems to be the only study quantifying the economic loss of depletion of specific resources, covering fourteen major abiotic resources. As this method is conceptually compatible with other damage costs, as for climate change and health effects, it is a prime candidate for using in the overall weighting method.

4.2.3 ExternE/NEEDS

Introduction

NEEDS stands for New Energy Externalities Developments for Sustainability. The objective of the NEEDS project is to evaluate the full costs and benefits (i.e. direct + external) of energy policies and of future energy systems, both at the level of individual countries and for the enlarged EU as a whole.

In this context NEEDS refines and develops the externalities methodology already set up in the ExternE project, through an ambitious attempt to develop, implement and test an original framework of analysis to assess the long term sustainability of energy technology options and policies, see: http://www.needs-project.org/ .

Weighting procedure

In NEEDS damages are valuated in monetary terms using willingness-to-pay as a base reference, but complementing this with other methods, like restoration costs (ecosystem health), damage cost or abatement cost (climate change), restoration costs (building materials) and yield (crops).

Modelling of cause-effect chain

The impact pathway approach is a full chain modelling of interventions to external costs. No intermediate results of midpoint and/or endpoint indicators are available. The models used to for impact assessment are different from the characterisation models used commonly in LCIA. For this reason the models are not further elaborated in the ILCD recommendations (Hauschild et al., in prep).

Model Ecosense calculates generalised values (euro/ton emission) per country for the most important pollutants. The model takes into account spatial specific dispersion, fate and exposure models, using concentration response functions and spatially differentiated monetary values.

Environmental interventions covered

The EcoSense model covers about 20 emissions to air.

MAJOR PARTS: SO_2, NOx, PM10, PM2.5, NH_3, NMVOC

These substances are emitted in large amounts. The valuation of the damage due to these emissions is site specific.

MINOR PARTS: Cd, As, Cr, Ni, Hg, Pb, Cr-VI, CH_2O, Dioxin

These substances are emitted in small amounts, but have very large damage per unit of emission. For the valuation of these emissions only generic monetary values are used.

EMISSION OF GREENHOUSE GASES: CO_2, CH_4, N_2O

Different approaches are available for estimation of external cost per ton of released greenhouse gas, e.g. damage cost, using different discounting methods and rates or abatement cost.

RADIO NUCLIDE EMISSIONS

LAND USE CHANGE

Impacts covered

The impacts on endpoints are not presented explicitly. Most likely the damage approach implicitly encompasses the following safeguard areas: (Damage) Costs due to impacts on human health, crops, building materials, ecosystems and due to climate change.

- Human health, loss of life (VOLY, Value Of Life Years)
  A new value for life year lost by air pollution is developed. The new value is the consensus of a team of experts, based on the results of a new contingent valuation method (CVM) that has been applied in 9 countries (total sample 1463). A procedure for transferring the results to other countries has also been developed and tested (Desaiges et al., 2007).

- Ecosystem health, loss of biodiversity (PDF, Potential Disappearing Fraction)
  Based on the work of Eco-indicator (1999) and Koellner (2002) to derive potentially disappeared fractions (PDF) due to certain land use changes as well as depositions of SOx, NOx and NH_3. The resulting PDF changes are then valued by using a restoration cost approach. The resulting external costs per unit of PDF change as well as per kg deposition of SOx, NOx and NH_3 are presented for 32 different European countries and validated with results from different WTP studies (Ott et al., 2006)

- Climate change (FUND model): Both damage and avoidance costs are estimated with the model of the Climate Framework for Uncertainty, Negotiation and Distribution (FUND). Damage cost estimates are controversial because of the many value-laden assumption behind them, for example on discounting, valuation of the risks of mortality, and aggregating impacts across countries with very different standards of living; because of the vast uncertainty; and because the use of marginal damage reflects weak sustainability. Therefore, also avoidance costs are presented (Tol, 2006; Anthoff, 2007).

- economic assets: the economic valuation of damages to crops and materials are estimated using market prices of major crops and restoration costs of building materials (Preiss and Klotz, 2007).

Geographical and temporal representation

Characterisation and damage assessment: spatial differentiating models are used. Impact factors depend on emission location and damage location. For the impact assessment of some effects (e.g. climate change) discounting is used. However, it is not clear whether discounting is consistently applied for all effects with long time horizon.

Weighting: Valuation is mainly based on country specific costs or WTP.

Normalisation level

No normalisation is needed, as monetisation is applied at the endpoint level.

Degree of being operational

Table 4-2 shows some cost used implicitly in the economic valuation of interventions in the impact pathway approach of NEEDS.

			Cost (€, 2000)	NEEDS source
Human health[1]	VOLY (€, 2000)	EU16	41000	Desaigues et al., 2007
		New member states	33000	
		EU25	40000	
Ecosystem health	Restoration cost (€/m2)	EU25	0.17-8.39 (dependent of target biotope)	Ott et al., 2006
Land use change	Restoration cost (€/(PDF*m2))	EU25	0.57-7.39 (dependent of target biotope)	Ott et al., 2006
Building materials	Maintenance cost (€/m2)	galvanised steel	14-45 (country specific)	Preiss & Klotz, 2008
		limestone	245	
		mortar	27	
		natural stone	245	
		paint	11	
		rendering	27	
		sandstone	245	
		zinc	22	
Crop losses	Market price (€/ton)	sunflower	273	Preiss & Klotz, 2008
		wheat	137	
		potato	113	
		rice	200	
		rye	99	
		oats	132	
		tobacco	2895	
		barley	93	
		Sugar beet	64	
Climate change[2]	Abatement cost ($/ton)	CO2	38-349 (dependent of target)	Tol, 2006 Anthoff, 2007
		CH4	793-7330 (dependent of target)	
		N2O	11708-108211 (dependent of target)	
	Damage cost ($/ton)	CO2, CH4, N2O, SF6	Dependent of discounting method and rate	Tol, 2006 Anthoff, 2007

Table 4-2 Cost used for economic valuation in NEEDS

Ad 1. The recommended VOLYs for EU16 and NMC are based on the results of a questionnaire. In general the WTP increases with income (Desaigues et al., 2007) and therefore a difference in WTP between EU16 and NMC can be explained. However, for cost-benefit analyses of EU directives and policies it is recommended to use the same value for EU25. Based on the VOLY value from the poled sample this value is: 40000 euro.

This approach will also be applied for cases of morbidity. The monetary values for morbidity used within NEEDS are listed in table 4-3 below.

Health endpoints	**Euro** **per case, or per day, or per YOLL**
Increased mortality risk (infants)	3,000,000
New cases of chronic bronchitis	200,000
Increased mortality risk - $YOLL_{acute}$	60,000
Life expectancy reduction - $YOLL_{chronic}$	40,000
Respiratory Hospital Admissions (RHA)	2,000
Cardiac Hospital Admissions (CHA)	2,000
Work loss days (WLD)	295
Net Restricted Activity Days (netRADs)	130
Minor Restricted Activity Days (MRAD)	38
Lower Respiratory Symptoms (LRS)	38
LRS excluding cough	38
Cough days	38
Medication use / bronchodilator use	1

Table 4-3 Monetary values for morbidity in NEEDS

These values cover the currently used health endpoints. However, if for a health endpoint a monetary value is missing, it is possible to convert the health endpoint into DALYs (disability adjusted life years) and then use the monetary values for a life year lost per DALY.

Ad 2. Climate change values are given for different discount methods (Ramsey, Weitzman) and discount rates (0, 1, 3%) rates with equity weighing or simple summation of regional values.

4.2.4 EXIOPOL

Introduction

EXIOPOL is a European FP6 project. Aim of the project is to develop a consistent method with European and global data for the development of Environmental Extended Input Output Tables (EEIOT), see their website: http://www.feem-project.net/exiopol/. Part of the project is also to weight the environmental interventions into one environmental score, mainly linking to the results of the NEEDS project. For the modelling of interventions into damages and valuation of damages into costs economic models are used.

These economic models are developed by many different institutes. These economic models do not (always) distinguish the different steps in environmental impact assessment and valuation. At this moment there is no unified conceptual framework for the modelling of:

- interventions into midpoint indicators (problem oriented characterisation),
- midpoint indicators into damages of endpoint indicators (damage assessment),
- valuation of damages into cost.

However, within the EXIOPOL project the expertise is available reflecting the state of the art for deriving external costs, including damages to the environment. And therefore the results of this project might be relevant.

Weighting procedure

In EXIOPOL damages are valuated in monetary terms. Different methods are used to valuate damages using different types of costs, e.g. prevention costs or damage costs, and different types of values, e.g. market values (loss, repair, replacement, out-of-service) or non-market values (contingent valuation method/willingness-to-pay (CVM/WTP), hedonic pricing (HP), travel cost method (TCM).

Modelling of cause-effect chain

EXIOPOL uses the impact pathway approach, meaning a full chain modelling of interventions to external costs. No intermediate results of midpoint and/or endpoint indicators are available.

Environmental interventions covered

The impact assessment and weighting method covers approximately 17 emissions using the model Ecosense (country specific externalities).

For Forestry and agriculture the weighting is not applied on interventions (emissions or extractions) but on activity (wood harvesting, manure, fertiliser, pesticide use).

Not covered are land use, water extraction and biotic and abiotic resources. Scarcity of resources is already factored in the price. There are no values for external costs of resource depletion.

Impacts covered

The impact pathway approach encompasses the following safeguard areas:

- Human health,
- Biodiversity,
- Economic assets, like crop losses and buildings.

Geographical and temporal representation

For damages on human health the following proposal is made for *geographical representation* of costs. Based on willingness-to-pay to reduce risk (which is the theoretical foundation for deriving values of life years lost) country specific values should be used. One way to obtain approximations to these values is to take estimates from one country and then adjust them up or down based on the ratio of the purchasing power parity (PPP) adjusted per capita income in the original country of estimation to the PPP adjusted per capita income in the country to which the transfers are being made. This assumes that the income elasticity of a Value Of Life Years (VOLY) is one. An alternative value that has been used in the literature for the income elasticity is 0.8. In that case the ratio of the VOLYs would be proportional to the ratio of the PPP per capita GDPs raised to the power 0.8.

However, for a number of reasons, we do not differentiate between VOLYs within the EU. One is that damages are occurring across national boundaries. A second is that over time we expect PPP-based GDP values to converge across the EU anyway and the third is that taking different values could exacerbate environmental inequalities if decisions on where polluting sources were to be located were based on differences in the value of the VOLYs.

Outside of the EU, we take the view that it is not appropriate to apply the EU value. Doing so would imply much more resources being allocated to pollution reduction in poor countries than is in fact the case, based on social values for environmental goods and services in these countries.

For these reasons, and based on a survey carried out in NEEDS, the use of the following VOLY value is recommended:

For EU + CH, Norway uniform value of 40,000 € (of year 2000)

For other parts of the world: the above figure, adjusting according to PPP adjusted income.

For an *increase of WTP over time* due to rise in wealth due to economic growth the following proposal is made. There is evidence that monetary values for health risks for future years increase with an inter-temporal elasticity to GDP per capita growth of 0,7 to 1,0. The expectation of economic growth of the EU is somewhat like 2% per year, however in our argument below we use 1,5% per year, so we recommend a combination of 1,5% per year growth rate and elasticity of 1 (for the next 30 years). The increases would be applied to disutility costs (WTP) and opportunity costs (productivity losses), not to medical costs. So the following formula holds for the WTP-based health costs and productivity losses:

$$WTP_t = WTP_{2000} * (1.015)^{(t-2000)}$$

Where WTP_t is the required WTP in year t and WTP_{2000} is the value in the year 2000 (E.g. €40.000 in the case of the VOLY for the EU+CH and Norway).

The following method is proposed for *discounting* of effects in future. For intragenerational damage (up to ca. 30-40 years into the future) estimates of the real social time preference rate of 3% should be used. The formula for the rate of discount “i” is given as:

$i = z + ng$

Where z is the rate of pure time preference (often described as being due to impatience), g is the rate of growth of real consumption per capita, and n is the percentage fall in the additional utility derived from each percentage increase in consumption. (n is referred to as the ’elasticity of the marginal utility of consumption’).

The value of 3% proposed above for “i” can be derived from using values for the components of the social time preference rate, substantiated by current empirical evidence of: z = 1.5 (pure time preference); n = 1; g = 1.5.

For damage occurring beyond the 40 year period (intergenerational impacts), e.g. for climate change impacts, radioactive waste disposal impacts and ingestion of heavy metals and POPs, a declining discount rate system is recommended. Weitzman argues that uncertainty about future interest rates provides a strong generic rationale for using certainty-equivalent social interest rates that decline over time.

Combining the discount rate justification provided above with that of Weitzman the following discount rates are proposed:

a) For the next 30 years from the present, use a real annual interest rate of around 3%

b) For the period from about 30 to about 75 years from the present, use a within-period instantaneous interest rate of around 2%.

c) For the period from about 75 to about 300 years from the present, use a within-period instantaneous interest rate of around 1%.

d) For more than about 300 years from the present, use a within-period instantaneous interest rate of 0.1%.

These discount rates are summarized in table 4-4 below.

Year	Uplift	Discounting
0-30	1.5%	3%
31-75	1%	2%
76-300	0.5%	1%
>300	0%	0.1%

Table 4-4 Discount rates in EXIOPOL

Normalisation level

No normalisation is needed, as monetisation is applied at the endpoint level.

Degree of being operational

The method is still in development. If there are no operational factors NEEDS will be used.

4.2.5 Conclusions on integrated modelling

Virtually all integrated modelling methods use some method of monetisation to bring effects into the same evaluation framework. However, from welfare theoretical considerations, only the methods specifying effects on income have the same meaning, either specified through willingness-to-pay or through hedonic pricing methods. These integrated modelling based methods cover health effects and depletion effects, and loss in market based production, like crop losses. The economic valuation of biodiversity loss has limited foundations yet; economic valuation mostly refers to landscape aspects of ecosystems.

4.3 Midpoint modelling and evaluation

4.3.1 TRACI/BEES/NOGEPA weighting

Introduction

In this section two weighting sets are introduced that can be applied on the level of midpoint impact categories, the BEES set and NOGEPA set. The BEES method (Lippiatt, 2007) is an impact assessment method based on TRACI with additional weighting by panel developed for the US building sector. TRACI was developed by the US EPA as a characterisation method for midpoint impact categories.

http://www.epa.gov/ORD/NRML/std/sab/traci/

http://www.bfrl.nist.gov/oae/software/bees/

The NOGEPA[11] study (Huppes *et al*., 2007) focuses on the development of a single environmental indicator for an eco-efficiency indicator. The method builds on the impacts assessment based on the Dutch Problem Oriented Approach (Guinée *et al*., 2002). The overall process for arriving at such an indicator is structured according to ISO 14042 about the life cycle impact assessment. For the last step in this process, the establishment of weighting factors across environmental themes, a panel method has been chosen, involving as stakeholders the government officials involved, the industry experts and independent experts from scientific institutes.

Weighting procedure

In BEES, the aggregation of impact category scores is optional. Weighting factors across impact categories are elaborated using the procedure of panel weighting. Scores may be aggregated by weighting each impact category by its relative importance to overall environmental performance. In BEES, the set of importance weights is selected by the user (so private stated). Several alternative weight sets are provided as guidance, and may be either used directly or as a starting point for developing user-defined weights (so collective stated). The alternative weights sets are based on an EPA Science Advisory Board study, a 2006 BEES Stakeholder (Building sector) Panel’s structured judgments, and a set of equal weights, representing a spectrum of ways in which people value diverse aspects of the environment.

Modelling of cause-effect chain

The weighting of impacts is applied on the midpoint level.

[11] The Netherlands Oil and Gas Exploration and Production Association (NOGEPA)

Environmental interventions covered

Characterisation: 3000 substances, (often) with characterisation factors for more than one impact category, or more than one compartment within an impact category.

Normalisation: A new version of TRACI is expected in 2008 which will be released with new normalisation data.

The most up to date normalisation factors for the Dutch Problem Oriented Approach are based on Wegener Sleeswijk et al. (2008). The Normalisation factors are derived from are approximately 1400 substance-compartment-interventions and are available for the world and the EU27.

Impacts covered

Table 4-5 presents the midpoint impacts as defined in TRACI and BEES.

Interventions	Midpoint level (TRACI)	Midpoint level (BEES)
Substance emissions	Global warming	Global warming
	Ozone depletion	Ozone depletion
	Acidification	Acidification
	Eutrophication	Eutrophication
	Photochemical ozone formation	Photochemical ozone formation
	Human health cancer	Human health
	Human health noncancer	
	Human health criteria pollutants	Human health criteria pollutants
	Ecotoxicity	Ecotoxicity
Resource extractions	Fossil fuel depletion	Fossil fuel depletion
		Water intake
Work environment		Indoor air quality
Land use		Habitat alteration

Table 4-5 Midpoint impacts as defined in TRACI and BEES (Lippiatt, 2007).

Geographical and temporal representation

For acidification, eutrophication and photochemical oxidant formation characterisation models are valid for North America. Global models are used for ozone depletion and global warming. Toxicity models are not site specific, but exposure factors and risk assessment values are based on US EPA values.

The impact assessment is based on present time emissions. The effects are based on long time horizons, e.g. a 100 year time frame is used for GWPs. The effects are not discounted over time.

Normalisation level

The normalisation data represent the USA in 1999. An update of data is planned.

Degree of being operational

Table 4-6 shows the panel weighting factors for BEES. In the table also the weighting factors of the NOGEPA[12] panel (Huppes *et al*., 2007) are presented.

Interventions	**Midpoint level (BEES)**	**EPA Science Advisory Board**	**BEES Stake-holder Panel**	**NO-GEPA**
		%	%	
Substance emissions	Global warming	16	29	32
	Ozone depletion	5	2	5
	Acidification	5	3	6
	Eutrophication	5	6	13
	Photochemical ozone formation	6	4	8
	Human health	11		16
	Human health cancerous		8	
	Human health non-cancerous		5	
	Human health criteria pollutants	6	9	
	Ecotoxicity	11	7	
	Fresh water ecotoxicity			6
	Marine ecotoxicity			8
	Terrestrial ecotoxicity			5
Resource	Fossil fuel depletion	5	10	

[12] NOGEPA, The Netherlands Oil and Gas Exploration and Production Association involving all major oil and gas producers in The Netherlands. The study focuses on the development of a single environmental indicator an eco-efficiency indicator. The overall process for arriving at such an indicator has been structured according to ISO 14042 about the life cycle impact assessment. For the last step in this process, the establishment of weighting factors across environmental themes, a panel method has been chosen, involving as stakeholders the government officials involved, the industry experts and independent experts from scientific institutes.

extraction			
	Water intake	3	8
Work environment	Indoor air quality	11	3
Land use	Habitat alteration	16	6
Total		100	100

Table 4-6 Relative importance weights based on EPA science advisory board study and BEES Stakeholder Panel judgements (Lippiatt, 2007) and the NOGEPA panel (Huppes *et al*., 2007)

Scientific quality and acceptance of the method

The relative importance weights based on EPA Science Advisory Board are based on an iterative panel process using a procedure of pairwise comparison of impact categories known as the Analytical Hierarchy Process (AHP). (ref)

Ten of the twelve BEES impact categories were included in the SAB lists of relative importance:

- Highest-Risk Problems: global warming, habitat alteration
- High-Risk Problems: indoor air quality, ecological toxicity, human health
- Medium-Risk Problems: ozone depletion, smog, acidification, eutrophication, criteria air pollutants

The SAB did not explicitly consider fossil fuel depletion or water intake as impacts. For this exercise, fossil fuel depletion and water intake are assumed to be relatively medium-risk and low-risk problems, respectively, based on other relative importance lists.

Verbal importance rankings, such as “highest risk,” may be translated into numerical importance weights by following ASTM standard guidance provided by a Multi-attribute Decision Analysis method known as the Analytic Hierarchy Process (AHP).47 The AHP methodology suggests the following numerical comparison scale:

1 Two impacts contribute equally to the objective (in this case environmental performance)

3 Experience and judgment slightly favour one impact over another

5 Experience and judgment strongly favour one impact over another

7 One impact is favoured very strongly over another, its dominance demonstrated in practice

9 The evidence favouring one impact over another is of the highest possible order of affirmation

2,4,6,8 When compromise between values of 1, 3, 5, 7, and 9, is needed.

Through an AHP process known as pairwise comparison, numerical comparison values are assigned to each possible pair of environmental impacts. Relative importance weights can then be derived by computing the normalized eigenvector of the largest eigenvalue of the matrix of pairwise comparison values.

Social acceptance of the content

Science Advisory Board

In 1990 and again in 2000, EPA's Science Advisory Board (SAB) developed lists of the relative importance of various environmental impacts to help EPA best allocate its resources.

BEES stakeholder Panel

Several interpretations and assumptions were required in order to translate SAB findings into numerical weights for interpreting LCA-based analyses. A more direct approach to weight development would consider a closer match to the context of the application; that is, environmentally preferable purchasing in the United States based on life-cycle impact assessment results, as reported by the BEES software.

In order to develop such a weight set, NIST assembled a volunteer stakeholder panel that met at its facilities in Gaithersburg, Maryland, for a full day in May 2006. To convene the panel, invitations were sent to individuals representing one of three "voting interests:" producers (e.g., building product manufacturers), users (e.g., green building designers), and LCA experts.

Nineteen individuals participated in the panel: seven producers, seven users, and five LCA experts. These "voting interests" were adapted from the groupings ASTM International employs for developing voluntary standards, in order to promote balance and support a consensus process.

4.3.2 Soft weighting, case specific

There are other methods to arrive at an overall score, based on soft weighting. With specific case outcomes and basic notions on relative importance, one often can come to a conclusion on preferred options. If for example normalised scores on climate change are very high and on eco-toxicity very low, for all alternatives concerned, the weak weighting assumption that climate change is at least as important as ecotoxicity suffices to come to an evaluation of the alternatives involved. This method has been developed by Lundie and Huppes (1999) and has come up again recently in Rogers & Seager (2009). They use case specific normalisation, which would make general statements on the relevance climate change and eco-toxicity quite impossible.

These methods are based on weak dominance analysis and are not fit for creating quantified scores and time series.

4.4 Endpoint modelling and evaluation

4.4.1 EPS

Introduction

The impact assessment in EPS translates emissions and extractions to endpoint impact category indicators. The method is designed to be used with Monte Carlo to include the level of uncertainty in decision making. EPS produces impact category indicators at damage level expressed in monetary units. The value is derived on the basis of willingness-to-pay (WTP). (Steen, 1999)

Weighting procedure

Damage assessment is performed using WTP. The methods used to estimate WTP vary from CVM, revealed preferences and restoration costs. Different types of costs are used as an expression to avoid negative changes in indicator values: future mining costs, willingness-to-pay to avoid YOLLs or willingness-to-pay for the protection of rare species. Different types of costs are added without further weighting. The uncertainty in quantifying WTP is estimated.

Modelling of cause-effect chain

EPS uses a procedure to calculate damages on the endpoint level.

Environmental interventions covered

Characterisation: approximately 200 substances (to a large extent extraction of resources).

Impacts covered

Table 4-7 shows the endpoint indicators across which weighting is applied. The endpoint indicators are added without further weighting into the safeguard areas.

Interventions	**Endpoint level**	**Safe guard area**
Substance emissions	Life expectancy	Human health
	Severe morbidity and suffering	
	Morbidity	
	Severe nuisance	
	Nuisance	
	Crop production capacity	Ecosystem production
	Wood production capacity	
	Fish and meat production capacity	
	Base cat-ion capacity	
	Production capacity for (drinking) water	
	Share of species extinction	Biodiversity
Resource extractions	Depletion of element reserves	Abiotic stock resources
	Depletion of oil	
	Depletion of gas	
	Depletion of coal	
	Depletion of mineral reserves	

Table 4-7 Endpoint indicators in EPS across which weighting is applied (Steen, 1999)

Geographical and temporal representation

Characterisation models are global, except for biodiversity where Swedish models are used.

The impact assessment is based on present time emissions and extractions. However for the modelling of the effects a very long (indefinite or near to indefinite) time horizon is used.

The WTP is measured in today's OECD population and applied to all those, who are affected by a change. No discounting for future effects are made as future generations have the same right to a good environment as we have (Rio Convention). The choice of the default reference state is the environment of today.

Normalisation level

There is no normalisation, because this is not needed in monetisation approaches.

Degree of being operational

Table 4-8 shows the weighting factors on endpoint level for EPS (Steen, 1999). There is a large emphasis on extraction of resources.

Safeguard subject	Impact category	Category indicator	Indicator unit	Weighting factor (ELU/ indicator unit)	Uncertainty factor
Human health	Life expectancy	YOLL	Person-years	85000	3
Human health	Severe morbidity	Severe morbidity	Person-years	100000	3
Human health	Morbidity	Morbidity	Person-years	10000	3
Human health	Severe nuisance	Severe nuisance	Person-years	10000	3
Human health	Nuisance	Nuisance	Person-years	100	3
Ecosystem production capacity	Crop growth capacity	Crop	kg	0.15	2
Ecosystem production capacity	Wood growth capacity	Wood	kg	0.04	1.4
Ecosystem production capacity	Fish and meat production capacity	Fish and meat	kg	1	2
Ecosystem production capacity	Soil acidification	Base cat-ion capacity of soil	mole H+ - equivalents	0.01	2
Ecosystem production capacity	Production capacity for irrigation water	Irrigation water	kg	0.003	4
Ecosystem production capacity	Production capacity for drinking water	Drinking water	kg	0.03	6
Abiotic stock resources	Depletion of oil reserves	Fossil oil	kg	0.506	1.4
Abiotic stock resources	Depletion of coal reserves	Fossil coal	kg	0.0498	2
Abiotic stock resources	Depletion of natural gas reserves	Natural gas	kg	1.1	2
Abiotic stock resources	Depletion of Ag reserves	Ag reserves	kg of element	54000	2.2

Safeguard subject	Impact category	Category indicator	Indicator unit	Weighting factor (ELU/ indicator unit)	Uncert ainty factor
Abiotic stock resources	Depletion of Al reserves	Al reserves	kg of element	0.439	2
Abiotic stock resources	Depletion of Ar reserves	Ar reserves	kg of element	0	1
Abiotic stock resources	Depletion of As reserves	As reserves	kg of element	1490	2.2
Abiotic stock resources	Depletion of Au reserves	Au reserves	kg of element	1190000	3
Abiotic stock resources	Depletion of B reserves	B reserves	kg of element	0.05	10
Abiotic stock resources	Depletion of Ba reserves	Ba reserves	kg of element	4.45	3
Abiotic stock resources	Depletion of Bi reserves	Bi reserves	kg of element	24100	2.2
Abiotic stock resources	Depletion of Be reserves	Be reserves	kg of element	958	3
Abiotic stock resources	Depletion of Br reserves	Br reserves	kg of element	0	1
Abiotic stock resources	Depletion of Cd reserves	Cd reserves	kg of element	29100	2.2
Abiotic stock resources	Depletion of Ce reserves	Ce reserves	kg of element	45.2	3
Abiotic stock resources	Depletion of Cl reserves	Cl reserves	kg of element	0	1
Abiotic stock resources	Depletion of Co reserves	Co reserves	kg of element	256	3
Abiotic stock resources	Depletion of Cr reserves	Cr reserves	kg of element	84.9	3
Abiotic stock resources	Depletion of Cs reserves	Cs reserves	kg of element	512	3
Abiotic stock resources	Depletion of Cu reserves	Cu reserves	kg of element	208	3
Abiotic stock resources	Depletion of Dy reserves	Dy reserves	kg of element	1020	3
Abiotic stock resources	Depletion of Er reserves	Er reserves	kg of element	1410	3
Abiotic stock resources	Depletion of Eu reserves	Eu reserves	kg of element	3130	3

Safeguard subject	Impact category	Category indicator	Indicator unit	Weighting factor (ELU/ indicator unit)	Uncertainty factor
Abiotic stock resources	Depletion of F reserves	F reserves	kg of element	4.86	3
Abiotic stock resources	Depletion of Fe reserves	Fe reserves	kg of element	0.961	2.2
Abiotic stock resources	Depletion of Ga reserves	Ga reserves	kg of element	212	3
Abiotic stock resources	Depletion of Gd reserves	Gd reserves	kg of element	1060	3
Abiotic stock resources	Depletion of Ge reserves	Ge reserves	kg of element	2120	3
Abiotic stock resources	Depletion of H reserves	H reserves	kg of element	0	1
Abiotic stock resources	Depletion of He reserves	He reserves	kg of element	0	1
Abiotic stock resources	Depletion of Hf reserves	Hf reserves	kg of element	512	3
Abiotic stock resources	Depletion of Hg reserves	Hg reserves	kg of element	53000	2.2
Abiotic stock resources	Depletion of Ho reserves	Ho reserves	kg of element	4790	3
Abiotic stock resources	Depletion of I reserves	I reserves	kg of element	0	1
Abiotic stock resources	Depletion of In reserves	In reserves	kg of element	48700	3
Abiotic stock resources	Depletion of Ir reserves	Ir reserves	kg of element	59400000	3
Abiotic stock resources	Depletion of K reserves	K reserves	kg of element	0.01	10
Abiotic stock resources	Depletion of La reserves	La reserves	kg of element	92	3
Abiotic stock resources	Depletion of Li reserves	Li reserves	kg of element	0.1	10
Abiotic stock resources	Depletion of Lu reserves	Lu reserves	kg of element	11000	3
Abiotic stock resources	Depletion of Mg reserves	Mg reserves	kg of element	0	1
Abiotic stock resources	Depletion of Mn reserves	Mn reserves	kg of element	5.64	3

Safeguard subject	Impact category	Category indicator	Indicator unit	Weighting factor (ELU/ indicator unit)	Uncert ainty factor
Abiotic stock resources	Depletion of Mo reserves	Mo reserves	kg of element	2120	3
Abiotic stock resources	Depletion of N reserves	N reserves	kg of element	0	1
Abiotic stock resources	Depletion of Na reserves	Na reserves	kg of element	0	1
Abiotic stock resources	Depletion of Nb reserves	Nb reserves	kg of element	114	3
Abiotic stock resources	Depletion of Nd reserves	Nd reserves	kg of element	115	3
Abiotic stock resources	Depletion of Ne reserves	Ne reserves	kg of element	0	1
Abiotic stock resources	Depletion of Ni reserves	Ni reserves	kg of element	160	2.2
Abiotic stock resources	Depletion of O reserves	O reserves	kg of element	0	1
Abiotic stock resources	Depletion of Os reserves	Os reserves	kg of element	59400000	3
Abiotic stock resources	Depletion of P reserves	P reserves	kg of element	4.47	3
Abiotic stock resources	Depletion of Pb reserves	Pb reserves	kg of element	175	2.2
Abiotic stock resources	Depletion of Pd reserves	Pd reserves	kg of element	7430000	3
Abiotic stock resources	Depletion of Pr reserves	Pr reserves	kg of element	471	3
Abiotic stock resources	Depletion of Pt reserves	Pt reserves	kg of element	7430000	3
Abiotic stock resources	Depletion of Rb reserves	Rb reserves	kg of element	27	3
Abiotic stock resources	Depletion of Re reserves	Re reserves	kg of element	7430000	3
Abiotic stock resources	Depletion of Rh reserves	Rh reserves	kg of element	49500000	3
Abiotic stock resources	Depletion of Ru reserves	Ru reserves	kg of element	29700000	3
Abiotic stock resources	Depletion of S reserves	S reserves	kg of element	0.1	5

Safeguard subject	Impact category	Category indicator	Indicator unit	Weighting factor (ELU/ indicator unit)	Uncertainty factor
Abiotic stock resources	Depletion of Sb reserves	Sb reserves	kg of element	9580	3
Abiotic stock resources	Depletion of Sc reserves	Sc reserves	kg of element	424	3
Abiotic stock resources	Depletion of Se reserves	Se reserves	kg of element	35800	3
Abiotic stock resources	Depletion of Sm reserves	Sm reserves	kg of element	632	3
Abiotic stock resources	Depletion of Sn reserves	Sn reserves	kg of element	1190	2.2
Abiotic stock resources	Depletion of Sr reserves	Sr reserves	kg of element	9.4	3
Abiotic stock resources	Depletion of Ta reserves	Ta reserves	kg of element	1980	3
Abiotic stock resources	Depletion of Tb reserves	Tb reserves	kg of element	5940	3
Abiotic stock resources	Depletion of Te reserves	Te reserves	kg of element	594000	3
Abiotic stock resources	Depletion of Th reserves	Th reserves	kg of element	288	3
Abiotic stock resources	Depletion of Ti reserves	Ti reserves	kg of element	0.953	3
Abiotic stock resources	Depletion of Tl reserves	Tl reserves	kg of element	3960	3
Abiotic stock resources	Depletion of Tm reserves	Tm reserves	kg of element	9900	3
Abiotic stock resources	Depletion of U reserves	U reserves	kg of element	1190	3
Abiotic stock resources	Depletion of V reserves	V reserves	kg of element	56	3
Abiotic stock resources	Depletion of W reserves	W reserves	kg of element	2120	
Abiotic stock resources	Depletion of Y reserves	Y reserves	kg of element	143	3
Abiotic stock resources	Depletion of Yb reserves	Yb reserves	kg of element	1980	3
Abiotic stock resources	Depletion of Zn reserves	Zn reserves	kg of element	57.1	2.2

Safeguard subject	Impact category	Category indicator	Indicator unit	Weighting factor (ELU/ indicator unit)	Uncertainty factor
Abiotic stock resources	Depletion of Zr reserves	Zr reserves	kg of element	12.5	3
Biodiversity	Species extinction	NEX	Dimension-less	1.10E+11	3

Table 4-8 Weighting factors for endpoint category indicators from EPS (Steen, 1999).

Scientific quality and acceptance of the method

For some category indicators (e.g. changes in production capacities), the market price is used to estimate WTP. The goal that was set up for the EPS system requires the result to be understandable for the designer. This speaks for a choice of a monetary value that is familiar to the designer: the price the buyer has to pay. Variations in market prices are included in the uncertainty measure of the weighting factor.

A method often used to estimate non-market environmental values is the CVM method. CVM stands for 'Contingent Valuation Method' and is widely used to measure WTP in various groups to various concepts, which are described to them. The CVM technique is based on interviews and is following a special procedure. In the EPS-system the CVM technique is used for category indicators of morbidity and nuisance and for recreation values. The precision of results obtained via the CVM technique varies.

For the WTP for indicators of the safe guard subject ‘abiotic stock resources', a market scenario was created, where the production cost of substances similar to the abiotic stock resources is used as an estimate of WTP. It is assumed that some of these stock resource materials always will be produced even if the volume decreases.

Treatment of uncertainty

Uncertainties in WTP for example due to fluctuation in market prices, regional differences and differences in CVM techniques are included as an uncertainty measure of the weighting factor.

4.4.2 LIME

Introduction

The LIME (Life cycle Impact assessment Method based on Endpoints) method is developed in Japan and mainly applied in Japan (Itsubo et al., 2004). LIME is a damage oriented impact assessment method. The impact assessment has a two step procedure. Interventions are translated into impact category indicators on the midpoint level. These midpoint indicators are next translated into damages on the endpoint level. Weighting of the damages is based on WTP for avoiding damages of every safeguard subjects.

Weighting procedure

Weighting of the damages is based on WTP for avoiding damages. Values for WTP are derived by comparison of importance among the four safeguard subjects by applying conjoint analysis (Itsubo et al., 2004).

> conjoint analysis
>
> Conjoint analysis is a general name for the methods of assessing individuals' preferences for each of a number of attributes (in this case safeguard areas). A choice-based type of questionnaire is prepared for the interview with the respondents selected by random sampling (400 respondents) WTP per quota can be determined by statistical simulation based on the random utility theory reflecting the responses to the questionnaires by random sampling.

Modelling of cause-effect chain

LIME uses a two step procedure to calculate impact category indicators on the midpoint level by characterisation and endpoint level by damage assessment.

Environmental interventions covered

Characterisation: 1000 substances, (often) with characterisation factors for more than one impact category, or more than one compartment within an impact category

Normalisation: A reference situation of the environmental state in Japan is used to derive the WTP. This reference situation is based on about 200 substance-impact-interventions.

Impacts covered

Table 4-9 shows the midpoint and endpoint impacts that are taken into account in LIME. The actual weighting is applied on 4 safeguard areas.

Midpoint level	Endpoint level	Safeguard subjects
Urban air pollution	cancer	Human health
Indoor air pollution	Respiratory disease	
Human toxicity	cataract	
noise	Thermal stress	
Ozone layer depletion	Infectious diseases	
Climate change	starvation	
Photochemical oxidant formation	disaster	
Ecotoxicity	Terrestrial species	Biodiversity
Eutrophication	Aquatic species	
Acidification	Crop	Primary production
Waste	Forestry	Social assets
Land use	Fishery	
Mineral resource	Land loss	
Fossil fuels	Energy	
Biotic resource	Materials, resources	

Table 4-9 midpoint and endpoint impacts and safe guard areas of LIME

Geographical and temporal representation

Characterisation factors: Japan, Global (Climate change, ozone layer depletion and resource depletion)

The impact assessment is based on present time emissions and extractions. However for the modelling of the effects different time horizons are used. Future effects are not discounted over time.

The WTP is derived from respondents in Japan only.

Normalisation level

No normalisation is needed, as monetisation is applied at the endpoint level.

Degree of being operational

Table 4-10 shows the monetised and otherwise dimensionless weighting factors as derived in LIME.

	normalisation value		weighting factor	WTP for annual damage	weighting factor
	Japan	unit	JY / a unit	JY / a	
human health	5.43E+05	Daly	9.70E+06	5.27E+12	0.33
social assets	2.29E+06	million JY	1.00E+06	2.29E+12	0.14
primary production	1.94E+08	Dry-ton	2.02E+04	3.92E+12	0.25
biodiversity	9.23E-01	EINES	4.80E+12	4.43E+12	0.28
			total	**1.59E+13**	

Table 4-10 weighting factors across safe guard areas is used in LIME

As stated before, normalisation is not necessary when the monetised weighting factors are used. Normalisation value refers to the reference situation of the environmental state in Japan and is used in the valuation procedure to derive the WTP.

Scientific quality and acceptance of the method

In Conjoint analysis a weighting set is developed for a total set of problems (e.g. human health, biodiversity, primary production, social assets). So in a questionnaire the respondents are confronted with scenarios addressing the total of environmental problems. Subsequently, this weighting set is translated into monetary terms. Whilst in CVM the monetary valuation of a problem is derived for each problem independently, often using different methods/questionnaires. It can be said that the conjoint analysis, which reflects the weighting results among endpoints on a single index, is closer to the idea of ISO for weighting than the approach of CVM, which eventually gives a single index by independently evaluating the environmental values of each endpoint. As a consequence Conjoint analysis can be used to derive both dimensionless and monetised weighting factors.

4.4.3 ReCiPe-weighting-using-damage-cost

Introduction

ReCiPe is a follow up of Eco-indicator99 and CML2002 methods. It integrates and harmonises a midpoint and endpoint approach in a consistent framework. All impact categories have been redeveloped and updated. In a sub project “ReCiPe weighting” attention is given to three weighting methods:

1) For endpoints a manual for panel weighting is available but no operational generic weighting sets have been developed (Pré, 2009 in prep).

2) For endpoints a monetisation on the basis of damage costs is provided (Heijungs, 2008). The method is described in this section 4.4.
3) For the midpoints a monetisation on the basis of prevention costs is provided (de Bruyn *et al*., 2007). This method is described section 4.6.

Weighting procedure

Damage costs are estimated at the endpoint level. The values are based on literature review using various valuation techniques. Weighting should be applied on the scores of the impact category indicators that are not normalised.

Three endpoints are to be subject to valuation:

- human health, with the category indicator damage to human health measured in terms of DALY;
- ecosystem quality, with the category indicator damage to ecosystem diversity measured in terms of PDF*time;
- resource availability, with the category indicator damage to resource costs measured in terms of surplus costs.

Modelling of cause-effect chain

ReCiPe uses a two step procedure to calculate impact category indicators on the midpoint and endpoint level.

Environmental interventions covered

Characterisation: 3000 substances, (often) with characterisation factors for more than one impact category, or more than one compartment within an impact category.

Normalisation: 1370 substance-compartment-interventions based on Wegener Sleeswijk et al. (2008).

Normalisation is only necessary in case the weighting across impact categories is based on panel weighting. For weighting based on damage cost the cost should be applied on non-normalised impacts.

Impacts covered

Table 4-11 shows the midpoint and endpoint impacts as defined in ReCiPe.

Midpoint level	Endpoint level
Ionising radiation	Human health
Ozone depletion	
Human toxicity	
Photochemical oxidant formation	
Particulate matter formation	
Climate change	Ecosystem quality (biodiversity)
Terrestrial acidification	
Terrestrial ecotoxicity	
Fresh water eutrophication	
Fresh water ecotoxicity	
Marine eutrophication	
Marine ecotoxicity	
Agricultural land occupation	Resources
Urban land occupation	
Natural land transformation	
Depletion of fossil fuel resources	
Depletion of mineral resources	
Depletion of fresh water resources	

Table 4-11 Midpoint and endpoint impacts (safe guard areas) in ReCiPe

Geographical and temporal representation

Characterisation factors: Europe, global (climate change, ozone layer depletion and resource depletion)

The impact assessment is based on present time emissions and extractions. However for the modelling of the effects different time horizons are used of 20, years, 100 years or indefinite, depending on the cultural perspective.

Cultural perspectives are used to distinguish three different sets of subjective choices; Hierarchist, Individualist and Egalitarian. A ‘short’ time horizon is for example used in the individualist perspective, whilst a long/indefinite time horizon is used for the other perspectives.

Normalisation level

Normalisation of impact scores is only relevant in case where panel weighting is applied. Normalisation data are available for the World and Europe in the year 2000. Normalisation for land transformation and fresh water depletion are not included. In case monetised weighting factors are used these factors should be applied to non-normalised impact scores.

Degree of being operational

Table 4-12 gives weighting factors based on WTP to avoid damages on the endpoint impacts derived from a literature survey for ReCiPe_CML (Heijungs, 2008).

Endpoint	indicator	unit	Weighting factor	Unit
Human health	DALY	yr	60.000	$/yr
Ecosystem quality	PDF	M2.yr	175.000.000.000	$/yr
Resource availability	Surplus cost	$	1	$/$

Table 4-12 Weighting factors based on WTP optional to use ReCiPe_CML (Heijungs et al., 2008)

This report (Heijungs, 2008) discusses weighting with an approach in which environmental damage is translated into monetary terms. Different interpretations of terms like "damage" and "cost" are discussed, and different techniques for estimating these costs are presented. A literature review of the estimates of monetary damage for the three endpoints of ReCiPe then leads to a summary table of weighting factors. At the same time, it is concluded that there is so much uncertainty, and that the uncertainty is moreover unknown itself, that users of these weighting factors are advised to be very careful.

4.4.4 Ecoindicator99

Introduction

The Ecoindicator99 was originally developed for the purpose of 'ecodesign'. Designers were deemed unable to work with 10-20 different impact indicator results as in the problem oriented approach (Guinée et al., 2002). Therefore the aim was to simplify interpretation and weighting of results by reducing the number of midpoint impact categories into 3 endpoint damage categories. The Ecoindicator99 expresses the environmental impact in one overall environmental impact score using a weighting set based on panel weighting. The two 'Dutch' impact assessment methods Ecoindicator99 (Goedkoop and Spriensma, 2000) and CML2002 (Guinée et al., 2002) are combined into one method ReCiPe, which is discussed in sections 4.4.3 and 4.6.1).

www.pre.nl/ecoindicator99

Weighting procedure

In Ecoindicator99, the aggregation of impact category scores is optional. Weighting factors across impact categories are elaborated using the procedure of panel weighting. Three scenarios perspectives are developed as a way to deal with subjective choices on endpoint level; Hierarchist, Individualist and Egalitarian. The authors recommend to use the Hierarchist version as the default method. The other value systems can be used as a form of sensitivity analysis.

Modelling of cause-effect chain

Scientific models are used to derive 'characterisation' factors. These factors are used to translate interventions into damages to endpoint categories. The weighting of impacts is applied on the endpoint level. Impacts on midpoint level are not separately reported.

Environmental interventions covered

Characterisation: approximately 400 substances, (often) with characterisation factors for more than one impact category, or more than one compartment within an impact category.

Normalisation: Normalisation is based on European emissions to air, water and soil (appr. 120) and extractions (appr. 10).

Impacts covered

Table 4-13 presents the endpoint impacts as defined in Ecoindicator99.

Interventions	Midpoint level	Endpoint level
Substance emissions	Ionising radiation	Damage to human health
	Ozone layer depletion	
	Acidification/Eutrophication	
	Carcinogenic effects	
	Respiratory organic effects	
	Respiratory inorganic effects	
	Climate change	Damage to ecosystems
	Ecotoxicity	
Land use and land conversion	biodiversity	
Resource extraction	Depletion of fossil fuel resources	Damage to resources
	Depletion of mineral resources	

Table 4-13 Midpoint and endpoint impacts in Ecoindicator99

Note that midpoint results are not reported separately. Interventions are directly translated into endpoint damages taking into account several midpoint mechanisms.

Geographical and temporal representation

Characterisation factors: Europe, global (climate change, ozone layer depletion and resource depletion)

The impact assessment is based on present time emissions and extractions.

Cultural perspectives are used to distinguish three different sets of subjective choices; Hierarchist, Individualist and Egalitarian. A short time horizon is for example used in the individualist perspective, whilst a long/indefinite time horizon is used for the other perspectives.

Normalisation level

Normalisation data are available for Europe mid nineties.

Degree of being operational

Table 4-14 gives weighting factors used in the Ecoindicator99. The weighting factors are based on the panel procedure. A questionnaire was sent to 365 respondents (22% responses), all were members of the Swiss discussion platform on LCA. The questionnaire also was aimed to distinguish between different cultural perspectives and ecocentric and anthropocentric attitudes.

Endpoint	indicator	unit	Weighting factor			
			average	individualist	egalitarian	hierarchist
Human health	DALY	yr	0.4	0.55	0.3	0.3
Ecosystem quality	PDF	% plant species M2.yr	0.4	0.25	0.5	0.4
Resources	Surplus energy	MJ surplus energy	0.2	0.2	0.2	0.3

Table 4-14 Weighting factors of the Ecoindicator99 per cultural perspective (Goedkoop and Spriensma, 2000)

The authors recommend to use the hierarchist version as the default method. The other value systems can be used as a form of sensitivity analysis.

Scientific quality and acceptance of the method

The Ecoindicator99 method is widely used by LCA practitioners. A large advantage of the Eco-indicator 99 is that category indicators are defined at the endpoint level, giving them greater environmental relevance. As it is this level that ultimately matters to society, the object of the weighting procedure is more immediate. The major uncertainties associated with modelling from midpoints to endpoints constitute a serious drawback, however.

4.5 Aggregation without evaluation: Distance-to-Target type

4.5.1 Ecological Scarcity method (Swiss Ecopoints 2006)

Introduction

In the Ecological Scarcity Method weighting factors are derived for different emissions into air, water and topsoil/groundwater. Also weighting factors are derived for the extraction of energy resources and fresh water. These eco-factors are based on the annual actual flows (current flows) and on the annual flow considered as critical (critical flows) in a defined area (country or region) over a specific time horizon. The critical flows are deduced from environmental policy targets. An implicit weighting takes place in accepting the various goals of the environmental policy and considering the goals of equal importance. (Frischknecht et al., 2008)

Weighting procedure

The Ecological Scarcity Method is a Distance-to-target (DTT) Method. Emissions that are most distant from stated critical loads receive the largest weights. The critical loads are derived from stated policy targets. As a consequence the Ecological Scarcity Method is region and time specific.

Modelling of cause-effect chain

In the Distance-to-target approach of the Ecological Scarcity Method the targets are set on the level of the interventions (emission or extraction). The endpoints are indirectly considered by the targets set by policy. In setting these targets, it is not only considerations of importance of effects which determine the target level but also other considerations like technical, economic, social ones.

Environmental interventions covered

The weighting method covers more than 400 substances.

Impacts covered

Table 4-15 presents the interventions and midpoint impacts on which level weighting factors are defined in the Ecoscarcity method (Frischknecht et al., 2008).

Interventions	Intervention level	Midpoint level
Substance emissions		Global warming
		Ozone depletion
		Acidification (SO2, HCl, HF etc, excl NH3, NOx)
	Photochemical ozone formation: NMVOC	
	Respiratory effects: PM10, PM2.5, black carbon	
	Air emissions: NOx, NH3, Pb, Cd, Zn, Hg, benzene, dioxins and furans	
	Surface water emissions: COD, Phosphorus, N-total, As, Hg, Cd, Pb, Cr, Cu, Zn, Ni, PAH, benzo(a)pyrene, AOX,	

	chloroform, radioactive emissions and endocrine disruptors	
	Sea water emissions: radionuclides	
	Ground water emissions: NO3-	
	Soil emissions: pesticides, Cd, Pb, Cu, Zn	
	Waste: landfilled municipal (reactive) wastes, hazardous wastes (stored underground), radioactive wastes	
	Endocrine disruptors	
Resource extractions	Primary energy resources	
	Water consumption	
	Gravel consumption	
Land use		Biodiversity losses due to land occupation

Table 4-15 Interventions and midpoint impacts on which level weighting factors are defined in the Ecoscarcity method (Frischknecht et al., 2008).

The ecological scarcity method partly is based on characterisation models used in LCA (e.g. global warming, ozone depletion, acidification, primary energy resources and land use). Other interventions are assessed individually (e.g. NH3, heavy metals) or as a group (e.g. NMVOC).

Geographical and temporal representation

The Distance-to-target method is by definition region and time specific. The method described in this paragraph refers to the Swiss situation (based on Swiss policy targets). Various other Ecological Scarcity Methods are derived for other countries, like the Netherlands, Belgium, Japan.

The actual flows refer to the 2004 situation, the critical flow correspond to policy objectives in 2005 and reflect targets to be achieved within 5 to 20 years.

Normalisation level

Normalisation data are available for the reference situation Switzerland, 2004.

Degree of being operational

In the original report (Frischknecht et al., 2008) there is a list available of approx. 40 ecofactors for individual substances or substance groups.

Scientific quality and acceptance of the method

The Ecofactor is for each intervention defined as:

$$\text{Ecofactor} = K * (1 * UBP/F_n) * (F/F_k)^2 * c$$

K = characterization factor

Fn = Normalisation total: the actual yearly intervention (in Switzerland)

F = Actual intervention: the actual yearly intervention (in a region). On the country level it will be equal to Fn (Switzerland)

Fk = Critical load: the critical yearly intervention (in a region)

C = Constante (10^{12} / year)

UBP = (Umweltbelastungspunkt) the unit of the weighted result

The formula presented can be separated in the 3 elements of the ISO standard 14042:

1. Characterisation is represented by K and is optional for some impact indicators
2. Normalisation is performed by ($1/F_n$).
3. the weighting factor is give by $(F/F_k)^2$

The description given above explains how the ‘weighting factor’ is derived. However, note that in the Ecopoint method the actual weighting across problems is missing since all the critical loads (Fk) are considered of equal importance. Besides this missing of the actual weighting the DTT methods also have other flaws like the limited geographical and temporal representation. Furthermore the critical loads (targets) are not only determined by importance of effects but also by other considerations like technical, economic, social ones.

4.5.2 EDIP97-weighting

Introduction

There are two version of the LCIA of EDIP, namely EDIP97 (Wenzel et al., 1997, Hauschild and Wenzel, 1998) and EDIP2003 (Hauschild and Potting, 2003, Potting and Hauschild, 2003). EDIP2003 is the follow up of EDIP97 methodology with inclusion of spatially differentiated impact assessment of non-global impact categories at midpoint level. EDIP2003 is not an update of EDIP97 but a spatially differentiated alternative. Only EDIP97 has weighting of environmental impacts included in the methodology and focuses on the global level. Therefore only EDIP97 is elaborated below.

The impact assessment on EDIP97 supports emission-related impact categories at midpoint level, resources and working environment. The method includes normalisation and weighting of environmental impacts based on political environmental targets.

Weighting procedure

Weighting is based a Distance-to-Target approach using political reduction targets (only binding targets) for environmental impacts and working environment impacts, and supply horizon for resources.

Modelling of cause-effect chain

In EDIP the impact assessment is performed on the Midpoint level. There is no modelling of impacts to endpoint impacts. Therefore the weighting is performed across midpoint impacts.

Environmental interventions covered

Characterisation: 3000 substances, (often) with characterisation factors for more than one impact category, or more than one compartment within an impact category

Normalisation: The number of interventions taken into account to estimate normalisation factors is unknown.

Impacts covered

Table 4-16 shows the impacts that are covered in EDIP97.

Interventions	**Midpoint level**
Substance emissions	Global warming
	Ozone depletion
	Acidification
	Eutrophication
	Photochemical ozone formation
	Human toxicity (3 sub categories)
	Ecotoxicity (3 sub categories)
Resource extractions	Resource depletion
Working environment	Working environment (7 sub categories)

Table 4-16 Impacts on midpoint level that are covered in EDIP97 (Wenzel et al., 1997; Hauschild and Wenzel, 1998).

Geographical and temporal representation

Characterisation: The characterisation factors are based on global models and present time interventions taking into account long term effects. The future effects are not discounted over time.

Weighting: The weighting factors are derived using international policy targets. The weighting factors are representative for a limited time horizon, reference year 1994-target year 2004 (see section “Scientific quality “).

Normalisation level

Normalisation data are available for two reference situations World, 1994 and Europe, 1994. There are no normalisation data for resources and working environment.

Degree of being operational

Table 4-17 shows the weighting factors based on the Distance-to-target method as derived for EDIP97 (Wenzel et al., 1997; Hauschild and Wenzel, 1998). Normalisation and weighting factors for working environment and resources seem to be not available.

Impact category	Unit	Normalization reference	Reference year	Weighting factor	Reference year	Reference region
Environmental impacts						
Global						
Global warming	kg CO_2-eq/pers/year	8.70E+03	1994	1.1	2004	World
Ozone depletion	kg CFC-11-eq/pers/ar	0.103	1994	63	2004	World
Regional and local						
Photochem oz. Form.	kg C_2H_4-eq/pers/year	25	1994	1.3	2004	EU-15
Acidification	kg SO_2-eq/pers/year	74	1994	1.3	2004	EU-15
Nutrient enrichment	kg NO_3^--eq/pers/year	119	1994	1.2	2004	EU-15
-N-equivalents	kg N-eq/pers/year	24	1994	1.4	2004	EU-15
-P-equivalents	kg P-eq/pers/year	0.4	1994	1	2004	EU-15
Ecotoxicity						
- water acute	m^3 water/pers/year	2.91E+04	1994	1.1	2004	EU-15
- water chronic	m^3 water/pers/year	3.52E+05	1994	1.2	2004	EU-15
- soil chronic	m^3 soil/pers/year	9.64E+05	1994	1	2004	EU-15
Human toxicity						
- via air	m^3 air/pers/year	3.06E+09	1994	1.1	2004	EU-15
- via water	m^3 water/pers/year	5.22E+04	1994	1.3	2004	EU-15
- via soil	m^3 soil/pers/year	1.27E+02	1994	1.2	2004	EU-15
Waste						
-bulk Waste	kg/pers/year	1350	1991	1.1	2000	Denmark
-hazardous waste	kg/pers/year	20.7	1991	1.1	2000	Denmark
-slag and ashes	kg/pers/year	350	1991	1.1	2000	Denmark
-nuclear waste	kg/pers/year	0.035	1989	1.1	2000	Sweden

Table 4-17 Weighting factors based on the Distance-to-target method as derived for EDIP97 (Wenzel et al., 1997; Hauschild and Wenzel, 1998).

Scientific quality and acceptance of the method

Weighting factors are determined by a method called “distance-to-target”. The weighting factor is defined for each environmental impact category as the ratio between the actual impact and the target impact:

WF = Actual impact in reference year x/ Target impact in year y (future)

The greater the difference between the actual impact and the target impact, the higher the weighting factor. There are several different options for target impacts to choose from:

- carrying capacity and sustainability
- political targets
- politically determined environmental space

In EDIP international (and national) political targets are used to derive the weighting factors. The size of the weighting factor will depend on the choice of the “reference year” and the “target year”. The weighting factors thus need regular updating and will change over time. The present weighting factors are based on reference year 1994 and target year 2004. There has been no update of weighting factors since.

4.6 Aggregation without evaluation: Cost methods

4.6.1 ReCiPe-weighting using abatement cost

Introduction

ReCiPe is a follow up of Eco-indicator99 and CML2002 methods. It integrates and harmonises a midpoint and endpoint approach in a consistent framework. All impact categories have been redeveloped and updated. In a sub project “ReCiPe weighting” attention is given to three weighting methods:

1) For endpoints a manual for panel weighting is available but no operational generic weighting sets have been developed (Pré, 2009 in prep).
2) For endpoints a monetisation on the basis of damage costs is provided (Heijungs, 2008). The method is described in section 4.4.
3) For the midpoints a monetisation on the basis of prevention costs is provided (de Bruyn *et al.*, 2007). This method is described this section 4.6.

ReCiPe is the follow up and combination of the IA methods Ecoindicator99 and CML2002. To a substantial degree the NOGEPA weighting method also can be applied on ReCiPe midpoint level, as is the case with BEES, all with conversion problems due to different normalisation methods as have been applied. ReCiPe-Pré, describes a panel weighting procedure but no operational weighting factors have been developed. For other IA methods, no directly linked operational weighting factors are available. Examples are the following: impact2002+, LUCAS, TRACI [in BEES]. These IA methods are comparable to or overlap with the problem oriented approach in ReCiPe and Bees.

Weighting procedure

Prevention cost on the level of midpoint indicators are derived based on MACC (Marginal Abatement Cost Curves) and set policy targets. Weighting should be applied on the non-normalised impact category indicators.

Modelling of cause-effect chain

For ReCiPe-weighting-using-abatement-cost impact category indicators are calculated on the midpoint level.

Environmental interventions covered

Characterisation: 3000 substances, (often) with characterisation factors for more than one impact category, or more than one compartment within an impact category.

Normalisation: 1370 substance-compartment-interventions

Normalisation is only necessary in case the weighting across impact categories is based on panel weighting. For weighting based on abatement cost the cost should be applied on non-normalised impacts.

Impacts covered

Table 4-18 shows the midpoint and endpoint impacts as defined in ReCiPe.

Midpoint level	Endpoint level
Ionising radiation	Human health
Ozone depletion	
Human toxicity	
Photochemical oxidant formation	
Particulate matter formation	
Climate change	Ecosystem quality (biodiversity)
Terrestrial acidification	
Terrestrial ecotoxicity	
Fresh water eutrophication	
Fresh water ecotoxicity	
Marine eutrophication	
Marine ecotoxicity	
Agricultural land occupation	Resources
Urban land occupation	
Natural land transformation	
Depletion of fossil fuel resources	
Depletion of mineral resources	
Depletion of fresh water resources	

Table 4-18 Midpoint and endpoint impacts (safe guard areas) in ReCiPe

Geographical and temporal representation

Characterisation factors: Europe, Global (Climate change, ozone layer depletion and resource depletion)

The impact assessment is based on present time emissions and extractions. However for the modelling of the effects different time horizons are used of 20, years, 100 years or indefinite, depending on the cultural perspective (see also section on Ecoindicator99). There is no discounting of effects over time.

Cultural perspectives are used to distinguish three different sets of subjective choices (see also section on Ecoindicator99).

All monetised values are based on Dutch policy targets and therefore representative for the Netherlands for the period 2000-2010. Further work on target based reduction cost values are in development but not operational yet.

Normalisation level

Normalisation of impact scores is only relevant in case panel weighting is applied. Normalisation data are available for the World and Europe in the year 2000.

Normalisation for land transformation and fresh water depletion are not included. In case monetised weighting factors are used these factors should be applied to non-normalised impact scores.

Degree of being operational

In De Bruyn et al. (2007) a set of monetised values is presented, see table 4-19. All monetised values are representative for the Netherlands for the period 2000-2010. Because of this temporal representiveness the present weighting factors are outdated. It is recommended not to use these outdated weighting factors (personal comment de Bruyn). Further work on target based reduction cost values are in development but not operational yet.

impact category	characterisation factor (Guinée et al., 2002)	monetised value (euro/kg eq.)		price level	policy target year	region
		based on sustainability target	Based on policy target			
Global warming	GWP100 (kg CO2 eq.)	0.091	0.05	appr. 2000	2010	Netherlands
Ozone layer depletion	ODPsteady state (kg CFC-11 eq.)	5724.69	30	appr. 2000	infinite	Netherlands
Photochemical oxidation	POCP (kg ethylene eq.)	4.402	2	appr. 2000	2010	Netherlands
Acidification	AP (kg SO2 eq.)	2.723	4	appr. 2000	2010	Netherlands
Eutrophication	EP (kg PO4 3- eq)	54.454	9	appr. 2000	2002	Netherlands
Human toxicity	HTPinf (1,4 db eq.)	0.048	0.09	appr. 2000	?	Netherlands
Fresh water ecotoxicity	FAETPinf (1,4 db eq.)	0.048	0.03	appr. 2000	?	Netherlands
Terrestrial ecotoxicity	TETPinf (1,4 db eq.)	0.048	0.06	appr. 2000	?	Netherlands

Table 4-19 Weighting factors based on abatement cost and policy targets for midpoint impacts ReCiPe_CE (de Bruyn et al., 2007) It is recommended not to use these outdated weighting factors (personal comment De Bruyn).

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Scientific quality and acceptance of the method

In De Bruyn et al. (2007) the following considerations are given for the development and use of the weighting factors based on abatement cost.

To construct a set of weighing prices for LCA using prevention costs involves three steps:

1. Construct the marginal cost curve.
2. Select targets and determine where targets and cost curves meet to find the marginal costs for the pollutant.
3. Translate the information on marginal costs for certain substances into the LCA-framework at midpoint level.

The weighting scheme based on the prevention costs gives, in essence, the economic valuation of policy targets given present targets and technologies for abatement. Therefore the values are policy (target and time horizon) and technology dependent (cost and time horizon) and will vary between regions and in time.

Targets can be set by policy, sustainability targets or expert panels.

The amount of emission reductions to be achieved is subject to the future growth in emissions. In order to determine the marginal costs of policy goals one must hence use scenarios that estimate future emissions in absence of policy measures.

Due to economies of scale and learning, techniques become more efficient and/or cheaper over time. If policy targets are formulated for years far in the future, costs must be corrected for the technological improvements.

Technologies and measures may affect different emissions simultaneously. In this case, costs cannot be simply attributed to a specific emission reduction. Hence, they must be allocated to all reduced emissions.

Targets can be set for <u>individual substances</u> or at <u>midpoint</u> level (e.g. global warming, acidification). Translation of reduction cost prices from substances to midpoints or vice versa may be troublesome because policy considerations and characterisation models may not match.

In relation to reproducibility of the weighting factors based on Marginal Abatement Cost Curves (MACC) De Bruyn et al. (2007) mention several drawbacks, like:

- MACC-studies are not harmonized and results from various studies are hardly comparable; mostly due to the used techniques and price levels of the techniques.
- MACC-studies in the past have given an overestimation of the costs of pollution reduction; international literature reports a difference of a factor 2-5.

As long as there is no bias in the overestimation this problem might not be relevant for shadow prices used for relative weighting. The prevention costs used for weighting are solely used for their relative information instead of the absolute values.

- MACC is difficult to obtain if techniques reduce more than one pollutant.

4.7 Aggregation without evaluation: one issue methods

4.7.1 Ecological Footprint

Introduction

The concept of the Ecological Footprint was developed by William Rees and Mathias Wackernagel in the early '90s. The Ecological Footprint is a resource accounting tool that measures how much biologically productive land and water area a population uses to produce the resources it consumes and to absorb the CO2 it generates by consumption of fossil fuels (and nuclear energy[13]). (Wackernagel et al., 2005). The most widely used methodology for calculating national Footprints are the National Footprint Accounts by the Global Footprint Network.

www.footprintnetwork.org

Weighting procedure

In the Ecological Footprint there is no explicit value-based weighting. The Ecological Footprint does not aggregate different environmental impacts, instead it calculates the area needed for resource consumption and CO2 assimilation. To compare and add up the different land areas, hectares are translated into global hectares, so that each global hectare represents the same amount of natural productivity.

Modelling of cause-effect chain

Renewable resource extractions are related to land use areas using country specific yield data.

[13] For nuclear energy it is assumed that one unit of nuclear energy has the same ecological footprint as one unit of average fossil fuel electricity based on the area needed to sequester CO2 emissions).

Environmental interventions covered

The scope of interventions taken into account by the ecological footprint is limited. It only takes into account extraction of renewable resources (crops, animal products, fisheries, forest products) and CO2 emissions. So non-renewable resources and other emissions, like other greenhouse gases, toxic substances, VOCs, acidifying and eutrophying substances are neglected.

Impacts covered

Intervention	**'Impact category'**
Extraction of renewable resources (crops, animal products, fisheries, forest products)	Biological productive area needed to produce these resources or to absorb CO2
CO2	

4-20 Interventions and impacts covered in the Ecological Footprint

Geographical and temporal representation

The Ecological Footprint is expressed in global hectares, which is a hectare with the world's average biological productivity. To translate actual into global hectares, equivalence factors and yield factors are applied. Equivalence factors represent the world's average potential productivity of a certain area to the world average potential productivity of all areas. Yield factors relate the productivity of an area in a particular country (country specific yield factors) to the global average productivity of that type of area. (Wackernagel et al. 2005).

Normalisation level

No normalisation is performed.

Degree of being operational

There is no value based weighting set available. The method does no correspond to the assessment of mid and endpoint impact categories as recommended by ILCD.

Scientific quality and acceptance of the method

The land necessary for accumulation of CO2 is debatable. It is assumed that 35% of CO2 is absorbed by oceans and that the remaining 65% is assimilated in forests. There is no scientific basis for these assumptions. For nuclear energy it is assumed that one unit of nuclear energy has the same ecological footprint as one unit of average fossil fuel electricity based on the area needed to sequester CO2 emissions. This assumption does not have a scientific basis and does not reflect real impacts related to nuclear energy.

The most widely used methodology for calculating national Footprints are the National Footprint Accounts by the Global Footprint Network. The Global Footprint Network (GFN) is the organization that promotes the application of Ecological Footprint accounts and is supported by more than 70 partner organizations. The National Footprint accounts are calculated annually for more than 150 countries. The Global Footprint standards (GFN 2006) have been initiated by the Global Footprint Network to reach consensus on a common calculation method for the Ecological Footprints. Partners of the Global Footprint Network are required to comply with the most recent Ecological Footprint standards. The Global Footprint Network is working on providing more transparency and standardizing the methodology by making a complete handbook available for the National Footprint Accounts 2008. Unfortunately, this is not yet available.

4.7.2 Other Footprint measures

The term 'footprint' is used much more extensively than in the Ecological Footprint of the Global Footprint Network as described above. Even within their Ecological Footprint there are other footprints like the Carbon Footprint, which covers the main content of the Ecological Footprint, as a "sub-footprint". Unluckily, others, with substantial authority, use the Carbon Footprint in a more mundane sense, as the carbon emissions (or $carbon_{equivalent}$ emissions) of a country or other group of economic activities. The JRC of the European Commission has as a definition "….a carbon footprint is a life cycle assessment with the analysis limited to emissions that have an effect on climate change." (EU-Platform on LCA, 2007). It can be expressed in GWPs. Also at a national level the carbon footprint is used widely. One main example is the Carbon Trust, set up by the UK government, see their international website, http://www.carbontrust.com/. They define the carbon footprint as "the total set of GHG (greenhouse gas) emissions caused directly and indirectly by an individual, organization, event or product", Carbon Trust 2008). See the survey by Wiedmann and Minx (2007) for many more applications of the term carbon footprint.

Footprints referring to other subjects have been set up as well, like the Water Footprint (http://www.waterfootprint.org/) and the pesticide footprint, see the FP6 FOOTPRINT project[14].

Such applications of the term are covered in principle by the methods considered in this report, not independently but as part of broader schemes of environmental interventions and characterisation methods. In that sense, they form part of the weighting schemes investigated, as these all cover climate changing emissions

More encompassing, for example Rood et al (2004) use the term Ecological Footprint for the ecological effects resulting from a broad range of environmental interventions. However, they then reduce the analysis to effects on biodiversity, through land-use for e.g. agriculture and loss of quality in the remaining natural area as a result of e.g. fisheries and logging, and climate change. So they are broader than what is usually covered in LCA but do not cover all routes to biodiversity loss and don't cover effects on other areas of concern. Their analysis hence does not form a basis for the weighting exercise developed here, but has been taken into account in the ReCiPe project.

14 **FOOTPRINT is a research project funded by the European Commission as part of its 6th sixth Framework Programme for Research and Technological Development (FP6). The project aims at developing computer tools to evaluate -and reduce- the risk of pesticides impacting on water resources in the EU (surface water and groundwater). The project started in January 2006 and benefits from DG Research support for 3.5 years, i.e. until June 2009.**

5 Summary of weighting approaches

For all weighting methods, and for all non-weighting methods for integration, operational methods have been surveyed, see the survey table in the appendix and the summary table 5-1.

A taxonomy of methods that aggregate environmental interventions into a single score has been developed. The methods explicitly having an aggregation step combining different factors can be further grouped according to how the modelling and aggregation steps have been structured. We distinguish between

- ***integrated modelling and evaluation***, see chapter 4.2;
- ***midpoint modelling and evaluation***, chapter 4.3; and
- ***endpoint modelling and evaluation***, see chapter 4.4.

For all of them, available weights in the step covering the evaluation are based on questioning panels of some composition.

There are two more categories:

a) The ***distance-to-target type*** methods touch on weighting but do not make that step explicitly or recognisably, see chapter 4.5. Apart from methodology issues like independence of irrelevant issues, they still lack the inter-effect factor which indicates how important some effect is relative to another. All targets are treated equal, that is without weighting.

b) Finally, several methods use ***cost as method of aggregation,*** like cost for emission reduction, cost for reaching targets and cost for compensating measures, see chapter 4.6. Such methods do not solve the weighting problem. Allowable costs can only be established based on view of the importance of impacts; one would need to conduct the weighting step first in order to define allowable costs.

The three main approaches covering evaluation can be transformed into midpoint weighting methods, using the ILCD midpoint definitions. Discrepancies in the interventions covered are to be resolved. This restructuring allows a direct comparison between methods and will be investigated in the next part of the project.

Method	ILCD recommendation	ExternE/NEEDS/ EXIOPOL\climat models	TRACI/BEES	NOGEPA	EPS	LIME	Ecoindicator99	ReCiPe_CML	EDIP	Ecological Scarcity method	Re-Ci-Pe_CE	Ecological Footprint
interventions												
air emission	x	x	x		x	x	x	x	x	x	x	CO_2
water emission	x	(x)	x		(x)	x	x	x	x	x	x	
soil emission	x	(x)	x		(x)	x	x	x	x	x	x	
minerals extraction	x				x	x	x	x	x		x	
fossil fuels extraction	x		x		x	x	x	x	x	x	x	
biotic resource extraction												x
water extraction	x		x							x		
land use	x		x				x	x		x	x	x
midpoint impacts												
climate change	x	x	x	x		i		i	x		x	
ozone depletion	x		x	x		i		i	x		x	
human toxicity effects	x	i, part.	x	x		i		i	x		x	
respiratory inorganics	x		x			i		i				
ionizing radiation	x							i				
photochemical ozone creation	x		x	x		i		i	x		x	
acidification	x	i	x	x		i		i	x		x	
eutrophication	x	i	x	x		i		i	x		x	
ecotoxicity	x		x	x		i		i	x		x1	
land use	x		x			i		i				x
abiotic resource depletion	x		x2			i		i	x			
work environment			x						x			
endpoint impacts (AoP)												
human health	DALY	VOLY, diff			x	DALY	DALY	DALY				
natural environment (biodiversity loss)	PDF	PDF5			NEX	EINES	PDF	PDF				
natural resources (abiotic resources and water)	?				x	x3	x	x				
Man-made environment												
building materials		diff				x3						
primary production		diff			x	x4						

Table 5-1 Scope of combined impact assessment and weighting methods

1 fresh water and terrestrial **2 fossil fuel and water**

3 part of social assets **4 partially: dry ton produced crop**

5 partially: covers only acidification and eutrophication

? unclear **i : implicit, midpoint effect is part of the impact assessment at endpoint level**

diff.: cost is differentiated e.g. country specific data for different types of hospital costs, restoration costs for materials, market prices for lost crops

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Glossary

Shadow price
Shadow prices constitute the opportunity cost of an activity or project to society, computed where the actual price is not known or, if known, does not reflect the real sacrifice made. Here used as: Any monetised value used where market prices are not available.

Conjoint analysis
Conjoint analysis is a general name for the methods of assessing individuals' preferences for each of a number of attributes (in this case safeguard areas). A choice-based type of questionnaire is prepared for the interview, with the respondents selected by random sampling. WTP per quota can be determined by statistical simulation based on random utility theory, reflecting the responses to the questionnaires by random sampling.

7 Annex 1. Survey table of characteristics of operational methods

IA and weighting method	Weighting procedure	Cause-effect chain		
		interventions	**impacts assessment**	
			midpoint (problem assessment)	**endpoint (damage assessment)**
Economic valuation (combined ExternE/NEEDS /EXIOPOL)	monetised values, using a mixture of valuation methods	20 air emissions, (some activities, some water and soil emissions), no extractions		intermediate results?
LIME	monetised values, stated damage cost (WTP for reducing damages (conjoint analysis))	1000 substances (several compartments)	appr. 15 indicators	appr. 15 indicators, representing 4 safeguard areas: human health, biodiversity, primary production, social assets
Ecoindicator99	panel weighting	400 substances		3 safeguard areas
ReCiPe-CE	monetised values, revealed prevention cost	3000 substances (several compartments)	8 midpoint indicators	
ReCiPe-CML	monetised values, stated damage cost (WTP)			3 safeguard areas: human health, biodiversity, resources
ReCiPe-Pré	panel weighting			3 safeguard areas: human health, biodiversity, resources
TRACI/BEES/ NOGEPA	panel weighting (collective stated preferences)	3000 substances (several compartments)	9-11 midpoint indicators	
EDIP97	Distance-to-target method (collective stated (political) targets)	3000 substances (several compartments)	9 midpoint indicators	
Ecological Scarcity Method	Distance-to-target method (collective stated (political) targets)	400 substances (several compartments)	no effect modelling	
EPS	monetised values, using a mixture of valuation methods	`200 substances (emphasis on extractions, some air emissions, and few water and soil emissions)		

Table 5-2a Overview of weighting methods (to be continued)

IA and weighting method	(damage) valuation	geographical representation		time
		impact assessment	valuation	
Economic valuation (combined ExternE/NEEDS /EXIOPOL)	full chain modelling, valuation encompasses human health, biodiversity (acidification, eutrophication), climate change, crops and building materials	EU, site specific	EU, site specific	present time interventions, long term effects, incl. discounting (discount rate different, not always explicit)
LIME	WTP for avoiding damages regarding 4 safeguard areas	Japan and global	Japan	present time interventions, long term effects
Ecoindicator99		NL, EU and global		present time interventions, long term effects
ReCiPe-CE	prevention cost, based on cost (benefit) curves and targets set on intervention or midpoint level	EU and global	Dutch policy targets and costs	present time interventions, long term effects, present cost for present abatement technologies
ReCiPe-CML	WTP for avoiding damages regarding 3 safeguard areas	EU and global	international literature survey	
ReCiPe-Pré	iterative process of panel weighting	EU and global		
TRACI/BEES NOGEPA	iterative process of panel weighting	USA and global NOGEPA Europe		present time interventions, long term effects
EDIP97	Distance-to-target method, based on political targets set on midpoint level	global	international targets	present time interventions, long term effects; DTT set on reference year 1994 and target year 2004
Ecological Scarcity Method	Distance-to-target method, based on political targets set on intervention level	no effect modelling	Swiss policy targets	present time interventions, DTT set on reference year 2005 and target for 5 to 20 years
EPS	full chain modelling, valuation most likely encompasses human health, biodiversity, ecosystem productivity, abiotic resources	global	WTP based on OECD population	present time interventions, long term effects; no discounting for future effects

Table 5-2b Overview of weighting methods (continued)

IA and weighting method	normalisation	operational
Economic valuation (combined ExternE/NEEDS /EXIOPOL)	not necessary	average EU data, list of euro / kg intervention
LIME	not necessary, reference interventions are used in procedure to valuate damages to 4 safeguard areas	weighting factors available, both monetised and dimensionless
Ecoindicator99	EU	Weighting sets available for different cultural perspectives
ReCiPe-CE	data available for EU and world, 2000; data for land transformation and water depletion missing	two sets of weighting factors available for 8 impact categories based on policy targets and sustainability targets
ReCiPe-CML		only indicative values available
ReCiPe-Pré		no operational panel weighting factors available
TRACI/BEES NOGEPA	data available for North America, 1999 NOGEPA; Data fro Netherlands and Europe	two sets of weighting factors available for 9 and 11 impact categories: EPA Science Advisory Board and Building sector stakeholder Panel
EDIP97	data available for EU and world, 1994; data for resources and working environment are missing	weighting factors available for ca. 8 impact categories
Ecological Scarcity Method	Switzerland, 2004	appr. 40 ecofactors available for individual substances or substance groups
EPS	not necessary	weighting factors available presented at intervention level (list of euros / kg intervention)

Table 5-2c Overview of weighting methods (continued)

IA and weighting method	remarks	
	weaknesses	strengths
Economic valuation (combined ExternE/NEEDS /EXIOPOL)	mixture of weighting methods, limited number of interventions, important interventions are missing, full chain modelling not easily compatible with ILCD	representative for EU
LIME	representative for Japan	one weighting method used for all safeguard areas, large number of interventions (air, water and soil emissions; mineral extractions, land use), to a large extend compatible with ILCD
Ecoindicator99		
ReCiPe-CE	representative for Dutch policy targets and costs, cost based on present technologies, target values are outdated, targets are considered equally important	one weighting method used for 8 midpoint indicators, large number of interventions (air, water and soil emissions; mineral extractions), to a large extend compatible with ILCD
ReCiPe-CML	only indicative values available, highly uncertain	
ReCiPe-Pré	not operational	
TRACI/BEES NOGEPA	representative for North America NOGEPA for EU	one weighting method used for 10 midpoint indicators, large number of interventions (air, water and soil emissions; mineral extractions), to a large extend compatible with ILCD
EDIP97	weighting factors are region and time specific, international targets are used but the time horizon is out dated; targets are obscured by social, economic and technical considerations, targets are considered equally important	one weighting method used for 8 midpoint indicators, large number of interventions (air, water and soil emissions; mineral extractions), to a large extend compatible with ILCD
Ecological Scarcity Method	representative for Swiss policy targets, not compatible with ILCD, targets are considered equally important	
EPS	mixture of weighting methods, limited number of interventions, important interventions are missing, full chain modelling not easily compatible with ILCD	

Table 5-2d Overview of weighting methods (continued)

European Commission

EUR 24997 EN – Joint Research Centre – Institute for Environment and Sustainability
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Abstract

This report presents the first part of the work carried out towards the development of a scheme for weighting indicators across the impact categories (climate change, acidification, resource depletion, human cancer effects, and others) that are commonly considered in life cycle assessment. Weighting is essential to derive a single indicator of the overall environmental impact of the EU-27 and to build the resources indicators as set out in the Thematic Strategy.

This report further details the above classification scheme and analyses a number of relevant weighting approaches. Each of the methods considered has been characterized in terms of methodological foundations, geographical representativeness, procedure for values definition, communication impact and major applications in the LCA practice.

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