11 Whole economy case study

In the previous Chapter, emissions from single facilities have been investigated. The second case study considers pan-European emissions into air for the situation in the year 1990. Beside the derivation of damage factors, this scenario investigates the overall external costs resulting from activities of whole societies or economies which will be described in the following.

11.1 Pan-European emission scenario for 1990

In order to be able to follow the Impact Pathway Approach for a whole economy, information on all releases of the contaminants to be assessed into the environment is needed. However, information on emissions that directly enter the media soil and/or water is rather scarce. A European-wide inventory of direct emissions to water and soil is planned by the European Union but is unfortunately not yet available. In the case of Germany, for instance, there are data available on releases of some heavy metals but only aggregated at the catchment scale (Umweltbundesamt, 2000). Therefore, only emissions into air will be analysed in the following.

The air emission scenario is defined as described in Droste-Franke et al. (2003). The air emission inventory provided by the German Federal Environment Agency and TNO for Europe (Berdowski et al., 1997) constitutes the base information. Where possible, these emissions are updated by using more actual data from EMEP (United Nations - Economic Commission for Europe and Co-operative programme for monitoring and evaluation of long range transmission of air pollutants in Europe, 2002). The data are spatially distributed using the scenario on air pollution emissions for 1990 by assuming that within each main source sector the emissions of the selected trace elements have the same distribution as particle emissions. The totals of the emission values used in the analysis thus obtained are given in Table 11-8.

As for the single facility scenarios, pulse emissions of a one year duration are analysed. The effective Intake Fractions for inhalation and ingestion exposures are derived separately in sections 11.3 and 11.4, respectively. The proper-

ties of the investigated contaminants are given in Appendix C. But before that, the development of the concentrations over the course of time under continuous emissions shall be demonstrated.

11.2 Tentative historic emission scenario and contamination increase in time

Records on historic emissions that have led to the present contamination of environmental media and food items are non-existing to the extent necessary for an appropriate assessment of anthropogenic emissions for instance since the industrial revolution. Recent findings indicate that emissions having occurred in the second half of the twentieth century have been the highest in the period of the modern times (Barbante et al., 2004). According to the concentrations in the investigated ice cores, the emissions of cadmium and chromium for the 1990ies even belong to the highest among the entire time period covered (ibid.). Arsenic and lead were not included in the analysis. An upper bound emission scenario shall be investigated assuming that releases took place for a one hundred year period at the level of the pan-European emission scenario for 1990 described above (section 11.1). This scenario serves two purposes. First, it may give an idea of the present contamination level of environmental media that can be attributed to trace element releases from larger facilities which occurred since the advent of industrialisation. The resulting concentrations in the environmental media and food items have been compared to reported values in section 9.3.2. Second, it shall provide insight into the dynamics related to trace element releases into the environment. This shall be done in the following for selected trace elements and compartments and/or zones. For information reasons, also the potential concentrations after 1000 years and at steady-state will be given. As for the case studies, the environmental setting is according to the 'food removal' scenario (see section 9.3.3), however, not investigating pulse-emissions but continuous releases as stated above.

The development towards the steady-state situation is fastest for cadmium and slowest for arsenic of the investigated trace elements (cf. sections 10.3 and 11.4) which is why these two elements will be analysed here. From Fig. 11-1 and 11-2, it can be seen that the concentrations of cadmium and arsenic in the arable land compartment approach slowly towards the steady-state. While the situation after a 1000 year release of cadmium already fairly well approximates the situation at steady-state, the degree of contamination by arsenic changes substantially between these two points in time. These differences can be attributed to the distinct behaviour of cadmium and arsenic in terms of their mobility in the environment, manifested in their solid-water partitioning coefficients (cf. Table C-1).

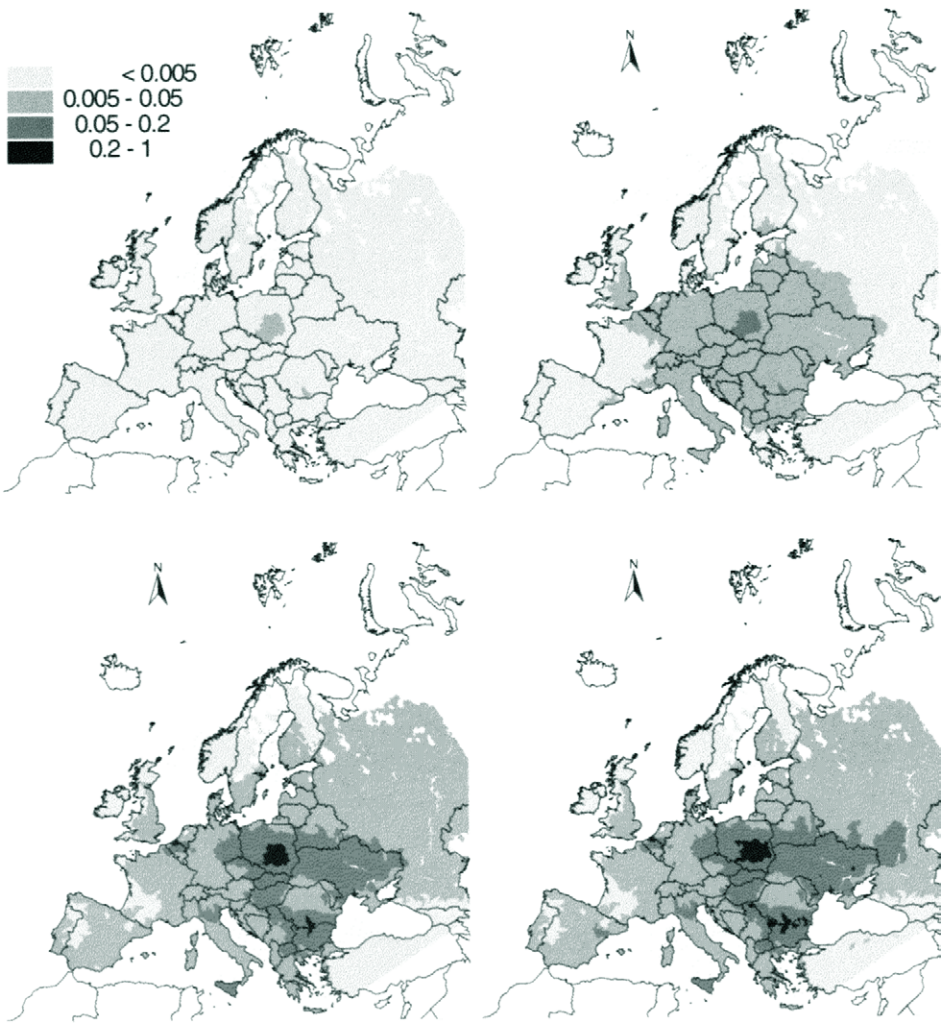

Fig. 11-1: Cadmium concentrations in arable land after 10 years (top left), 100 years (top right), 1000 years (bottom left) and at steady-state (bottom right) according to the pan-European emission scenario for 1990 (continuous releases) [mg/kg]

In order to demonstrate that the development towards the steady-state is rather different between the environmental compartments, the concentrations of arsenic in freshwater bodies are also given at different points in time (Fig. 11-3). It may be surprising to see that not only the concentrations in the arable land compartment rather slowly approach towards the steady-state situation but also those

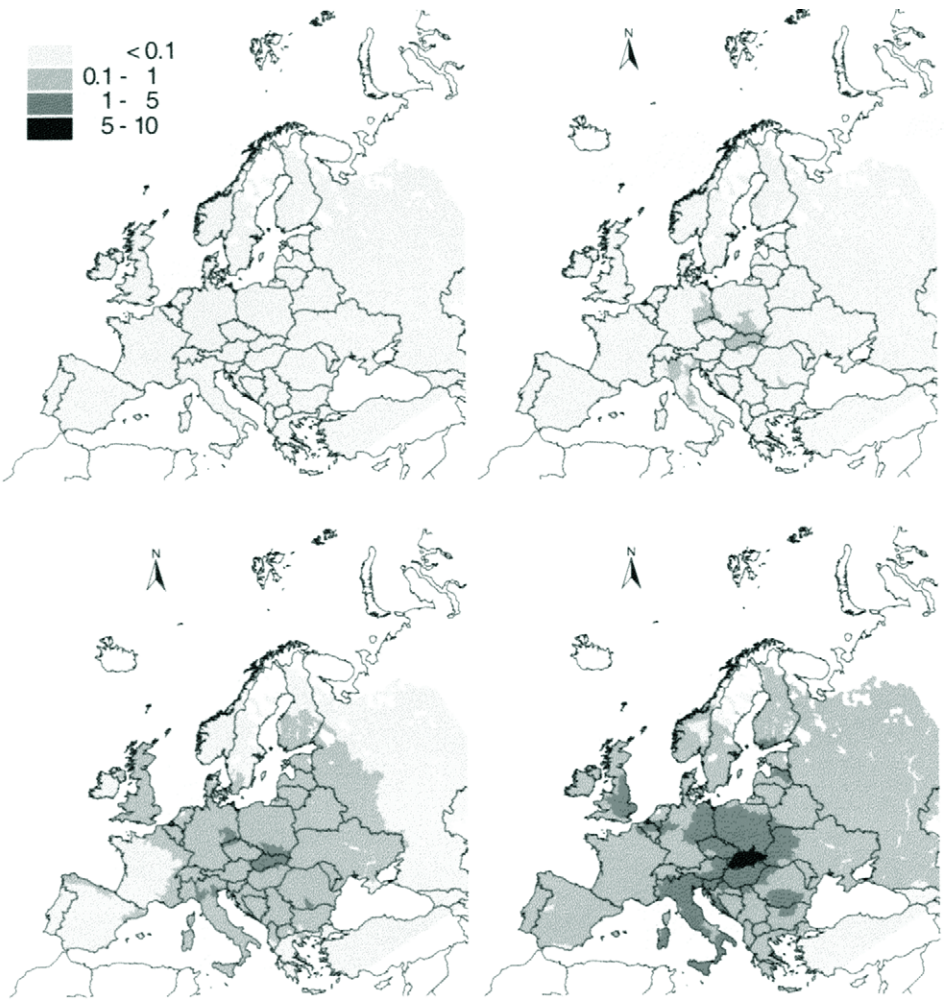

Fig. 11-2: Arsenic concentrations in arable land after 10 years (top left), 100 years (top right), 1000 years (bottom left) and at steady-state (bottom right) according to the pan-European emission scenario for 1990 (continuous releases) [mg/kg]

in freshwater bodies. In fact, substances in freshwater bodies usually have shorter residence times than in soils. However, these may only reach a steady-state once all the other compartments that deliver to the freshwater bodies are in this state. This is visualized for the temporal development of arsenic concentrations in one

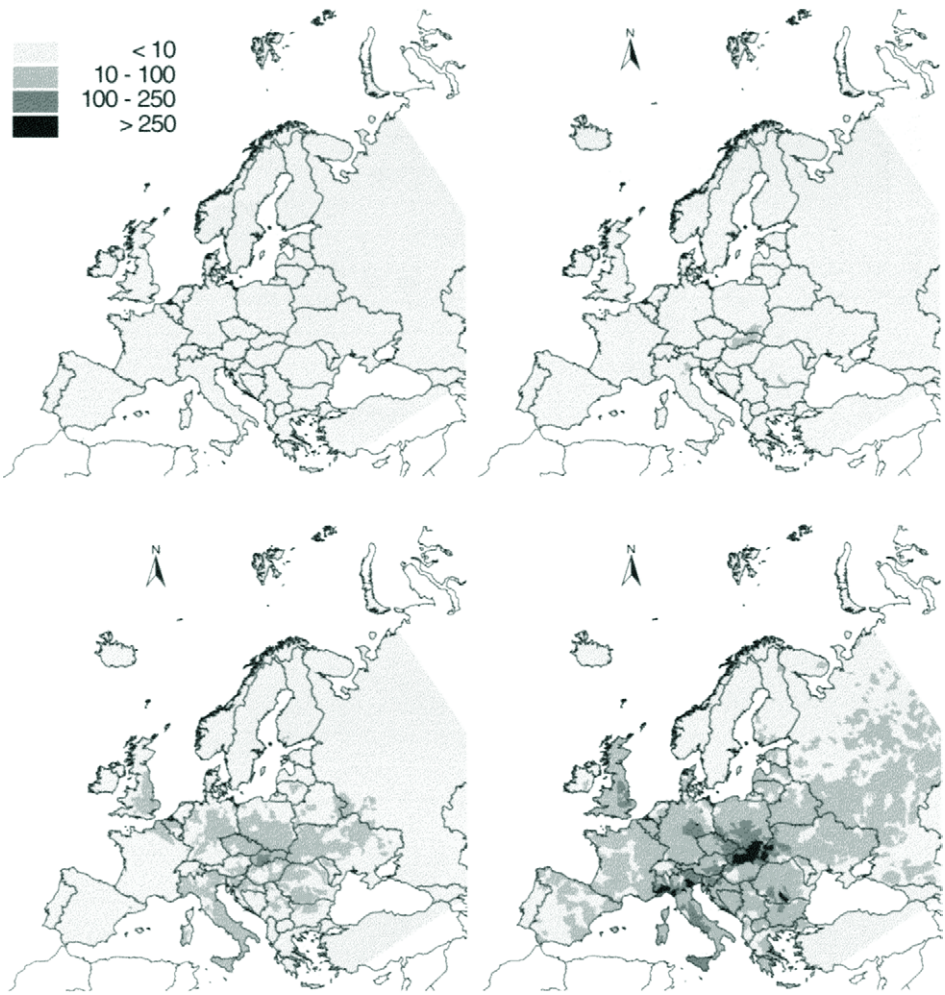

Fig. 11-3: Arsenic concentrations in freshwater bodies after 10 years (top left), 100 years (top right), 1000 years (bottom left) and at steady-state (bottom right) according to the pan-European emission scenario for 1990 (continuous releases) [pg/l]

zone. The zone selected is the Hron River, a tributary to the Danube located in the centre of the Slovakian Republic, for which the highest arsenic concentrations in both the arable land and freshwater compartment are assessed (cf. Fig. 11-2 and 11-3, respectively). The temporal development of the arsenic concentrations is

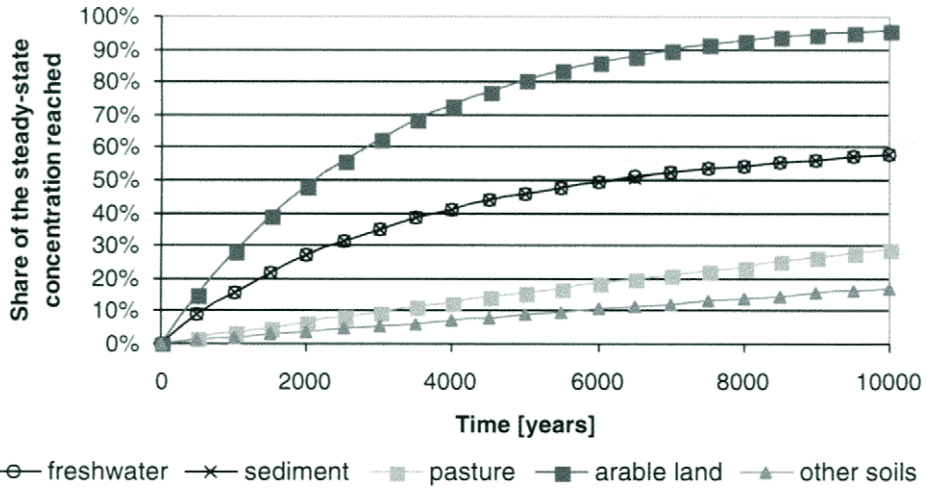

Fig. 11-4: Development of arsenic concentrations towards the steady-state in the Hron River catchment in central Slovakia according to the pan-European emission scenario for 1990 (continuous releases); the values are given relative to the steady-state situation (cf. Table 11-1) [-]

Table 11-1: Arsenic concentrations at steady-state in the Hron River catchment in central Slovakia according to the pan-European emission scenario for 1990 (continuous releases)

Compartment	Unit	Concentration
Freshwater	[ng/l]	0.72
Sediment	[µg/kg]	0.89
(Semi-) natural ecosystems ('other soils')	[mg/kg]	146
Pasture	[mg/kg]	80
Arable land	[mg/kg]	8.2

given in relation to the corresponding concentrations at steady-state in Fig. 11-4. This is because the concentrations at steady-state span orders of magnitude in this situation according to the analysed scenario (Table 11-1).

Table 11-2: Effective Intake Fractions [$kg_{inhaled}$ per $kg_{released}$] and resulting cancer and non-cancer associated Disability Adjusted Life Years (DALYs) per kilogram of a trace element released [years lost-equivalents per $kg_{released}$] due to inhalation caused by a one year pulse emission into air according to the pan-European emission scenario for 1990

Trace element	Effective Intake Fraction [$kg_{inhaled}$ per $kg_{released}$]	DALYs [years lost-equivalents per $kg_{released}$]	
		cancer	non-cancer
Arsenic	$3.2 \cdot 10^{-6}$	$2.20 \cdot 10^{-4}$	n/a
Cadmium	$1.5 \cdot 10^{-5}$	$4.41 \cdot 10^{-4}$	n/a
Chromium	$7.2 \cdot 10^{-8}$	$1.37 \cdot 10^{-5}$	$2.00 \cdot 10^{-7}$
Lead	$4.4 \cdot 10^{-6}$	n/a	n/a

Again the different dynamics between the different compartments become obvious. In the terrestrial environment, the arable land compartment is the quickest in approaching towards the steady-state. This is due to a higher outflow by means of the water soil erosion process out of this compartment followed by pastures and other land uses. In general, less than 5 % of the steady-state concentrations are reached within the first 100 years of continuous releases. While this is still the case for pastures and semi-natural ecosystems after 1000 years of continuous emissions, the share of the steady-state concentrations may amount to 16 % in the freshwater environment and up to 28 % for arable soils. After 2000 years, none of the compartments will have exceeded concentration levels that are half of those at steady-state.

Fig. 11-4 may help to explain why the influence of non-zero discounting on the case study results especially of arsenic is rather substantial (cf. sections 10.3 and 11.4) although different types of emission scenarios are investigated (continuous releases vs. pulse emission). While the arsenic concentrations in pasture soils develop rather slowly towards the steady-state situation, pasture-based food items, i.e., especially beef and dairy products, dominate the ingestion exposure towards arsenic in the long run (cf. Fig. 10-3, 10-4 and 11-6). Exactly these exposures are not taken into consideration when performing non-zero discounting owing to quasi-zero discount factors applicable to damages occurring in the intermediate to long term.

Table 11-3: Damage factors due to inhalation for a one year pulse emission according to the pan-European emission scenario for 1990 [$€_{2000}$ per $kg_{released}$]

Trace element[a]	Type of effect	Type of monetary value	Discount rate 0 %	Discount rate 3 %
Arsenic[b]	cancer	WTP	16.5	8.78
		COI	4.34	2.32
Cadmium[b]	cancer	WTP	33.1	17.6
		COI	8.72	4.65
Chromium	cancer	WTP	1.02	0.546
		COI	0.27	0.144
	non-cancer	WTP	0.015	0.00801
		COI	n/a	n/a

a.No effect information available for lead.
b.No non-cancer effect information available.

As a result, the analysis of the contamination development emphasizes the long time scales potentially involved when releasing persistent substances such as trace elements.

11.3 Impacts due to inhalation exposure

The impact assessment due to inhalation is based on the corresponding (effective) Intake Fractions. The values are given in Table 11-2. The values for arsenic and lead are in the same range as for the single facilities investigated in the previous section (Table 10-4). The reason why the Intake Fraction for cadmium and chromium tend to be larger or smaller, respectively, than for the single facility emission scenarios is seen in the distribution of hot-spot concentrations entirely over land or to some degree over the sea. According to Figure 9.2 as given in Droste-Franke et al. (2003), areas with the highest cadmium concentrations in air are located in central eastern Europe, west of the Black Sea, and along the Po river. Figure 9.3 (ibid.) shows that the highest air concentrations of chromium are on the French coast around Marseilles so that much of the chromium present in air is deposited into the Mediterranean Sea without leading to human inhalation expo-

Table 11-4: Quantifiable external costs due to inhalation of selected trace elements and in total caused by a one year pulse emission according to the pan-European emission scenario for 1990 [10^6 €$_{2000}$/yr]

Trace element[a]	Type of effect	Type of monetary value	Discount rate 0 %	Discount rate 3 %
Arsenic[b]	cancer	WTP	19.5	10.4
		COI	5.14	2.74
Cadmium[b]	cancer	WTP	16.8	8.95
		COI	4.43	2.36
Chromium	cancer	WTP	2.29	1.22
		COI	0.605	0.322
	non-cancer	WTP	0.034	0.018
		COI	n/a	n/a
Sum			*48.8*	*26.0*

a.No effect information available for lead.
b.No non-cancer effect information available.

sures. This explains the lower (effective) Intake Fraction for chromium inhalation exposures according to the pan-European emission scenario. This effect may actually be counterbalanced by tourists temporarily staying in this area especially in summer. However, migration of people is not taken into account which would also lead to some absenteeism among the local population. The difference between the pan-European scenario and those for the single facilities presented in section 10.1 carries on through the steps from exposure to impact and to valued damages in the following.

The effective Intake Fraction due to inhalation translates into DALYs (Table 11-2) according to the respective cancer and non-cancer slope factors. Missing effect information is indicated as not available ('n/a'; for further discussion refer to section 10.2). Based on the DALYs per emitted kilogram of the respective trace element, the external costs are derived next.

The external costs per kilogram emitted and those for the overall pan-European emission situation are given in Tables 11-3 and 11-4, respectively. As for the single facilities (cf. section 10.2), the damage factors estimated here are about

in the same order of magnitude as those for the classical air pollutants, noting that the classical air pollutants are emitted in much larger quantities. There is, however, a change in the ranking of the trace elements in terms of leading to the highest quantifiable specific external costs via inhalation (cf. Table 10-6 and 11-3).

The total quantifiable external costs due to inhalation of the classical air pollutants released according to a pan-European emission scenario amount to about $210 \cdot 10^9$ €$_{2000}$ in EU15 when discounted at 3 % (Droste-Franke and Friedrich, 2003). One has to note that the assessment performed by Droste-Franke and Friedrich (2003) has experienced an update in terms of both the dose-response functions and the monetary valuation (European Commission, 2004). This update leads to even higher estimates by about 10 %. Comparing the resulting $230 \cdot 10^9$ €$_{2000}$ to the high-end total quantifiable external costs due to the inhalation of arsenic and cadmium which are in the order of 10^7 €$_{2000}$ shows that the inhalation-induced human health damages due to the release of the investigated trace elements contribute only marginally.

11.4 Impacts due to ingestion exposure

The assessment of the ingestion-related human exposures is performed based on the environmental settings of the 'food removal' scenario (see section 9.3.3). As for the single facilities, the four trace elements analysed show marked differences in terms of both the absolute value of the effective Intake Fraction due to ingestion exposures and the temporal development of this measure (Fig. 11-5, note the logarithmic scale). The amount of cadmium taken in is highest reaching 0.12 % of the overall amount released. Arsenic, chromium and lead show effective Intake Fractions due to ingestion of about one order of magnitude less than cadmium in the long run. According to the present assessment, there will be a relatively and absolutely higher exposure of cadmium and lead within the next 100 years upon release of the trace elements. This means that 17.7 %, 8.2 %, 1.6 % and 1.0 % of the time-integrated effective Intake Fraction due to ingestion will have been obtained within this period of time for cadmium, lead, chromium and arsenic, respectively. These figures are much smaller for the first 10 year period (2.9 %, 1.3 %, 0.25 % and 0.12 %, respectively). The contribution of the inhalation-related Intake Fraction is insignificant in the long run (Fig. 11-5). This also holds for the short term except for arsenic and lead, indicating once more that the exposure through the media soil and water are most significant at least in the intermediate to long term.

As could already be demonstrated for the single facilities, not only the total amounts taken in vary considerably in time but also the most contributing food items (top vs. bottom of Fig. 11-6). Again note the different contributions of cattle

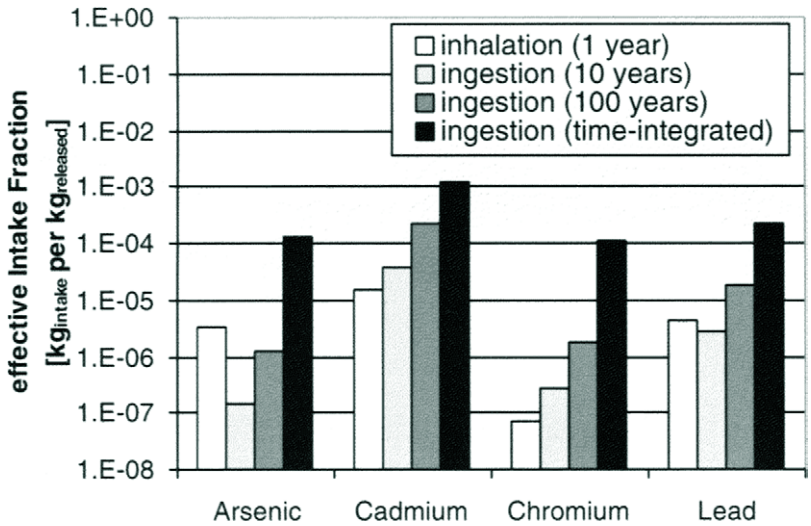

Fig. 11-5: Effective Intake Fraction of selected trace elements via inhalation after one year and via ingestion of food after 10 and 100 years, and time-integrated for a one year pulse emission according to the pan-European emission scenario in 1990 (note the logarithmic scale) [kg_{intake} per $kg_{released}$]

products and spinach for the situations after 10 years and when integrating exposure over infinite time. The human exposure towards cadmium and lead primarily occurs primarily through cereals and to some extent through potatoes while dairy products are also important for chromium and even more so for arsenic paralleling the picture for the single facilities. Beef only constitutes a remarkable share of chromium's Intake Fraction in the long run (Fig. 11-6) while freshwater fish and other animal products such as pork, poultry and eggs contribute insubstantially.

Generally, non-cancer effects dominate the DALYs assessed due to ingestion (Table 11-5) which is also due to lack of cancer effect information (cf. section 7.3). While the DALYs due to non-cancer effects are more than one order of magnitude larger than due to cancer for arsenic, this discrepancy amounts to about three orders of magnitude for lead. Exposure towards lead and cadmium leads to the highest human health impacts amounting to about 22 and 37 year lost-equivalents per tonne emitted, respectively.

Valuing the impacts assessed to occur via ingestion yields the damage factors (Table 11-6). These vary considerably according to the different discount rates employed due to the slow dynamics and long persistency of the substances investigated. The damage factors are at the lower end of those for the single facil-

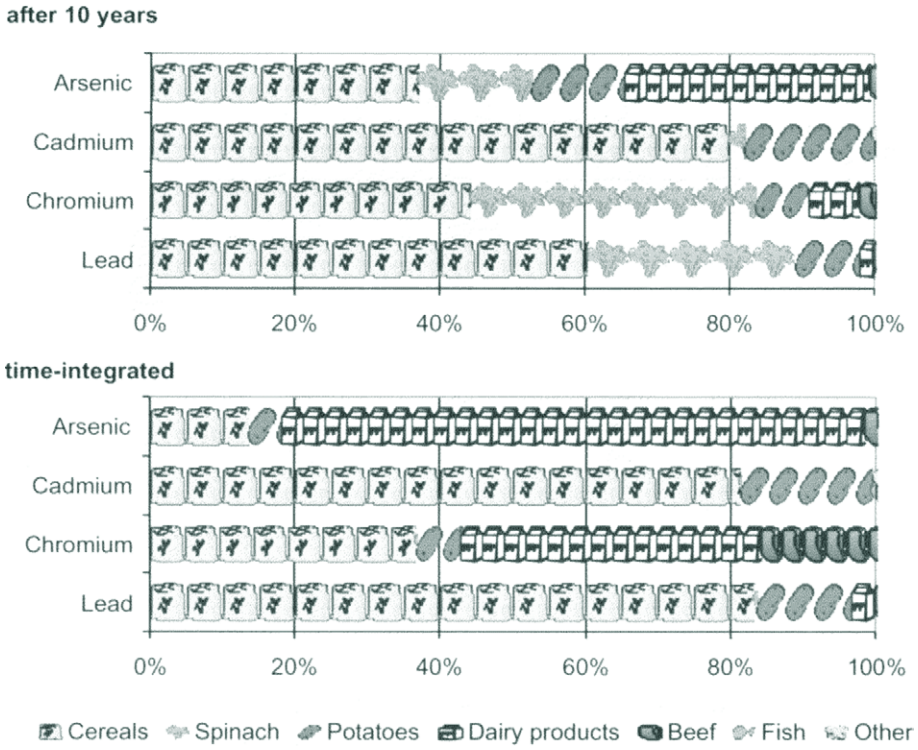

Fig. 11-6: Relative contribution of the different food items to the effective Intake Fraction (ingestion) of selected trace elements after 10 years (top) and time-integrated (bottom) for a one year pulse emission according to the pan-European emission scenario to air in 1990 (cliparts by Corel Corporation, 2002)

Table 11-5: Time-integrated Disability Adjusted Life Years (DALYs) per kilogram of trace element released due to cancer and non-cancer effects upon ingestion exposure caused by a one year pulse emission according to the pan-European emission scenario for 1990 [years lost-equivalents per $kg_{released}$]

Trace element	Cancer effect	Non-cancer effect
Arsenic	$3.35 \cdot 10^{-4}$	$7.11 \cdot 10^{-3}$
Cadmium	n/a	$3.71 \cdot 10^{-2}$
Chromium	n/a	$1.14 \cdot 10^{-5}$
Lead	$6.06 \cdot 10^{-5}$	$2.24 \cdot 10^{-2}$

ities, i.e., equal or below those for the power plant in the UK (cf. Table 10-13). This is also reflected in the estimated DALY values (Table 10-9 vs. Table 11-5). This can partly be explained by depositions taking place into the sea to a large extent (especially for chromium). The impact pathway cannot be followed through the marine environment at present as discussed in section 7.1. Additionally, the smaller impact of the pan-European emission scenario may be explained by the 'mismatch' of elevated atmospheric depositions and highly productive agricultural areas. In line with the findings by MacLeod et al. (2004), this has been identified to be one reason why, for instance, exposure through potatoes are more substantial for the emission sites in Germany and in Belgium as compared to those in the UK and France (cf. section 10.3).

The damage factors are significantly smaller when non-zero discounting is performed, owing to the long-lived nature of these pollutants (cf. Hellweg, 2000; van den Bergh et al., 2000; Huijbregts et al., 2001; de Vries et al., 2004). Depending on the dynamics of the respective pollutant (cf. Fig. 11-5), the effect is more (e.g., arsenic) or less (e.g., cadmium) pronounced. Related to the issue of discounting, it is interesting to see that its impact is variable for the investigated scenarios. For instance, when computing the ratio between the 3 % discounted and 0 % discounted damage factors for cadmium of the facility in the UK and the pan-European scenario, these ratios amount to 6.4 % and 4.0 %, respectively. This difference hints at different temporal distributions of impacts attributable to variable properties of the receiving environments leading to different dynamics for the metal to reach the human population from different release sites. This stresses once more that it does not mean that the site of release is almost irrelevant as argued by Spadaro and Rabl (2004) despite trade may lead to homogeneous levels in the food items under consideration.

A comparison of the damage factors to the one for arsenic found in the literature has been performed in a previous section (10.3) and shall thus not be repeated here.

The quantifiable external costs through ingestion exposures due to the emissions of the considered trace elements to air in 1990 range from about two million to some ten billion Euros when discounting at a rate of 0 % (Table 11-7). The highest values are obtained for lead caused by hypertension followed by those for cadmium due to kidney damage which are about a factor of 40 smaller. The damages due to arsenic and chromium contribute about a factor of two and about three orders of magnitude less than those for cadmium, respectively. When comparing the total quantifiable external costs due to ingestion discounted at 3 % to the most dominant damage costs due to inhalation, i.e., those of the classical air pollutants, discounted at the same rate (Droste-Franke and Friedrich, 2003), the contribution is marginal. The highest external costs quantified for lead constitute

Table 11-6: Damage factors due to ingestion for a one year pulse emission according to the pan-European emission scenario for 1990 [$€_{2000}$ per $kg_{released}$]

Trace element	Type of effect	Type of monetary value	Discount rate 0 %	Discount rate 3 %
Arsenic	cancer	WTP	25.1	0.0446
		COI[a]	17.1	0.0303
	non-cancer	WTP	533	0.946
		COI	n/a	n/a
Cadmium[b]	non-cancer	WTP	2785	110
		COI	n/a	n/a
Chromium[b]	non-cancer	WTP	0.855	0.00262
		COI	n/a	n/a
Lead	cancer	WTP	4.55	0.0740
		COI[a]	1.52	0.0247
	non-cancer	WTP	1681	27.4
		COI[c]	62	1.01

a.COI for average cancer.
b.Cancer effect information not available.
c.COI for hypertension.

about 0.4 % of those of the classical air pollutants in 1990 of about $230 \cdot 10^9$ $€_{2000}$ (value updated, only impacts in EU15 valued, cf. section 11.3). When discounting at 0 %, a different picture is obtained. The total quantifiable external costs due to inhalation of classical air pollutants increase only little to $240 \cdot 10^9$ $€_{2000}$ (derived according to Droste-Franke, 2005) whereas those due to ingestion exposure towards the trace elements increase substantially (cf. Table 11-7). As a result, the quantifiable external costs due to lead add about 25 % to those due to the classical air pollutants. Three aspects need to be noted related to this comparison. First, the estimated relative contributions by the trace elements discussed so far and in the following may be overestimated as the external cost assessments by Droste-

Table 11-7: Quantifiable external costs due to ingestion of selected pollutants and in total caused by a one year pulse emission according to the pan-European emission scenario for 1990 [10^6 €$_{2000}$/yr]

Trace element	Type of effect	Type of monetary value	Discount rate 0 %	Discount rate 3 %
Arsenic	cancer	WTP	29.7	0.0528
		COI[a]	20.2	0.0359
	non-cancer	WTP	631	1.12
		COI	n/a	n/a
Cadmium[b]	non-cancer	WTP	1410	55.8
		COI	n/a	n/a
Chromium[b]	non-cancer	WTP	1.91	0.00586
		COI	n/a	n/a
Lead	cancer	WTP	158	2.57
		COI	52.8	0.860
	non-cancer	WTP	58500	952
		COI[c]	2160	35.2
Sum			63000	1050

a.COI for average cancer.
b.Cancer effect information not available.
c.COI for hypertension.

Franke (2005) for the classical air pollutants only consider the damages within the area that is determined by the 15 European Union member countries (EU15; i.e., the countries being member of the European Union as of 1995: Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, The Netherlands, Portugal, Spain, Sweden and the United Kingdom). Taking only the damages occurring in these countries into account leads to total quantifiable external costs due to lead exposure that are about half of those reported in Table 11-7. On the other hand, the presented assessment does not contain all exposure pathways so that the damages caused by the investigated contaminants need to be con-

Table 11-8: Total trace element emissions in Europe in 1990 and 2000 estimated according to Droste-Franke et al. (2003) and ESPREME (2004), respectively

Trace element	Emissions [t/yr] in ...		Relation
	1990	2000	[%]
Arsenic	1184	654	55
Cadmium	508	269	53
Chromium	2238	1658	74
Lead	34797	10923	31

sidered lower bound estimates. Further work is needed in order to find out to what extent these incompleteness of both analyses lead to a bias and into which direction. Second, the β_{ED10} slope factor for hyptertension due to lead must be considered particularly uncertain (cf. Table 7-7). Thus, the next highest total quantifiable external costs are those for cadmium adding only 0.59 % or 0.024 % to those of the classical air pollutants when discounting at 0 % and 3 %, respectively. These contributions are reduced by about 60 % when only taking the damages in the EU15 countries into account. Third, when transferring the assessment to more recent years the change in the emission situation needs to be borne in mind. The emissions of contaminants into air have considerably declined since the early 1990s. In the case of the investigated trace elements, this particularly concerns lead (cf. Table 11-8). Assuming the same emission distribution at the reduced level of the year 2000, the total quantifiable ingestion-related external costs from lead, thus, only amount to $19 \cdot 10^9$ €$_{2000}$ (0 % discounting). These can be compared to the external costs assessed for an emission scenario of NO_x, SO_2, NH_3, NMVOC and primary particles for 1998 (Droste-Franke, 2005) which may serve as a proxy for the year 2000. The external costs discounted at a rate of 0 % and updated as described above amount to about $150 \cdot 10^9$ €$_{2000}$. As a result, the contribution by lead to the external costs caused by the emission of classical air pollutants is about 13 %. When using a discount rate of 3 %, again only a very small relation is obtained, i.e., 0.2 %. This emphasizes once more that the exposure towards persistent substances such as the trace elements investigated here involves very long time horizons, substantially lowering their monetised damages when performing non-zero discounting.

Inhalation-related annual external costs due to a one year pulse emission at the level of 1990 have been presented in Table 11-4. These are not only negligible when compared to the classical air pollutants (cf. section 11.3) but also when

compared to ingestion exposure of the same trace elements (comparing the totals in Table 11-4 with those in Table 11-7). Discounting the damages following ingestion exposures with up to a discount rate of 3 % does not change this conclusion.

Including substances other than the classical air pollutants in the assessment, thus, leads to an increase in the total quantifiable external costs at most in the order of less than one percent when discounting at 3 % according to the presently available information. A similar picture has been obtained for ingestion exposures to dioxins and dioxin-like substances for selected countries for which the contribution amounts to at most a few percent (Droste-Franke et al., 2003). Assuming that the time preference of society is indifferent towards when a benefit is realized, i.e., when discounting at 0 %, the contribution of the investigated trace elements to the total quantifiable external costs related to human health impacts becomes more substantial. These are assessed to contribute at most 25 % to the total quantifiable external costs caused by the classical air pollutants. Thus, depending on society's time preference, the impact by the investigated trace elements may be substantial.

Some general concluding remarks on the estimated DALYs and external costs for both case studies will be given in sections 12.3.2 and 12.4.