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Arbeitsbericht

Parameterisation of the environmental fate and exposure assessment of WATSON

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Abbreviations and Acronyms

BAU	Business As Usual
BMD_{10}	Benchmark Dose
DALY	Disability Adjusted Life Years
ED_{10h}	Effective Dose
EMEP	Co-operative programme for monitoring and evaluation of long range transmission of air pollutants in Europe
EROS Data Center	Earth Resources Observation and Science Data Center
EU	European Union
EUROSTAT	Statistical Office of the European Communities
ExternE	Externalities of Energy
HM	Heavy Metal
IPA	Impact Pathway Approach
ISO	International Organization for Standardization
K_d	Solid-water partitioning coefficient
K_{ow}	N-octanol-water partitioning coefficient
LCIA	Life Cycle Impact Assessment
LOAEL	Lowest Observed Adverse Effect Level
MOE	Margin of Exposure
MSC East	Meteorological Synthesizing Centre East
MSCE-HM	Heavy Metal model of the Meteorological Synthesizing Centre East
NEEDS	New Energy Externalities Development for Sustainability
NOAEL	No Observed Adverse Effect Level
NUTS	Nomenclature des Unités Territoriales Statistiques (Nomenclature of Territorial Units for Statistics)
red.	Reduction
US-EPA	United States – Environmental Protection Agency
WATSON	WAter and SOil environmental fate and exposure model of noxious substances at the European scale
YLD	Years of Life lived with a Disability
YOLL	Years of Life Lost

Summary

Widespread public concern exists about the hazards of heavy metals (HMs) that are released into different environmental media, i.e. air, water and soil, by industrial and agricultural activities as well as by transport. Humans are exposed to these contaminants via multiple pathways, especially through inhalation and ingestion. However, for most HMs we do not yet fully understand their behaviour in environmental media and their effects on human health and the environment at different concentration levels, especially at large scales, e.g. for the whole of Europe.

To be able to assess the impacts of HM releases for all relevant pathways in a spatially resolved way, an integrated assessment of selected HMs at the European scale has been performed by means of the multimedia modelling framework WATSON. Aside from a so-called base scenario (Business as Usual, BAU) single emission reduction scenarios have been performed for each country of concern as well as a country-weighted scenario for the member states of the European Union (EU) as of 2008. The aim of this assessment was to derive simplified relationships in both space and time from various model runs that follow the pathways of the contaminants through the full chain, accounting for the highly complex and sometimes non-linear fate and exposure processes that the contaminants go through. As result, the input for the full chain approach which represents the human activities by means of emissions released into the environment, is directly linked to the output of the full chain approach, expressed e.g. in number of human health incidences or monetary terms, i.e. external costs. The simplified relationships between input and output, hereafter referred to as parameterised values, are readily available for integration into other assessment frameworks that aim to comprise a chemical's pathway not only through air, but also through water and soil, and thus, also including the ingestion pathway into their exposure assessment on the base of various food items as well as of drinking water. Finally, the results of this parameterisation runs can be also interpreted as characterisation factors when using them within the context of life cycle impact assessment (LCIA) according to life cycle assessment ISO standards.

1 Introduction

The present documentation aims at giving an overview of the simplification process of the integrated environmental fate and exposure/impact assessment modelling framework WATSON. The modelling approach has been performed in order to derive parameterised relationships between different output parameters, such as external costs, and human activities, hereafter represented by a certain emission amount caused by different industrial and agricultural processes as well as by transport. All this has been done at the European scale and in a spatially resolved way with the result that the highly complex and sometimes non-linear fate and exposure processes that the contaminant goes through could be depicted in a very simplified way of directly relating the outcome of the different modelled scenario runs to the emission caused by human activities.

All simplified relationships between input and output of the modelling system are readily available for integration into other assessment frameworks that aim to comprise a chemical's pathway not only through air, but also through water and soil, and thus, also including the ingestion pathway into their exposure assessment on the base of various food items as well as of drinking water. The parameterised relationships are at the moment presented as linear source receptor relationships, but may be also available in the future as non-linear relationships when another series of reduction scenarios has been carried out.

The composition of this documentation follows a logical structure by first giving a brief introduction into the used modelling framework (Chapter 2) and what processes and other parts of this modelling framework have been covered by the parameterisation procedure (Chapter 3). As a second step, the various scenarios that have been defined within the frame of parametrising the fate and exposure processes of the WATSON model are introduced and, in addition, how they are connected to each other (Chapter 4). Finally, all the different parameters relevant either as input to the simplification process (Chapter 5) or as output from the various scenarios (Chapter 6) are described in detail and how they are available as well as how they are to be interpreted for further use in other integrated modelling systems. Together with the further development of the modelling system and/or the introduction of additional parameterisation runs, respectively, this documentation will be updated accordingly.

2 Modelling Framework

The multimedia modelling framework termed WATSON ('WAter and SOil environmental fate and exposure model of noxious substances at the European scale') performs environmental fate and exposure assessment including impact assessment and monetary valuation in a bottom-up analysis. It is based on the impact pathway approach (IPA) which has been developed within the series of ExternE Projects on 'External Costs of Energy' funded by the European Commission /Bickel; Friedrich 2005/.

The framework covers the media water and soil and was originally implemented by T.M. Bachmann /Bachmann 2006/ as an extension of the integrated air quality and impact assessment model EcoSense /European Commission 2003/. Unlike other fate models, WATSON is spatially explicit according to information for about 3.400 catchments, covering the whole of Europe.

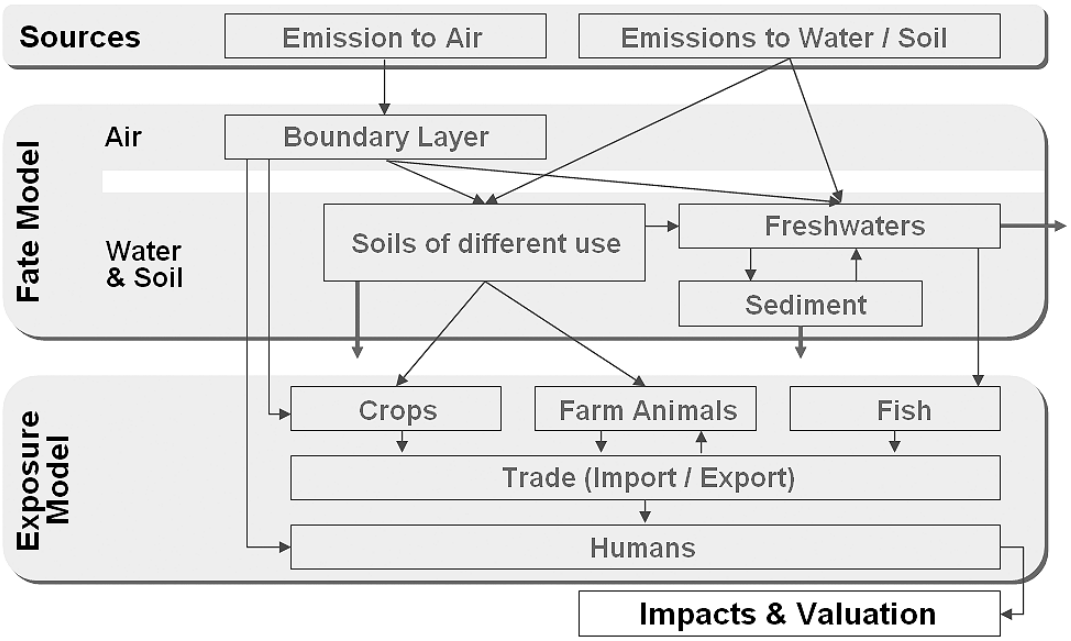

Figure 2-1: Conceptual structure of WATSON. Arrows connecting boxes denote a substance's environmental pathway, arrows not connecting boxes indicate ultimate removal processes from the model's scope.

WATSON facilitates the coverage of exposures towards hazardous substances through ingestion of various food items as well as through drinking water in a spatially resolved pan-European setting. A contaminant's environmental fate is described with the help of a climatological box model similar to Mackay level III/IV models /Mackay 1991/.

The subsequent fate model assumes long-term average conditions in order to describe the environment. The implemented exposure assessment for ingestion is complex due to both

the variety of food items to which human beings might be exposed and the spatial distribution of the food production. The estimation of ingestion-related exposures builds on the guidance documents for a site-specific risk assessment approach recommended by the US-EPA for hazardous waste combustion facilities /United States – Environmental Protection Agency 1998/. Trade is seen as an extension of the (natural) environmental fate. According to the conceptual structure shown in Figure 2-1 WATSON is at present able to calculate the exposure, impacts and external costs of HMs due to ingestion.

A more detailed description of the multimedia modelling framework WATSON can be found in /Bachmann 2006/.

3 Parameterised Parts of the Modelling Framework

The multimedia modelling framework WATSON comprises different parts of a full chain approach. As a first step the distribution, dispersion, and chemical transformation in the environmental media of concern, e.g. (fresh) water including sediment and soil, is implemented in the environmental fate model and provides concentrations in the media as output. Secondly, these concentrations serve as input for the exposure assessment model, which traces the pathway of the contaminants from the environmental media through the different exposure pathways, i.e. via drinking water and via the food chain. The exposure assessment model, thus, ends up with concentrations in humans or, if required, in further receptors. These concentrations in humans are then linked to contaminant and exposure pathway specific health end-points as part of the impact assessment part of WATSON, resulting in end-point specific number of incidences that are further aggregated by means of different severity measure, such as Disability Adjusted Life Years (DALY). Finally, the aggregated results can be valued in order to end up in monetary terms in the last step of the modelling framework, the external cost assessment part.

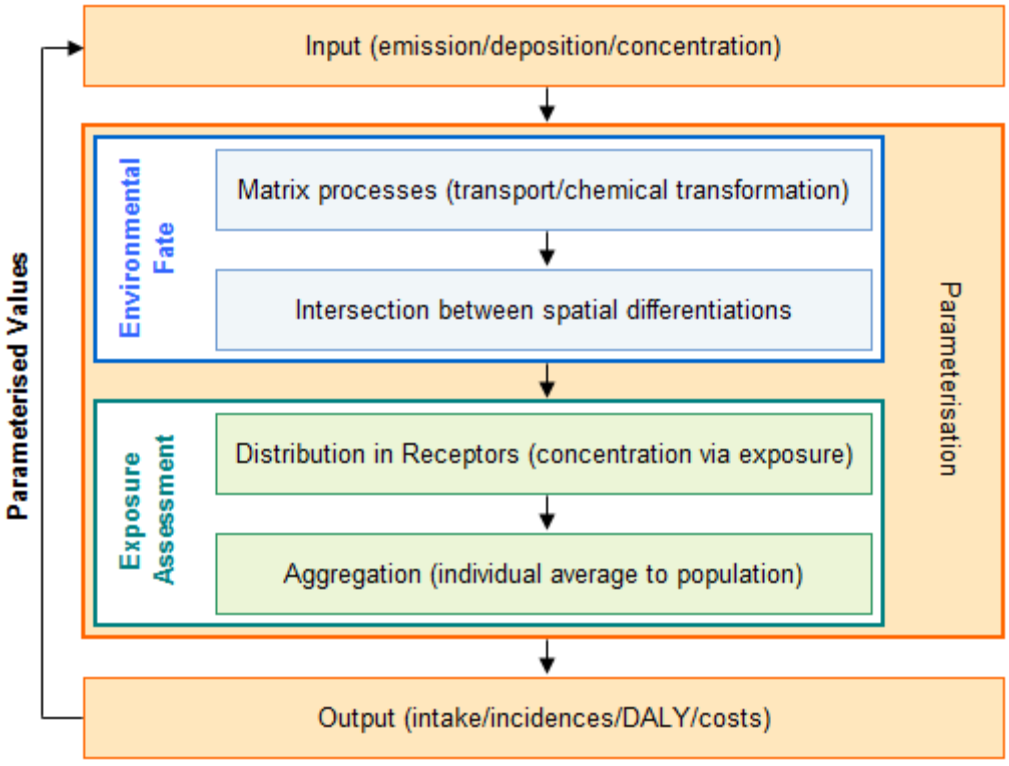

Figure 3-1: Processes and other parts of the multimedia modelling framework that have been parameterised. The linear linkages of the impact and external cost assessment are integrated in the output box of this figure, which is linked to the input in order to describe the characteristics of the parameterised values.

While the first two parts of the modelling framework, the environmental fate and the exposure assessment, are highly complex and contain loads of linear and non-linear processes, the last parts of the framework, the impact and external cost assessment, only are represented by linearly linking the outcome of the exposure assessment to dose-response relationships, then to the respective severity measure, and finally to a monetary value. Hence, the parameterisation can mainly be seen as a simplification of the environmental fate and exposure assessment of the model as described in the following Sections 3.1 to 3.3 and shown in Figure 3-1. As a last point, the different output variables are described in the context of what parts of the overall simplification they comprise (Section 3.4).

3.1 Processes of the environmental fate module

The part of the modelling framework that deals with the environmental fate of the contaminants in the terrestrial and aquatic environment is formulated like a spatially-resolved Mackay-type multimedia model based on homogeneous compartments at equilibrium and first order kinetics for the exchange between compartments and respective loss processes. As further described in detail in /Bachmann 2006/, the mass balance is based on concentrations. In matrix notation, the respective inhomogeneous system of ordinary linear first order differential equations reads as follows:

$$\vec{v} \cdot \frac{d\vec{c}}{dt} = A \times \vec{c} + \vec{p} \qquad \textbf{Equation 3-1}$$

where

A : coefficient matrix of dimension n x n [$m^3 \cdot s^{-1}$] representing the process rates between different source and receiving compartments

$\vec{v}$: volume vector of dimension n [m^3]

$\vec{c}$: concentration vector of dimension n [$kg \cdot m^{-3}$]

$\vec{p}$: perturbation vector of dimension n with exogenous inputs considering atmospheric deposition or direct releases into the considered compartments or media [$kg \cdot s^{-1}$]

t : time [s].

In this equation, the volume vector and the coefficient matrix contain information about various process rates of the processes listed in the following, which depend on nature and substance-specific properties. The perturbation vector defines the emission scenario and serves as input to the environmental fate model. The coefficient matrix can be defined in a very flexible way which allows the inclusion or exclusion of selected compartments/media as well as of selected processes. This system of linear differential equations can be solved for the steady-state situation or dynamically, referred to as level III or IV Mackay models, respectively /e.g. Mackay 1991/.

The following environmental fate processes are included in the modelling framework and have been covered by the parameterisation procedure:

- Degradation
- Radioactive decay
- Water soil erosion
- Overland flow
- Ice melt
- Matrix leaching
- Vertical substance transport in soils due to stochastic processes (preferential transport)
- Uptake by biota and removal
- Discharge
- Water circulation in large lakes
- Sedimentation (or sediment deposition) in fresh-water compartments
- Resuspension of bottom sediment matter
- Sediment burial
- Diffusion from fresh-water body to sediment
- Diffusion from sediment to fresh-water body.

All listed environmental fate processes are described in detail in /Bachmann 2006/ and can be combined in different ways as so-called process sets. Note, that emission into the atmospheric compartment is not considered as direct input, but the present modelling framework can be linked to an air quality model that calculates the fate processes of contaminants within air and then provides deposition and concentration in air as input for the WATSON fate model. Hence, processes that require the presence of an atmospheric compartment, e.g. volatilisation, are not considered either in the present framework.

3.2 Intersection between different spatial differentiations

When developing a framework that follows the pathway of a contaminant from its release into the environment through different media and receptors to finally end up e.g. in humans, different spatial differentiations are required to consider for different parts of the framework. For the interface between an atmospheric quality model and the environmental fate model of the WATSON framework the spatial differentiation typically is a (polar-stereographic) grid, e.g. on the base of the 50 km x 50 km EMEP grid developed by the Co-operative programme for monitoring and evaluation of long range transmission of air pollutants in Europe (EMEP). This grid can for example be used for integrating the output of the air quality model that leaves the boundaries of the model, e.g. dry and wet deposition as the part of the contaminant that does not remain in air but enters other compartments generally not being considered in an atmospheric model, such as different terrestrial regions.

These terrestrial regions, to which also rivers, lakes and swamps are counted, are distinguished according to the drainage basins to which they contribute. These drainage basins were taken from the HYDRO1k geographic database developed at the EROS Data Center. The HYDRO1k database consists of topographically derived datasets, including streams, drainage basins and ancillary layers based on the USGS' 30 arc-second digital elevation model of the world (GTOPO30). Within this dataset, the drainage basins are organized following the Pfaffstetter code. This code allows identifying whether and where a region is situated within a drainage basin. According to this code, each drainage basin of larger rivers is subdivided into nine sub-basins if at least four larger tributaries can be identified. These are coded with even numbers from downstream to upstream. The drainage areas between these basins (called inter-basins) assume the respective odd numbers and constitute the main stem of the subdivided river. This procedure can be repeated for each basin and inter-basin if again at least four tributaries can be identified.

While the spatial differentiation in the context of the environmental fate of contaminants can be applied as described, for assessing exposures to humans and related impacts within the present modelling framework another spatial differentiation needs to be taken into consideration with respect to the spatial distribution of different receptors, such as humans. The spatial differentiation of the exposure and impact assessment is, hence, based on administrative units mostly according to the Nomenclature of Territorial Units for Statistics (Nomenclature des Unités Territoriales Statistiques, NUTS) used by the Statistical Office of the European Communities (EUROSTAT). According to this differentiation, the information that is available in a spatially-resolved way is attributed to the different administrative levels distinguished, such as countries or municipalities.

As on the one hand the results with respect to atmospheric depositions from the air quality model, which are given on the EMEP grid, form the basis for the indirect input to the terrestrial and aquatic environment, this information needs to be transformed to match the regions distinguished for the terrestrial environment. On the other hand, as the results of the environmental fate model, information for regions distinguished for the terrestrial environment, serve as basis for the exposure and impact assessment, this information needs to be transformed to match the administrative units used within the exposure and impact assessment. Transforming such spatial information from one spatial differentiation into another requires an intersection of the respective data. This is done on an area-based weighting scheme without distinguishing between different land uses in the spatial differentiation of terrestrial regions. Hence, the intersection between grid cells and drainage basins as well as between drainage basins and administrative units can be performed in either direction according to a general formulation represented by Equation 5-2 and is thus also part

of the parameterisation procedure. This allows for setting up a coherent and continuous approach throughout the impact pathway of a contaminant of concern.

3.3 Processes of the exposure assessment module

In the exposure assessment of the modelling framework different exposure pathways are considered. Here, exposure pathway (or food chain) means any combination of (a) an environmental medium concentration as predicted by the fate model (e.g. agricultural soil concentration) based on which (b) different interrelated intermediate food concentrations are derived (e.g., wheat consumed by cow, milk) finally being (c) taken in by humans. This brings about that human exposure for example to milk is composed of the exposure pathways based on the ingestion of soil particles, forage, silage and grains by milk cattle. Each of the linkages or steps of an exposure pathway is termed an exposure transfer, i.e. a transfer from one medium, substrate, or receptor to another.

The environmental fate model results in bulk concentrations that are given in weight of a substance per volume of a medium (kg per m^3). However, many of the equations in the exposure assessment of terrestrial food chains are based on concentrations that are given in weight of a substance per (dry or fresh) weight of the medium (kg per $kg_{fresh\ weight}$ or kg per $kg_{dry\ weight}$) according to /United States – Environmental Protection Agency 1998/. The food chains in the aquatic environment, in contrast, are based on the dissolved fraction of the substance. Consequently, different unit conversions need to be performed in order to arrive at a specific exposure as described in detail in /Bachmann 2006/.

The following exposure pathways are included in the modelling framework and have been covered by the parameterisation procedure:

- Inhalation exposure pathway ‘air – humans’
- Food exposure pathway ‘atmospheric deposition – aboveground exposed produce – humans’, e.g. spinach
- Food exposure pathway ‘atmospheric deposition – forage/silage – cattle – humans’
- Food exposure pathway ‘arable land – aboveground protected produce – humans’, e.g. cereals
- Food exposure pathway ‘arable land – aboveground exposed produce – humans’
- Food exposure pathway ‘arable land – belowground produce – humans’, e.g. potatoes
- Food exposure pathway ‘pasture/arable land – feed – milk cattle – humans’
- Food exposure pathway ‘pasture/arable land – feed – beef and veal cattle – humans’
- Food exposure pathway ‘pasture (soil particles) – animal products – humans’; ‘animal products’ could mean cattle milk, beef and veal, poultry meat, eggs from laying hens as well as pork

- Food exposure pathway ‘freshwater – fish – humans’
- Drinking water exposure pathway ‘dissolved substances in groundwater – humans’
- Drinking water exposure pathway ‘dissolved substances in surface water – humans’
- Drinking water exposure pathway ‘particle-bound substances in surface water – humans’

All listed exposure assessment pathways except the ingestion of drinking water are described in detail in /Bachmann 2006/.

3.4 Accounting for different output variables

When dealing with parametrising a modelling framework as described in Sections 3.1 to 3.3, it is as a first step required that the variables are defined that are to be linked in a way that exactly one input variable is linked to exactly one output variable. However, when taking different problems of using and applying such input-output-linkages into account, it is necessary to strictly distinguish between all complex and possibly non-linear processes and linear relationships that are all covered by the parameterisation procedure.

As an example, when at the very end of the parameterisation procedure a parameter, such as a linear dose-response relationship, is introduced, then it might be helpful to not only provide the results that already comprises the dose-response relationship, but also to provide an intermediate result that allows the user to apply his own linear dose-response relationship. Another example is related to the monetary valuation of human health impacts when one wants to apply different discounting schemes for different purposes or for comparative studies. When only providing parameterised values that already include a specific discounting scheme, then it is hard if at all possible to apply another discounting scheme without knowing the input-output relationships without the undiscounted values.

However, while with respect to the first problem in this parameterisation runs different output variables are taken into consideration for providing simplified results at different levels of the full chain, it was so far not possible to provide undiscounted parameterised values that at the same time can be used as basis for applying different discount schemes than these which have been already implemented. This is due to the fact that applying a discount scheme to the output of the parameterisation runs can not be seen as a linear relationship or process.

Generally, the first output variable of the parameterisation procedure that links a selected input (e.g. anthropogenic emissions) to an output of the modelling framework is the effective intake fraction as described in Section 6.1. As this variable can be interpreted as the relationship between the amount of a substance taken in by a receptor and the amount of the

same substance released into the environment, it is the base for all further output variables of the parameterised full chain approach. Each further output variable, such as the number of human health end-point specific incidences, only covers an additional linear step of the impact pathway, which thus easily can be replaced by the user in order to apply his own values of such a parameter to the output variable that does not yet comprise this step. Thus, providing a predefined set of output variables allows for using the results of the parameterisation runs in a wide context, starting at effective intake fraction and following the impact pathway step by step by covering one additional linear relationship between two output variables, finally ending up at external costs.

4 Parameterisation runs: scenarios

The overall objective behind the simplification of the complex processes and pathways related to the environmental fate and exposure behaviour of HMs after being emitted into the environmental media air, water, and soil, is to relate a specific amount of each emitted trace element at a specific site within the geographical scope of Europe to the amount of impacts and thus damages that it causes either at the same and/or other sites, respectively. However, this methodology must not only be seen as evaluation purpose of state of the art or business as usual situations, but also as a possibility to predict future situations, when also accounting for sustainability concerns, which is highly relevant in the frame of decision support for current legislation. In order to make a point in the context of international regulations, the simplification has been adopted at the national level, i.e. the question 'How much will it cost any country in terms of human health impacts to emit one unit of a selected HM into one specific country?' will be answered in as appropriate a way as possible.

Altogether 43 different scenarios have been defined within the process of simplifying the fate and exposure model. As a first step, an emission situation for the selected HMs arsenic, cadmium, chromium, nickel, and lead has been set up for the year 2010, which is hereafter referred to as business as usual scenario ('BAU 2010 base'). Note that when performing (human) impact assessment, it is of importance whether the chemical forms in drinking water or in the edible portions of the food items are available to humans and that these available forms have the potential to cause an adverse effect on a receptor. Examples are inorganic versus organic arsenic /Agency for Toxic Substances and Disease Registry 2000b/, mercury compounds /Boening 2000/, and chromium VI versus chromium III /Agency for Toxic Substances and Disease Registry 2000a/. However, as no effect information related to ingestion of different chemical forms of chromium has been available for the present modelling approach, chromium has been modelled as sum of all chemical forms, i.e. chromium III and chromium VI.

Based on the 'BAU 2010 base' situation the anthropogenic emissions into the atmospheric compartment have been reduced individually by a certain amount in all countries listed in Table 4-1, e.g. one ton reduction of each trace element in Germany, while the emissions in all other countries remain constant, i.e. the same as in the 'BAU 2010 base' scenario. In most of the countries, exactly one ton of emission reduction has been determined as far as this has made sense in relation to the overall emission in the country. This means, wherever the overall amount of a country's emission has been estimated as too low for a one ton reduction, only a reduction of 10 kg has been defined (see Table 4-1). In order to be able to evaluate the range of the results of all country-specific reduction scenarios, an additional evaluation scenario has been conducted (hereafter referred to as 'EU27+2' scenario) where a

certain amount of each trace element, i.e. around one ton, has been reduced in a country-weighted way, comprising 29 countries (all member states of the European Union as of 2008 as well as Norway and Switzerland) according to Equation 4-1:

$$E_{air}(p,b,s) = \frac{\sum E_{air}(p,b,s) \cdot A(b)}{\sum A(b)} \qquad \textbf{Equation 4-1}$$

where

E_{air} : emission reduction of substance p in the atmospheric compartment of country b due to direct releases of source category s [$t_{released} \cdot yr^{-1}$]

ΣE_{air} : total emission reduction of substance p in the atmospheric compartment of as sum of the emission reduction in all countries b due to direct releases of source category s [$t_{released} \cdot yr^{-1}$]

A : area of country b [m^2]

ΣA : total considered geographical scope as sum of the area of all countries b [m^2].

Table 4-1: Emission scenarios considered for the parameterisation of WATSON. The base year for all considered scenarios is 2010.

Scenario	Reduction	Remarks	Scenario	Reduction	Remarks
BAU 2010 base	-	base scenario (emission defined to as 100% as reference for all other scenarios)	IE	10 kg	-
			IS	-	no deposition input due to missing matrices from MSC East
EU27+2	1 ton	evaluation scenario (country area weighted reduction in EU27+2 countries; see text for further explanation)	IT	1 ton	-
			LT	10 kg	-
			LU	10 kg	-
			LV	10 kg	-
AL	10 kg	-	MC	-	emission too low to be reduced
AT	1 ton	-			
BA	10 kg	-	MD	10 kg	-
BE	1 ton	-	MK	10 kg	-
BG	1 ton	-	MT	-	no deposition input due to missing matrices from MSC East
BY	10 kg	-			
CH	1 ton	-	NL	10 kg	-
CY	-	no deposition input due to missing matrices from MSC East	NO	1 ton	-
			PL	1 ton	-
CZ	1 ton	-	PT	1 ton	-
DE	1 ton	-	RO	1 ton	-
DK	1 ton	-	RU	1 ton	-
EE	10 kg	-	SE	10 kg	-
ES	1 ton	-	SI	10 kg	-

Scenario	Reduction	Remarks	Scenario	Reduction	Remarks
FI	10 kg	-	SK	1 ton	-
FR	1 ton	-	TR	1 ton	-
GR	1 ton	-	UA	1 ton	-
HR	10 kg	-	UK	1 ton	-
HU	1 ton	-	YU	1 ton	-

The evaluation scenario ‘EU27+2’ has been conducted for comparative reasons as the results of the different country-specific scenarios show a variance for each output parameter; the variance between all country-specific scenarios is shown exemplarily for the effective intake fraction in Figure 4-1.

However, the ‘EU27+2’ scenario also has been defined to be able to derive a general European average value for all relevant output parameters, such as the effective intake fraction. This is due to the fact that for some questions, it would rather be sufficient to only get a representative European average value than values for one or more selected countries. Therefore, the country area weighted results of the evaluation scenario have been compared to the arithmetic mean of the results of all country-specific scenarios and it was found that the values of the ‘EU27+2’ situation can be seen as the representative average of the country-specific values (see Figure 4-1).

The arithmetic mean of the resulting output variables of all country-specific scenarios has been calculated exemplarily for the effective intake fraction via ingestion of food according to Equation 4-2:

$$\overline{iF}_{food}(p,i,b) = \frac{1}{b_{total}} \cdot \sum_{b=1}^{b_{total}} iF_{b,\,food}(p,i,b) \qquad \textbf{Equation 4-2}$$

where

$\overline{iF}_{food}$: arithmetic mean of the effective intake fraction of substance *p* via the food ingestion pathway for a certain population *i* in all countries *b* of the selected geographical scope [kg_{intake} per $kg_{released}$]

b_{total} : total number of countries *b* contributing to the overall effective intake fraction in the selected geographical scope [-]

$iF_{b,\,food}$: effective intake fraction of substance *p* via the food ingestion pathway for a certain population *i* in country *b* [kg_{intake} per $kg_{released}$].

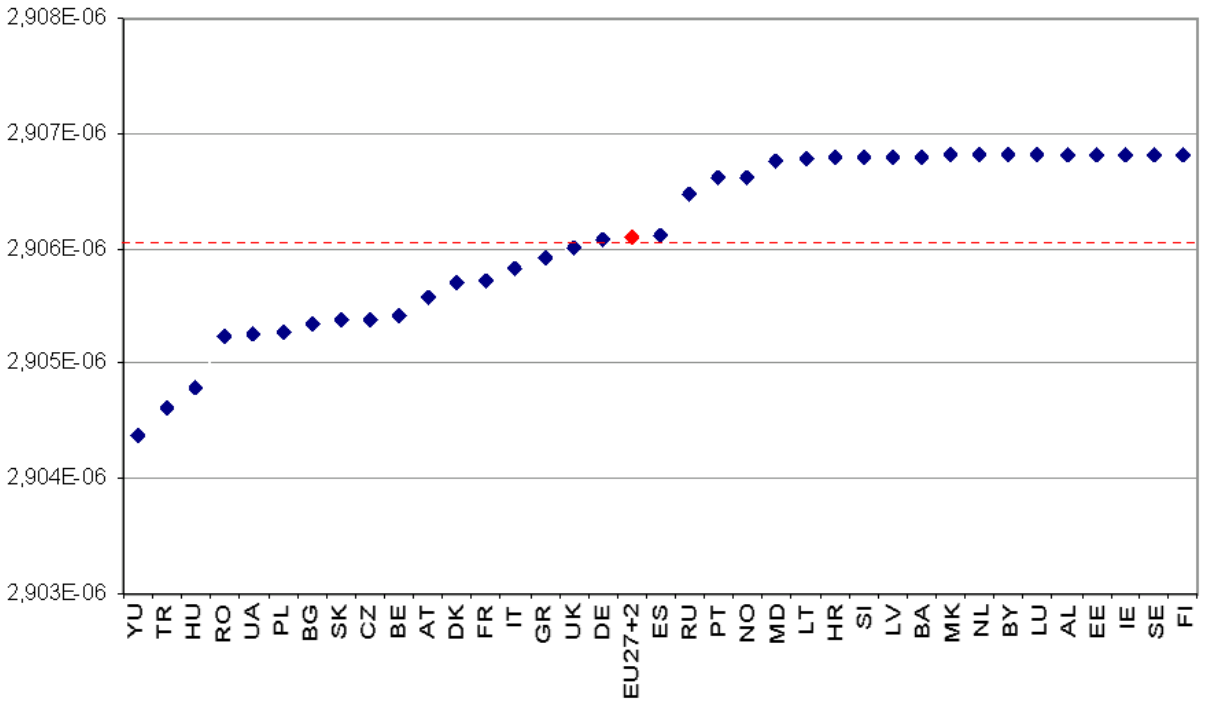

Figure 4-1: Comparison of the range of effective intake fractions [kg_{intake} per $kg_{released}$] between the country-specific scenarios and the country area weighted effective intake fraction of the evaluation scenario 'EU27+2'. The red dotted line denotes the calculated arithmetic mean of the country-specific effective intake fractions.

Note, that while for the country-specific model runs as well as for the evaluation scenario different reduction amounts (either one ton or 10 kg) have been implemented for different reasons, all the output variables to be found in Chapter 6 are based on a normalised emission of one kg, i.e. the output values have been divided by the total emission amount [kg] of each scenario to end up in a comparative base that is set to one kg of emitted substance per scenario. That normalised emission base can be seen either as reduction or as additional release. This means, when proceeding from an emission reduction as base for deriving an output, then the results are to be interpreted as reduced costs, while they have to be interpreted as additional cost when proceeding from an additional emission release. Exceptions are the variables 'effective intake fraction' and 'accumulated intake' as these parameters are based on the overall emission amount of a scenario (see Sections 6.1 and 6.2).

5 Parameterisation runs: input

All scenario runs performed in the present parameterisation study are based on the general parameter set which is used in the water and soil model WATSON. For a detailed insight of the modelling framework, its set of variables, processes and formulation please refer to /Bachmann 2006/. For the context of parametrising, out of a defined set of processes within the modelling framework, only the most important input parameters will be described in the following, thereby distinguishing between the two main modules, the environmental fate module on the one hand and the exposure/impact assessment module on the other hand.

The following set of general input data has been identified as important for the subsequent environmental fate module:

- Wet and/or dry atmospheric deposition (anthropogenic and/or natural), if relevant
- Concentrations in air, if relevant
- Direct and indirect releases into soil and/or (fresh) water, if relevant
- Anthropogenic emissions into air; note that air emissions only are used as basis for calculating the intake fraction and can be defined either as a one year pulse emission or as a continuous emission over several years

The following set of general input data has been identified as important for the subsequent exposure/impact assessment module:

- Concentration in all compartments distinguished (provided by the fate module)
- Receptor information (e.g. food production, consumption data for different food items and drinking water), generally provided by the accompanying database
- Monetary values for all processed end-points

As a third aspect also substance-specific data have to be taken into consideration, if new substances shall be modelled that are not contained in the database as yet:

- Concerning environmental fate: solid-water partitioning coefficient (K_d) or n-octanol-water partitioning coefficient (K_{ow}), Henry's law constant, etc.
- Concerning exposure assessment: bio-accumulation factors for all processed food items, bio-concentration factors
- Concerning effect assessment: exposure/intake-response information (dose-response functions).

In the following only two main aspects of input parameters will be discussed in detail, namely the anthropogenic emissions into the atmospheric compartment and the effect information related to the ingestion of different food items as well as of drinking water. This

is due to the fact that on the one hand only the anthropogenic emissions into air have been selected to be reduced in the different scenarios and on the other hand, the effect data, generally derived from epidemiological studies, significantly influence the output of the modelling framework.

While the water and soil model directly starts its environmental fate calculations on the base of atmospheric depositions (see above), the amount emitted into the atmospheric compartment is required as basis for the calculation of the intake fractions that are described in more detail in Section 6.1. Hence, the emissions into air are relevant for the parameterisation process as link between the model's different output parameters based on the intake fraction and the human activities, i.e. the anthropogenic emissions and, thus, will be described in the following.

5.1 Releases into the environment

For many substance categories, anthropogenic emissions, i.e. direct or indirect releases of a chemical into the atmosphere by human activities, e.g. industrial processes and transport, are commonly estimated on the base of emission factors according to Equation 5-1:

$$E_{air}(p,z,s) = \sum A(p,z,s) \cdot emi_{air}(p,z,s) \qquad \textbf{Equation 5-1}$$

where

E_{air} : emission of substance *p* into the atmospheric compartment of grid cell or zone *z* due to direct releases of source category *s* [$t_{released} \cdot yr^{-1}$]

A : activity rate that is associated with the release of substance *p* into grid cell or zone *z* by source category *s* [e.g. $t_{produced\ product} \cdot yr^{-1}$]

emi_{air} : emission factor of substance *p* for the atmospheric compartment of grid cell or zone *z* from source category *s* [e.g. $t_{released} \cdot yr^{-1}$ per $t_{produced\ product} \cdot yr^{-1}$].

This common methodological approach combines information about the quantity of a sector- or source category-specific human activity taking place with coefficients that quantify the emissions per a particular unit of this human activity. Those coefficients, independently of dealing with (stationary) point, mobile or area sources, are referred to as emission factors and are available e.g. for greenhouse gases /Intergovernmental Panel on Climate Change 2006/, 'classical' air pollutants United States – Environmental Protection Agency 1995/ / United States – Environmental Protection Agency 2006/ and other hazardous compounds, such as HMs /European Environment Agency 2007/. In many cases, these emission factors are averages of all available activity data of acceptable quality, and are generally assumed to be representative for all facilities in the concerned source category as long as only atmosphere is taken into consideration as receiving environmental compartment.

The anthropogenic emissions for the selected HMs have been provided by means of the emission model MSCE-HM /Meteorological Synthesizing Centre – East 2005/ for the BAU 2010 scenario situation at a spatial resolution of 50 km x 50 km polar-stereographic grid cells /Klein 2003/. All grid cells have been considered to at least partly be covered by a respective country by following the intersection procedure as described in Equation 5-2:

$$c_g(g,k) = \sum_{\{b/b\cap g\neq 0\}} \left(\frac{A_{b\cap g}(g,b)}{\sum A_g(g,b,k)} \cdot c_b(g,b,k) \right) \qquad \textbf{Equation 5-2}$$

where

c_g : concentration [$\mathrm{kg \cdot m^{-3}}$] in compartment *k* of each grid cell *g*

$A_{b\cap g}$: fraction of the area of a grid cell *g* [$\mathrm{m^2}$] that it shares with the country *b* in which it is located

ΣA_g : area of a grid cell *g* [$\mathrm{m^2}$] as sum of all areas of which the grid cell shares different adjacent countries *b*; note that only those countries are considered here in which the respective compartment *k* is present, thus, $\sum A_g \leq A_{g,\,total}$ where $A_{g,\,total}$ is the total area of the grid cell *g*

c_b : concentration [$\mathrm{kg \cdot m^{-3}}$] in compartment *k* of each country *b*

$b/b\cap g\neq 0$: index of all countries *b* which share the same grid cell *g*.

As an example, the direct anthropogenic emissions of arsenic into the atmospheric compartment are shown in Table 5-1. Having in mind, that the different scenarios are listed in columns while the countries that receive the different emissions for each scenario are listed in rows (this refers to all subsequent tables that are extracted from either input or output tables of the WATSON model), the values in this table can thus be interpreted as follows:

- The total amount of emissions of arsenic in the scenario ‘1t reduction in Belgium’ is spatially distributed in a way that the emissions in all countries except Belgium equal the emissions in the ‘BAU 2010 base’ scenario whereas in Belgium the emissions have been reduced by exactly one ton compared to the ‘BAU 2010 base’ scenario (see marked cells in Table 5-1). Note that the ‘1t red.’ for this reduction scenario only refers to the defined reduction for the respective country, i.e. one ton of reduction only in Belgium compared to ‘BAU 2010 base’.

Table 5-1: Anthropogenic emissions of arsenic into air [kg per year] per scenario, steady state. For further details, see text.

Emission [kg/yr]	BAU2010	EU27+2	AL	AT	BA	BE	BG	CH
	-	1t red.	10kg red.	1t red.	10kg red.	1t red.	1t red.	1t red.
AL	2.455E+02	2.455E+02	2.355E+02	2.455E+02	2.455E+02	2.455E+02	2.455E+02	2.45E+02
AT	5.891E+03	5.870E+03	5.891E+03	4.891E+03	5.891E+03	5.891E+03	5.891E+03	5.89E+03
BA	1.624E+03	1.624E+03	1.624E+03	1.624E+03	1.614E+03	1.624E+03	1.624E+03	1.62E+03
BE	1.048E+04	1.045E+04	1.048E+04	1.048E+04	1.048E+04	9.482E+03	1.048E+04	1.05E+04
BG	4.403E+03	4.388E+03	4.403E+03	4.403E+03	4.403E+03	4.403E+03	3.403E+03	4.40E+03
BY	3.262E+03	3.262E+03	3.262E+03	3.262E+03	3.262E+03	3.262E+03	3.262E+03	3.26E+03
CH	2.648E+03	2.639E+03	2.648E+03	2.648E+03	2.648E+03	2.648E+03	2.648E+03	1.65E+03
CY	5.332E+02	5.313E+02	5.332E+02	5.332E+02	5.332E+02	5.332E+02	5.332E+02	5.33E+02
CZ	8.921E+03	8.890E+03	8.921E+03	8.921E+03	8.921E+03	8.921E+03	8.921E+03	8.92E+03
DE	3.871E+04	3.858E+04	3.871E+04	3.871E+04	3.871E+04	3.871E+04	3.871E+04	3.87E+04
DK	1.756E+03	1.750E+03	1.756E+03	1.756E+03	1.756E+03	1.756E+03	1.756E+03	1.76E+03
EE	4.824E+03	4.807E+03	4.824E+03	4.824E+03	4.824E+03	4.824E+03	4.824E+03	4.82E+03
ES	3.701E+04	3.688E+04	3.701E+04	3.701E+04	3.701E+04	3.701E+04	3.701E+04	3.70E+04
FI	5.121E+03	5.103E+03	5.121E+03	5.121E+03	5.121E+03	5.121E+03	5.121E+03	5.12E+03
FR	2.678E+04	2.669E+04	2.678E+04	2.678E+04	2.678E+04	2.678E+04	2.678E+04	2.68E+04
GR	6.954E+03	6.930E+03	6.954E+03	6.954E+03	6.954E+03	6.954E+03	6.954E+03	6.95E+03
HR	1.921E+03	1.921E+03	1.921E+03	1.921E+03	1.921E+03	1.921E+03	1.921E+03	1.92E+03
HU	5.545E+03	5.525E+03	5.545E+03	5.545E+03	5.545E+03	5.545E+03	5.545E+03	5.54E+03

5.2 Human health end-points

In order to assess the physical impacts/health effects from exposures to substances, it is generally preferable to use dose- or exposure-response relationships in order to estimate effects that can be derived based on observations on human populations. Combining these effects with an appropriate measure of severity then yields impacts. As experiments at least with human beings are not ethically defendable, the best information available is provided by epidemiological studies. Epidemiologically derived exposure-response functions are widely used in the context of human health assessments for policy decision purposes /e.g., European Commission 1999a/ and /Friedrich; Bickel 2001a/. For more information, /Bachmann 2006/ discusses the availability and feasibility of information for exposure-response functions in detail.

In the following, all exposure/dose-response relationships used within the parameterisation runs are listed, which are either applicable for ingestion of different food items (Table 5-2) or for ingestion of drinking water (Table 5-3).

Table 5-2: Physical end-points of human health effects due to ingestion of food used within the parameterisation procedure. Note, that only for the HMs arsenic, cadmium and lead appropriate end-points were available; however, for chromium and nickel at least exposures could be calculated.

Pollu-tant	End-Point		Risk Group	Expos. Time	Unit Risk	Severits Measure			Costs per Case
-	Name	Fatality	-	[years]	[risk/kg_{intake}]	YOLL	YLD	IQ Points	[€/case]
As	skin cancer	85%	all	70	7.83E-01	5.18	0.16	0	213,520
As	bladder cancer	85%	all	70	3.91E+00	3.94	0.31	0	169,660
As	cardiovascular mortality	100%	all	35	2.35E+02	12.80	0.00	0	512,000
As	still birth (babies)	100%	all	1	2.86E+02	0.00	1.28	0	51,200
Cd	osteoporosis	100%	all	35	6.26E+02	0.00	0.28	0	11,200
Cd	renal dysfunction	100%	all	35	3.13E+01	0.00	0.64	0	25,600
Pb	anaemia	100%	all	1	1.32E+02	0.00	0.64	0	25,600
Pb	IQ points loss (children)	100%	0-1 years	1	1.15E+03	0.00	0.00	1	8,600

Table 5-3: Physical end-points of human health effects due to ingestion of drinking water used within the parameterisation procedure. Note, that only for the HMs arsenic, cadmium and lead appropriate end-points were available; however, for chromium and nickel at least exposures could be calculated.

Pollu-tant	End-Point		Risk Group	Expos. Time	Unit Risk	Severits Measure			Costs per Case
-	Name	Fatality	-	[years]	[risk/kg_{intake}]	YOLL	YLD	IQ Points	[€/case]
As	skin cancer	85%	all	70	7.83E-01	5.18	0.16	0	213,520
As	bladder cancer	85%	all	70	3.91E+00	3.94	0.31	0	169,660
As	cardiovascular mortality	100%	all	35	2.35E+02	12.80	0.00	0	512,000
As	still birth (babies)	100%	all	1	2.86E+02	0.00	1.28	0	51,200
Cd	osteoporosis	100%	all	35	6.26E+02	0.00	0.28	0	11,200
Cd	renal dysfunction	100%	all	35	3.13E+01	0.00	0.64	0	25,600
Pb	anaemia	100%	all	1	2.74E+02	0.00	0.64	0	25,600
Pb	IQ points loss (children)	100%	0-1 years	1	1.15E+03	0.00	0.00	1	8,600

For each particular human health end-point the following information are available and required for the impact assessment as performed in the present study:

- Exposure route: either ingestion of food stuff or ingestion of drinking water; note that within the used set of exposure-response information the risk value related to food stuff and to drinking water only differ for one end-point.
- Fatality factor: fraction of a given individual life-time risk which ultimately leads to death among the affected population when related to cancer; note that when such an information has not been available, a fatality of 100% has been assumed, which is not related to death but to the end-point itself.
- Risk group: the fraction of a population which in fact is affected by a particular end-point (e.g. children, babies, pregnant women, etc.); note that when such an information has not been available, it has been assumed that the whole population would have been affected.
- Exposure time or exposure duration: the time in years to which a certain population has been exposed to the substance of concern; note that wherever such an information has not been available and at the same time the unit of the individual risk has been denoted as life-time risk, an exposure time of 70 years has been assumed for the population of concern; however, this is not assumed to be transferable to the risk for individuals.
- Severity measures ‘Years of Life Lost’, ‘Years of Life lived with a Disability’ and ‘IQ Points loss’ per case of a particular end-point are given as base for calculating the severity-specific damage costs per case; the severity-specific costs are shown in Table 5-4. The severity measure ‘Disability adjusted life years’ has not been listed as it is calculated as the sum of ‘Years of Life Lost’ and ‘Years of Life lived with a Disability’ (see Equation 6-6 for more details) and thus refers to the sum of the costs of ‘Years of Life Lost’ and ‘Years of Life lived with a Disability’ per case.

In order to finally arrive at external costs a monetary value is applied to each severity measure used in the parameterisation procedure as shown in Table 5-4.

The monetary values for the aggregated severity measure Disability Adjusted Life Year (DALY) are assumed to equal the monetary values of Years of Life Lost (YOLL) as well as of Years of Life lived with a Disability (YLD) throughout the parameterisation scenario runs. For further information about the relationship between DALY, YOLL and YLD please refer to Section 6.4.

Table 5-4: Monetary values of different severity measures, i.e. Years of Life Lost, Years of Life lived with a Disability and IQ Points loss.

Severity Measure	Costs per Unit of Severity Measure	[unit]
Years of Life Lost (YOLL)	40,000[a]	[$Euro_{2000}$ per YOLL]
Years of Life lived with a Disability (YLD)	40,000[a]	[$Euro_{2000}$ per YLD]
IQ Points loss	8,600[a]	[$Euro_{2000}$ per IQ Point]

[a] Values only valid for Europe as decided in the frame of the NEEDS international project as of 2007. Note that e.g. one lost IQ Point in the United States costs 8,000 $Euro_{2000}$ according to /GREENSENSE 2004/.

6 Parameterisation runs: output

Among the various output variables of both the environmental fate module and the exposure/impact assessment module of the applied modelling framework only the most important are described in detail in the subsequent sections as only those are relevant for the linkage between the anthropogenic emissions into the environment and the impacts on human health. Starting from the direct connection between the emission amount released into air and the human intake via the ingestion pathway, which will be represented by the effective intake fraction (Section 6.1), different intermediate output variables will be described, such as accumulated intake (Section 6.2), human health end-point specific incidences (Section 6.3) and end-point specific severity measures (Section 6.4), before finally arriving at end-point specific external costs in Section 6.5.

6.1 Effective intake fraction

The overall exposure of a population is assessed by means of the population-based source-to-intake measure 'intake fraction' /Bennett et al. 2002/, sometimes also referred to as exposure efficiency /Evans et al. 2002/. It is the fraction of a substance's mass released into the environment that is ultimately taken in by the human population as a result of food and/or drinking water consumption, inhalation and/or dermal exposure. In case of food ingestion, this implies that it aggregates the exposure towards different produces, which may become contaminated due to different causes. Each such cause-exposure chain starting at the result of the environmental fate model is hereafter termed exposure pathway. For the purpose of the parameterisation runs, the intake fraction due to ingestion exposure is calculated as:

$$iF(p,e,i,b) = \frac{\sum IR_{personal}(p,e,r,b) \cdot n_{population}(b)}{S(p,k,b_{total})} \qquad \textbf{Equation 6-1}$$

where

iF : effective intake fraction of substance p via a specific exposure pathway e for a certain population i in country b [kg_{intake} per $kg_{released}$]

$IR_{personal}$: effective personal intake rate of substance p via a specific exposure pathway e related to produce/food item r in country b [$kg_{intake} \cdot capita^{-1} \cdot s^{-1}$]

$n_{population}$: population in country b [capita]

S : source strength of a substance p into compartment k of all countries b_{total} [$kg_{released} \cdot s^{-1}$].

The concept of the intake fraction here only covers that portion of the emission of a substance to which exposure occurs and which then may lead to an adverse effect. This portion is referred to as effective intake fraction. The effective intake fraction, thus, can be interpreted as the fraction of a substance released into the environment (emission into a source compartment) that leads to the effects as described in Section 5.2 due to ingestion by a

population of concern through a given exposure pathways, e.g. ingestion of drinking water, meat, milk, fish, etc. (intake by humans). According to these definitions, the accumulated effective intake fraction for arsenic is exemplarily shown in Table 6-1 and can be interpreted as follows:

- For 'kg' as base unit (note that due to the fact that this parameter is actually a fraction, any mass unit can be taken as base, such as ton or mg): The total amount of emissions of arsenic in the scenario '1t reduction in Belgium' will lead to an accumulated effective intake fraction of 0.347 mg_{intake} per $kg_{released}$ in Germany due to the ingestion of different food items under steady state conditions (see marked cell in Table 6-1). Note that the '1t red.' for this reduction scenario only refers to the defined reduction for the respective country, i.e. one ton of reduction only in Belgium compared to 'BAU 2010 base'.

Table 6-1: Accumulated effective intake fraction [kg_{intake} per $kg_{released}$] due to food ingestion of arsenic per scenario, steady state. For further details, see text.

Intake Fraction [kg_{in}/kg_{emi}]	BAU2010 -	EU27+2 1t red.	AL 10kg red.	AT 1t red.	BA 10kg red.	BE 1t red.	BG 1t red.	BY 10kg red.
AL	8.50E-09	8.48E-09	8.48E-09	8.48E-09	8.48E-09	8.47E-09	8.47E-09	8.48E-09
AT	4.15E-08	4.14E-08	4.14E-08	4.14E-08	4.14E-08	4.14E-08	4.14E-08	4.14E-08
BA	7.09E-09	7.08E-09	7.08E-09	7.07E-09	7.08E-09	7.07E-09	7.07E-09	7.08E-09
BE	4.22E-08	4.21E-08	4.21E-08	4.21E-08	4.21E-08	4.21E-08	4.21E-08	4.21E-08
BG	1.48E-08	1.48E-08	1.48E-08	1.48E-08	1.48E-08	1.48E-08	1.48E-08	1.48E-08
BY	3.90E-08	3.89E-08	3.89E-08	3.89E-08	3.89E-08	3.89E-08	3.89E-08	3.89E-08
CH	2.65E-08	2.64E-08	2.64E-08	2.64E-08	2.64E-08	2.64E-08	2.64E-08	2.64E-08
CY	3.63E-09	3.62E-09	3.62E-09	3.62E-09	3.62E-09	3.62E-09	3.62E-09	3.62E-09
CZ	4.66E-08	4.65E-08	4.65E-08	4.65E-08	4.65E-08	4.65E-08	4.65E-08	4.65E-08
DE	3.48E-07	3.47E-07	3.47E-07	3.47E-07	3.47E-07	3.47E-07	3.47E-07	3.47E-07
DK	2.68E-08	2.67E-08	2.67E-08	2.67E-08	2.67E-08	2.67E-08	2.67E-08	2.67E-08
EE	6.16E-09	6.14E-09	6.14E-09	6.14E-09	6.14E-09	6.14E-09	6.14E-09	6.14E-09
ES	1.88E-07	1.88E-07	1.88E-07	1.88E-07	1.88E-07	1.88E-07	1.88E-07	1.88E-07
FI	1.97E-08	1.97E-08	1.97E-08	1.97E-08	1.97E-08	1.97E-08	1.97E-08	1.97E-08
FR	2.67E-07	2.66E-07	2.67E-07	2.66E-07	2.67E-07	2.66E-07	2.66E-07	2.67E-07
GR	4.22E-08	4.21E-08	4.21E-08	4.21E-08	4.21E-08	4.21E-08	4.21E-08	4.21E-08
HR	1.55E-08	1.54E-08	1.55E-08	1.54E-08	1.55E-08	1.54E-08	1.54E-08	1.55E-08
HU	5.10E-08	5.08E-08	5.08E-08	5.08E-08	5.08E-08	5.08E-08	5.08E-08	5.08E-08

6.2 Accumulated intake

In order to arrive at the effective accumulated intake related to a respective exposure pathway, within this assessment the corresponding effective intake fraction of a selected pollutant is multiplied by the emission of the same pollutant into the environment as described in the following formulation:

$$U_{accumulated}(p,e,i,b) = \sum_{t=1}^{t_{total}} \left(iF(p,e,i,b) \cdot E_{air}(p, b_{total}, s_{total}) \right) \qquad \textbf{Equation 6-2}$$

where

$U_{accumulated}$: accumulated intake of substance *p* via a specific exposure pathway *e* for a certain population *i* in country *b* [kg_{intake}]

t : year for which an annual intake has been calculated [yr]

t_{total} : time span for which the accumulated intake has been calculated [yr] as sum of all annual intakes within the considered time span [yr]

iF : effective intake fraction of substance *p* via a specific exposure pathway *e* for a certain population *i* in country *b* [kg_{intake} per $kg_{released}$]

E_{air} : overall considered emission of substance *p* into the atmospheric compartment of all countries b_{total} due to direct releases of all source categories s_{total} [$kg_{released}$]; note that the fraction of how much a certain source category contributes to the overall exposure cannot be extracted from any output parameter.

According to Equation 6-2, the accumulated intake over a specific time span for arsenic is exemplarily shown in Table 6-2. Taking into account that the base for the calculation of the accumulated intake refers to the overall emission in a country and not to a particular unit (e.g. one kg), the values in Table 6-2 can be interpreted as follows:

- The total amount of emissions of arsenic in the scenario ‘1t reduction in Belgium’ will lead to an accumulated intake of 0.211 kg_{intake} in Germany due to the ingestion of different food items under steady state conditions (see marked cell in Table 6-2). Note that the ‘1t red.’ for this reduction scenario only refers to the defined reduction for the respective country, i.e. one ton of reduction only in Belgium compared to ‘BAU 2010 base’.

Table 6-2: Accumulated intake [kg] due to food ingestion of arsenic per scenario, steady state. For further details, see text.

Intake [kg]	BAU2010	EU27+2	AL	AT	BA	BE	BG	BY
	-	1t red.	10kg red.	1t red.	10kg red.	1t red.	1t red.	10kg red.
AL	5.16E-03	5.15E-03	5.15E-03	5.15E-03	5.15E-03	5.14E-03	5.14E-03	5.15E-03
AT	2.52E-02	2.51E-02	2.51E-02	2.51E-02	2.51E-02	2.51E-02	2.51E-02	2.51E-02
BA	4.31E-03	4.30E-03	4.30E-03	4.30E-03	4.30E-03	4.29E-03	4.29E-03	4.30E-03
BE	2.56E-02	2.56E-02	2.56E-02	2.56E-02	2.56E-02	2.56E-02	2.56E-02	2.56E-02
BG	9.01E-03	8.99E-03	8.99E-03	8.99E-03	8.99E-03	8.99E-03	8.99E-03	8.99E-03
BY	2.37E-02	2.36E-02	2.36E-02	2.36E-02	2.36E-02	2.36E-02	2.36E-02	2.36E-02
CH	1.61E-02	1.60E-02	1.60E-02	1.60E-02	1.60E-02	1.60E-02	1.60E-02	1.60E-02
CY	2.20E-03	2.20E-03	2.20E-03	2.20E-03	2.20E-03	2.20E-03	2.19E-03	2.20E-03
CZ	2.83E-02	2.82E-02	2.82E-02	2.82E-02	2.82E-02	2.82E-02	2.82E-02	2.82E-02
DE	2.11E-01	2.11E-01	2.11E-01	2.11E-01	2.11E-01	2.11E-01	2.11E-01	2.11E-01
DK	1.63E-02	1.62E-02	1.62E-02	1.62E-02	1.62E-02	1.62E-02	1.62E-02	1.62E-02
EE	3.74E-03	3.73E-03	3.73E-03	3.73E-03	3.73E-03	3.73E-03	3.73E-03	3.73E-03
ES	1.14E-01	1.14E-01	1.14E-01	1.14E-01	1.14E-01	1.14E-01	1.14E-01	1.14E-01
FI	1.20E-02	1.19E-02	1.19E-02	1.19E-02	1.19E-02	1.19E-02	1.19E-02	1.19E-02
FR	1.62E-01	1.62E-01	1.62E-01	1.62E-01	1.62E-01	1.62E-01	1.62E-01	1.62E-01
GR	2.56E-02	2.55E-02	2.55E-02	2.55E-02	2.55E-02	2.55E-02	2.55E-02	2.55E-02
HR	9.40E-03	9.38E-03	9.38E-03	9.38E-03	9.38E-03	9.38E-03	9.38E-03	9.38E-03
HU	3.09E-02	3.09E-02	3.09E-02	3.09E-02	3.09E-02	3.09E-02	3.09E-02	3.09E-02

6.3 Impacts – end-point specific incidences

When the amount of a substance taken in by humans is linked to one of the human health end-points, which are further described in Section 5.2, one arrives at the cumulative risk of a whole population of concern as sum of the individual lifetime risk of a human individual to be affected by the respective end-point. According to M. Matthies (personal communication) and /Bachmann 2006/ most of the effect information is given as thresholds like No Observed Adverse Effect Levels (NOAEL) or Lowest Observed Adverse Effect Levels (LOAEL). Hence, Such threshold based end-points may not be extrapolated from individual risk to population risk due to the fact that such measures bring about two main problems in the context of pan-European external cost assessments:

- During marginal external cost assessments, a threshold-based effect measure ’punishes’ the human activity that emits the final amount of a substance causing the threshold to be exceeded by holding it responsible for all effects to occur. However, there were usually other human activities as well that used up the ’assimilative capacity of the environment’ /Pearce; Turner 1990/ or from an exposure perspective the human population’s ability to accommodate emissions, i.e., the ’erosion of the available Margin Of Exposure’ (MOE, /Crettaz et al. 2002/). Comparative analyses are, thus, hampered.
- In order to decide whether a threshold exceedance is likely to occur, true environmental concentrations need to be estimated. Due to limited resources and imperfect

information for instance on all emissions and processes influencing the environmental fate of a substance, a modelling exercise at the regional scale needs to fail to predict true environmental concentrations if it does not succeed by accident.

Due to the fact that true environmental concentrations are so far not predictable in the frame of the present parameterisation study, it is consequently necessary to look for alternatives. Thus, an approach has been proposed to also convert threshold effect information into linear so-called slope factors for cancer /Crettaz et al. 2002/ and non-cancer effects /Pennington et al. 2002/ based on /Crettaz 2000/. In these studies, the effective dose ED_{10h} is the maximum likelihood (rather than the 95% lower confidence limit for the BMD_{10}) estimate of the dose corresponding to 10% response of humans over background. It is derived by fitting a steady model through a discrete set of measured dose-response data employing a so-called linear multistage model. The ED_{10h} is taken as the point of departure in order to extrapolate to lower doses. It is assumed that the dose-response curve is linear and crosses at the origin for substances not showing thresholds in their effects. The slope factor, thus, represents a measure for the population-averaged excess individual risk of an effect per unit daily dose for a lifetime exposure. The linearisation is based on a non-threshold assumption. In contrast to non-carcinogenic effects /Pennington et al. 2002/, it is generally well accepted that there is no threshold for genotoxically acting carcinogens even at the individual level /World Health Organisation 2000/; /Tennant 2001/.

On the other hand, the linearisation is considered justified irrespective of the type of effect due to the growing recognition that ’no evidence’ does not necessarily mean ’no effect’ and that bioassays cannot give real insights on linearity or non-linearity at low doses, which only depend on the extrapolation model adopted. While toxicologists argue that mechanistic threshold concentrations or doses may exist for human health effects for many (non-genotoxically carcinogenic) substances, usually it has not been possible to establish the existence of mechanistic thresholds in epidemiological studies. Populations consist of individuals that show different susceptibilities or sensitivities to develop the investigated diseases even at low ambient levels /Hurley; Miller 2001/. Additionally, it may be argued not to assume thresholds from a precautionary principle perspective which is adopted by the European Council /European Commission 2000/. Due to these facts care must be taken not to bias the assessment through rather conservative approaches.

However, all considered human health end-points have been extrapolated linearly from individual risks to population risk, even when thereby overestimating those impacts that are based on end-points with thresholds.

The number of incidences of a particular end-point in the whole population can be derived from taking the accumulated intake of the population and multiply it with the unit

risk factor of the health end-point of concern. While the unit risk factors of all considered end-points are described in detail in Section 5.2, the calculation of the number of end-point specific incidences is described by the following formulation:

$$n_{incidences}(p,e,h,i,b) = \sum_{t=1}^{t_{total}} \left(U_{accumulated}(p,e,i,b) \cdot rf(p,h,e) \right) \qquad \textbf{Equation 6-3}$$

where

$n_{incidences}$: number of incidences of human health end-point *h* due to intake of substance *p* via a specific exposure pathway *e* by a certain population *i* in country *b* [$risk_{population}$]; note that the risk for the whole population is assumed to equal the sum of all individual risks (by linear extrapolation of individual risks)

t : year for which an annual number of incidences has been calculated [yr]

t_{total} : time span for which the total number of incidences has been calculated [yr] as sum of all annual numbers of incidences within the considered time span [yr]

$U_{accumulated}$: accumulated intake of substance *p* via a specific exposure pathway *e* for a certain population *i* in country *b* [kg_{intake}]

rf : unit risk factor of human health end-point *h* due to intake of substance *p* via a specific exposure pathway *e* [$risk_{personal} \cdot kg_{intake}^{-1}$].

According to Equation 6-3, the number of incidences per end-point over a specific time span for arsenic is exemplarily shown in Table 6-3 and can be interpreted as follows:

- When emitting one $kg_{released}$ of arsenic in Belgium, it will lead to a number of skin cancer incidences in Germany of 2.71E-07 due to the ingestion of different food items under steady state conditions (see marked cell in Table 6-3). This number refers to the overall risk to the population of Germany as linear extrapolation of the individual risk of getting skin cancer. Note that the '1t red.' for this reduction scenario only refers to the defined reduction for the respective country, i.e. one ton of reduction only in Belgium compared to 'BAU 2010 base', whereas the number of incidences is normalised and thus based on the emission of one kg.

Table 6-3: Number of incidences for skin cancer due to food ingestion of arsenic per 1 kg of released arsenic, steady state. For further details, see text.

Number of Cases	BAU2010	EU27+2	AL	AT	BA	BE	BG	BY
	-	1t red.	10kg red.	1t red.	10kg red.	1t red.	1t red.	10kg red.
AL	6.65E-09	6.64E-09	6.64E-09	6.63E-09	6.64E-09	6.63E-09	6.63E-09	6.64E-09
AT	3.25E-08	3.24E-08	3.24E-08	3.24E-08	3.24E-08	3.24E-08	3.24E-08	3.24E-08
BA	5.55E-09	5.54E-09	5.54E-09	5.54E-09	5.54E-09	5.54E-09	5.54E-09	5.54E-09
BE	3.31E-08	3.30E-08	3.30E-08	3.30E-08	3.30E-08	3.30E-08	3.30E-08	3.30E-08
BG	1.16E-08	1.16E-08	1.16E-08	1.16E-08	1.16E-08	1.16E-08	1.16E-08	1.16E-08
BY	3.05E-08	3.05E-08	3.05E-08	3.05E-08	3.05E-08	3.05E-08	3.05E-08	3.05E-08
CH	2.07E-08	2.07E-08	2.07E-08	2.07E-08	2.07E-08	2.07E-08	2.07E-08	2.07E-08
CY	2.84E-09	2.83E-09	2.83E-09	2.83E-09	2.83E-09	2.83E-09	2.83E-09	2.83E-09
CZ	3.65E-08	3.64E-08	3.64E-08	3.64E-08	3.64E-08	3.64E-08	3.64E-08	3.64E-08
DE	2.72E-07	2.71E-07	2.72E-07	2.71E-07	2.72E-07	2.71E-07	2.71E-07	2.72E-07
DK	2.10E-08	2.09E-08	2.09E-08	2.09E-08	2.09E-08	2.09E-08	2.09E-08	2.09E-08
EE	4.82E-09	4.81E-09	4.81E-09	4.81E-09	4.81E-09	4.81E-09	4.81E-09	4.81E-09
ES	1.48E-07	1.47E-07	1.47E-07	1.47E-07	1.47E-07	1.47E-07	1.47E-07	1.47E-07
FI	1.54E-08	1.54E-08	1.54E-08	1.54E-08	1.54E-08	1.54E-08	1.54E-08	1.54E-08
FR	2.09E-07	2.09E-07	2.09E-07	2.09E-07	2.09E-07	2.09E-07	2.09E-07	2.09E-07
GR	3.30E-08	3.29E-08	3.29E-08	3.29E-08	3.29E-08	3.29E-08	3.29E-08	3.29E-08
HR	1.21E-08	1.21E-08	1.21E-08	1.21E-08	1.21E-08	1.21E-08	1.21E-08	1.21E-08
HU	3.99E-08	3.98E-08	3.98E-08	3.98E-08	3.98E-08	3.98E-08	3.98E-08	3.98E-08

6.4 Severity measures

From a valuation perspective, it is necessary to distinguish between diverse human health effects as 'effects' in general may lead to consequences with different severities like acute death or some short-lived skin irritation. In case of fatal diseases, the concept of Years of Life Lost (YOLL) is recommended in different contexts (e.g. /Krewitt et al. 2002/). The YOLL indicator measures the reduction in life expectancy resulting from an increased level of exposure to pollutants in the environment. In order also to account for effects related to morbidity, /Crettaz et al. 2002/ and /Pennington et al. 2002/ make use of the Disability Adjusted Life Years (DALY) concept. It comprises the effects measured by the YOLL indicator and adds the measure Years of Life lived with a Disability (YLD). Although the approach has some disadvantages related to the derivation of the YLD and when applied to non-cancer effects (see below), it is deemed a step towards a more differentiated assessment of cancers for whose valuation only one generic monetary value for any type of cancer is used according to the latest ExternE methodology /European Commission 2004/.

$$n_{severity}(p,h,e,i,b) = \sum_{t=1}^{t_{total}} \left(n_{incidences}(p,e,h,i,b) \cdot smf(p,h,e) \right) \qquad \textbf{Equation 6-4}$$

where

$n_{severity}$: number of a specific severity measure per human health end-point h due to intake of substance p via a specific exposure pathway e by a certain population i in country b [YOLL; YLD; IQ Points loss]; note that $n_{severity}$

only refers to ‘years of life lost’, ‘years of life lived with a disability’ and ‘IQ Points loss’, while ‘disability adjusted life years’ are calculated according to Equation 6-5 as no severity-measure factors for DALYs have been applied directly

t : year for which an annual amount of external costs has been calculated [yr]

t_{total} : time span for which the total external costs have been calculated [yr] as sum of all annual external costs within the considered time span [yr]

$n_{incidences}$: number of incidences of human health end-point h due to intake of substance p via a specific exposure pathway e by a certain population i in country b [-]

smf : severity-measure factor per incidence of human health end-point h due to intake of substance p via a specific exposure pathway e [YOLL·incidence^{-1}; YLD·incidence^{-1}; IQ Points lost·incidence^{-1}].

$$n_{DALY}(p,h,e,i,b) = n_{YOLL}(p,h,e,i,b) + n_{YLD}(p,h,e,i,b) \qquad \textbf{Equation 6-5}$$

where

n_{DALY} : number of ‘disability adjusted life years’ per human health end-point h due to intake of substance p via a specific exposure pathway e by a certain population i in country b [DALY]

n_{YOLL} : number of ‘years of life lost’ per human health end-point h due to intake of substance p via a specific exposure pathway e by a certain population i in country b [YOLL]

n_{YLD} : number of ‘years of life lived with a disability’ per human health end-point h due to intake of substance p via a specific exposure pathway e by a certain population i in country b [YLD].

According to Equation 6-4, the number of a particular severity measure per human health end-point over a specific time span for arsenic is exemplarily shown in Table 6-4 and can be interpreted as follows:

- When emitting one $kg_{released}$ of arsenic in Belgium, it will lead to an accumulated ‘years of life lost’ for skin cancer of 1,41E-06 YOLL in Germany due to the ingestion of different food items under steady state conditions (see marked cell in Table 6-4). Note that the ‘1t red.’ for this reduction scenario only refers to the defined reduction for the respective country, i.e. one ton of reduction only in Belgium compared to ‘BAU 2010 base’, whereas the numbers of severity measure are normalised and thus based on the emission of one kg.

Table 6-4: Number of YOLLs for skin cancer due to food ingestion of arsenic per 1 kg of released arsenic, steady state. For further details, see text.

Number of YOLL	BAU2010	EU27+2	AL	AT	BA	BE	BG	CH
	-	1t red.	10kg red.	1t red.	10kg red.	1t red.	1t red.	1t red.
AL	3.44E-08	3.43E-08	3.44E-08	3.43E-08	3.44E-08	3.43E-08	3.43E-08	3.43E-08
AT	1.68E-07	1.68E-07	1.68E-07	1.68E-07	1.68E-07	1.68E-07	1.68E-07	1.68E-07
BA	2.87E-08	2.87E-08	2.87E-08	2.87E-08	2.87E-08	2.87E-08	2.87E-08	2.87E-08
BE	1.71E-07	1.71E-07	1.71E-07	1.71E-07	1.71E-07	1.71E-07	1.71E-07	1.71E-07
BG	6.01E-08	6.00E-08	6.00E-08	6.00E-08	6.00E-08	6.00E-08	6.00E-08	6.00E-08
BY	1.58E-07	1.58E-07	1.58E-07	1.58E-07	1.58E-07	1.58E-07	1.58E-07	1.58E-07
CH	1.07E-07	1.07E-07	1.07E-07	1.07E-07	1.07E-07	1.07E-07	1.07E-07	1.07E-07
CY	1.47E-08	1.47E-08	1.47E-08	1.47E-08	1.47E-08	1.47E-08	1.46E-08	1.47E-08
CZ	1.89E-07	1.88E-07	1.88E-07	1.88E-07	1.88E-07	1.88E-07	1.88E-07	1.88E-07
DE	1.41E-06	1.41E-06	1.41E-06	1.41E-06	1.41E-06	1.41E-06	1.41E-06	1.41E-06
DK	1.09E-07	1.08E-07	1.08E-07	1.08E-07	1.08E-07	1.08E-07	1.08E-07	1.08E-07
EE	2.50E-08	2.49E-08	2.49E-08	2.49E-08	2.49E-08	2.49E-08	2.49E-08	2.49E-08
ES	7.64E-07	7.62E-07	7.62E-07	7.62E-07	7.62E-07	7.62E-07	7.62E-07	7.62E-07
FI	7.99E-08	7.97E-08	7.97E-08	7.96E-08	7.97E-08	7.96E-08	7.96E-08	7.97E-08
FR	1.08E-06	1.08E-06	1.08E-06	1.08E-06	1.08E-06	1.08E-06	1.08E-06	1.08E-06
GR	1.71E-07	1.70E-07	1.70E-07	1.70E-07	1.70E-07	1.70E-07	1.70E-07	1.70E-07
HR	6.28E-08	6.26E-08	6.26E-08	6.26E-08	6.26E-08	6.26E-08	6.26E-08	6.26E-08
HU	2.07E-07	2.06E-07	2.06E-07	2.06E-07	2.06E-07	2.06E-07	2.06E-07	2.06E-07

6.5 Damages – external costs

After having calculated all basic and intermediate output parameters for the considered scenarios it is the overall goal of the present study to finally end up in the linkage between the human activity, which is herein represented by the amount of anthropogenic emissions released into the atmospheric environment, and the external effects due to this emissions, represented by monetary values. This is due to the fact that it was not only tried to assess the impacts of HMs on human health but also to value them in order to e.g. support cost-benefit analysis. Monetised externalities are termed external costs when they are negative and external benefits when they are positive. However, as the impacts calculated here per emission of a contaminant are negative, we finally arrive at external costs by multiplying the number of end-point specific incidences with the applied number of a representative severity measure per end-point, i.e. DALY as sum of YOLL and YLD, or IQ Points loss (please refer to Equation 6-4 and Equation 6-5) and finally with the costs that are assigned to one unit of the respective severity measure:

$$EC(p,h,e,i,b) = \sum_{t=1}^{t_{total}} \left(n_{severity}(p,e,h,i,b) \cdot scf(p,h,e) \right) \qquad \textbf{Equation 6-6}$$

where

EC : external costs of human health end-point h due to intake of substance p via a specific exposure pathway e by a certain population i in country b [$Euro_{2000}$]

t : year for which an annual amount of external costs has been calculated [yr]
t_{total} : time span for which the total external costs have been calculated [yr] as sum of all annual external costs within the considered time span [yr]
$n_{severity}$: number of severity measure per end-point h due to intake of substance p via a specific exposure pathway e by a certain population i in country b [YOLL; YLD; IQ Points loss]; note that $n_{severity}$ only refers to 'years of life lost', 'years of life lived with a disability' and 'IQ Points loss', while 'disability adjusted life years' are calculated according to Equation 6-5 as no severity-cost factors for DALYs have been applied directly
scf : severity-cost factor per incidence of human health end-point h due to intake of substance p via a specific exposure pathway e [$Euro_{2000}$ per YOLL; $Euro_{2000}$ per YLD; $Euro_{2000}$ per IQ Points loss].

According to Equation 6-6, the accumulated external costs per end-point over a specific time span for arsenic is exemplarily shown in Table 6-5 and can be interpreted as follows:

- When emitting one $kg_{released}$ of arsenic in Belgium, it will lead to accumulated external costs for skin cancer of 0.058 $Euro_{2000}$ in Germany due to the ingestion of different food items under steady state conditions (see marked cell in Table 6-5). Note that the '1t red.' for this reduction scenario only refers to the defined reduction for the respective country, i.e. one ton of reduction only in Belgium compared to 'BAU 2010 base', whereas the external costs are normalised and thus based on the emission of one kg.

Table 6-5: External costs for skin cancer [$Euro_{2000}$] due to food ingestion of arsenic per 1 kg of released arsenic, steady state. For further details, see text.

Costs [€2000]	BAU2010 -	EU27+2 1t red.	AL 10kg red.	AT 1t red.	BA 10kg red.	BE 1t red.	BG 1t red.	BY 10kg red.
AL	1.42E-03	1.42E-03	1.42E-03	1.42E-03	1.42E-03	1.42E-03	1.42E-03	1.42E-03
AT	6.93E-03	6.91E-03	6.92E-03	6.91E-03	6.92E-03	6.91E-03	6.91E-03	6.92E-03
BA	1.19E-03	1.18E-03	1.18E-03	1.18E-03	1.18E-03	1.18E-03	1.18E-03	1.18E-03
BE	7.06E-03	7.04E-03	7.04E-03	7.04E-03	7.04E-03	7.04E-03	7.04E-03	7.04E-03
BG	2.48E-03	2.47E-03	2.48E-03	2.47E-03	2.48E-03	2.47E-03	2.47E-03	2.48E-03
BY	6.52E-03	6.51E-03	6.51E-03	6.50E-03	6.51E-03	6.50E-03	6.50E-03	6.51E-03
CH	4.42E-03	4.41E-03	4.41E-03	4.41E-03	4.41E-03	4.41E-03	4.41E-03	4.41E-03
CY	6.06E-04	6.04E-04	6.05E-04	6.04E-04	6.05E-04	6.04E-04	6.04E-04	6.05E-04
CZ	7.79E-03	7.77E-03	7.77E-03	7.77E-03	7.77E-03	7.77E-03	7.77E-03	7.77E-03
DE	5.81E-02	5.80E-02	5.80E-02	5.80E-02	5.80E-02	5.80E-02	5.80E-02	5.80E-02
DK	4.48E-03	4.47E-03	4.47E-03	4.47E-03	4.47E-03	4.47E-03	4.47E-03	4.47E-03
EE	1.03E-03	1.03E-03	1.03E-03	1.03E-03	1.03E-03	1.03E-03	1.03E-03	1.03E-03
ES	3.15E-02	3.14E-02	3.14E-02	3.14E-02	3.14E-02	3.14E-02	3.14E-02	3.14E-02
FI	3.29E-03	3.29E-03	3.29E-03	3.29E-03	3.29E-03	3.29E-03	3.29E-03	3.29E-03
FR	4.46E-02	4.45E-02	4.45E-02	4.45E-02	4.45E-02	4.45E-02	4.45E-02	4.45E-02
GR	7.05E-03	7.03E-03	7.03E-03	7.03E-03	7.03E-03	7.03E-03	7.03E-03	7.03E-03
HR	2.59E-03	2.58E-03	2.58E-03	2.58E-03	2.58E-03	2.58E-03	2.58E-03	2.58E-03
HU	8.52E-03	8.50E-03	8.50E-03	8.50E-03	8.50E-03	8.50E-03	8.49E-03	8.50E-03

Unless dealing with acute health effects, a delicate question arises when dealing with long time spans that are highly relevant for e.g. the ingestion pathway and thus play a role in

the present assessment. The question is “How can we compare future external costs to present external costs?” Economists usually employ discounting in order to give future benefits or costs present values. In addition, the European Commission recommends the involvement of discounting “whenever positive and negative impacts can be expressed in monetary terms” /European Commission 2002, p. 16/. Thus, the application of different discount schemes to the external cost output of the parameterisation runs will be discussed in the following section.

6.6 Discounting – the value of future damages

Generally, when one wants to perform a cost-benefit analysis, the question of ’how are human health effects to be valuated’ comes up. When dealing rather with chronic than with acute health effects, for a cost-benefit analysis it is required to compare future damages or benefits with present damages or benefits. In economy, thus, discounting as a weighting scheme is employed in order to convert future effects into present values, starting from the human activity, such as an emission, following the whole impact pathway via environmental fate and exposure to various receptors up to the damages, e.g. expressed in monetary terms. As an example also the European Commission recommends to use a discount rate ”whenever positive and negative impacts can be expressed in monetary terms” /European Commission 2002/.

Generally, discounting only will be applied to the evaluation part of the full chain and thus only monetary terms can be presented as discounted values while all other output variables, such as accumulated intake, cannot be interpreted properly when applying discounting here.

Nevertheless, for some technical reasons it makes sense to apply discounting to the first output variable which is available, i.e. in the present study the accumulated exposure, in order to simplify the calculation process when e.g. adapting another set of dose-response relationships as base for the external cost assessment. This means, that when starting to discount the monetary terms rather than the exposures, one has to completely rerun the whole scenarios for receiving discounted costs for every new dose-response relationship. When discounting the exposures instead, one always has a discounted basis for the whole impact and damage assessment, independently from what dose-response relationships will be used.

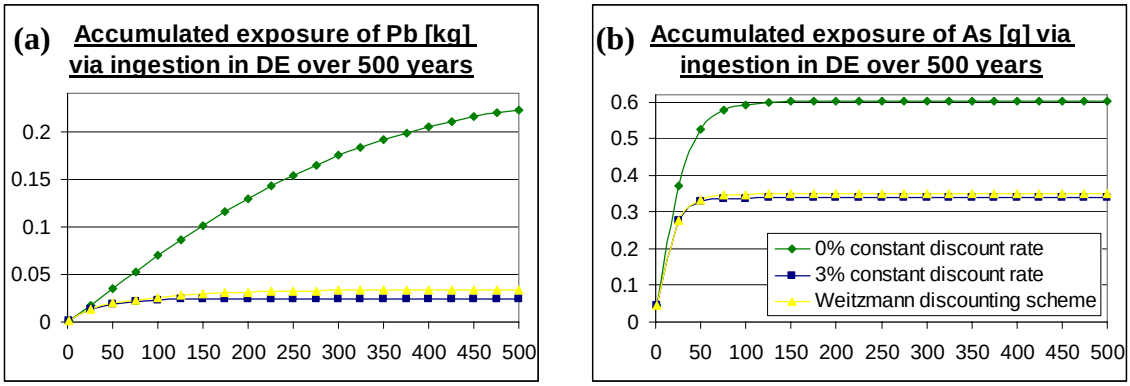

Figure 6-1: Comparison between different discounting schemes for the accumulated exposure via ingestion of lead (a) and arsenic (b) in Germany ('BAU 2010 base' scenario).

Note, that discounting is always conducted when valuing effects at different points in time because 'no discounting' simply means to use a discount rate of 0%. Thus, discounting is always performed when dealing with effects occurring over a longer time span, either explicitly or implicitly. As an example of how different contaminants accumulate in the human body over a given time span, for the two HMs lead and arsenic the time dependency is shown in Figure 6-1. In Figure 6-1 (a) lead accumulates very slowly over the first 500 years in the human body and thus discounting the exposure does highly influence valuation of related damages. In contrast to that in Figure 6-1 (b) the accumulation over time differs entirely for arsenic, which is ingested up to 95% within the first 100 years. Hence, discounting the exposure occurring in the nearer future does not influence the accumulated exposure as much as it does for lead.

According to the fact that there are several considerations why the valuation of deep-future damages and benefits should be valued differently from those occurring in the near future different discount schemes may be applied. In the context of long-term effects, /Azar, Sterner 1996/ as well as /Weitzman 1999/ conclude that there is no rationale for a constant discount factor in time and therefore suggest different discount rates for different time periods. As shown in Table 6-6, Weitzman introduced a variable discount rate that declines depending on the time period considered in the future due to increasing uncertainty about the predictability of future interest rates.

Table 6-6: Declining discount rate scheme suggested by /Weitzman 1999/ and used within WATSON for human health damages via ingestion.

Time Horizon [years]	Discount rates suggested by Weitzman (1999)
0-25	'low-normal' real annual interest rate of around 3-4%
25-75	within-period instantaneous interest rate of around 2%
75-300	within-period instantaneous interest rate of around 1%
> 300	within-period instantaneous interest rate of around 0%

When assuming 3.5% for the first 25 years according to Table 6-6, the resulting discount factors for all years in the future are computed as shown in Table 6-7.

Table 6-7: Approach of calculating different discount factors for different time periods implemented in WATSON according to /Weitzman 1999/.

Equation for calculating the discount factor W_t	Equation is valid for the time period t
$W_t = \frac{1}{(1+0.035)^t}$	for: $0 < t \le 25$
$W_t = \frac{1}{(1+0.035)^{25}} \cdot \frac{1}{(1+0.02)^{t-25}}$	for: $25 < t \le 75$
$W_t = \frac{1}{(1+0.035)^{25}} \cdot \frac{1}{(1+0.02)^{50}} \cdot \frac{1}{(1+0.01)^{t-75}}$	for: $75 < t \le 300$
$W_t = \frac{1}{(1+0.035)^{25}} \cdot \frac{1}{(1+0.02)^{50}} \cdot \frac{1}{(1+0.01)^{225}} \cdot 1$	for: $t > 300$

When dealing with comparing future effects or benefits to present effects or benefits via the ingestion pathway it is necessary to distinguish between different parts of the whole chain, i.e. the time lag between the human activity, i.e. the emission of contaminants into environmental media, and a respective exposure as well as the time lag between the exposure and the corresponding physical impacts, such as a certain cancer. For the implementation of the parameterisation scenarios only the part between emission and exposure is taken into consideration as part of the modelling approach by applying the steady-state solution to the environmental fate model, which can be seen as a 'time-integrated exposure' and thus as discounted exposure.

The calculation of the time-integrated exposure is further described in /Bachmann 2006/. Steady-state is a situation in which no changes in concentration occur over time within a certain zone. This means that all outputs of a compartment in this zone, such as arable land, equal the inputs of this compartment in the same zone. Furthermore, /Heijungs 1995/ has shown that the steady-state situation can also be used for time-integrated exposure assessments of pulse emissions which are used for the present scenarios. When computing a

time-integrated exposure as a kind of discounted exposure over time then a dynamic computation is not further required unless additional discount schemes are required to be used /Heijungs 1995/. However, in the present study the exposure and thus impacts and damages due to ingestion have been calculated for the following discount schemes: 0% discounting, 3% constant discount rate, and discount rates according to /Weitzman 1999/.

In order to also consider discounting for the part between exposure and the occurrence of the corresponding impacts, hereafter referred to as end-point specific latency; the user is recommended to additionally apply a desired discount scheme as described at the beginning of this chapter, when employing monetary values on the impacts, i.e. different physical end-points like cancers. Note again that although several output parameters exist as discounted values, e.g. number of end-point specific incidences, number of end-point specific severity measures and external costs, only the monetary values, i.e. external costs, are recommended for use as discounted values. This is due to the fact that theoretically all parameters can be discounted, but only the monetary terms are likely to explain when applying discounting.

The reason why all output parameters are available for different discount schemes is that for the parameterisation runs a variable called ‘discounted accumulated exposure’ has been introduced as base for all further output parameters in order to let the user finally choose either to use the pre-calculated discounted damages or to apply his own, different discount scheme(s) to the discounted exposure.

7 Conclusions and Outlook

The aim of this assessment was to derive simplified relationships in both space and time from various model runs that follow the pathways of the contaminants through the full chain, accounting for the highly complex and sometimes non-linear fate and exposure processes that the contaminants go through. As result, the input for the full chain approach which represents the human activities by means of emissions released into the environment, is directly linked to the output of the full chain approach, expressed e.g. in number of human health incidences or monetary terms, i.e. external costs. The parameterised relationships between input and output are readily available for integration into other assessment frameworks that aim to comprise the pathway of a chemical not only through air, but also through water and soil, and thus, also including the ingestion pathway into their exposure assessment on the base of various food items as well as of drinking water.

In addition, by providing a predefined set of output variables allows for using the results of the parameterisation runs in a wide context, starting at effective intake fraction and following the impact pathway step by step by covering one additional linear relationship between two output variables, finally ending up at external costs. This also is helpful when integrating the parameterised values into other assessment frameworks as e.g. their own dose-response relationships can be applied as long as they differ from the dose-response relationships used throughout the present assessment.

As a next step, the parameterisation procedure of the modelling framework is to be extended to not only link input to output in a linear way but to introduce a non-linear term as relationship between input and output in order to more accurately describe the linkage between what goes into the model and what comes finally out. This may then especially be helpful neither to underestimate nor to overestimate the output values when increasing or reducing the input values, respectively.

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