Annex: update monetisation of the MMG method (2014)

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This note describes the methods and data used to monetise the LCIA impacts as a further development of the approach used in MMG2012 (Debacker et al., 2012). First, we discuss some general changes to methods and data. Second, we discuss for each impact category the approach, data and results. We focus on new developments (i.e. MMG2014) compared to MMG2012, and refer to previous studies for the details.

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1 Introduction

Objective: the analysis in Debacker et al. (2012) indicates that the monetisation step can have a big impact on the (aggregated) MMG results. Monetising impacts both reveals the uncertainty inherent in LCIA and adds uncertainty. In 2013-2014, we further developed the monetization step, especially to make the approach consistent with the indicators used in ILCD recommendations and CEN guidelines, and to clarify and narrow down the uncertainties.

This note describes the methods and data used to monetise the LCIA impacts as a further development of the approach used in MMG2012 (Debacker et al., 2012). First, we discuss some general changes to methods and data. Second, we discuss for each impact category the approach, data and results. We focus on new developments (i.e. MMG2014) compared to MMG2012, and refer to previous studies for the details. In the following paragraphs 'MMG2012' will refer to the original assessment method developed in 2012. With 'MMG2014' we will refer to new developments.

2 Summary overview of methods and results

2.1 Methods

The methods and data build further on methods and data selected in the context of the MMG2012 project (Debacker et al, 2012; De Nocker, 2012).

In the context of the current project, the methods and data are updated and adapted to account for:

- Consistency of the indicators for LCI inventory and LCIA impact assessment, in line with the CEN and CEN+ recommendations. The list of impacts to be covered and the selected metrics and LCIA models/methods are given in table 1 and 2 for the CEN and CEN+ indicators respectively. The LCIA models for the CEN indicators are based on the latest version of the EN 15804 standard (CEN 2013). The LCIA models for the CEN+ indicators are based on ILCD recommendations (JRC 2011), with the exception of the land use biodiversity indicators.
- For 'Land Transformation' and 'Land Occupation' the sub indicator 'biodiversity' has been added. Based on Allacker et al. (2014) and Debacker et al. (2012), land use impacts related to changes in biodiversity are significant for some processes and products within the built environment.
- In addition, the scope and characteristics of the LCIA models in these steps may have impacts for the selection of the monetary value.
- Update of new information related to monetization of environmental impacts.
- A more coherent approach to discounting, using a 3 % discount rate as the central estimate.
- A more consistent approach to deal with uncertainty for different impact categories and to present a central, low and high estimate.
- Monetary values of each environmental indicator have been determined for three regions: Western Europe, Flanders/Belgium and the rest of the world. Only the Western European values are taken into account for the publically available MMG method, because most processes related to the life cycle of building products are related to this geographical area. The monetary values for the Flemish/Belgian region and the rest of the world are determined for sensitivity analyses. The uncertainty related to the monetary value is the smallest for the Flemish/Belgian region (or equal to the Western European region and the rest of the world).

2.2 Results

Tables 3 and 4 give an overview of the results for Western Europe. In sections 4 and 5, we also report the values for Flanders/Belgium and for the 'rest of the world' category.

Environmental indicator (CEN)	Unit	Selected LCIA model
1. Global warming	kg CO2 eqv.	EN 15804+A1 *
2. Depletion of the stratospheric ozone layer	kg CFC-11 eqv.	EN 15804+A1 *
3. Acidification of land and water sources	kg SO2 eqv.	EN 15804+A1 *
4. Eutrophication	kg (PO4)3- eqv.	EN 15804+A1 *
5. Formation of tropospheric ozone photochemical oxidants	kg etheen eqv.	EN 15804+A1 *
6. Abiotic depletion of non-fossil resources	kg Sb** eqv.	EN 15804+A1 *
7. Abiotic depletion of fossil resources	MJ, net caloric value	EN 15804+A1 *

Table 1: Selected CEN environmental indicators, the corresponding units and LCIA models

* EN 15804+A1 = EN 15804+A1 (as used in CML version 2012)
** Sb: antimony

Environmental indicator (CEN+)	Unit	Selected LCIA model
8. Human toxicity a. cancer effects	CTUh***	Rosenbaum et al., 2008 (as used in USEtox)
b. non-cancer effects	CTUh***	Rosenbaum et al., 2008 (as used in USEtox)
9. Particulate matter	kg PM2,5 eqv.	Rabl et al, 2004 (as used in RiskPoll)
10. Ionising radiation, a. human health	kg U235 eqv.	Frishknecht et al., 2000 (as used in ReCiPe midpoint)
b. ecotoxicity	CTUe**** (per kBq)	Garnier-Laplace et al. as (2008, 2009)
11. Ecotoxicity[1]: a. terrestrial	-	-
b. freshwater	CTUe****	Rosenbaum et al., 2008 (as used in USEtox)
c. marine	-	-
12. Water scarcity	m^3 water eqv.	Frishknecht et al, 2006 (as used in Swiss Ecoscarcity – water)
13. Land use: occupation: a. soil organic matter	kg C deficit	Milà i Canals et al., 2007
b. biodiversity [2]	m^2a	Köllner, 2000 + characterisation factors set on "1"
14. Land use: transformation a. soil organic matter	kg C deficit	Milà i Canals et al., 2007
b. biodiversity	m^2a	Köllner, 2000 + characterisation factors set on "1" or "-1"

Table 2: Selected CEN+ environmental indicators, the corresponding units and LCIA modelsEnvironmental indicator (CEN+) Unit Selected LCIA model

*** CTUh: comparative toxic units for humans

**** CTUe: comparative toxic units for ecosystems

1For terrestrial and marine ecotoxcity, the PEF Guide does not recommend an LCIA model

2 Biodiversity impacts related to land use are not taken into account in the PEF. However, due to its importance in the built environment, the Köllner 2000 model as used in Eco-Indicator 99 (in PDF*m²yr) is proposed as a best proxy to take into account biodiversity impacts related to land use for the individual MMG scoring. Impacts of land occupation and land transformation expressed in m²a have proven to be a better basis to calculate the related environmental costs. For this reason, the land occupation and transformation processes (expressed per m²a) considered in the Eco-Indicator 99 method are taken into account, but characterisation factors are set to "1" or "-1" for the calculation of the environmental costs. For the calculation of the individual indicators the Köllner 2000 model as used in Eco-Indicator 99 (in PDF*m²yr) is used.

Environmental indicator (CEN)	Unit	Central (€/unit)	Low (€/unit)	High (€/unit)
1. Global warming	kg CO2 eqv.	**0.100**	0.050	0.200
2. Depletion of the stratospheric ozone layer	kg CFC-11 eqv.	**49.10**	25	100
3. Acidification of land and water sources	kg SO2 eqv.	**0.43**	0.22	0.88
4. Eutrophication	kg $(PO4)^{3-}$ eqv.	**20**	6.60	60
5. Formation of tropospheric ozone photochemical oxidants	kg etheen eqv.	**0.48**	0	6.60
6. Abiotic depletion of non-fossil resources	kg Sb eqv.	**1.56**	0	6.23
7. Abiotic depletion of fossil resources	MJ, net caloric value	**0**	0	0.0065

Table 3: Overview of West-European monetary (central, low and high) values for the CEN indicators

Environmental indicator (CEN+)	Unit	Central (€/unit)	Low (€/unit)	High (€/unit)
8. Human toxicity a. cancer effects	CTUh	**665109**	166277	2660434
b. non-cancer effects	CTUh	**144081**	28816	720407
9. Particulate matter	kg PM2,5 eqv.	**34**	12.70	85
10. Ionising radiation, a. human health	kg U235 eqv.	**9.7E-04**	3.2E-04	2.9E-03
b. ecosystems	CTUe (per kBq)	**3.70E-05**	7.39E-06	1.85E-04
11. Ecotoxicity: a. terrestrial	-			
b. freshwater	CTUe	**3.70E-05**	7.39E-06	1.85E-04
c. marine	-			
12. Water scarcity	m^3 water eqv.	**0.067**	0.022	0.20
13. Land use: occupation: a. soil organic matter	kg C deficit	**2.7E-06**	6.8E-07	1.1E-05
b. biodiversity b1. urban: loss ES*	m^2.a	**0.30**	0.07	2.35
b2. agricultural	m^2.a	**6.0E-03**	1.5E-03	2.4E-02
b3. forest: biodiversity	m^2.a	**2.2E-04**	5.5E-05	8.8E-04
14. Land use: transformation a. soil organic matter	kg C deficit	**2.7E-06**	6.8E-07	1.1E-05
b. biodiversity b1. urban	m^2	**n.a.**	n.a.	n.a.
b2. agricultural	m^2	**n.a.**	n.a.	n.a.
b3. forest, excl. tropical	m^2	**n.a.**	n.a.	n.a.
b4. tropical rainforest	m^2	**n.r.**	n.r.	n.r.

Table 4: Overview of West-European monetary (central, low and high) values for the CEN+ indicators

*ES: Ecosystem Services

n.a. = not available; n.r. = not relevant

3 Methodology: selection of data and uncertainties

3.1 Selection of methods and data

Table 5 gives an overview of the available information for monetisation per impact category, taking into account the selected environmental indicators (tables 1 and 2). It distinguishes methods based on damage and prevention costs. The background to the environmental costing methods used is discussed in detail by Debacker et al. (2012).

3.1.1 Indicators for social costs based on damage costs

As damage cost methods calculate in detail how emissions or use of resources causes damages to man and the economy, it is the preferred method. Table 5 illustrates that for most impact categories, there is information on damage costs, although the amount and quality of information differs, which is reflected in the + to +++ scores. We have to look into the details of the available information per impact category to appreciate its strengths and weaknesses. We summarise this info below, a more profound analysis including references is provided in the sections per impact category.

— Damage cost methods calculate how emissions of air pollutants disperse in the atmosphere, affects agriculture and public health which leads to welfare losses in terms of additional costs (e.g. liming, medicine costs), loss of income (production losses, sick leaves) or comfort (e.g. pain). This requires fate and effect models, to account for dispersion, interaction and for impact assessment. The monetary valuation of these impacts uses different methods and data, appropriate to the type of impact. It includes market data (to value loss of agricultural production, medicine costs or loss of income) and data from scientific literature to value pain or loss of life expectancy, based on revealed or stated willingness to pay. Damage costs information is available for most impact categories, and especially if impacts on public health and agriculture are important impact categories (e.g. depletion of the stratospheric ozone layer, tropospheric ozone, human toxicity, particulate matter). A higher number of studies with converging insights is reflected in the scores in table 5 (+++ to ++).

— For impact categories related to depletion of natural resources and with dominant impacts on ecosystems, there is less and more divergent info on damage costs. This is reflected in the + qualification in table 5. Here, it also requires fate and effect modelling to quantify impacts that can be valued. The monetary valuation of impacts on ecosystems for acidification and ecotoxicity is based on stated willingness to pay for biodiversity, (and on prevention costs (see below)). For land occupation, valuation is based on the loss of ecosystem services due to urban land use in Flanders and Europe. The loss of regulatory ecosystem services is valued based on available information per service, either damage cost (air quality, noise) or prevention costs (water quality). The loss of cultural services is based on methods to value health impacts and revealed and stated willingness to pay to live and recreate in a green environment.

— Damages for resource depletion refer to additional costs for future generations, e.g. for mining or importing water over larger distances. These additional costs are based on market prices for e.g. additional energy use.

— For global warming, although there is a lot of information on damage costs, it is qualified with a single +, because this information is less complete and less convergent compared to other impact categories.

Environmental indicator (CEN)	Damage Costs (DC)	Prevention Costs (PC)	Central estimate	Range low-high estimate
1. Global warming	+	++	PC	DC, PC
2. Depletion of the stratospheric ozone layer	++	+	DC	DC, PC
3. Acidification of land and water sources	++	+	DC	DC, PC
4. Eutrophication	+	+	DC, PC	DC, PC
5. Formation of tropospheric ozone photochemical oxidants	+++	-	DC	DC
6. Abiotic depletion of non-fossil resources	+	-	DC	DC, PC
7. Abiotic depletion of fossil resources	+	+	DC	DC, PC
Environmental indicator (CEN+)				
8. Human toxicity, a. cancer b. non-cancer effects	 ++ +	 + +	 DC DC	 DC, PC DC, PC
9. Particulate matter	+++	+	DC	DC
10. Ionising radiation, a. human health b. ecosystems	 ++ +	 + -	 DC DC	 DC DC
11. Ecotoxicity: a. terrestrial b. freshwater c. marine		+	DC	
12. Water scarcity	+	-	DC	DC
13. Land use: occupation: a. soil organic matter b. biodiversity b1 : urban b2 : agriculture b3: forestry	 + + 	 + - + +	 DC, PC DC PC PC	 DC,PC DC PC PC
14. Land use: transformation a. soil organic matter b. biodiversity b1 : urban b2 : agriculture b3: forestry (excl. rainforest) b4 : rainforest (World estimate)	 + - - - +	 + - - - -	 DC, PC DC	 DC, PC DC

Table 5: Overview of available methods per impact category and methods used for central, low and high estimations

3.1.2 Indicators for social costs based on prevention costs

Prevention cost methods (also referred to as control or abatement costs methods) values an impact based on marginal cost to meet the policy objective for this impact. This requires a clear policy objective (either in terms of emission reductions or environmental quality objectives (such as ppm in ambient air). In addition, information about all potential prevention measures in different sectors is ranked in terms of their cost-effectiveness, and expressed e.g. in €/kg pollutant. The costs of the least cost-efficient measure to meet the target is an indicator of the value society is willing to pay or impose on citizens or firms to control the environmental problem.

As indicated in table 5, prevention costs are less available, because it requires clear policy objectives and a cost-effectiveness analysis. It is available for global warming and European air quality issues. This is reflected in the ++ in table 5 for global warming, for which a long term global target is available, which is used in different cost-effectiveness studies and that info is used in several studies and policy documents as an indicator for the social costs of carbon. For impacts related to European air quality (acidification, ozone formation, particulate matter), we qualified the information as + because air quality problem is a multi-source and multi-effect problem which makes it more difficult to assess prevention costs for single effects, because the targets reflect short term compromises rather than long term objectives and because this information is seldom used as indicators for social costs of air pollution.

For impact categories related to ecosystems and biodiversity, there is not enough information for a proper prevention cost method. As a proxy, the costs for a typical measure or the amount of environmental taxes are used. The restoration costs in the context of agri-environmental measures in Europe is taken as an indicator of the value European society puts on safeguarding biodiversity and soil organic manner. For the monetary value of biodiversity, it translates into €/PDF, which is further used to value ecotoxicity and some impacts of acidification or eutrophication. For land use occupation, the info is translated into €/m².

3.1.3 Selection of information for central estimate and law-ligh estimates

As reflected in table 5, the central estimate for most impact categories is based on damage cost approaches. The estimation of low-high estimates account for uncertainty and information from other sources and methods, including that based on prevention costs. As discussed in these sections, both methods give comparable results.

For global warming, the central estimate is based on prevention costs, because damage costs are highly uncertain and prevention cost information is good.

For soil organic matter, the central estimate is based on both methods. For impacts of land use on biodiversity, and ecotoxicity, we can only provide some monetary indicators on one method with less info and larger uncertainties.

Overall, table 5 reflects that there is more information to value CEN indicators, and that for CEN+ indicators, the availability of information is very divers.

3.1.4 Comparison with literature

Overall, approaches and data compare rather well with the LCIA assessment of European Energy sector (Alberici, 2014). This study distinguishes between a similar list of 18 impacts and use a similar mix of studies, mainly based on damage costs, except for climate change. The study builds on indicators and data from ReCiPe, ExternE-NEEDS and De Bruyn (2010). Although some indicators are different compared to MMG2014 (e.g. depletion), it builds on the same studies for its monetarisation (e.g. ReCiPe).

3.2 Discounting

The value of future impacts is discounted using a social discount rate. The choice of the discount rate has been subject to many studies and debate, but there is no overall agreement on which discount rate to use (Pearce, 2003; Arrow, 2012). These debates have indicated that discount rates may differ, depending on the objective of the analysis (Goulder, 2012), on the time horizon of the impacts (Arrow, 2012, Ochelen, 2010; Lowe 2008; Commissariat Général du Plan, 2005), ethical issues including equity weighting (Anthoff, 2009) and age weighting (WHO, 2014), uncertainty and risk premiums (Romijn, 2013, Kousky, 2011, van den Bergh, 2007). Consequently, the effect of different discount rates on the overall result has to be reflected in the uncertainty analysis (see below).

For the central estimate, we use a 3 % social discount rate. This discount rate is in line with the recommendations of different governmental institutions and studies that give input to the MMG2014 framework. It is in line with:

- the recommendation of the Flemish government, i.e. a 4 % discount rate to be applied for the first 30 years (within one generation) and a declining discount rate for the rest of the period, that results in a 2 % discount rate for the very long term (Ochelen, 2010) . Such a system results in an average discount rate of 3 %, if we assume an equal yearly impact over a time horizon of 300 years.
- The principle of a declining discount rate is in line with recommendations from the French and UK government, although the latter starts from a lower discount rate (3.5 %) (Commissariat Général du Plan, 2005; Lowe 2008).
- Is close to the recommendation for CBA in the Netherlands, i.e. a base line risk-free discount rate of 2,5 % + a risk premium depending on the type of project (van den Bergh, 2007).
- The discount rate used in ReCiPe, e.g. for the assessment of depletion (Goedkoop, 2013).
- The discount rate used by WHO for the calculations of DALYs.

The discount rate is a very important factor for the assessment of the social costs of global warming, and it is more discussed in detail in that context (see below, paragraph 4.1). Discount rates are also relevant for the quantification and valuation of chronic human health impacts. If monetary valuation is based on the value of loss of DALYs, discounting of future health is already accounted for in the quantification of DALYs using a discount rate of 3 % (WHO, 2014). For valuation of health impacts from emissions of particulate matter, the value of future health impacts is accounted for in the approaches for valuation loss of life expectancy (value of a life of year lost) (Alberini, 2007).

3.3 Uncertainty

For the new environment assessment method (MMG2014) we have updated and improved the MMG2012 approach to deal with the uncertainty.

3.3.1 Background

Uncertainty in LCA, LCIA and impact assessment is determined by a large number of factors (Lloyd, 2008; Huybrechts, 2000). For impact assessment and its valuation, it is important to distinguish between uncertainty due to variation (e.g. different locations of emissions) and uncertainty due to limited understanding of impacts of limits in modelling all the interactions (Webster, 2003, Spadaro 2008, Holland, 2013). Studies that account for a systematic and scientific approach for all sources of uncertainty come up with uncertainty ranges that are very large. In the MMG2012 study we used a systematic approach to assess uncertainty covering all steps in the analysis, following the approach used by Spadaro and Rabl (2008) for assessment of damages from air pollutants. This approach has also been used in the framework of the ALPHA methodology, for assessment of benefits of air pollution for the EC (Holland, 2013). In this context, it has been fully applied within the TUBA framework (Treatment of Uncertainty for Benefit Assessment) in order to test robustness of specific answers to specific questions, i.e. are benefits greater than the costs. We used an estimate of the standard deviation (σg) as the indicator to assess and calculate the uncertainty range. The range reflected a 68 % confidence interval, and was calculated as explained in table 5.

In MMG2012, the standard deviation is assessed for each impact category, and it varies between 3 (for the relative more certain impacts of particulate matter on public health) to 5 (for global warming or biodiversity). This range reflects variation in spatial characteristics, and uncertainty due to limitations in our understanding, models and data. This approach is well justified from a scientific point of view.

Most documents or guidelines that develop indicators and data to assess and monetise environmental impacts however only account for selected sources of variation and/or uncertainty (Lloyd, 2008). For example, the handbook on external costs for transport does discuss the uncertainty related to different impact categories (climate change, particulate matter) but accounts only for variation in two parameters (technologies and location) for the reported values (Ricardo, 2014). The Dutch handbook for shadow prices reports two sets of values, reflecting uncertainty related to the method (damage costs versus prevention costs), but does not account for the uncertainty of the values due to other factors (De Bruyn, 2010). The technical document supporting the US recommendation for the social costs of carbon accounts for variation in two parameters (year of GHG-emission and discount rate), but does not account for the full variation of model uncertainty etc., as done in the more scientific studies (e.g. Tol, 2012). The assessment of damages of industrial activities follows for air pollution the approach of the ALPHA assessment method (Holland, 2013). However, for assessment of uncertainties, it does not include the full range of uncertainty assessments (the TUBA method, as mentioned above) but only accounts for one source of uncertainty, i.e. for valuation of health impacts from air pollutants. Consequently, low and high estimates only differ a factor of 2. Although the technical tools to support the EC air quality policy (particulate matter) allows for a full analysis of uncertainties, the policy documents only report uncertainty due to one parameter (valuation of mortality), with a bandwidth of factor 4 (Holland, 2014).

3.3.2 Treatment of uncertainty in MMG2014

In MMG2014, our approach to uncertainty is closer to how this is dealt with in policy studies and we have adapted our approach in different ways:

— First, we distinguished 'variability' and 'uncertainty'. Variability of the monetary values reflects that emissions and impacts in different locations have a different value, reflecting differences in the physical environment (e.g. average temperature or dominant soil types), and in the socio-economic environment (e.g. number of people exposed to pollution, differences in habits or diets, differences in income and preferences). To account better for this variation, we made distinction between emissions and burdens in Flanders/Belgium, Western Europe and the rest of the world. The data for Western Europe are used as the central data set for the public version of the MMG method.

— Second, we narrowed the confidence interval for the presentation of the results. We used a more narrow band for the low and high estimate, which is more in line with ranges used in documents to support policy analysis. For some impact categories, e.g. eutrophication, we used the central value in different studies to define the low and high estimate. For other impact categories, e.g. global warming, we looked at the bandwidth (BW= high estimate/low estimate) used in other guidelines and policy studies.

  We noted that these typical ranges are about half the ranges of the 68 % confidence interval used for the MMG2012. To be more in line with standard practice for policy studies, we adapted our approach to calculate the low and high estimate.

— Third, we have reviewed the assessment of the scientific analysis of uncertainty, because we separated variation and uncertainty and because for some impact categories the indicators/units and LCIA models used have changed.

	MMG2012	MMG2014
	Variation and uncertainty	**Uncertainty**
Parameter used	σg = standard deviation	BW = Bandwidth, expert judgement ≈ σg (for some impact categories)
Central value	µg = expected or central value	µg = expected or central value
Low estimate	µg/σg	µg/ (√BW)) ≈ µg / (σg / 2)
High estimate	µg * σg	µg * (√BW) ≈ µg * (σg / 2)
		Spatial Variation
Central value		Western Europe
		Flanders / Belgium
		Rest of the world ***

Table 6: Approach to uncertainty in MMG2012 and MMG2014.

* Depending on σg = standard deviation, determined for each impact.

** Depending on the location of the activity and its impact

*** For emissions in rest of the world, a specific approach to low and high estimate is applied (see text).

The value for Western Europe is the central value, with additional values for Flanders and the rest of the world. The low and high estimate for Western Europe and Flanders are based on the bandwidth around the central estimate, calculated as explained in table 5. Compared to MMG2012, the bandwidth is four times smaller. For emissions in the rest of the world, the bandwidth may be much larger if the impacts vary between countries. Therefore, we apply a broader range. The low estimate may be zero. In general, the damages in Flanders and Western-Europe are higher compared to World average, due to high population density, high environmental pressures on ecosystems and high GDP/capita. Therefore, these damages are representative for the higher estimates at world level. Consequently, the high estimate for world level is based on the maximum of the high estimate for Western Europe and Flanders/Belgium.

	MMG2012M	MMG2014	
Environmental indicator (CEN)	**Og (1)**	**Bandwith (2)**	**√BW (3)**
1. Global warming	5	4	2
2. Depletion of the stratospheric ozone layer	4	4	2
3. Acidification of land and water sources	4	4	2
4. Eutrophication	5	9	3
5. Formation of tropospheric ozone photochemical oxidants	4	4	2
6. Abiotic depletion of non-fossil resources	5	16	4
7. Abiotic depletion of fossil resources	/	16	/
Environmental indicator (CEN)			
8. Human toxicity, a. cancer b. non-cancer effects	 4 4	 16 25	 4 5
9. Particulate matter	4	6,7	2,6
10. Ionising radiation, a. human health b. ecosystems	 4 /	 9 25	 3 5
11. Ecotoxicity: a. terrestrial b. freshwater c. marine	4	25	5
12. Water scarcity	/	9	3
13. Land use: occupation: a. soil organic matter b. biodiversity	5	 16 16	 4 4
14. Land use: transformation a. soil organic matter b. biodiversity	5	 16 16	 4 4

Table 7: Overview of the uncertainty values for all CEN and CEN+ indicators

(1) Standard deviation for expected result, assuming log-normal distribution, and used to calculate min. and max estimate within the MMG2012 method

(2) BW = Bandwidth between low and high estimate = high estimate/low estimate, in case low estimate is zero, bandwidth = standard deviation of MMG2012

(3) √BW = square root of BW, used to calculate low (/√BW) and high estimates (* √BW)

3.4 Comparison with the Dutch environment assessment methods for buildings

In the Netherlands, a similar approach is used to quantify and assess the environmental impacts of buildings and its constituting parts. Although environmental costing is also used as aggregation method, the number of individual LCIA indicators is nevertheless restricted. The development of the Dutch assessment method is performed by SBK (2011). For the definition of the monetary values SBK refers mainly to a study performed by TNO performed in 2006, in order of the Dutch Ministry of Infrastructure and Environment.

The values used for monetary valuation in the Dutch SBK project (2011) are similar to the values for OVAM MMG2014. The SBK data are based on prevention costs. For global warming, ozone depletion, eutrophication, formation of tropospheric ozone and depletion of non-fossil resources, the Dutch SBK values are between the low and high range of OVAM. For global warming, the SBK value is on the lower boundary, which rather reflects the prevention costs in the short run. The OVAM MMG2014 value is more in line with recent studies, such as the guidelines for monetary valuation of external costs of transport for the EC (Ricardo-AEA, 2014). It should be noted that SBK takes depletion of fossil fuels into account, whereas MMG2014 does not, and one of the arguments is that global warming is a more stringent constraint then depletion.

For acidification, the SBK value is much higher, but it attributes all the prevention costs of reduction of SO2 emissions to the 'acidification', whereas these costs should only be partly attributed to this problem and partly to the problem of health impacts from secondary particles. The arguments to justify further cuts in SO2 emissions mainly build on the latter health impacts (e.g. costs benefit studies of EU air quality policies (CAFE). Attributing prevention costs to both themes, unlike MMG, explains the high value of 4 €/kg SO2 eq.

3.4.1.1

Environmental Indicator		Method MMG OVAM			SBK
(CEN)	Unit	Central	Low	High	
Global warming	kg CO2 eqv.	0.10	0.05	0.20	0.05
Depletion of the stratospheric ozone layer	kg CFC-11 eqv.	49	25	100	30
Acidification of land and water sources	kg SO2 eqv.	0.43	0.22	0.88	4
Eutrophication	kg (PO4)$^{3-}$ eqv.	20	6.60	60	9
Formation of tropospheric ozone photochemical oxidants	kg etheen eqv.	0.48	0	6.60	2
Abiotic depletion of non-fossil resources	kg Sb* eqv.	1.56	0	6.23	0.16
Abiotic depletion of fossil resources	MJ, net caloric value	0	0	0.0065	
	kg Sb* eqv.				0.16*

Table 8: Comparison of indicators for monetary valuation in OVAM MMG and Dutch MMG

*Sb: antimony

4 Monetary indicators for CEN indicators

4.1 Global Warming

4.1.1 Monetary value

Monetary indicator for global warming

Global Warming		Value € / kg CO_2 eq		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	0.100	0.050	0.200
	Flanders / Belgium	0.100	0.050	0.200
	Rest of world	0.100	0.050	0.200
More detailed numbers per phase				
Construction	Western Europe	0.045	0.023	0.090
Use Phase	Western Europe	0.110	0.055	0.220
End of life	Western Europe	0.140	0.070	0.280

Table 9: Comparison of indicators for monetary valuation in OVAM MMG and Dutch MMG

Source: VITO 2014

4.1.2 Methods and data used

4.1.2.1 MMG2012

As for MMG2012, the monetary valuation is based on an analysis of prevention costs. The values reflect the costs for the global economy to limit the emission of greenhouse gases to levels that limit global warming to 2°C. The cost assessment is based on a meta-analysis of different models to reach this target (Kuik, 2008) and on other studies that estimate these costs. The details are described in MMG2012 (Debacker et al., 2012) and De Nocker et al (2010). This estimate is compared to the available information about the expected damages from global warming. The latter is more uncertain, and the damage estimates are incomplete. Nevertheless, the information on damages that can be estimated using the damage cost approach confirms the estimates of the prevention cost approach.

This approach and data are checked against more recent information on damage and prevention costs. In addition, a better procedure to estimate a single value has been developed.

4.1.2.2 Indicator

The indicator (kg CO2 eq) remains unchanged.

4.1.2.3 Update on literature on social costs of carbon

First, there are a number of new studies or publications on social costs of carbon based on damage costs estimates, including both new original model studies and review of literature (Waldman, 2014, Hope, 2013; Ackerman, 2011; Tol, 2012; Tol, 2011; Tol, 2009). The new estimates confirm the (large) bandwidth of previous studies, with central estimates that range from 7.3 €/ton CO2eq (range 1 -72 €/ton CO2eq; based on Fund 3.9 model) (Waldman, 2014; Tol, 2012) to 86 €/ton CO2eq (Page 09 model, Hope, 2013) (all values converted to euros based on exchange rates of November 2014). A meta-analysis of different studies confirms the large bandwidth, around a central estimate of 30 €/ton CO2eq (fitted distribution, median) (Tol, 2009). It should be reminded that these studies do not cover all the impacts, and especially those from more extreme global warming scenarios or uncertain events may be important damage categories (Ackerman, 2011; Stern, 2006; Watkiss, 2008).

Second, recent studies based on the prevention costs confirm that the UN objective to limit climate change to a maximum of 2°C can be achieved if worldwide strong measures are taken in all sectors and countries (Edenhofer, 2010; MC Kinsey & Company, 2009; IEA, 2010, Van Vuuren, 2011). It will require big but achievable investments in the coming decades. All studies indicate that the marginal costs of measures will increase over time, in order to remain on a path towards 2°C stabilization towards 2100. There is also a trade-off; fewer measures in the coming decades (and thus lower marginal costs) will imply more costly measures and higher marginal costs in later decades. The range of marginal costs is similar to the range in the study of Kuik (2009). The meta-analysis of Edenhofer (2010) shows the largest range (60 – 550 $/ton CO2eq), with lower band for coming decades and higher band for later decades (2060). The report by McKinsey (2012) focusses on measures in short-medium term (towards 2030) with marginal costs of 60 €/ton CO2eq (for 2030). After 2030, more expensive measures are required with marginal costs up to 100 €/ton CO2eq.

Thirdly, there are updates guidelines on the use of social costs of carbon. There is an update of the Handbook for External costs of transport for the EC, DG Mobility and Transport (Ricardo-AEA, 2014). This study builds on the same studies as De Nocker (2010) and MMG2012 report (Debacker et al., 2012) and especially on Kuik (2009) for prevention costs of global warming. The study recommends a central value of 90 €/ton CO2eq for emissions in 2015. The UK has updated its guidelines for social costs, that are based on prevention costs ranging from 79 €/ton CO2eq for 2015 to 281 £/ton CO2eq for 2050. The US government recommends a social cost of carbon for use in cost-benefit analysis with a range of 4 € to 53 €/ton CO2eq (US DOE, 2010). The difference between the US and European recommendations reflects the differences in climate change policies, in line with the prevention cost method.

4.1.2.4 Estimation of the monetary indicator

As explained in MMG2012, the external costs of emissions of greenhouse gases will increase in the coming decades with 6 % per year, reflecting both that marginal costs of meeting GHG-emission targets will become more costly (as the world's economy grows and targets become tighter) and that marginal damages increase as the world gets warmer. For MMG2012, we recommended to use a starting value of 0.02 euro/kg CO2eq in 2010 and an annual increase of 6.2 %.

For the MMG2014 method, we developed a better procedure to estimate a single value, based on the following approach:

- The MMG2012 value of 0.02 euro/kg CO2eq in 2010 with an annual increase of 6.2 %.
- Distribution of emissions of GHG over time, following the attribution of emissions to different phases as in Debacker et al. (2013) (30 % to the production and replacement, 65 % to the use phase and 5 % to end of life).
- Accounting for past inflation, expected future economic growth (based on IPCC scenarios) and discounting (3 %), in line with the assumptions and approach in Ricardo-AEA (2014). This approach is also consistent with how future impacts of current emissions (e.g. chronic health impacts) are dealt with for other impact categories.

We rounded the result of the analysis to 0.1 €/kg CO2eq or 100 €/ton CO2eq This result is similar to the 90 €/ton CO2eq from the most recent guidelines for external costs of transport for the EC, DG Mobility and Transport (Ricardo-AEA, 2014). The figure in our study is higher because the processes we look at partly occur further in the future, for which higher costs apply. Our figure is also in line with the methodological guidelines for Germany, that recommend a value of 80 €/ton CO2eq.

In addition, we calculated specific numbers for the construction, use and end of life phase. These impacts are different as we assume these emissions occur in different periods (2013 for construction; 2013-2063 for use phase and after 2063 for end of life). Numbers have been rounded.

4.1.2.5 Spatial variation

The location of the emissions of GHG has no impact on the damages, nor on the prevention costs (as the targets relate to total worldwide emission limits, and assuming least cost approaches). Consequently, there is no need to adapt the monetary value to the location of the emissions.

4.1.2.6 Uncertainty ranges and minimum and maximum estimates

The above mentioned guidelines (Ricardo-AEA, 2014) and UBA (2012) use a bandwidth of a factor 3 and 3.5 between the minimum and maximum value. The latter reflects the span between the minimum and maximum range of the meta analysis of Kuik (2008).

If we use half of the standard deviation, estimated at 5 in MMG2012, the bandwidth would be 6.2. We account for the fact that in recent years there is more consensus between guidelines on what values to use, and for MMG2014, we use a bandwidth of 4.

A bandwidth of 4 between our minimum and maximum estimate means that:
Low estimate = central estimate / √4 = central estimate / 2
High estimate = central estimate * √4 = central estimate * 2

4.2 Depletion of the stratospheric ozone layer

4.2.1 Monetary value

Depletion of ozone layer		Value € / kg CFC-11 eq.		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	49.1	25	100
	Flanders / Belgium	49.1	25	100
	Rest of world	49.1	25	100

Table 10: Monetary indicator for depletion of the stratospheric ozone layer

Source: VITO 2014

4.2.2 Methods and data used

4.2.2.1 Indicator and new information

The indicator and central values are the same as in MMG2012. They are based on damage costs for health, agriculture and materials (Goedkoop, 2008; De Bruyn, 2010).

4.2.2.2 Spatial variation.

Similar to global warming, there is no need to adapt the monetary value to the location of the emissions.

4.2.2.3 Uncertainty ranges and minimum and maximum estimates.

We apply the general approach, based on standard deviation estimated in MMG2012 ($\sigma g = 4$)
This results in a bandwidth of 4 between our minimum and maximum estimate:
Low estimate = central estimate / $\sqrt{4}$ = central estimate / 2
High estimate = central estimate * $\sqrt{4}$ = central estimate * 2

4.3 Acidification of land and water sources

4.3.1 Monetary value

Acidification land water		Value € / kg SO2 eq.		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	0.43	0.22	0.88
	Flanders / Belgium	1.01	0.5	2.02
	Rest of world	0.17	0.00	2.02

Table 11: Monetary indicator for acidification of land and water sources

Source: VITO 2014

4.3.2 Methods and data used

4.3.2.1 Indicator and new information

The indicator and information are the same as in MMG2012, but the new figures account more specifically for spatial variation. Whereas the central figure in MMG2012 is more close to the figure for Flanders; within this latest version the central figure for Western Europe is now lower.

The data are based on the ExternE-method (EC, 2005). It includes damage costs for impacts on building materials and ecosystems, assessed based on the Ecosense model (ExternE-Needs) and impacts on ecosystems (PDF) valued based on restoration costs (Ott, 2006). It does however not include impacts from SO2 on agriculture.

4.3.2.2 Spatial variation

Differences between Western-Europe and Flanders reflect differences in impacts from emissions due to differences in fate modelling (e.g. wind direction and speed), land use, precipitation, building materials used, etc. Data are calculated by the EcoSense-model, as reported for the cases project. Data for Western Europe are based on average EU 27 and data for Flanders are based on Belgium.

World values are based on data for Western Europe, adapted for differences in GDP/capita between Europe and the world average, while accounting for differences in price levels (Purchase Power Parity (PPP) World Bank). Consequently, the value for the rest of the world is 40 % of the value for Western Europe.

4.3.2.3 Uncertainty ranges and minimum and maximum estimates

For Western Europe and Flanders/Belgium, we apply the general approach, based on standard deviation estimated in MMG2012 ($\sigma g = 4$). This results in a bandwidth of 4 between our minimum and maximum estimate:

Low estimate = central estimate / $\sqrt{4}$ = central estimate / 2)
High estimate = central estimate * $\sqrt{4}$ = central estimate * 2

For the low and high value for rest of world region we assume that values can be as low as zero and be as high as the maximum value of the Flemish region, where acidification of land and water sources is (still) problematic.

4.4 Eutrophication

4.4.1 Monetary value

Eutrophication		Value € / kg $(PO4)^{3-}$ eqv.		
Indicator	Region	Central	Low	High
Full life cycle	Western Europe	20	6.6	60
	Flanders / Belgium	40	9.2	175
	Rest of world	8	0	175

Table 12: Monetary indicator for eutrophication

Source: VITO 2014

4.4.2 Methods and data used

4.4.2.1 Indicator and assessment models

In the MMG2014 method, the indicator for eutrophication (kg (PO4)3- eqv and LCIA model to assess this indicator (CML 2012) have changed compared to MMG2012 (ReCiPe).

The environmental costing methods and data to monetise the indicator are the same as in MMG2012. It is based on a literature review of studies that use different methods, including 'Willingness to pay' to avoid eutrophication impacts (Gren, 2008), fate and impact modelling to estimate impacts on biodiversity (ExternE-EcoSense) that valued using 'restoration costs' (Ott, 2006) and studies on the 'prevention costs' to meet the objectives for freshwater quality, as required by the European water framework directive (Broekx et al, 2009). The results are expressed in €/kg N, and are calculated to € / kg (PO4)3- eq. based on characterization factors of CML (1 kg N= 0.42 kg (PO4)3-).

4.4.2.2 Central value, uncertainty ranges and spatial variation

The procedure to estimate central, low and high value is somewhat different compared to other impact categories. As there is a limited number of studies with results that vary widely, we have used these results to determine the low and high value. Assuming a lognormal distribution of the values, we estimate the central value.

The central value for Western-Europe is 20 € / kg (PO4)3- eq., with a bandwidth of 6.6 to 60. The low value is based on the impact of emissions of N on ecosystems (PDF), which are valued based on restoration costs from Ott (2006). The high estimate is based on the Willingness to pay to avoid eutrophication (Gren, 2008). The bandwidth between the low and high estimate is a factor of 9. This wide range reflects the big uncertainties involved in estimating and valuing eutrophication impacts. This bandwidth does not reflect all uncertainties, e.g. uncertainties related to the use of different indicators (N versus P) and their relative weights as reported in CML-2012, ReCiPe (Goedkoop et al., 2008) and MIRA (2007) is not included.

Assuming a lognormal distribution and the low and high values, the estimate for the central value for Western Europe is 20 (= 60 / 3).

For Flanders, the low value is based on the same source as for Western Europe, but because of spatial variability, the result is 50 % higher. The high cost estimate is based on the prevention costs to meet the objectives of the Water Framework Directive for freshwater in Flanders (Broekx et al, 2009). These costs are relatively high because the pressures are high in Flanders, and the more expensive measures are required to meet the WFD targets.

For the rest of the world, we adapt the value of Western Europe for differences in GDP per capita (PPP). For the low and high value for rest of world region we assume that values can be as low as zero and be as high as the maximum in Flanders.

4.5 Formation of tropospheric ozone photochemical oxidants

4.5.1 Monetary value

Ozone (low level) formation		Value € / kg ethene eq.		
Indicator	Region	Central	Low	High
Full life cycle	Western Europe	0.48	0	6.6
	Flanders / Belgium	3.3	0	6.6
	Rest of world	0.18	0	6.6

Table 13: Monetary indicator for formation of tropospheric ozone photochemical oxidants

Source: VITO 2014

4.5.2 Methods and data used

4.5.2.1 Indicator and assessment model

In the MMG2014 method the indicator for low level ozone formation (kg ethene eqv; (C2H4 eqv) and LCIA model to assess this indicator (CML 2012) have changed compared to MMG2012 (i.e. ReCiPe midpoint, expressed in kg NMVOC). The environmental costing methods and data to monetise the indicator are the same as in MMG2012, but we have used a different reasoning to account for variation and uncertainty.

4.5.2.2 Central value, uncertainty ranges and spatial variation

The data are based on the ExternE-method that estimates the impacts of ozone formation, especially on public health (EC, 2005). The fate modelling is based on the Ecosense model (W-Europe) and on the Beleuros model (data for Flanders). The impact assessment and their valuation is for both based on the ExternE method. This method results in different estimates for different ozone precursors (NOx and NMVOC).

As the process of ozone formation is a complex one, in which ozone precursors (pollutants that contribute to ozone formation (NOx and NMVOC) can contribute to both ozone formation and decomposition, the marginal impact of different pollutants varies a lot in size, over space and time. Our approach to the low and high values and spatial variation accounts for these differences.

The low values are based on the contribution of emissions of NOx. This contribution is nil, because in some situations the addressed substances do not lead to photochemical oxidant formation (ozone precursors can have both the effect of formation as defragmentation of photochemical ozone). The high values are based on impacts of ozone precursors with net contribution to ozone formation (NMVOC). This approach is different compared to MMG2012, in which we only used data for NMVOC.

The data for Western Europe are based on data from the ExternE project (EC, 2005), as calculated by Ecosense and reported for the case projects for EU 26 (IER, 2008). The data for Flanders are based on calculations by De Nocker et al (2010). For the rest of the world, values for Western Europe were adapted for differences in GDP/capita (PPP) between Europe and the rest of the world.

To estimate the high value for Western Europe and Flanders, we applied the general approach, based on a bandwidth of 16.

Central estimate = high estimate /√4 = high estimate / 2)

For Europe and for the rest of world, we assume that the high value can be as high as the maximum for Flanders (i.e. 6.6 € / kg ethene eq.).

4.6 Abiotic depletion of non-fossil resources

4.6.1 Monetary value

Depletion of non-fossil resources		Value € / kg Sb* eqv.		
Indicator	Region	Central	Low	High
Full life cycle	Western Europe	1.56	0	6.23
	Flanders / Belgium	1.56	0	6.23
	Rest of world	1.56	0	6.23

Table 14: Monetary indicator for abiotic depletion of non-fossil resources

*Sb: antimonium

Source: VITO 2014

4.6.2 Methods and data used

4.6.2.1 Indicator and assessment model

The methods and data to monetise abiotic depletion of non-fossil resources are based on ReCiPe, as in MMG2012. However, as in ReCiPe the indicator is based on $ / kg Fe eq., and as there is no conversion factor given for Sb, we have to add an additional step to express the costs in terms of €/kg Sb eq. In addition, we have used a different reasoning to account for variation and uncertainty.

4.6.2.2 Central value, uncertainty ranges and spatial variation

Low estimate

The low value is nil, and reflects the point of view that resource depletion costs are internalized in prices (De Bruyn, 2010). This argument is further discussed in MMG2012.

High estimate

The high estimate is based on the additional costs for future generations for extraction of raw materials. The assessment and valuation methods are discussed in MMG2012. It is based on ReCiPe (Goedkoop, 2008) and the additional costs of energy required for extraction that can be interpreted as a 'resource depletion cost'. It amounts to 0.0518 €/kg Fe, which corresponds to 0.1 % of the market price of Fe (market price of 1 kg Fe equivalent = 72 $/kg).

The resource depletion costs differ between resources, depending on e.g. the current grade of metals or minerals in the ore. In ReCiPe, the resource depletion costs are calculated for 20 substances. Expressed as a % of the market price of the substances, the resource depletion cost varies from 0.1 % to + 600 %, with a median value of 3.4 % and an average of 88 % (see table below).

ReCiPe does not give a value for Sb. If we were to apply the characterisation factor from ILCD (1 kg Fe= 1.66 E-6 Sb), the resource depletion cost of Sb would amount to 42.2 k€/kg Sb, which is more than 4000 times the market price. If we were to use this characterisation factor as a basis for assessment of resource depletion costs, we would on average overestimate the resource depletion costs with many orders of magnitude (see table 12, method 1, columns 4 to 6). From the 13 substances for which we can make a direct comparison with ReCiPe, only for Fe, Cu and Mn this resource depletion cost estimate is in the same order of magnitude compared to those of ReCiPe. For all other materials, this approach would - compared to ReCiPe - overestimate the resource depletion costs with on average 4 orders of magnitude. If we were to use this approach, the average resource depletion costs would amount to 384 times the market prices. This cannot be justified based on the calculations from ReCiPe.

Therefore, we estimate the resource depletion costs for Sb based on the average value for the ratio between resource depletion costs - as estimated in ReCiPe - and the market price. On average, this ratio is 82.9 % (column (3) of table 15 and 16). At a market price of 9.4 $/kg Sb, this results in a resource depletion cost of 7.8 $/kg Sb or 6.23 €/kg Sb. This approach results in resource depletion costs for the different substances that are on average within the same order of magnitude compared to those of ReCiPe. For the majority of the substances, this estimate is lower compared to the ReCiPe estimate.

To estimate the central value, we apply the general approach, based on the standard deviation estimated in MMG2012 ($\sigma g = 5$). We use a bandwidth of 16 between the low and high estimate. Thus we calculate:

— Central estimate = high estimate / $\sqrt{16}$ = high estimate / 4
— Central estimate = 6.23 €/kg Sb / 4 = 1.56 €/kg Sb

As the market for raw materials are global markets and depletion is a world-wide problem, we do not apply different values for different countries or regions.

	RDC ReCiPe $/kg	market price $/kg	ratio =(1)/(2) %	RDC Method 1 $/kg	ratio =(4)/(1) %	ratio =(4)/(2) %
	(1)	(2)	(3)	(4)	(5)	(6)
Ag	20,48					
Al	0,0064	1,3	0,49%	1,07	16670%	82%
Ay	5,006					
Co	0,072	46	0,16%	1079,5	1499331%	2347%
Cr	1,78	9,6	19%	0,83	46%	8,6%
Cu	3,06	7	44%	105,4	3445%	1506%
Fe	0,07	72	0,10%	0,07	100%	0,10%
Ir	6,62	42000	0,02%	560,8	8472%	1,3%
Mn	5,48	1,5	365%	0,99	18%	66%
Mo	14,85	30	50%	2998,19	20190%	9994%
Ni	0,9	40	2,25%	176,3	19585%	441%
Os	464	77000	0,60%			
Pb	0,13	2,5	5,20%	632,5	486562%	25301%
Pd	273	58330	0,47%	394699	144578%	677%
Pt	11652	5400	216%	383313	3290%	7098%
Rh	1455	80000	1,82%			
Ru	144	14000	1,03%			
Sn	90,99	13	700%	4849,4	5330%	37303%
Zn	0,16	3,5	4,57%	153,9	96197%	4398%
Sb	na	9,4		42.168,7	na	448603%
Average			82,9%		164.558%	38.416%

Table 15: Resource depletion costs (RDC) for different substances, based on characterization factor ILCD (method 1)

na = not available
(1) RDC = Resource depletion costs, as estimated in ReCiPe (Goedkoop, 2013) p 117, in $/kg
(2) Market prices, in $/kg (Wikipedia 2014, compiled from different sources)
(4) RDC = Resource depletion costs, using method 1 = based on the RDC for Fe, the characterization factors from ILDC for Sb (1 kg Fe= 1.66 E-6 Sb) and for other metals.

	RDC ReCiPe $/kg	market price $/kg	ratio =(1)/(2) %	RDC Method 2 $/kg	ratio =(4)/(1) %	ratio =(4)/(2) %
	(1)	(2)	(3)	(4)	(5)	(6)
Al	0,0064	1,3	0,49%	0,0002	3,1%	0,02%
Co	0,072	46	0,16%	0,200	277%	0,43%
Cr	1,78	9,6	19%	0,0002	0,0%	0,00%
Cu	3,06	7	44%	0,019	0,6%	0,28%
Fe	0,07	72	0,10%	0,00001	0,0%	0,00%
Ir	6,62	42000	0,02%	0,10	1,6%	0,00%
Mn	5,48	1,5	365%	0,0002	0,0%	0,01%
Mo	14,85	30	50%	0,55	3,7%	1,85%
Ni	0,9	40	2,25%	0,033	3,6%	0,08%
Os	464	77000	0,60%			
Pb	0,13	2,5	5,20%	0,117	89,9%	4,68%
Pd	273	58330	0,47%	72,95	26,7%	0,13%
Pt	11652	5400	216%	70,84	0,6%	1,31%
Rh	1455	80000	1,82%			
Ru	144	14000	1,03%			
Sn	90,99	13	700%	0,896	1,0%	6,89%
Zn	0,16	3,5	4,57%	0,028	17,8%	0,81%
Sb	na	9,4		7,8		82,9%
Average			82,9%	10,24	30,4%	6,6%

Table 16: Resource depletion costs (RDC) for different substances, based on ratio to market prices (method 2)

(1) RDC = Resource depletion costs, as estimated in ReCiPe, Goedkoop, 2013, p 117, in $/kg
(2) Market prices, in $/kg (Wikipedia 2014, compiled from different sources)
(4) RDC = Resource depletion costs, using method 2 = based on the average ratio RDC/market price and applied to Sb market prices for Fe, the characterization factors from ILDC for Sb (1 kg Fe= 1,66 E-6 Sb) and for other metals.

4.7 Abiotic depletion of fossil resources

4.7.1 Monetary value

Depletion of fossil resources		Value € / MJ, net caloric value		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	0	0	0.0065
	Flanders / Belgium	0	0	0.0065
	Rest of world	0	0	0.0065

Table 17: Monetary indicator for abiotic depletion of fossil resources

Source: VITO 2014

4.7.2 Methods and data used

4.7.2.1 Indicator and assessment model

As in the MMG2012 method our best estimate for the central value is a zero value (Debacker et al., 2012), we have added now a value for the high estimate.

4.7.2.2 Central value, uncertainty ranges and spatial variation

The central and the low value is zero. This reflects the point of view that resource depletion costs are internalized in market prices (De Bruyn, 2010, Aguilera 2009, Aguilera 2013). In addition, MMG2014 uses a valuation of greenhouse gas emissions based on prevention costs, which assume that we limit emissions of greenhouse gasses to limit global warming to a maximum of 2°C, in line with the UN objective. This emission path limits the use of fossil fuels, irrespective of its availability (De Nocker, 2010).

The high value is based on the Eco-indicator 99 method. (Allacker et al, 2012) and costs of military action to secure energy supply (NDCF, 2007; Stern, 2010 as cited in VTPI, 2013).

As energy markets are global markets and depletion is a world-wide problem, we do not apply different values for different countries or regions.

5 Monetary indicators for CEN+ indicators

5.1 Human toxicity, cancer effects

5.1.1 Monetary value

Human toxicity cancer		€/ CTUh*		
	Region	Central	Low	High
	Western Europe	665,109	166,277	2,660,434
	Flanders/Belgium	680,451	170,113	2,721,802
	rest of world	266,043	66,511	2,721,802

Table 18: Monetary indicator for human toxicity cancer

*CTUh = (comparative toxic units human health)
Source: VITO 2014

5.1.2 Methods and data used

In the MMG2012 method, impacts on human health are assessed based on ReCiPe (Goedkoop et al., 2008), and expressed in terms of DALY (disability adjusted life years).

In the MMG2014 method, the impacts are assessed based on the USEtox method (Rosenbaum, 2008) and expressed in CTUh (comparative toxic units human health).

Quantification: The CTUh case = 1 cancer = 11.5 DALY (Rosenbaum, 2008; JRC 2011, p 31.)

Valuation :
Costs of cancer = cost of medical care (a) + loss of production value (b) + loss of life expectancy (c)

(a) + (b) are estimated based on a study of total costs of cancer in EU = 51,429.6 € (Luengo-Fernandez, 2013)

(c) Assuming 1 DALY related to cancer corresponds to 1 YOLL (years of life lost), then the value of a life year lost (VOLY) = 53,363.5 €/DALY (W-Europe)

CTUh cancer = 51,429.6 € + (11.5 x 53,363.5) = 665,109 € / CTUh.

5.1.3 Further explanation

The assessment is based on the scientific consensus model USEtox, developed in the framework of the UNEP–SETAC Life Cycle Initiative (Rosenbaum, 2008).

The characterisation factor for human toxicity (human toxicity potential) is expressed in comparative toxic units (CTUh). CTUh describes the estimated increase in morbidity in the total human population per unit mass of a chemical emitted (cases per kilogram), assuming equal weighting between cancer and non-cancer due to a lack of more precise insights into this issue (Rosenbaum, 2008). The cases can be converted to DALYs based on information of the ILCD Handbook (2011), making a distinction between cancer and non-cancer cases.

The total costs of a cancer case include:

(a) health care costs;
(b) loss of productivity for patient and family;
(c) loss of value of life expectancy for the patient.

The costs related to health care (a) + loss of productivity (b)

— These costs are quantified based on a recent study of total costs of cancer for the EU published in the Lancet Oncology (Luengo-Fernandez, 2013). It estimates total costs in the EU to be 126 billion (2009) of which 40 % for health care costs. These total costs per case of cancer (diagnosed) is 51,429 euro.
— This figure is substantially lower than the 450,000 euro/cancer case used in ExternE, but the sources for these latter data are not well documented (Eftec, 2004). Health care costs for the EU given by Luengo-Fernandez (2013) are in the same order of magnitude as health care costs for the US (Yabroff, 2007 cited in Mariotto, 2011).
— We assume the same value for Western Europe and Flanders. Despite the similarities between Europe and the US, we adapted the value for the rest of the world based on the differences in GDP/capita (PPP) between Europe and world.

The costs related to loss of value of life expectancy for the patient (c)

— In addition, it is estimated that on average a cancer case results in 11.5 DALYs loss of years of health life (JRC 2011), based on Huijbregts (2005). We assume that the value of 1 DALY for cancer equals 1 YOLL (year of life lost). It has been discussed in literature to which extent one should apply a 'cancer premium'. Some studies have found such premium which reflects a 'dread' factor and accounts for more pain and suffering associated with cancer (Eftec, 2004; Cropper et al, 2011). We do not apply such a cancer premium, because the evidence in literature is mixed. Whereas e.g. Hammit (2004) and Van Houtven (2008) found an important cancer premium (at least a factor 1.3), other recent studies Hammitt and Haninger (2010) Adamowicz et al. (2009) did not.

Valuation of a DALY:

We assume that the value of 1 DALY (disability adjusted life year) for cancer equals 1 YOLL (year of life lost), which is well justified for valuation of health impacts related to cancer. It has also been demonstrated that for the valuation of the major morbidity impacts the results using DALYs or using other data from economic literature results in identical impacts (Desaigues, 2007).

For the valuation of a YOLL, we use the figures for EU-25, based on survey in different EU countries in the context of the EU Needs project (40000 € in €2000 prices) (Desaigues et al, 2011) and corrected for inflation to 2012 price levels based on Eurostat HICP data).

— Western Europe = 1 DALY = 1 YOLL = 53,363 € (based on values for Europe in Desaigues et al, 2011)
— Valuation data for Flanders are based on EU wide data. As values depend on income per capita, the NEEDS study recommends to adapt the figures for local GDP/capita. GDP/capita in Flanders is 2.5 % above EU-average. 1 DALY = 54,697 €
— Rest of World: 1 DALY = 1 YOLL = 21,345 €/=YOLL (based on EU2013 value and differences in GDP/capita (PPP) between Europe and world).

For comparison: the economic valuation of a DALY accounting for all information on mortality and morbidity gives much broader ranges (from 5,000 € to 400,000 €). Central estimates vary from 10.000 to 100,000 €.

5.1.4 Low and high values

Rosenbaum reports uncertainty boundaries that reflect a standard deviation of 20 (Rosenbaum, 2008 and JRC, 2011). We have no yardsticks to compare this uncertainty assessment with that of other impact categories. The large uncertainty boundaries for human toxicity reflects that this impact category involves a wide range of exposure routes for a wide range of chemicals, and that for a large number of these chemicals the information of potential effects on human health is scarce, and has to be estimated based on e.g. animal experiments. In this context, it is common to work with uncertainty boundaries covering several orders of magnitude (Rosenbaum, 2008). A particular issue is that for a number of substances, information is missing, and impacts are estimated as zero impacts (JRC, 2011). Since the publishing of the assessment of USEtox method within the framework of the ILCD Handbook (2011) more chemical substances have been modelled.

Because impacts are both relatively uncertain and highly variable, we use a relative high bandwidth of 16.

Low value = central value / √16 = central value / 4
High value = central value * √16 = central value * 4

The estimate for Flanders and for the rest of the world are based on the figure for Western Europe, adapted for differences in GDP per capita. For rest of the world, we assume that the maximum value = max value for Flanders.

5.2 Human toxicity, non-cancer effects

5.2.1 Monetary value

Human toxicity non-cancer		€/ CTUh*		
	Region	Central	Low	High
	Western Europe	144.081	28.816	720.407
	Flanders/Belgium	147.683	29.537	738.417
	rest of world	57.633	11.527	738.417

Table 19: Monetary indicator for human toxicity non-cancer

* CTUh = comparative toxic units human health
Source: VITO 2014

5.2.2 Methods and data used

The environmental costing method is similar to that for human toxicity cancer. The main difference is the interpretation of a CTUh for non-cancer.

Quantification: 1 CTUh case = 1 non-cancer = 2.7 DALY (Rosenbaum, 2008; JRC, 2011, p 31)
The valuation of DALY is discussed above.

The uncertainties are larger for non-cancer compared to cancer (Rosenbaum, 2008 and JRC, 2011). To reflect this difference, we assume a bandwidth of 25 between the low and high value:

Low value = central value / √25 = central value / 5
High value = central value * √25 = central value * 5

5.3 Particulate matter

5.3.1 Monetary value

Particulate matter		Value € / kg PM 2.5		
Indicator	Region	Central	Low	High
Full life cycle	Western Europe	34.0	12.7	85.0
	Flanders / Belgium	57.8	21.7	144.5
	Rest of world	7.7	2.9	19.2

Table 20: Monetary indicator for Particulate matter

Source: VITO 2014

5.3.2 Methods and data used

In the MMG2012 assessment method, assessment and valuation were based on ReCiPe (Goedkoop et al., 2008). The data were based on the application of the ExternE accounting framework (Preiss and Klotz, 2008) to Flanders, in the context of a project for VMM and using specific air quality models and data for Flanders (De Nocker, 2010). In the MMG2014 assessment method, environmental costing models are also based on the ExternE accounting framework (Preiss and Klotz, 2008), as applied to EU 28 countries, using specific air quality models and data for EU 28.

Because the indicator and LCIA model have changed, we explain more in detail how the values were calculated. This level of detail is also justified because impacts of PM on public health are likely to be an important part of total monetized impacts for most applications.

Step 1: fate, exposure and effect analysis

— We use the results from the European LC-Impact project (2010-2013) (Preiss, 2013). In the LC-Impact project, estimates for primary PM and for PM precursors have been estimated for emissions from all regions (countries) in the world, building on earlier work and using the models that have been identified by ILCD as the best to use (RiskPoll, Humbert) (Preiss, 2013, ILCD, 2008). Emissions of precursors (SO2, NOx and NH3) will lead to an increase of PM aerosols in ambient air, and to related health impacts.

— The main progress of the LCA-Impact model relates to the impact factor, estimating the impact of emissions on intake by humans, while accounting for differences in population density exposed to the emission, distance between emission and population exposed (typically stack height) and meteorological conditions, especially wind speed and atmospheric mixing height. These differences are dealt with using 'archetypes' (remote, rural, urban) for emissions, as developed by Humbert (2011) (and reflecting the work of the UNEP/SETAC) Life Cycle Initiative). The impacts are modelled combining several dispersion models.

— The effect factor builds on the dose-response functions as developed in the EU-Heimsta project (Torfs et al, 2007) used for many assessments by the EU and WHO, and their aggregation into DALYs (as defined by WHO, 2012). The effect estimates account for differences in toxicity between primary PM and PM precursors and differences between countries related to age distribution.

— As final deliverables were not yet available, we used the results of the project as presented at SETAC 2013. European values are based on the average of values for Western and Eastern Europe (although differences are small). Impacts for emissions in Flanders are higher compared to EU-average (on average 1.8 times). This is based on the relationship between estimates for Belgium and EU average, as assessed by LC-Impact (PM (average for low and high stacks) or Cases (SO_2, NO_x, NH_3 (unspecified stack heights).

—

Step 2: valuation

— As the major impact from particulate matter relates to impacts on mortality (loss of life expectancy), we use the information on valuation of mortality impacts from PM to value a DALY in monetary terms. This valuation is based on the value of a YOLL (Year of life lost). We assume that 1 DALY equals 1 YOLL, which is well justified for valuation of health impacts from PM. It has also been demonstrated that for the valuation of the major morbidity impacts the results using DALYs or using other data from economic literature results in identical impacts (Desaigues, 2007).

— For the valuation of a YOLL, we use the figures for EU-25, based on survey in different EU countries in the context of the EU Needs project (40000 € in €2000 prices) (Desaigues et al, 2011) and corrected for inflation to 2012 price levels based on Eurostat HICP data.

  – Western Europe: 1 DALY = 1 YOLL = 53.363 €
  – Valuation data for Flanders are based on EU wide data. As values depend on income per capita, the NEEDS study recommends to adapt the figures for local GDP/capita. GDP/capita in Flanders is 2.5 % above EU-average. 1 DALY = 54.697 €
  – Rest of the world: 1 DALY = 21.345 €/=YOLL (based on EU2013 value and differences in GDP/capita (PPP) between Europe and world).

For comparison: the economic valuation of a DALY accounting for all information on mortality and morbidity gives much broader ranges (from 5.000€ to 400.000€. Central estimates vary from 10.000 to 100.000 €).

Results: step 1 x step 2

See table 21 to table 24

5.3.3 Low and high values

The uncertainty boundaries for PM 2.5 especially reflect a wide variation in impacts, depending on variations in fate and exposure assessment.

There is a wide variety in impacts and effects, depending on the variation in situations (population density, stack height, etc.). The variation is very high (two orders of magnitude) for emissions of different sectors and countries, but there is less information about the variation if applied in LCIA analysis, where emissions are always a combination of emissions from different countries, regions and/or sectors.

Although uncertainties remain related to impact assessment and their valuation, there is more consensus and certainty about the impacts per kg inhaled, as these elements have been subject to many studies and policy decisions over the last 20 years. This is e.g. reflected in the fact that most assessment models use similar dose-response functions, at least for the most important impacts.

To calculate low and high values, we have applied an uncertainty range of a factor of 2 for impact assessment, which reflects the differences between models (Preiss, 2013). In addition, we have used a range of +/- 25 % to account for variation and uncertainty in valuation. If we apply these ranges to the estimates of DALY/ton and value/DALY, it results in a bandwidth of 6.7 between the high and low estimate.

This is a wider range compared to most other impact categories. It does not mean that these impacts are more uncertain, but that there is a larger variation in impact between sites (emissions in rural versus urban areas) and between sectors (high stack emissions versus low stack emissions (transport)).

Discussion:

For PM2.5, the results relate very well to the estimates based on the more detailed EcoSense runs, as used in and reported for the EU CASES project (Feem, 2008). For SO2, NOx and NH3, LC-Impact results are lower compared to LC-Impact, but fall within the uncertainty boundaries as calculated in table 21.

It has to be noted that the uncertainty boundaries are similar to those for global warming, although the rationale behind is rather the opposite. For global warming, there is a consensus and little uncertainty about fate and exposure for different GHG emissions from different sectors and countries (contribution to global warming), but large uncertainties related to the impacts and valuation of global warming. For PM, there is a lot of consensus about effects and their valuation, but large variation in fate and exposure for emissions from different sectors and countries.

Particulate matter				
€/PM 2.5 kg	**Region**	**Central**	**Low**	**High**
	Western Europe	34.0	12.7	85.0
	Flanders	57.8	21.7	144.5
	World	7.7	2.9	19.2
€/NO_x kg	**Region**	**Central**	**Low**	**High**
	Western Europe	5.1	1.9	12.8
	Flanders	7.2	2.7	17.9
	World	1.3	0.5	3.3
€/SO_2 kg	**Region**	**Central**	**Low**	**High**
	Western Europe	5.5	2.1	13.7
	Flanders	9.0	3.4	22.4
	World	2.9	1.1	7.3
€/NH_3 kg	**Region**	**Central**	**Low**	**High**
	Western Europe	17.2	6.4	42.9
	Flanders	47.0	17.6	117.4
	World	2.6	1.0	6.6

Table 21: Damage costs per kg emitted, for primary particulate matter and particulate matter precursors

Particulate matter : background information, impact assessment (DALY/ ton emitted)				
DALY/PM 2.5 ton	**Region**	**Central**	**Low**	**High**
	Western Europe	0.64	0.32	1.27
	Flanders	1.06	0.53	2.11
	World	0.40	0.20	0.79
DALY/ NO_x ton	**Region**	**Central**	**Low**	**High**
	Western Europe	0.10	0.05	0.19
	Flanders	0.13	0.07	0.26
	World	0.07	0.03	0.14
DALY/ SO_2 ton	**Region**	**Central**	**Low**	**High**
	Western Europe	0.10	0.05	0.21
	Flanders	0.16	0.08	0.33
	World	0.15	0.08	0.30
DALY/ NH_3 ton	**Region**	**Central**	**Low**	**High**
	Western Europe	0.32	0.16	0.64
	Flanders	0.86	0.43	1.72
	World	0.14	0.07	0.27

Table 22: Impacts per ton emitted, for primary particulate matter and particulate matter precursors

Method: damage based: Data sources: LC-Impact (Preis, 2013)

Valuation of DALY	**background information, valuation (€/ DALY)**		
Region	**Central**	**Low**	**High**
Western Europe	53,363	40,023	66,704
Flanders	54,698	41,023	68,372
World	19,816	6,605	68,372

Table 23: Valuation of health impacts, DALY (disability adjusted life years)

Low (high) values = central value - / + 0.25 * central value

External costs		**In € / kg**	
Pollutant	Sector	EU 27	Belgium
PM 2.5	Unspecified sector and stack height	31.2	60.9
	Low stacks (households)	32.8	61.4
	High stacks (power, industry	15.5	26.3

Table 24: Impacts from emissions calculated by EcoSense: for year 2015, in EU 27 and Belgium

Source: Cases, FEEM, 2008

5.4 Ionising radiation: human health effects

5.4.1 Monetary value

Ionising radiation, human health		Value € / kg U235 eqv.		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	9.7E-04	3.2E-04	2.9E-03
	Flanders / Belgium	9.9E-04	3.3E-04	3.0E-03
	Rest of world	3.6E-04	1.2E-04	3.0E-03

Table 25: Monetary indicator for ionising radiation, human health

Source: VITO 2014

5.4.2 Methods and data used

In the MMG2012 assessment method the fate and impact assessment was based on the endpoint methods within ReCiPe. For valuation, the monetary indicator for DALY was used.

In the MMG2014 assessment method, quantification is based on midpoint assessment (Frishknecht et al, 2000) as implemented in ReCiPe midpoint assessment.

Quantification

The exposure and fate factor for 1 kg of uranium U235 eqv. is 1.40E-08 Man Sievert (Man.Sv) per kBq (Goedkoop et al. 2008, Table 9.1).

Second, the impact factor per man.Sv is 1.17 DALYs (Goedkoop et al. 2008, Table 9.3).

The total number of DALYs per kg U235 eq. is 1.64E-08.

Valuation

The valuation of a DALY in this context needs to account for the type of health impacts. The health impacts from exposure to Ionising radiation relate to cancers (fatal and non-fatal) and severe hereditary effects.

For hereditary effects, the loss of life expectancy is estimated at 59 years (Goedkoop et al. 2008), and – in line with valuation of life expectancy for other health impacts – loss of 1 year is estimated at 53 k€. The valuation of cancer accounts - As explained above (paragraph 5.8) - for loss of life expectancy (part A) and in addition the costs of medicine and loss of labour income (part B). These costs amount to 51 k€ (as explained in paragraph 5.8), divided by the number of DALYs, i.e. 11.5 for fatal cancers and 2.7 for non-fatal cancers.

Accounting for the relative share of these endpoints in the total numbers of DALYs per man.Sv, the average value per DALY is 69,305 €.

Consequently, the value per kg U235 eqv. is 9.7E-04 (central estimate for Western Europe).

	Quantification DALYs /mSv	Valuation € /DALY			Valuation € /mSv	
	(1)	Part a life exp. (2)	Part b medicine (3)	Total (4)	Total (5)	%
Fatal cancers	0.45	53,363	4,472	57,836	26,131	38%
Non-fatal cancers	0.25	53,363	19,048	72,411	18,435	27%
Severe hereditary effects	0.,46	53,363	0	53,363	24,739	36%
Total per m.Sv	**1.,17**				69,305	100%

Table 26: Valuation of DALY for ionising radiation, for Western Europe

(1) Goedkoop et al., 2008, p. 85
(2) Costs related to loss of life expectancy (Assuming 1 DALY = 1 VOLY, valuation, see table 23, for W-Europe)
(3) Costs related to health care and loss of productivity (51 €)/11.7 DALYs for fatal cancers and 2.7 DALYs for non-fatal cancers.
(4) (4)= (2) + (3)
(5) (5)= (1) x (4) t.

5.4.2.1 Uncertainty and low and high estimates

We use a bandwidth of 9, which is larger than for e.g. particulate manner and that reflects that the valuation of severe hereditary effects based on DALYs is very uncertain. It is narrower than for human toxicity, because for ionising radiation the impacts are associated with a limited number of substances.

The estimates for Flanders and for the rest of the world are based on the figure for Western Europe, adapted for differences in GDP per capita. For rest of the world, we assume that the maximum value equals the max value for Flanders.

5.5 Ionising radiation: ecosystems

5.5.1 Monetary value

Ionising radiation, ecosystems		Value €/CTUe*		
Indicator	Region	Central	Low	High
Full life cycle	Western Europe	3.70E-05	7.39E-06	1.85E-04
	Flanders / Belgium	3.70E-05	7.39E-06	1.85E-04
	Rest of world	1.34E-05	2.68E-06	1.85E-04

Table 27: Monetary indicator for ionising radiation on ecosystems

* CTUe = comparative toxic unit for ecosystems
Source: VITO 2014

5.5.2 Methods and data used

See below: valuation for ecotoxicity, freshwater.

5.6 Ecotoxicity, freshwater

5.6.1 Monetary value

Ecotoxicity, freshwater		Value €/CTUe*		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	3.70E-05	7.39E-06	1.85E-04
	Flanders / Belgium	3.70E-05	7.39E-06	1.85E-04
	Rest of world	1.34E-05	2.68E-06	1.85E-04

Table 28: Monetary indicator for Ecotoxicity, freshwater

* CTUe = comparative toxic unit for ecosystems
Source: VITO 2014

5.6.2 Methods and data used

In the MMG2012 assessment method the fate and impact assessment was based on the endpoint method ReCiPe, expressed in PDF/m² (potentially disappeared fraction) and PDF was valued based on restoration costs (Ott, 2006). The indicator used for quantification was expressed in kg 1.4 dichlorobenzene (DB) equivalent.

In the MMG2014, assessment method the characterization factor for aquatic ecotoxicity (ecotoxicity potential) is expressed in comparative toxic units for ecosystems (CTUe) and provides an estimate of the potentially fraction of affected species (PAF) integrated over time and volume per unit mass of a chemical emitted (PAF m3 day kg−1) (Rosenbaum, 2008, p. 7).

Quantification: 1 CTUe, freshwater = 1 PAF/m³ day/kg.

Valuation

We use the value €/PDF as discussed in the MMG2012 assessment method, i.e. 0.46 €/PDF. This value corresponds to the value attached to protection of terrestrial biodiversity, and is based on the costs citizens say are willing to pay to protect biodiversity. The underlying assumption is that all terrestrial species are valued equally.

To value ecotoxicity in freshwater, we have to expand the assumption, and assume that freshwater species are valued identical to terrestrial species. In addition, we assume that the values discussed above relate to all terrestrial species, and that aquatic species are valued on top of these, and that the value as discussed above can be used for aquatic species. This corresponds to 3.7 E-05 €/ CTUe. The details of the calculation are given in Table 29.

Indicators	correction factor	Value
Economic value of terrestrial biodiversity in PDF m².year/kg		0,46
from PDF m² terrestrial to PDF m³ freshwater (1) from PDF to PAF (2) from PAF year to PAF day (3)	0,053 0,55 0,00273973	0,025 0,013 0,0000370
Value ecotoxicity , €/CTUe (in PAF.m³.day/kg)		3,70E-05

Table 29: Correction factors used for valuation of Ecotoxicity, freshwater

(1) Assuming equal value of terrestrial and freshwater species, number of species based on Goedkoop, 2008, update 2011
(2) Based on Goedkoop, 2008 (based on Van Zelm 2007 and 2009)
(3) Accounting for 365 days/year

5.6.2.1 Uncertainty and low and high estimates

Given the wide range of species and substances involved, the uncertainty and variation for assessment and valuation of ecotoxicity is at least as great as that for human toxicity cancer. In addition, valuation of impacts on biodiversity is much more uncertain compared to that of human health. Consequently, the uncertainty range is at least as high as that for human-health non-cancer. This explains the bandwidth of 25 to express this relative large uncertainty and variation.

Low value = central value / √25 = central value / 5
High value = central value * √25= central value * 5

5.7 Water scarcity

5.7.1 Monetary value

Water scarcity		Value € / m³ water eqv.		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	0.067	0.022	0.200
	Flanders / Belgium	0.079	0.026	0.238
	Rest of world	0.012	0.004	0.238

Table 30: Monetary indicator for Water use

Source: VITO 2014

5.7.2 Methods and data used

Introduction

In the MMG2012 assessment method, this impact category was not monetized. Therefore, we give more details about the calculation of the monetary values in the current assessment method.

As water is a renewable source, the environmental and socio-economic impacts of the use of water within the limits of the resource availability, are very limited. The costs of water use above these limits may be very high, depending on the value of the foregone water use. The impacts may be environmental (low water impacts in rivers and drought impacts on habitats) but also socio-economic, e.g. limiting the use of water in other sectors that may add more value to a m³ of water compared to the current uses.

From a socio-economic perspective, the use of water is very different compared to e.g. the use of mineral resources or fossil energy sources. Whereas the economist may argue that scarcity of mineral and fossil resources will be reflected in prices and markets, this is not the case for the use of water because there are no real markets for water. The right to use water and the prices paid for it depend on (often complex) historical rights and specific institutional settings and agreements. In general, the price of water that users pay reflects – at maximum – the financing investments and operational costs of water treatment and water distribution systems, but not for the use of the water resource itself. As a consequence, the institutional arrangements (e.g. permits) and the price mechanisms do not reflect water scarcity. On the other hand, there is a lot of evidence that water users do not always pay the full financial costs and water use is subsidized (FAO, 2004; Anderson, 2008). This will lead to inefficient use of water, and limits the use of water savings or alternative water sources.

Water scarcity is a local, and sometimes temporary problem, which is difficult to deal with in generic terms. Furthermore, the understanding of the issue, indicators to measure water scarcity or legal frameworks to deal with it are less far developed compared to water quality (EC, 2007). As a consequence, LCIA methods have not much indicators, standards or criteria to build upon.

Step 1: fate, exposure and effect analysis

Compared to e.g. particulate matter, there is no history of complex models used to account in detail for variation in water scarcity. We use the Swiss Ecoscarcity method, as this method offers on the one hand some regional differentiation, and is well accepted (Kounina, 2013). It has however to be noted that more recently other methods have been developed, that go a step further in terms of a more detailed assessment of water use and accounting for regional differences in water scarcity (Kounina, 2013).

The Swiss Ecoscarcity method accounts for the water scarcity in the region where water is used. To this purpose, it calculates a weighting factor that is equal to: (Frischknecht, 2006)

$$\text{weighting factor} = \left(\frac{Water\ consumed\ in\ a\ region\ A\ (m^3)}{(Renewable\ water\ resouce\ in\ region\ A\ (m^3))x\ (\ rate\ of\ sustainable\ use\ (\%))} \right)^2$$

It further assumes the rate of sustainable use to be 20 %, based on OECD that states that regions with 20 % use of renewable water sources have a medium water stress (Frischknecht, 2006). This % is in line with criteria used by the European Environmental Agency (EEA, 2009)

The weighting factor varies from 0.0625 for countries or regions with a “low water pressure” (e.g. Switzerland) to 2.25 for countries with a "medium water pressure" (e.g. Spain and Italy). For countries with an "extreme high water pressure", the weighting can go up to 56. (See table 31)

Step 2: Valuation

Step 2 a: costs per m³ of water

First, we look at indicators for the marginal impact from 1 m³ of additional use of freshwater, in a region with a medium water scarcity or water stress. In that case, other users will have to look for alternative water sources and/or authorities will oblige other water users to take additional measures for water saving or alternative water sources. We use the additional costs of these alternative water sources and water saving technologies as a proxy for the resource costs of water use.

Table 32 lists a range of different options and technologies that are actually used in countries with moderate to high water stress. We estimate their costs at average 0.15 €/m³, with a range of 0.05 to 0.5 €/m³. This value can also be compared with the central estimate of 0.2 €/m³ used by Alberici (2014) to estimate the external costs of energy use in Europe.

Step 2 b: costs per ecopoint

The above mentioned costs are for a situation in countries with a medium water scarcity and thus weighting factor of 2.25. The costs per ecopoint are thus

Costs per ecopoint = costs per m³ / weighting factor = 0.067

Step 3: Values for different regions and low and high estimates

The variation in water scarcity between regions is accounted for the weighting of ecopoints. So the differences in valuation between regions only account for differences in GDP and sector.

substance	factor	unit
Water, river	0,162	m3 water eq / *m³*
Water, unspecified natural origin, AT	0,012	m3 water eq / *m³*
Water, unspecified natural origin, AU	0,0385	m3 water eq / *m³*
Water, unspecified natural origin, BE	**2,84**	m3 water eq / *m³*
Water, unspecified natural origin, CA	0,00401	m3 water eq / *m³*
Water, unspecified natural origin, CZ	0,619	m3 water eq / *m³*
Water, unspecified natural origin, DE	1,52	m3 water eq / *m³*
Water, unspecified natural origin, DK	0,736	m3 water eq / *m³*
Water, unspecified natural origin, ES	1,66	m3 water eq / *m³*
Water, unspecified natural origin, extreme water stress	**36,8**	m3 water eq / *m³*
Water, unspecified natural origin, FI	0,0082	m3 water eq / *m³*
Water, unspecified natural origin, FR	0,619	m3 water eq / *m³*
Water, unspecified natural origin, IT	0,87	m3 water eq / *m³*
Water, unspecified natural origin, low water stress	**0,0401**	m3 water eq / *m³*
Water, unspecified natural origin, LU	2,84	m3 water eq / *m³*
Water, unspecified natural origin, medium water stress	**1,47**	m3 water eq / *m³*
Water, unspecified natural origin, moderate water stress	0,368	m3 water eq / *m³*
Water, unspecified natural origin, MX	0,468	m3 water eq / *m³*
Water, unspecified natural origin, NL	0,124	m3 water eq / *m³*
Water, unspecified natural origin, OECD	0,162	m3 water eq / *m³*
Water, unspecified natural origin/kg	0,000162	m3 water eq / *kg*
Water, unspecified natural origin/m3	0,162	m3 water eq / *m³*
Water, unspecified, very high water stress	10,4	m3 water eq / *m³*
Water, well, in ground	0,162	m3 water eq / *m³*

Table 31: Examples of weighting factors for water scarcity for selected countries

Source: Frischknecht, 2006

The values for Flanders/Belgium are adapted accounting for a higher GDP compared to EU 27 (GDP Belgium = 1.19 x GDP EU 27).

The values for the world account for differences in GDP compared to EU 27. In addition, we assume that in general, water use outside the EU is likely to be dominated by agricultural sectors, and that for this sector the costs for water savings and alternative supplies are lower. Therefore we estimate the costs are 50 % lower.

The range of values in the table on costs (table 32) suggests a bandwidth of 9
Low value = central value / √9 = central value / 3
High value = central value * √9 = central value * 3

Country	Type of measure	Costs In €/m³	Source
Belgium, Flanders,	Additional costs for drinking water companies to import water from other regions or companies	0.2 (0.05 – 0.35)	Aquaflanders, 2012
Belgium, Flanders,	tax paid by users of freshwater, as a proxy for resource costs	0.04 (0.02-0.06	
Germany	Incentive paid by water companies to improve water infiltration	0.05	Greiber et al., 2009
France	Incentive paid by private company to improve water infiltration (Bionade)	0.17	VITO
Spain	Costs of water saving measures	0.12-0.21	WWF, cited in Ecologic, 2008
Spain	Costs of water transfer from other basins	0.5 0.3-0.7	
US	Reuse of waste water	0.75 -15	UNEP, 2006
Spain	Building dams to ensure water use	0.06	
France	Making water irrigation systems more efficient (from gravity to pressurized)	0.05	
Spain	Extra fee for extra water use, within a more sophisticated water irrigation system with individual quota and counters	0.06	Cited in Ecologic, 2008
Flanders, Belgium	Additional costs for industrial users that switch from groundwater to 'grey water' (treated waste water)	0.5	
Flanders, Belgium	Additional costs to produce drinking water based on waste water re-use (infiltration of treated waste water)	0.5	
Malta	Costs of producing drinking water based on desalination	1.3	
Europe	Costs of alternative supply measures in EPS valuation	0.003- 0.03	Steen, 2000

Table 32: Costs of different options for water saving and alternative water supply

5.8 Land use occupation

5.8.1 Monetary value

Land use occupation		.		
Indicator a : soil organic matter (C)		**Value € / kg C deficit**		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	2.7E-06	6.8E-07	1.1E-05
	Flanders / Belgium	2.7E-06	6.8E-07	1.1E-05
	Rest of world	2.7E-06	6.8E-07	1.1E-05
Indicators related to biodiversity				
Indicator b1 : loss of ecosystem services for land use urban and industry		Value € / m²a		
Indicator	**Region**	**Central**	**Low**	**High**
land use urban, industry, traffic	Western Europe	0.30	0.07	2.35
	Flanders / Belgium	0.59	0.15	2.35
	Rest of world	n.a.	n.a.	n.a.
Indicators b2 and b3 : loss of ecosystem services for land use agriculture and forestry		Value € / m²a		
Indicator	**Region**	**Central**	**Low**	**High**
land use agriculture	Western Europe	6.0E-03	1.5E-03	2.4E-02
	Flanders / Belgium	6.0E-03	1.5E-03	2.4E-02
	Rest of world	n.a.	n.a	n.a.
land use forestry	Western Europe	2.2E-04	5.5E-05	8.8E-04
	Flanders / Belgium	2.2E-04	5.5E-05	8.8E-04
	Rest of world	n.a.	n.a.	n.a.

Table 33: Monetary indicators for land use occupation

n.a. = not available
Source: VITO 2014

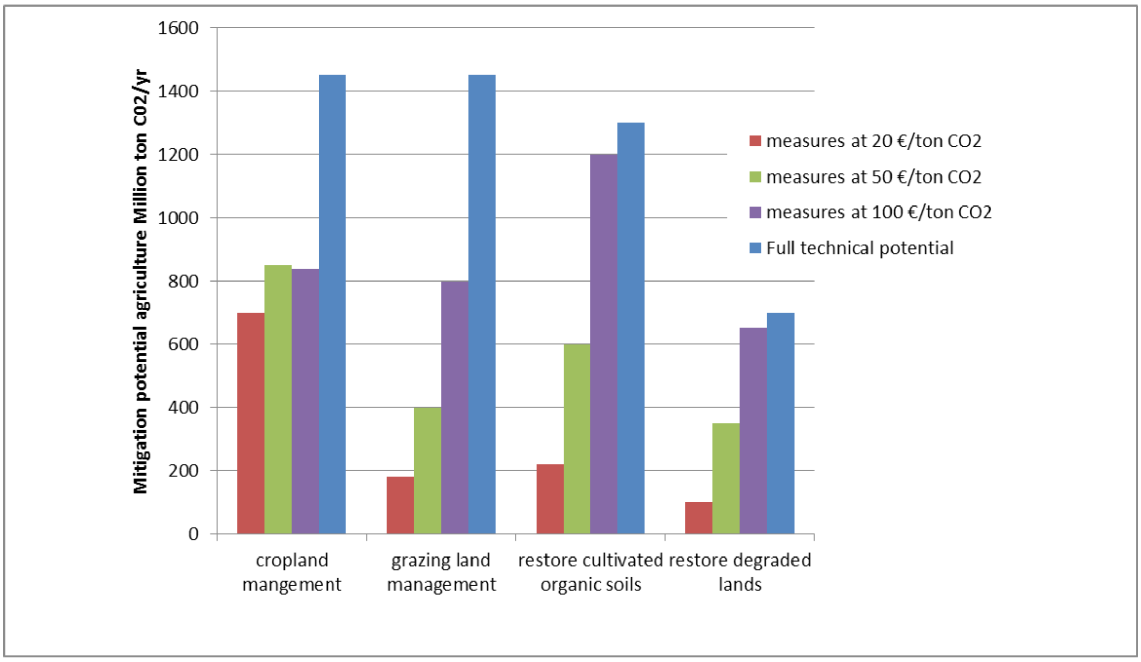
5.8.2 Methods and data used

The impacts from land occupation and land transformation are very diverse. Impact assessment of land use is still in the first stages of development, capturing the most important impacts (Mila I Canals, 2007; 2008). We distinguish four different impacts.

5.8.2.1 Soil quality and soil organic matter

The first indicator relates to the loss of soil quality. The loss of soil organic matter is a major concern in EU and the rest of the world, both from a perspective of climate change and from loss of soil quality (Smith et al, 2007; EEA, 2012; Schils, 2008, Louwagie, 2009). Loss of soil organic carbon is valued based on the prevention costs, related to measures in agricultural sector to prevent the losses, e.g. zero tillage.

It has been estimated that a carbon tax at the equivalent of 100 $ per ton CO2 equivalent would ensure that the most efficient measures will be implemented, and that for cropland and grazing land about half of the technical potential measures will be taken (figure 1). As we value global warming emissions at 100 euro/ton CO2eq, we also use this cost indicator for the valuation of loss of soil quality. As loss of soil quality is expressed in euro/kg C deficit, we have to correct for the different units (1 kg of CO2 = 0.27 kg of C).



Figuur 1: Illustration of the mitigation potential for worldwide reduction of GHG measures in agriculture

Source: based on Smith, 2007

The uncertainties related to the valuation of soil quality are inherent to the valuation of carbon as a greenhouse gas. The variation of soil quality impacts depend of measures between countries and different kinds of land uses. We therefore use a bandwidth of 16.

A bandwidth of 16 between our minimum and maximum estimate means that
Low estimate = central estimate / √16 = central estimate / 4
High estimate = central estimate * √16 = central estimate * 4

5.8.2.2 Biodiversity

Land occupation (urban and industrial)

The impact of land use changes on biodiversity is a very complex issue, and there are no direct tools or data to quantify and monetise these impacts in a systematic way that is relevant for different locations, land uses and species. We therefore follow the approach from TEEB (the economics of ecosystems and biodiversity) (TEEB, 2010). In this worldwide initiative, supported by the EC, land use changes are assessed by accounting for the quantity and value of the ecosystems goods and services from different land uses. They include production services (e.g. agricultural products or wood), regulation services (e.g. carbon sequestration, pollination, etc.) and cultural services (recreation, health, etc.) (see table 34 for the complete list). Whereas production services are mostly valued in private markets, regulation and cultural services are not. Although these services are not a direct indicator for biodiversity, more natural land uses are often associated with both more biodiversity and more regulation and cultural services (Schneiders, 2012). The concept op ecosystem services is used by the Flemish government for the report on the state of nature in Flanders (INBO, 2014)

Land use occupation for urban, industry or transport will result in a loss of unpriced ecosystem services that are associated with open, green spaces. These ecosystem services relate to regulation functions such as removal of pollutants from air by trees and grassland, reduction of noise, global climate regulation (storage of C in biomass), water quality (denitrification). Second, they relate to the cultural services such as recreation, a more pleasant living environment and health benefits. The loss of production functions (agricultural products and wood) will have been compensated when the land was purchased for industry or urban occupation. The loss of other services has not been compensated.

These services have been identified and valued for open green areas in Flanders. The average for a mixed land uses (agriculture with forests and green areas) is 5900 euro/ha.year. These values represent the situation in Flanders, with on average high levels of pollutants (e.g. air pollution) and a population density three times higher compared to the average for the EU. To estimate a value for Europe, we have corrected these estimates for Flanders, accounting for differences in population densities (see table 34).

Ecosystem service	**Flanders (€/ha) (A)**	**Europe (€/ha) (B)**
Production services		
production agricultural products	pm	pm
wood production	pm	pm
Regulation services		
Air quality: capture by plants	1.950	635*
Carbon sequestration in biomass	625	625
Carbon sequestration in soils	946	946
Noise nuisance reduction	0,2	0.1*
Flood prevention	na	na
Water supply	13	13
Nutrient removal	261	85*
Erosion prevention	na	na
Pollination	na	na
Cultural services		
Recreation and tourism	1.400	456*
Quality of living environment	135	44*
Health effects of contact with nature	554	180*
Total (€/ha.a)	5.884	2.984
Total (€/m².a)	0,59	0,30

Table 34: Ecosystem services delivered by open green areas in Flanders and Europe

na: not available, pm (pro memory, not to be included to value loss of land use)

* adapted from values from Flanders, accounting for differences in population density.

Source: based on Broekx, 2013

Expressed per ha of land use, these values are similar to the estimates in MMG2012, based on valuation of PDF. The current estimate is more reliable as it avoids building on a single study for valuation of species restoration and the assumption that all species have the same value.

The assessment of ecosystem services includes a wide range of different impacts, and uncertainties for assessment and valuation are high. In addition, for each service, the local situation (soil and habitat type, population density, etc.) may vary significantly. To express both uncertainty and variation, we use a bandwidth of 16.

A bandwidth of 16 between our minimum and maximum estimate means that:
Low estimate = central estimate / √16 = central estimate / 4
High estimate = central estimate * √16 = central estimate * 4

We do not have comparable data to make reliable estimates for impacts in the rest of the world.

Land occupation (agriculture and forestry)

For agricultural land use we use the prevention costs approach, based on the costs of the measures related to the implementation of target 2 of the EU biodiversity strategy, which is that "By 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15% of degraded ecosystems". This will require additional measures for land used by agriculture. The costs of these measures have been estimated by Tucker (2013).

From this study, the average cost, expressed per ha of agricultural land (cropland and grassland), is 60 €/ha. We use this figure for the valuation of biodiversity loss on agricultural land.

For forests, this value is estimated at 2.2 €/ha.year.

Uncertainty range:

The authors stress that these numbers are minimum estimates, and the costs per ha may vary a lot depending on country, type of land use and measures required. To express both uncertainty and variation, we use a bandwidth of 16.

A bandwidth of 16 between our minimum and maximum estimate means that
Low estimate = central estimate / √16 = central estimate / 4
High estimate = central estimate * √16 = central estimate * 4

We do not have comparable data to make reliable estimates for impacts in the rest of the world.

5.9 Land use transformation

5.9.1 Monetary value

Land use transformation		.		
Indicator a : Soil organic matter (C)		**Value € / kg C deficit**		
Indicator	**Region**	**Central**	**Low**	**high**
Full life cycle	Western Europe	2.7E-06	6.8E-07	1.1E-05
	Flanders / Belgium	2.7E-06	6.8E-07	1.1E-05
	Rest of world	2.7E-06	6.8E-07	1.1E-05
Indicators related to biodiversity				
Indicator b1 : transformation from urban land		Value € / m²a		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	n.a.	n.a.	n.a.
	Flanders / Belgium	n.a.	n.a.	n.a.
	Rest of world	n.a.	n.a.	n.a.
Indicator b2 : transformation from agricultural land		Value € / m²a		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	n.a.	n.a.	n.a.
	Flanders / Belgium	n.a.	n.a.	n.a.
	Rest of world	n.a.	n.a.	n.a.
Indicator b3 : transformation from forest, excl. rainforest		Value € / m²a		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	n.a.	n.a.	n.a.
	Flanders / Belgium	n.a.	n.a.	n.a.
	Rest of world	n.a.	n.a.	n.a.
Indicator b4 : transformation from tropical rainforest		Value € / m²a		
Indicator	**Region**	**Central**	**Low**	**High**
Full life cycle	Western Europe	n.r.	n.r.	n.r.
	Flanders / Belgium	n.r.	n.r.	n.r.
	Rest of world	27	6.9	110

Table 35: Monetary indicator for land use transformation

n.a. = not available
n.r. = not relevant
Source: VITO 2014

5.9.2 Methods and data used

5.9.2.1 Soil organic matter (C): the data are the same as for land occupation.

5.9.2.2 Biodiversity: transformation from rainforest (b4)

The transformation of tropical rainforest leads to a loss of valuable areas for biodiversity, with high biodiversity values and very long restoration times. Based on the TEEB study, this loss can be estimated at 8000 €/ha.year or 0.8 €/m².year (Braat, 2008, Chiabai, 2011). To calculate the present value of these annual losses, we have to make assumptions about the discount rate. For the central estimate, we use a discount rate of 3 %, which leads to an indicator of 27.45 €/m².

These numbers are very uncertain, and we use a bandwidth of 16.

Loss of tropical rainforest is not relevant for Europe or Flanders.

Annex 1: List of tables

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Annex 3: Bibliography

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