HEALTH RISK ASSESSMENT GUIDANCE FOR METALS

FACT SHEET

HERAG 03

INDIRECT EXPOSURE VIA THE ENVIRONMENT AND CONSUMER EXPOSURE

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1. Introduction

This fact sheet summarises the experience gained by industry in previous and current ESR and voluntary risk assessments on the aspects of "indirect exposure via the environment" and "consumer exposure". Whereas both topics are not strictly regarded to be metal-specific, it was considered as an outcome of the HERAG discussions that there are several issues which are particularly relevant for metals and which are not appropriately covered by the standard TGD assessment approaches. Therefore, the aim of this document is to capture these metal-specific issues in order to facilitate exchange of such knowledge on a broader basis for the metals industry.

In chapter 2 general aspects for the exposure assessment and risk characterisation for “indirect exposure” as well as metal specific issues in the assessment of "indirect exposure" are discussed.

The issue of consumer exposure is then further addressed in chapters 3, including a tabular summary of previously encountered "consumer exposure" issues, and followed by a discussion of "consumer exposure" issues of common interest and with particular relevance to metals.

For reference purposes, the sections on both topics start with a brief summary of the principles of the “conventional” assessment of "indirect exposure of humans via the environment" and of "consumer exposure" (i.e. subsections 2.1. and 3.1.)

Conclusions and recommendations are then given in chapter 4.

For further reference detailed summaries on the assessment of “indirect exposure” and "consumer exposure" as conducted in previous or ongoing metal risk assessments are presented in the appendices. However, these presentations vary in extent and detail according to the input received from industry and to the state of the respective RAR or VRAR, some of which were only in a draft stage at the time of compilation of this fact sheet.

2. Indirect exposure via the environment

2.1. Current principles for the assessment of indirect exposure via the environment

Indirect exposure of humans may occur via the typical exposure pathways inhalation, ingestion and dermal contact. The TGD (Part I, Chapter 2, Section 2.4) lists the following exposure scenarios and gives detailed guidance on their assessment:

- exposure via inhalation of air

- exposure via drinking water

- exposure via food consumption
  concentrations in food are derived using the concepts:
  - bioconcentration in fish
  - biotransfer from soil and air to plants
  - biotransfer to meat and milk

Exposure via soil ingestion and dermal contact is discounted for in the indirect exposure section of the TGD because exposure through these routes is considered to be very unlikely. Noted as exceptions are extremely polluted soils e.g. land fill sites or accidental releases.

Specific calculation methods and lists of default input parameters for the calculation of the indirect exposure via the pathways listed above are given in Annex III to Part I of the TGD. In general, the assessment of indirect exposure is carried out following a step-wise procedure:

- Assessment of the concentration in intake media (food, water, air, soil)
- Assessment of the intake rate of each medium
- Combination of the concentration and the intake rate, and if necessary a factor for the bioavailability of the substance through the respective route of intake
- Finally, an internal dose is derived which is taken forward to the risk characterisation.

The concentrations in environmental compartments and intake media used to derive the daily intake should preferably be based on monitored data. As an alternative, modelling of environmental concentrations can be used by applying approaches described in Part II, Section 3 of the TGD, which addresses Environmental Risk Assessment. Methods are given to derive the concentration in intake media from concentrations in environmental compartments.

The determination of reasonable average consumption rates is usually a complex task, because human food patterns vary to a great extent not only between individuals but also between cultures or countries in general, and a common assessment of the total EU population might not be possible.

Indirect exposure is principally assessed on a local and on a regional spatial scale. In this context, local and regional environments are not actual sites or regions, but standardised environments as defined in the before mentioned TGD section on Environmental Risk Assessment. The local scale, in which all food products are derived from the vicinity of one point source, represents a worst-case situation. In contrast, the regional scale assessment depicts a highly averaged exposure situation, which cannot ensure protection of the individual.

In case the properly conducted local assessment does not indicate a potential risk, there is no reason for concern. In contrast, when the regional assessment indicates a risk, there is a clear need for refinement of the assessment. The situation is less clear when the local assessment gives reason for concern and the regional assessment does not. In this case, a further analysis of the major exposure sources in the local scenario should be used to investigate its realism.

2.2. General aspects for exposure assessment and risk characterisation for "indirect exposure"

This section gives an overview of aspects that are not explicitly metal specific but require attention when conducting exposure assessment and risk characterisation for indirect exposure via the environment.

Accuracy of the assessment at different levels

In general terms, a risk characterisation for indirect exposure via the environment can be conducted at several different levels, as shown below in hierarchical order of complexity and relevance:

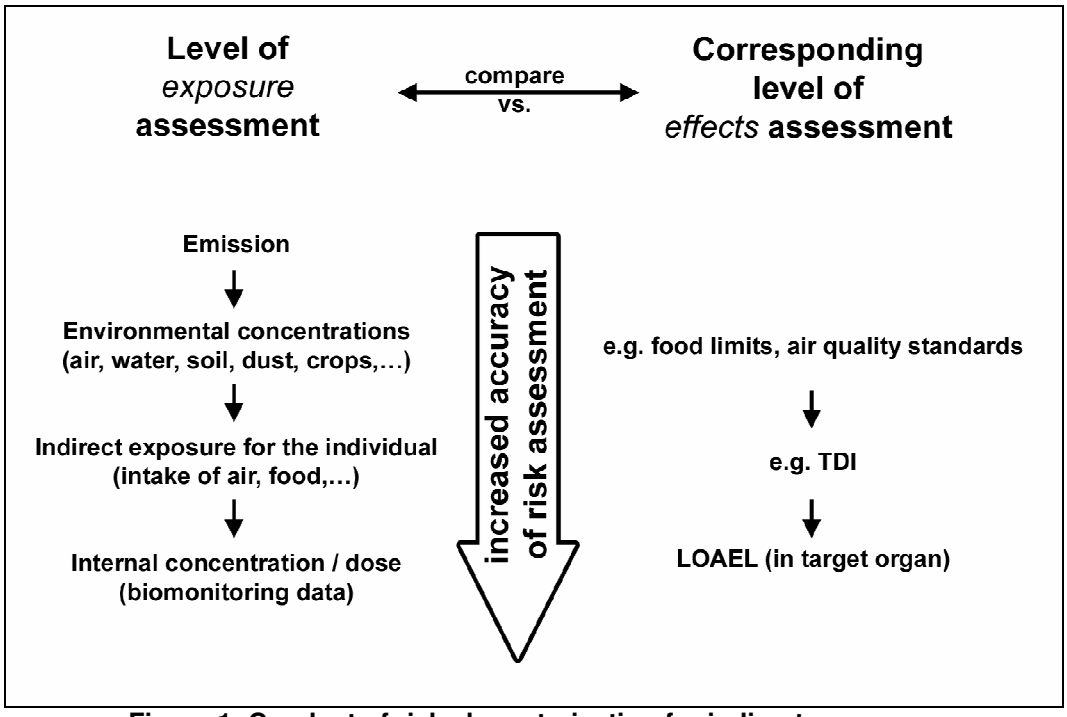


Figure 1: Conduct of risk characterisation for indirect exposure at different levels of accuracy.

If regulatory limit values like air quality standards or food limits exist for a given substance, a direct comparison with environmental concentrations may be feasible at the first level. As a more refined approach, a comparison of actual intake with limit values such as a TDI (tolerable daily intake) could be performed. At the final and principally most accurate level, the actual internal dose (biomarkers) of individuals is compared with the lowest observe adverse effect level (LOAEL), preferably in a specific target organ for repeated dose toxicity.

In general, preference should be given to internal over external concentrations for risk characterisation. Measured concentrations are preferred over predicted exposure values.

Methods for the assessment of exposure via the ingestion of food

For the specific case exposure route "ingestion of food" several tools are available for the determination of this intake, which can be ranked according to their level of accuracy as indicated in figure 2 below.

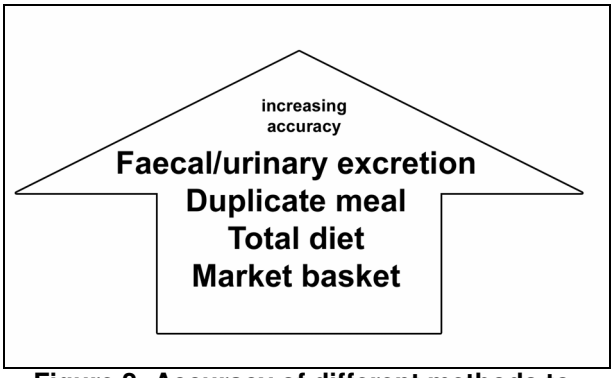


Figure 2: Accuracy of different methods to assess actual intake via food

Conversion of external exposures to internal doses using absorption factors:

For a reliable conversion of external exposures (via food, water and air) into systemic uptakes, accurate absorption factors are required. It is not uncommon that absorption will vary as a function of the level of intake (i.e., saturation; for examples, refer to the risk assessments on Zinc and Copper). Thus, considerably different oral absorption factors are likely be required to assess intake from dietary sources (relevant for consumers and indirect exposure) and under occupational circumstances, for example. For more details, please refer to the separate fact sheet on GI uptake and toxicokinetic models.

Similarly, inhalation uptake will vary considerably between the occupational setting and situations where humans are exposed via ambient air. In the latter situation, chemicals are commonly assumed to be adsorbed onto fine (sub-micron) particulate matter, which is assumed to penetrate to a high extent to the pulmonary region of the lung, with potentially very high uptake rates, for which often 100% absorption is assumed by default (for examples, please refer to the risk assessment on Lead). In contrast, workplace aerosols are often characterised by mass median aerodynamic diameters of airborne particles well above 10 microns, so that the bulk of inhaled material will be redirected to the GI tract, with absorption rates usually much lower that 100% (for more details, see fact sheet on inhalation exposure/absorption).

In summary, the assessor needs to choose on which level and with which tools to conduct the risk assessment, depending on the required level of accuracy, the available or obtainable measured data and on the quality of available prediction and modelling methods.

2.3. Biomonitoring data

In chapter 2.2 above, preference is given to biomonitoring data over an assessment of external exposure, for the reasons that any assessment of external exposure requires (among other aspects) a certain level of conservatism in view of inherent variations between exposures of individuals and the rate of uptake. The uncertainty in the two latter aspects are minimised when instead of external exposures and a translation into an estimated systemic doses, the actual body burden is more specifically measured by adequate biomonitoring data.

Biomonitoring data offer several advantages over environmental monitoring (e.g. air monitoring) to evaluate internal dose and hence to estimate overall integrated health risks. The first advantage of biological monitoring is the fact that the biological parameter of exposure is more directly related to the adverse health effects that one attempts to prevent than any environmental measurement. Secondly, biological monitoring takes into consideration absorption by all routes (lung, skin, gastrointestinal tract). Non-occupational background exposure, often much more difficult to assess quantitatively, can be expressed at a biological level as the organism integrates this external (residence, dietary habits, smoking, leisure activity,…) environmental exposure into a common systemic burden.

In previous EU risk assessments, such biomonitoring has only been considered for hazard and exposure characterisation for the two metals cadmium and lead, the basis of which is briefly summarised in the two subchapters below.

2.3.1. Blood lead biomonitoring

In the VRA on lead and lead compounds, a detailed description of the use of blood lead biomonitoring in the assessment of indirect, consumer and occupational exposure is given. Since in this exceptional case, all human health effects are also based on an integrated measure of systemic exposure (blood lead levels) reflective of multi-media exposure sources and human studies, the risk characterisation could be conducted with a low level of uncertainty.

Bioavailability of lead may vary as a function of multiple factors such as chemical speciation, age of the exposed individual, level of exposure, the matrix within which the lead was contained and nutritional status. Children are known to absorb lead at higher rates than adults: default uptake rates of 50% and 5-10% are assumed for children and adults, respectively. The higher uptake rates observed in children compared to adults is assumed to be related to uptake pathways for essential minerals (e.g. calcium and Iron) which are more active in children than in adults.

The mineral content of food is one contributing factor that reduces absorption of lead when ingested with a meal. For example, the presence of calcium and phosphate in a meal will depress the absorption of ingested lead, perhaps by competition for binding sites that mediate uptake. In addition, lead absorption in children is affected by nutritional iron status: children who are iron deficient have higher blood lead concentrations than similarly exposed children who have ample stores of iron (Wasserman et al. 1994). Dietary calcium can also affect lead absorption if intake levels are inadequate and calcium deficiency results.

Apart from diet matrix effects, the absorption of lead from ingested soil is reduced in comparison (30% for children and 6% for adults). This adjustment is based upon empirical observations and is presumed to be due to the high adsorptive affinity of soil particles for lead.

Indirect exposure via the environment is largely a function of dietary intake levels for adults and soil/dust ingestion levels for children. Uptake of lead is known to occur via efficient saturable active transport pathways (intended for the uptake of essential nutrients) and non-saturable passive diffusion mechanisms. Nonlinearity of uptake results as lead exposure increases and uptake shifts from saturable active transport to passive diffusion processes. This non-linearity of uptake as a function of exposure intensity is reflected in computerised exposure simulation models (such as the Integrated Exposure Uptake Biokinetic Model for children - for more details, please refer to the separate fact sheet on gastrointestinal uptake and absorption and toxicokinetic models).

Ambient exposures to airborne lead are usually to fine particles, for which a mean mass median aerodynamic diameter (MMAD) in the range of 0.5 µm can be assumed, and which are deposited primarily in the alveolar sacs or the lower respiratory tract of the lung, where they are almost completely absorbed.

For reasons of completeness, it is noted here that inhalation exposure under occupational circumstances involves predominantly relatively coarse aerosol particles which deposit in the upper airways with subsequent translocation to the GI tract (with correspondingly lower absorption rates). The use of lead in air measurements to monitor occupational lead exposure has a long history. The lead industry relied on voluntary standards to reduce their employees' exposure to lead beginning as early as the 1920s.

However, it is also worth mentioning that the prediction of the air/blood lead relationship in the workplace is further complicated by ingestion as a relevant route of exposure via hand-to-mouth transfer, and other inadvertent oral uptake (deposition on facial skin).

2.3.2. Urinary cadmium biomonitoring

In the EU RAR on cadmium, the risk characterisation of man indirectly exposed to Cd via the environment has been based on internal concentration data (Cd body burden of the general population expressed as urinary Cd, i.e. Cd-U). The Cd-U values were either measured or derived from Cd uptake under different scenarios for the general population. The conversion of Cd intake to Cd-U required a one-compartment model, derived from the Nordberg-Kjellström model (assuming a Cd $t_{1/2}$ of 13.6 y, 1/3 of the body burden in the kidney and a daily urinary excretion of 0.016 % of Cd kidney content). It was calculated that a continuous uptake of 1 µg Cd/day was equivalent to a urinary excretion of about 0.5 µg Cd/24 hours or 0.5 µg/g creatinine at the age of 50 years. The Cd-U values related to daily uptakes calculated by this approach can be compared for validation with the data reported in large epidemiological studies conducted in Europe. The calculated Cd-U intakes were subsequently compared with the critical Cd-U concentration (LOAELs) for the different effects.

Since cadmium is a cumulative toxicant, the use of a biological marker of body burden (i.e. Cd-U) allows to integrate long-term exposure. However, caution must be applied to the accuracy of analytical methods when using biomonitoring data. For example, in the past, reports on Cd blood levels have often been unreliable owing largely to the difficulties encountered in the analysis of this element in blood. The assessment of the indirect exposure to Cd via the environment - as well as the risk characterisation - has also allowed to validate the proposed gastro-intestinal absorption factors. In the Cd RAR, iron deficiency was identified as a major factor that may influence the fraction of food Cd absorbed by the GI tract. The fraction of food Cd absorbed as reported in published literature varied between 3 and 10 %, the highest values being associated with deficiencies of Fe, Zn or Ca, for example. The one compartment model that was used to convert the Cd intakes into internal concentrations showed that a 3 % absorption factor could be used in the risk characterisation to predict the internal concentration rather than 10%.

Two independent data sets were used to validate the model by comparing calculated dietary and measured Cd-U data in the general population. Median Cd-U data were adequately predicted using $t_{1/2}$= 13.6 years - as in the Nordberg-Kjellström model - and a gastrointestinal absorption rate of maximally 5 %. GI absorption rates of 5 and 10 % and the kidney Cd half life of 13.6 years overestimated median and upper values of observed Cd-U in these two reliable databases.

The model was also validated with the data of a group of women with a vegetarian/high fibre diet. Adequate predictions of Cd-U were also obtained with a GI absorption rate of 3 % and the kidney Cd half life of 13.6 years. The largest observed Cd-U was 1.8 fold overestimated with a 5 % absorption rate and 3.8-fold overestimated with a 10 % absorption rate. This demonstrated that the upper ranges of internal dose were best described when selecting a 3 % GI absorption rate for a kidney Cd half-life of 13.6 years (predicted/observed ratio 0.9-1.3).

Overall, these results might reflect the fact that, while increased GI absorption rates up to 10 % may exist during certain periods of iron deficiency (e.g. late pregnancy), this status does not persist constantly during the whole life and that using an absorbed food fraction of Cd of 10% during 50 years would be inadequate for risk characterisation.

2.4. Indirect exposure via the environment - metal specific issues and scenarios

The introduction to this fact sheet places foremost emphasis in the precision of exposure assessment on biomonitoring data. However, two aspects need to be considered beyond this:

- In the absence of adequate biomonitoring, the only fallback position is the conduct of an external exposure assessment, which should consider all relevant routes and sources of exposure and be as accurate as possible.

- Even when biomonitoring is available, the quantitative assessment of external exposures is nevertheless essential to identify the most relevant sources of exposure, so that if required, correct recommendations for appropriate risk reduction measures can be given.

In order to facilitate the indirect exposure assessment for metals, previous risk assessments reports conducted under the ESR or as voluntary risk assessments were screened for issues of "indirect exposure via the environment", and in addition key metal industries provided input on their experience with these issues.

Detailed extracts of the indirect exposure assessments of individual metals are given as background information in appendix A 1 of this fact sheet. In contrast, this section focuses on deviations from the TGD approach and attempts so summarise metal specific issues (by route of exposure) which have been identified and require attention in future risk assessments.

2.4.1. Indirect exposure via the route of inhalation

2.4.1.1. Exposure related to local point sources

For local exposure assessments, conventional TGD plume modelling based on stack emissions has largely been used for metals so far. As a result, for metals such as Zn, Pb, Cu and Cd, environmental inhalation exposure via this route has been shown to be quantitatively insignificant.

However, air concentrations derived from TGD models represent levels 100 m from the local source emission stack. In reality, the spatial layout of the vast majority of industrial sites most often causes the nearest residential populations to be located at significantly greater distances, whereas there is currently no practically relevant concept for the assessment of fugitive emissions.

2.4.1.2. Exposure related to smoking

<u>General aspects:</u>

Smoking has been shown to constitute a relevant intake route for metals, for example in the EU RAR on cadmium and cadmium oxide. Concerning the latter, the conclusion of the indirect exposure assessment was that a risk could not be completely ruled out for the general population, but also that the "higher percentiles" were likely to be represented by smokers.

For this reason, the contribution of smoking to the overall systemic exposure to metals is subjected to a detailed review in this fact sheet. In previous metal risk assessments, several discrepancies or deficiencies in the assessment of smoking have been noted:

- Conflicting assignment to either "consumer" (i.e., copper VRA) or "indirect exposure" (i.e., cadmium ESR RAR) section.
- Passive smoking may also be of relevance, but is not accounted for.
- Heterogeneity between allocation of typical and RWC exposure (cigarette consumption).
- Lack of information on cigarette consumption of workers compared to the general population.

Quantitative aspects relevant for the exposure assessment:

The content of a particular "metal" in cigarette tobacco may often be known, as perhaps also the concentration in smoke. This amount can show significant variation as a function of the geographic source of tobacco used, cultivation practices, soil properties and anthropogenic sources that might result in deposition of metals upon tobacco leaf surfaces. However, for quantitative exposure assessment purposes, only the relative fraction of the metal retained after inhalation from the smoke should be considered for assessment (where possible), instead of the total concentration in the smoke. For some metals, such as lead and cadmium, it is also possible to quantify the impact of smoking upon exposure through the measurement of metals in relevant biological compartments such as blood and/or urine.

It should also be noted that apart from direct uptake of a metal contained in cigarette smoke, smoking may also impact the uptake of metals through other mechanisms: constituents of cigarette smoke can inhibit mucociliary clearance mechanisms and/or result in the reduction of ciliated surfaces through the induction of squamous metaplasia. This can impair the clearance of metals from the respiratory tract, including metals/compounds that might be present in cigarette smoke or metals derived from other sources (e.g. such as in the occupational setting). A quantitative individual contribution of the above mentioned aspects is however not possible in the absence of accurate exposure biomarkers. Further understanding of this mechanism may explain the observation that for some metals, smoking is associated with higher systemic exposure.

Data on cigarette consumption in the EU:

There is considerable heterogeneity in the assumption of "typical" and "realistic worst case" numbers for total number of cigarettes/day between the various metal risk assessments. Not only did maximum assumed daily consumptions vary between metal reports, but also between population sub-groups (general population vs. workers), and in one case, "typical" consumer behaviour was assumed to be "non-smoking". It was therefore suggested to obtain more substantiated data on average and RWC cigarette/tobacco consumption. From a multitude of data sources, the following three were considered most relevant:

WHO: Detailed annual per capita WHO consumption statistics are available at the following website: http://www.cdc.gov/tobacco/who. However, the most recent data on cigarette consumption dates back to 1992. Since tobacco control policies have been stepped up in recent years, other information sources were also investigated.

Giskes et al., 2005: In a recent multi-national study, data on smoking status and daily cigarette consumption were obtained from nine European countries (Norway, Sweden, Denmark, Finland, United Kingdom, Netherlands, Germany, Italy and Spain) involving a total of 451 386 participants (non-institutionalised men and women, 25–79 years old). As a brief summary it can be concluded that whereas the overall number of smokers may have gradually declined in the period 1985-2000, the average consumption/day of smokers did not change to any relevant extent: in the countries studied (which do not cover those EU countries with the highest cigarette consumption), women tended to consume 10-12 cigarettes per day, whereas men consumed between 13-15 cigarettes per day on average.

EEIG, 2003: A recent project of the EU Commission has provided very recent data on cigarette consumption in the EU, which is presented graphically below:

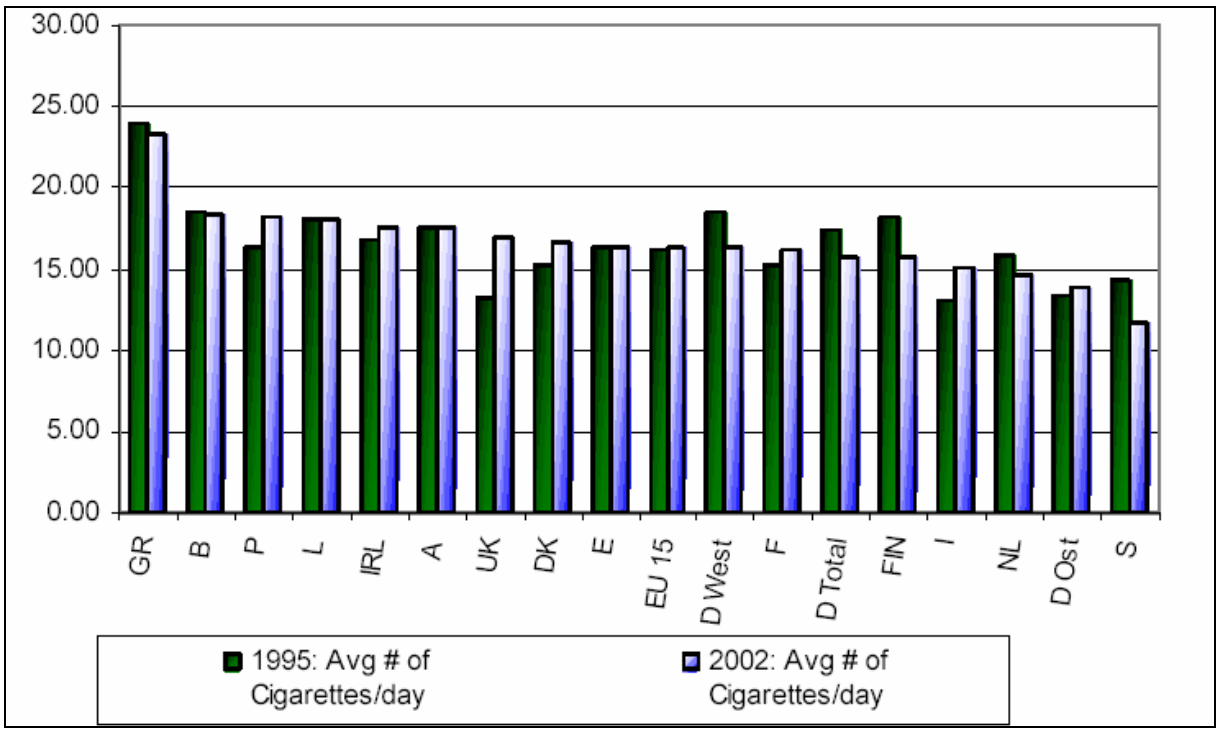


Figure 3: Daily "EU-15" cigarette consumption in the years 1995 and 2002 (EEIG, 2003)

Individual data on cigarette consumption were not available in this report, but only country-by-country averages. From these, median and 90th percentiles were derived (see below) for an assessment of the variation in cigarette consumption across Europe:

	Cigarettes per day
Min	11.8
Median	16.9
P90	18.6
Max	23.6

The following overall conclusions on smoking are drawn from the data and argumentation presented above:

(i) Daily cigarette consumption varies between 12-24 in the "EU 15" countries. Median (16.9) and 90th percentile (18.6) consumption rates across Europe are not that far apart that they would merit in distinguishing for this parameter.

(ii) The average prevalence of smokers in the EU is 39.4 %. It is therefore a borderline argumentation to designate non-smokers (60 %) vs. smokers (40 %) as "typical" of the EU population.

(iii) Compared to the EU average, this does not provide arguments in support of assuming a much higher smoking rate for workers: EEIG (2003) states that the top two groups in terms of smoking in 2002 were the unemployed (53.8 %) and manual workers (51.5 %). However, with respect to daily cigarette consumption (EU average: 16.35 cigarettes/day), the heaviest smokers are the self-employed with 18.42 cigarettes/day; manual workers/labourers were not separately identified.

2.4.2. Indirect exposure via the route of dermal contact

Exposure via dermal contact is discounted for in the indirect exposure section of the TGD because exposure through these routes is considered to be very unlikely. This is especially true for metals, where percutaneous transfer from water or soil may be considered to be negligible, considering the low ambient concentrations and the low dermal absorption rates for metals (for details, see separate fact sheet on dermal exposure and absorption). Noted as exceptions are extremely polluted soils e.g. land fill sites or accidental releases.

Nevertheless, as a result of possible hand-to-mouth transfer of contaminated soil, especially by children, the correct assessment of dermal exposure is relevant for the assessment of exposure via ingestion, which is addressed in detail in subchapter 2.4.3.1 below.

2.4.3. Indirect exposure via the route of oral intake

2.4.3.1. Ingestion of soil and dust

The TGD does not provide any sophisticated guidance on this, but instead states: "exposure via soil ingestion and dermal contact is not addressed in this guidance because they represent significant exposure routes only for specific situations of soil pollution".

However, soil ingestion by children via hand-to-mouth transfer is particularly relevant for metals, whereby not the entire fraction of a metal in soil may be considered as anthropogenic pollution in view of the ubiquitous nature of many metals.

Largely based on the experience gained in the VRA on lead and on copper, in which the relevance of this pathway has been dealt with comprehensively, the following tiered approach is suggested to assess indirect exposure via the environment from ingestion of soil/dust for children:

Tier I calculations:

Indirect exposure via ingestion of soil/dust can at a screening level be calculated based on the IEUBK model[1] default assumptions, which are summarised in the table overleaf and which in themselves are considered conservative (for reasons given below).

The IEUBK (Integrated Exposure and Uptake Biokinetic) Model was developed to predict children's exposure to lead, but the exposure defaults may also be considered applicable to other metal pollutants ingested via soil and dust.

Tier II calculations:

For the IEUBK model, the parameters "soil ingestion rate" and soil lead bioavailability have the greatest influence on the model predictions. However, by cross-validation of previous model assumptions for uptake of lead, it has been shown that these do not necessarily match with measured blood lead data, when applying the IEUBK defaults.

Lead may be seen as an ideal example for such a model exercise, since (i) children have higher GI uptakes of lead than adults, and (ii) detailed information is available for the bioavailability of lead from soil.

[1] Reference US EPA (2005). A detailed review of the IEUBK model may also be found in the separate HERAG fact sheet on "Gastrointestinal uptake and absorption and catalogue of toxicokinetic models".

Table: default IEUBK input parameters useful in assessing exposure of children

Time spent outdoors		**hours / day**
Age =	0-1 year (0-11 month)	1
	1-2 years (12-23 month)	2
	2-3 years (24-35 month)	3
	3-7 years (36-83 month)	4
Ventilation rate (given in m³/day)		**m³ / day**
Age =	0-1 year (0-11 month)	2
	1-2 years (12-23 month)	3
	2-3 years (24-35 month)	5
	3-4 years (36-47 month)	5
	4-5 years (48-59 month)	5
	5-6 years (60-71 month)	7
	6-7 years (72-84 month)	7
Drinking water ingestion rate		**liters / day**
Age =	0-1 year (0-11 month)	0.20
	1-2 years (12-23 month)	0.50
	2-3 years (24-35 month)	0.52
	3-4 years (36-47 month)	0.53
	4-5 years (48-59 month)	0.55
	5-6 years (60-71 month)	0.58
	6-7 years (72-84 month)	0.59
Percentage of total water intake *		**%**
	first draw water	50
	flushed water	100 minus first draw and fountain
	fountain water	15
Soil/dust ingestion		**g / day**
Age =	0-1 year (0-11 month)	0.085
	1-2 years (12-23 month)	0.135
	2-3 years (24-35 month)	0.135
	3-4 years (36-47 month)	0.135
	4-5 years (48-59 month)	0.100
	5-6 years (60-71 month)	0.090
	6-7 years (72-84 month)	0.085
* Applicable for metals present in water pipe material. First draw water, i.e. water standing in the pipe over night, might exhibit higher concentrations due to release of the metal by the pipe.		

As a result of the comparison of monitored data with model predictions, the IEUBK default soil ingestion rates had to be modified for lack of agreement with measured EU blood lead data, i.e. by reducing the soil ingestion rate considerably for specific (e.g. urban) exposure environments. These modifications to the default IEUBK default soil ingestion rates were not unexpected in that IEUBK was developed to predict lead exposure at specific, predominantly rural, contaminated sites in the United States. Problems can emerge when the model is then applied in other settings. For example, many EU urban environments contain limited areas of soil accessible to children. Thus, in the absence of bare soils, IEUBK soil ingestion estimates will likely be too high. Limited amounts of soil in urban environments in the United States, where IEUBK was developed and validated, are similarly suspected to result in model over-prediction of blood lead levels in urban settings. In the Voluntary Lead Risk Assessment, more accurate concordance between model predictions and observed blood leads was obtained when estimated levels of soil and dust ingestion by children in urban environments were reduced from 135 to 40 mg per day. Although the level of soil and dust ingestion assumed was significantly lower than that assumed by IEUBK, 40 mg was in fact close to the average ingestion rate suggested by more recent observational studies of faecal excretion by children with "soil tracer" substances (LDAI, 2005). Soil ingestion rates of 135 mg per day were retained for rural environments even though modest overestimation of blood lead levels was observed.

Model calculations were then made under "typical" and "worst case" exposure scenarios, and the predicted average blood lead level under each circumstance compared to observational blood lead studies. Model input parameters were judged to be adequate if predictions of average blood lead levels were within 20 % of those observed. Intraindividual variability in blood lead, resulting from dietary and behavioural variables, was also estimated via the application of a geometric standard

deviation to the central (median) estimate of the population blood lead average. General population blood lead distributions are typically log-normal in shape and can be predicted via the application of a geometric standard deviation that captures the cumulative impacts of the multiple factors believed to modify exposure and uptake. Thus, in addition to the average blood lead levels resulting under typical and worst-case exposure scenarios, estimates were also made of the upper 90th percentile of the blood lead distribution under each exposure scenario. Whenever possible, observed and modelled upper 90th percentile estimates were compared to validate model use.

Factors to consider in the derivation of concentrations in soil/dust

Concentrations of a particular metal are often modelled based on outdoor soil concentrations. However, two observations need to be considered for a refined assessment:

(i) distinct differences between outdoor and indoor dust levels are known to exist, and
(ii) similarly, there can be considerable margins between rural and urban soil/dust concentrations (example: early vs. late phase-out of lead in petrol)

Further, quality and relevance control of soil monitoring data should be documented when deriving outdoor dust concentrations from ambient soil monitoring data, since these may be influenced by historical (local) pollution, and should therefore not be considered for an assessment directed at current industrial practice. Finally, many EU data bases are oriented towards the ascertainment of metal levels in agricultural soils but residential soils are of greatest relevance to the exposure of children. Estimates of soil concentrations should therefore be based upon soils to which children would most likely be exposed.

If available, preference should be given to quality-screened analytical data on metal contents in household dust samples. In the absence of measured data, exposure assessment models such as IEUBK will estimate a contribution of metal to soil to the metal content of household dust. However, as with ingestion estimates for soil, the limited amount of soil present in many urban areas limits the accuracy of IEUBK default assumptions and predictions.

2.4.3.2. Ingestion of foods

The conventional partitioning-based TGD methodology for determining air-to-soil and soil-to-plant factors in the assessment of human uptake pathways may be considered as mostly inapplicable to metals.

Whereas model calculations may still be useful for the prediction of aerial deposition on vegetables, for example, an alternative approach to the use of "biotransfer factors" (BTFs) needs to be considered for metals, as summarised briefly below. The reasoning for this approach is basically provided by the findings of a few selected RAs on this subject:

VRA Lead

In the period 1975 – 2005, lead in food intake has declined considerably, largely as a result of phase-out of lead in petrol, but also of other food-related uses (such as banning the use of lead for soldering on food cans, and wine bottle seals).

Food groups that generally dominate the total dietary Pb intake are cereals, beverages and vegetables. Whereas Pb concentrations in fruit and vegetables may be influenced by food preparation (i.e., washing) and storage, they do not necessarily reflect uptake by plants. Experiments demonstrate that the Pb concentrations in agricultural food-crops are predominantly derived from airborne Pb, even in grains, and that translocation of soil Pb from roots to shoots is limited.

Concentrations of Pb are generally highest in kidneys and liver, crustaceans, molluscs and shellfish, egg whites, dried fruit, wild edible fungi, some oils, spices and wine. However, since these products are consumed only in small amounts in an average diet, they have not necessarily a large impact on total dietary Pb exposure.

Numerous dietary intake studies based on Pb levels in food and consumption patterns are available for lead, including market basket[2] studies, total diet[3] studies, and duplicate meal[4] studies. Further, faecal output studies of Pb have been used to estimate daily intake, assuming that about 10% of the intake is absorbed. Estimates based upon the latter method must, however, consider the extent to which biliary excretion pathways are used to eliminate existing body burden for a given metal (LDAI, 2005).

VRA Copper

In deviation from the TGD food basket approach, the VRA on copper also bases its assessment on duplicate diet and market basket studies. For vegetables, only modelled aerial deposition is considered, assuming 10 years of continuous deposition in addition to natural background and excluding any accumulation.

Aerial deposition was considered in detail because it was found to constitute the majority of the contaminant, to which leafy vegetables (i.e. lettuce) are particularly susceptible – any deposited metal contaminant may therefore hypothetically make its way into processed food via incomplete washing procedures. In consequence, lettuce may be a conservative surrogate for vegetables based on these considerations.

In contrast, uptake and translocation of copper into the edible parts of plants appeared to have been largely unrelated to soil copper levels, as particularly observed for plants grown under conditions of high metal soil concentrations.

Finally, attention is drawn to the quality of the conduct of such studies, since copper was found to be heavily concentrated in the non-edible parts of the plant (in particular roots) which are frequently contaminated with soil particles, so that incomplete washing procedures may have introduced bias.

EU RAR Cadmium

In uncontaminated areas, dietary Cd and smoking are the main pathways of Cd exposure.

Dietary data were analysed for each country, and ranges for mean values for dietary intake were taken forward to risk characterisation. It is particularly noteworthy that upper percentiles were not used in the indirect exposure assessment via food, since the corresponding extreme values - mainly reported in old studies - were not considered as being representative of a lifetime exposure; while for cadmium any risk is indeed related to cumulative dose. Moreover, more recent studies performed in countries with similar dietary habits failed to report similar values.

It is also worthwhile to note that foods differ in bioavailability of Cd and that some of the products containing elevated total Cd levels are lower in available Cd. As a result, elevated dietary intake due to preference for products like seafood or mushrooms may not necessarily induce a higher health risk for the consumers.

The relationship between soil Cd and dietary Cd intake was calculated using food consumption data and food Cd concentrations that are predicted from soil Cd concentrations and the appropriate soil-plant transfer factors (TFs). This calculation must consider the amount of dietary Cd that is impacted by the level of Cd in the soil.

Factors that affect crop Cd concentrations were examined in detail in the Cd RA. The crop Cd is mainly derived from soil, although atmospheric Cd can contaminate crops through direct interception by plants:

[2] In market basket studies, individual food items are sampled from retail outlets and are analysed.
[3] In total diet studies, food items are processed for consumption, subsequently analysed individually or in food groups, and intake is calculated as the product of the contaminant level in food and the amount consumed.
[4] In duplicate meal studies, duplicate samples of meals, snacks and beverages are collected and analysed.

The available evidence indicates that airborne Cd may be a significant source of Cd for crops grown in areas where atmospheric Cd is at least tenfold higher and where soil Cd is not high (or not available).There are, however, few studies that enable this contribution of airborne metal to be appropriately estimated. Overall, from the available data, it can be anticipated that the fraction airborne can be neglected in crops grown in contaminated soils and if the atmospheric Cd deposition is low (e.g. soils contaminated by high metal sludge).

The risk of increased soil Cd on elevated Cd concentrations in crops was assessed using appropriate slopes of the dose-response curves and derivation of TFs. Plant Cd concentrations increase linearly with increasing soil Cd if the Cd is added to the soil as a Cd^{2+} salt within the environmental range (up to ~2 mg/kg, see figure below). However, plant Cd increases slightly more than proportionally to soil Cd, meaning that the Cd added to the soil from diffuse sources (P-fertiliser, deposition, alluvial deposits) is somewhat more available than Cd present in the soil. This should be taken into consideration when TFs are derived for the risk assessment. The use of TFs of soils at background Cd concentrations (i.e. the slope of the proportional increase on the figure below) disregards the fact that recently added Cd may be more available to plants than Cd present in the soil. Since the annual addition rates from these diffuse sources are generally small (typically about 1 % of the amount present in soil), and as with the available data it was impossible to reconstruct exact dose-response curves from historic data for these diffuse sources, it was, however, hypothesised that the Cd risk assessment based on the actual Cd TFs in soils at background Cd levels was not underestimating plant Cd concentrations in soils contaminated with diffuse sources of Cd.

These Cd TFs should not be used, however, for assessing the risks of sludge-born Cd on food-chain contamination with Cd, as a curvilinear increase was often found. Increasing sludge levels in soil increase the metal absorption capacity of the soil; availability was therefore reduced compared with Cd added as Cd^{2+} salt. The Cd transfer factors should then preferably be calculated from paired observations of soil and plants.

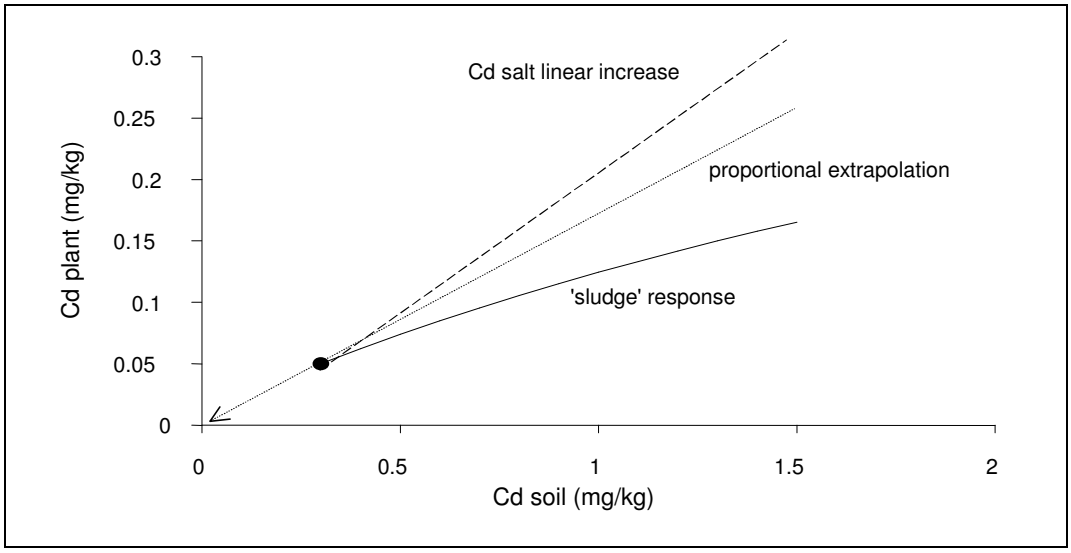


Figure 4: Schematic representation of the increase in plant Cd upon adding Cd to the soil as Cd salt or as sludge containing Cd. The proportional extrapolation is based on soil and plant Cd concentrations in soils with background Cd levels (black dot).

2.4.3.3. Ingestion of drinking water

Factors relevant for the assessment of metal leaching from water piping etc.

Apart from bottled (mineral) waters, tap waters are the most quantitatively relevant sources of Cu and Pb in drinking water, both resulting predominantly from the use of piping and fitting materials for plumbing. The following discriminators are essential to avoid a multiplication of worst-case exposure levels:

- Water composition: water hardness and other water constituents have a severe impact on the dissolution of metals form the inner linings into the water flow. The highest lead concentrations have been found with very "soft" waters, whereas for copper, various degrees of "aggressiveness" have been defined. In contrast, water hardness conveyed by carbonate concentrations is usually the modifier that restricts copper and lead concentrations to a minimum, plus pH and DOC content.
- Standing time: redox/dissolution and subsequent precipitation processes will yield concentrations that will build up to an equilibrium with time in stagnant waters, which will be cleared after flushing a tap for several seconds.

In reflection of this, several sampling procedures for drinking water exist that intrinsically will influence the measured concentration:

- *First draw* samples taken after a period of stagnation of the water (such as during the night).
- Random daytime samples taken during the day without knowledge of withdrawal frequency/incidence.
- Fully flushed samples in which a tap is flushed for several minutes before taking the sample.

Thus, recognition in the assessment of exposure via drinking water must be given to first draw samples (which may be more appropriate for acute effects assessments) versus composite sampling after extended flushing of a tap (which may be more appropriate for long-term, typical exposure assessment).

Default exposure values

It is noted that the TGD provides a default drinking water consumption value for adults of 2 L/day, whereas the US EPA (1997) states a mean adult consumption of 1.4 L/day. Since the TGD does not assign any age-specific drinking water consumption values for children and adolescents, the following levels may be taken from the latter source instead:

Drinking water consumption, default values		
Age [years]	**EU [L/day]**	**US EPA [L/day]****
0 - 3	*	**0.61**
3 - 5	*	**0.87**
1- 10	*	**0.74 [35 mL/kg/d]**
11 - 19	*	**0.97 [18.3 mL/kg/d]**
adults	**2.0**	**1.40 [21 mL/kg/d]**

* no EU value proposed; ** EPA Exposure Factors Handbook, Chapter 3, Table 3-30

2.4.3.4. Ingestion as a result from "pica" behaviour

Pica can be described as an abnormal appetite for some things that may be considered as foods or food ingredients (e.g., flour, raw potato, starch), but is of more concern when directed at eating dirt, clay, etc. Pica is seen in all ages, particularly in pregnant women and small children, especially among children who are developmentally disabled where it is the most common eating disorder. This behaviour occurs at a level which is considered developmentally inappropriate, and in its extreme forms is regarded as a medical disorder. For the latter reason, the existence of this phenomenon is addressed here, but its applicability for a large cross-section of any sub-population is questionable.

3. Consumer Exposure

3.1. Current principles for the assessment of consumer exposure

In the current version of the TGD (2003), a step-wise procedure for the assessment of consumer exposure to both existing and new substances is described in Part I, Chapter 2.3. The assessment of consumer exposure starts with an initial screening to determine if the substance under investigation is actually used as or in consumer products or whether the expected exposure is so low that it can be neglected in the risk characterisation phase. If the consumer exposure is not considered to be negligible, then a quantitative exposure assessment is desirable.

In the initial screening, the consumer product categories in which the substance of interest occurs have to be identified. The TGD (Part I, Appendix II, Section 2) provides a list of categories which can be used for this identification. In brief, the following consumer product categories are defined:

- cleaner / polish
- adhesives / sealant
- printing / writing material
- painting material and additives
- fuels
- bleaches / disinfectants / sterilizers
- removers
- photographical chemicals
- textile chemicals
- vehicle maintenance
- cosmetic / personal hygiene product
- contamination of food
- air contaminant / pollutant
- toy / joke / children's plaything
- other categories not mentioned otherwise

The TGD then presents simplified model approaches for the assessment of exposure via inhalation, dermal contact and oral intake. For a refinement of these assessments, it then proposes to look at the release mechanisms of a substance, as follows:

- evaporation from a liquid surface
- evaporation from a layer/coating
- contact of layer (liquid/semi-liquid/semi-solid) with body surface
- contact of skin with solid article
- migration from articles
- spraying
- contaminations
- solid particles in air

<u>Shortcomings of the TGD approach with regard to consumer exposure to metals/metal compounds</u>

In those cases where a liquid preparation may contain some form of inorganic metal compound, then any consumer exposure assessment is not likely to be different from the approach suggested in the TGD. However, where models are used as in chapter 4 of the TGD appendix, these are based on principles that are not likely to be helpful in any way for a quantitative consumer exposure assessment to metals and their inorganic compounds. Similarly, whereas chapter 5 of the TGD appendix gives useful information on anthropometric data, the information on time budgets for exposure and "typical" product composition data is limited to specific consumer products (e.g. laundry products and cosmetics). Only in rare cases are metals or metal compounds contained in such products (e.g. zinc oxide in sun cream, aluminum chlorohydrate in anti-perspirants). Therefore this data provided by the TGD has limited applicability for metal risk assessments.

Finally, direct contact with metal containing objects (e.g. jewellery) and the exposure resulting from the potential release from such objects are not addressed in the TGD at all.

3.2. Consumer exposure - previous experiences in EU risk assessments

The currently available risk assessments reports on metals and metal compounds were screened for consumer exposure issues which were deemed quantitatively relevant for human health risk assessment.

The consumer exposure scenarios that have previously been established for metals are summarised in the following overview table, sorted by route of exposure, and including an allocation of a relevance ranking in either "high", "moderate" or "negligible" exposure levels, according to their (current) assessment in their respective risk assessments.

More detailed extracts of the consumer exposure section of all available RARs are given in appendix A 2. It is suggested to refer to these extracts for examples for such consumer exposure scenarios.

Those cases which are metal-specific and have in the past been handled differently between the individual ESR-RARs and VRAs and required commenting are discussed in chapter 3.3 below.

Table: Consumer exposure scenarios that have previously been established for metals, including a relevance ranking in either “high”, “moderate” or “negligible” exposure levels as derived in the respective risk assessment report.

Consumer product / scenario	Substance	Relevance*
Route of exposure: inhalation		
Smoking (cigarettes)	Copper, Nickel (Cadmium)	2
Candles with Lead wick cores	Lead	2
Brazing operations with metal alloys	Cadmium metal / CdO	2
Use as pigment in household paint (dust)	Lead	2
Indoor use of ammunition (firing ranges)	Lead	2
Electrical soldering	Lead	3
Reloading of ammunition	Lead	3
Flame retardant synergist in textile fibre (dust formation)	Antimony trioxide	3
Route of exposure: dermal		
Jewellery	Nickel metal	1
Manipulation of metal objects (coins, tools etc.)	Nickel metal	2
Metal plating	Cadmium metal	2
Cosmetics and toiletries	Copper	2
Hair care products	Copper	2
Handling of coins	Copper	2
Jewellery	Copper	2
Jewellery (alloy constituent)	Cadmium metal	2
Hair dye	Lead	2
External use of Lead sheet (children, hand-to-mouth)	Lead	2
Garden ornamentals (children, hand-to-mouth)	Lead	2
Ni-Cd batteries	Cadmium / CdO	3
Alloys as Brazing Material	Cadmium metal / CdO	3
Sitting on flame retardant-treated upholstery fabrics	Antimony trioxide	3
Pigments (glass & enamels, plastics, artist’s paint)	Cadmium sulphide & Cadmium sulphoselenide	3
Stabiliser in polymers	Cadmium laurate & Cadmium stearate	3
Weights/sinkers	Lead	3
Artist Materials	Lead	3
Electrical soldering	Lead	3
Use of stabilisers in exterior PVC	Lead	3
External use of Lead sheet (run-off)	Lead	3
Decorative ceramics	Lead	3
Decorative crystal	Lead	3
Cosmetics: deodorants	organic Zinc compounds	3
Cosmetics: dandruff shampoo	organic Zinc compounds	3
Cosmetics: Eye shadow	Zinc distearate	3
Impregnating agents	Zinc naphthenate	3
Cosmetics: Sunscreen	Zinc oxide	3
Baby care ointment	Zinc oxide	3
Zinc oil for treatment of skin disorders	Zinc oxide	3
Paint (anti corrosive primer)	Zinc phosphate	3

Table (continued): exposure scenarios that have previously been established for metals, including a relevance ranking in either "high", "moderate" or "negligible" exposure levels as derived in the respective risk assessment report.

Route of exposure: oral		
Food contact materials	Nickel metal	1
Food packaging **	Tin metal **	2**
Water pipes	Lead	2
Crystal stemware, decanters	Lead	2
Pigment in household paint (dust ingestion)	Lead	2
Ceramics (glazes/pigments) (non-ISO / illicit imports)	Lead	2
Ceramics (glazes/pigments) (ISO compliant)	Lead	3
External use of Lead sheet (children, hand-to-mouth)	Lead	2
Garden ornamentals (children, hand-to-mouth)	Lead	2
Dietary supplements	Copper	2
	Zinc (most commonly as gluconate)	3
	Nickel sulphate	4
Drinking from a PET-bottle	Antimony trioxide	3
Sucking on upholstery fabric (children/infants)	Antimony trioxide	3
Hunting (shot/bullets)	Lead	2/3
External use of Lead sheet (adult exposure)	Lead	3
Crystal flatware,	Lead	3
Gargle / oral rinses	Zinc Chloride	3
Use of stabilisers in water piping materials (PVC)	Lead	3
Use of stabilisers in interior plastics	Lead	3
Route of exposure: other		
Contraceptive devices (intrauterine)	Copper	3
Eye drops (absorption through the eye)	Zinc sulphate	3
orthodontic appliances, surgical implants and other non-dental medical devices (Iatrogenic / systemic)	Nickel	4
Further Lead scenarios with negligible consumer exposure These scenarios are mentioned in the consumer exposure section in the Lead VRA but (i) are not expected to result in exposures of concern as a result of consumer use in the intended fashion, or (ii) from which there is no actual route of exposure (oral/dermal/inhalation) for the consumer.		
Lead acid batteries	Lead	3
Other automotive batteries	Lead	3
Consumer electronics	Lead	3
Cable sheathing	Lead	3
Plastic stabilisers in wires/cables	Lead	3
Pigment in paint (commercial use)	Lead	3
Radiation shield	Lead	3

* Relevance rating based on the quantitative exposure assessment in the respective ESR-RAR or VRA:
(1) high (2) moderate (3) negligible exposure (4) not available / not given
** Tin has not yet been the subject of an official ESR RA or VRA. This item was derived from information made available by industry (see also appendix A 2.7 of this fact sheet). Note:
(1) a ranking of "high" was assigned if the exposure already from this single source gave rise to concern; "negligible" was taken from the assessment in the respective RA documents; a ranking with "moderate" was assigned if this source of exposure was not deemed negligible in the respective RA, but would on its own not constitute a cause for concern. It is explicitly noted in this context that generally applicable criteria for assigning a particular source of exposure as "negligible" do not exist.
(2) Entries in this table concerning Lead were taken directly from the Lead VRA. There, a typical- and a worst-case exposure estimate is listed for each scenario, with the typical exposure being "negligible" for all scenarios. In cases where the worst-case estimate is rated negligible as well, the rating (3) is given here. In cases, where a worst-case exposure estimate was derived as a concrete numerical value in the Lead RAR, the rating (2) is given here. A rating (2) was also given here, when "inadequate" was listed in the worst-case column in the original table in the Lead RAR. There, a specification of "inadequate" is an indication that insufficient information was available to make a quantitative evaluation of worst-case exposure potential but that some level of exposure might plausibly be postulated to occur during consumer use.

3.3. Consumer exposure - metal specific issues and scenarios

Generally, the compilation of consumer exposure scenarios as presented in appendix A 2 may be useful when assessing another metal or another inorganic compound of any particular metal, and it is suggested to explore use of these examples in analogy, where appropriate.

It is explicitly noted here that none of the possible routes of exposure should be excluded or deemed irrelevant from the onset in a future assessment of any metal or metal compound. Each conceivable scenario should be assessed quantitatively.

In this chapter, the consumer exposure issues of common interest and with relevance to metals are addressed sorted by route of exposure with a reflection of discrepancies between various metal risk assessments. Where possible, suggestions have been made how to deal with these discrepancies.

3.3.1. Consumer exposure via the route of inhalation

3.3.1.1. Smoking

Smoking can contribute to a relevant extent to overall systemic exposure to metals. Although smoking is usually a deliberate choice made by an individual and might therefore be considered a "consumer exposure" issue, the fact that it entails an inhalation exposure to metals is again something that is beyond the influence of an individual. This is why it is considered more appropriate to address this exposure scenario under "indirect exposure via the environment" (see above).

3.3.1.2. Use of metal compounds as pigments in household paints

This issue is currently focused on lead pigments, but in future may become an issue of general relevance for other metals. Consumers may be exposed primarily from paint rub-off in household dust via inhalation, but for children hand-to-mouth transfer is similarly relevant.

Considering that this particular use has been phased out for lead, it has been suggested to better assess this source of exposure under "indirect exposure via the environment", included within "house-dust".

3.3.2. Consumer exposure via the route of dermal contact

3.3.2.1. NiCd batteries

It is merely noted here that this aspect is dealt with in the cadmium RAR under consumer exposure ("… it can be concluded that consumer exposure to cadmium(oxide) from batteries is non-existent or negligible"). The respective section of the Nickel RAR does not assess this.

3.3.2.2. Dermal exposure from manipulation of metal/alloy objects

Consumer exposure from handling of metal objects or items, as well as exposure through jewellery is assessed with widely differing concepts in the various RARs and VRAs. It was noted that often (seemingly unsupported) model assumptions on release from metal/alloy surfaces were made.

The following step-wise approach may be taken for future consumer exposure assessment in such cases:

(i) For initial screening purposes, default assumptions as given in the various examples reviewed in appendix A 2 of this fact sheet may be used.

(ii) For a more refined assessment, the discussion within the HERAG group yielded the recommendation to give preference to measured release rates in physiological media (like artificial sweat, for example) of metal ions from a particular object or alloy product (for a detailed example of such testing (cobalt) refer to Stopford et al. 2003).

3.3.2.3. Dermal transfer rates for transformation products layers

Several metals are known to rapidly form an oxide layer at their surface which subsequently may turn into a form of "patina"", most commonly as a consequence of reaction to hydroxide and carbonate forms.

Such layers may be prone to "rub-off"" during dermal contact, and thus represent not only a source of dermal exposure relevant to adults, but particularly also for oral exposure of children with respect to potential "hand-to-mouth" transfer.

In those cases, it is essential to conduct dermal transfer monitoring experiments (as currently being undertaken by the lead industry, for example).

3.3.3. Consumer exposure via the route of oral intake

3.3.3.1. Food-packaging and food contact

Specific EU legislation for food contact materials exists. An overview of this legislation, as well as addresses of national authorities and further (guidance) documents are available at the website of the European Union: *http://europa.eu.int/comm/food/food/chemicalsafety/foodcontact/documents_en.htm*.

In general, where leaching standards for a metal ion from a particular material have been established, these may be used in a first screening step for consumer exposure via ingestion, realising that these constitute performance criteria for marketed products, and therefore may result in overestimates for actual exposure.

However, where for example food market surveys exist (example: leaching of antimony from PET bottles into soft drinks, see appendix A 2.1), preference should be given to such measured data for the assessment of consumer exposure.

3.3.3.2. Dietary supplements

Dietary supplements containing not only vitamins etc. but also mineral trace elements are widely available in drugstores and/or pharmacies. The consumer use of these products may occur under poorly controlled circumstances.

Despite the fact that this use may also be seen as a pharmaceutical one, the general availability of dietary supplemental products suggests that the assessment under the aspect of consumer exposure is nevertheless warranted.

The composition of market products and the frequency of use in the general population must be established.

3.3.3.3. Exposure through drinking water (water pipes)

This aspect is of particular relevance for copper (present major use), but also for lead (as a now largely historical application).

However, it is noted that in the VRA on lead, this issue is addressed both under consumer exposure (but with negligible exposure to the typical consumer for reasons of phase-out) and indirect exposure via drinking water, whereas the copper VRA assesses it (in line with the TGD) only under "indirect exposure".

4. Summary and conclusions

4.1. Indirect exposure via the environment

General recommendations:

The assessment of indirect exposure to humans via the environment employs a considerable overlap with the assessment of possible risk to environmental ecosystems. Therefore reference is made at this point to the MERAG project. This project aims to provide the regulatory community at regional and international level with scientific and regulatory guidance on the most advanced status of environmental risk assessment concepts for metals and inorganic metal compounds. Especially the concepts of bioavailability, bioconcentration, bioaccumulation and biomagnification of metals and/or metal compounds are also applicable when assessing the possible exposure of humans via the environment, e.g. via ingestion of food. The scientific and technical recommendations by MERAG can be found on the internet at http://www.euras.be/eng/project.asp?ProjectId=67 .

A further general conclusion is that the model approaches suggested by the TGD which are based on partition equilibria are applicable to metals only to a very limited degree. Depending on the level of data available, some metals may be in the position to assess indirect exposure at a rather sophisticated level (i.e. using biomonitoring data) as compared to basic modelling of concentrations in air, water and food (see section 2.2.).

Where relevant, the assessment of indirect exposure via the environment should be performed separately for susceptible or particularly sensitive subpopulations (for an example, please refer to the VRA Lead, where exposure of children is addressed in detail).

The general use of 90th percentiles of quantitative exposure measures as in the TGD is likely not applicable to metals, because a multiplication of such values for concentration in environmental media and their intake rates may result in overestimates of internal exposure. However, the use of RWC values may under certain circumstances allow a consideration to be given to (potentially sensitive) known subpopulations.

Most key metals are ubiquitous in the environment: Reflection must therefore be given to a correct distinction between natural ambient and anthropogenic concentrations. This will be of particular relevance when CMR labelled metals which are ubiquitous in the environment are discussed for risk through indirect exposure because of the conflict with the "no threshold concept"" for genotoxicity and carcinogenicity.

Relevant compartment-specific conclusions:

- For the assessment of indirect exposure via air (inhalation), previous and current metal risk assessments have not deviated substantially from the approach of the TGD, for lack of plausible alternatives.

- For the assessment through the intake of drinking water, a set of definitions and defaults have been proposed that may allow a more consistent approach of metals for which this is a relevant route of exposure. In addition, default (age-dependant) uptake factors beyond the abbreviated approach in the TGD are proposed for drinking water.

- Uptake from soil is a pathway particularly relevant for children, for which default and conservative uptake factors from the US EPA IEUBK model (previously not considered in the TGD) are proposed. A more refined assessment scheme (example: lead) based upon toxicokinetic modelling and comparison with biological monitoring data is presented in subchapter 2.4.3.1.

When attempting to use soil monitoring data, care should be exercised to avoid any bias introduced by historical soil pollution (out of scope) and by a potential over-representation of "hot spots".
In the validation of the assessment with measured data, variations of ambient concentrations in time need to be carefully considered (example: phasing-out of lead in petrol).

- Ingestion via food is a key pathway for indirect exposure to metals via the environment, for which also some specific conclusions were derived:

    - The tools of market basket, total diet, duplicate diet and excretion analysis for the assessment of intake via food are not described in the TGD; these have the distinct advantage of being derived from measured values instead of hypothetical considerations based on plant uptake factors.

    - Where known, metal-metal interactions need to be addressed, since nutritional (metal) status or food composition with respect to metal content has been shown to influence uptake (examples: calcium as a modulator of lead absorption, zinc vs. copper interactions, etc.).

<u>General recommendations with respect to repeated dose toxicity:</u>

Where possible, the assessment should be conducted at the principally most accurate level, where the actual internal dose (biomarkers, e.g. urinary cadmium, blood lead) of individuals is compared with the lowest observe adverse effect level (LOAEL), preferably in a specific target organ. Where such biomarkers of exposure are not available, data of external exposure (intake, air concentrations) need to be considered.

4.2. Consumer exposure

Although in general, the assessment of consumer exposure is not strictly regarded as a metal-specific issue, several individual product types or consumer exposure scenarios include issues particularly relevant for metals and/or metal compounds, and upon discussion within the HERAG project were not considered adequately covered by the current approach suggested by the TGD.

This fact sheet therefore summarises the experience gained by industry in this field during previously undertaken risk assessments under the existing substances regulation (ESR), or within "voluntary risk assessments".

A tabular summary of consumer exposure scenarios potentially relevant for metals is given, as well as a discussion of aspects of common interest for which discrepancies between various previous metal risk assessments have been observed.

In the appendix to this fact sheet, more extensive summaries of previously undertaken consumer exposure assessments - partly still in draft status - are given as a background document, from which more detailed information may be extracted as guidance for future assessments.

The following general guidance for the future assessment of consumer exposure to metals and metal compounds can be summarised from this fact sheet:

- It is recommended to consider every conceivable product or scenario in detail, and to conduct a quantitative exposure assessment for each; in other words, the *a priori* exclusion of a particular scenario, or the designation of lack of relevance to any potential route of exposure (inhalation/dermal/oral) should be avoided.

- As with "indirect exposure" the assessment of sub-populations might be applicable also for certain consumer articles/scenarios.

- The ConsExpo model is considered not applicable for metals and metal compounds.
- Experience in previous risk assessments for metals has shown that the designation of a particular scenario as "negligible" inevitably opens a discussion on the "cut-off" value for this, since the various existing routes of exposure may cumulate to critical total body burdens.
- For the assessment of exposure via dermal contact, a tiered approach should be followed:
    - Default assumptions for release rates as used in some of the examples in the appendix should be used for initial screening purposes only.
    - For a refined assessment, it is recommended to rely on release rates of the metal from the product, to be experimentally determined in appropriate physiological media.
- The compilation of examples given in this fact sheet is not exhaustive, and therefore other product types or consumer exposure scenarios may need consideration for other metals.

5. References and abbreviations

References

B (2004)	EU RARs on Cadmium and Cadmium oxide, final drafts September 2004.
DK (2005)	EU RARs Nickel and Nickel sulphate, final drafts June 2005.
EC (1998c)	Directive 98/79/EC of the European Parliament and of the Council of 27 October 1998 on in vitro diagnostic medical devices. OJ. L331, 7.12.1998, p. 1 - 37
ECB (2004)	EU RARS on Zinc and Zinc compounds, final reports 2004.
ECI (2005)	VRA on Copper and Copper compounds, first draft report May 2005.
EEC (1990a):	Council Directive 90/385/EEC of 20 June 1990 on the approximation of the laws of the Member States relating to active implantable medical devices. OJ. L189, 20.07.1990, p. 17 – 36. Amended by EEC, 1993c, 1993d.
EEC (1993c)	Council Regulation (EEC) 793/93 of 23 March 1993 on the evaluation and control of the risks of existing substances. OJ. L84, 5.4.1993, p.1 -75.
EEC (1993d)	Council Directive 93/42/EEC of 14 June 1993 concerning medical devices. O.J. L 169, 12.07.1993 p. 1 – 43.
EEIG (2003)	Smoking and the Environment: Actions and Attitudes, Directorate General Health and Consumer Protection, European Commission, Special Eurobarometer 183 / Wave 58.2, European Opinion Research Group EEIG, 2003.
Giskes et al. (2005)	Trends in smoking behaviour between 1985 and 2000 in nine European countries by education, *J. Epidemiol. Community Health* 2005 (59), 395-401
LDAI (2005)	Voluntary risk assessment report on lead and lead compounds, first draft report May 2005.
Stopford et al. (2003)	Bioaccessibility testing of cobalt compounds, *J. Environ. Monit.* 5, 675-680, 2003.
TGD (2003)	European Commission: Technical Guidance Document in support of Commission Directive 93/67/EEC on Risk Assessment for new notified substances, Commission Regulation (EC) No 1488/94 on Risk Assessment for existing substances and Directive 98/8/EC of the European Parliament and of the Council concerning the placing of biocidal products on the market (2nd Edition, 2003).
US EPA (1997)	Exposure Factors Handbook, US Environmental Protection Agency, National Center for Environmental Assessment, Office of Research and Development:, 1997 available at http://cfpub.epa.gov/ncea/cfm/efprog.cfm
US EPA (2005)	Integrated Exposure Uptake Biokinetic Model for Lead in Children, Current Windows® version dated September 2005: IEUBKwin v1.0 build 262. Information and download available at http://www.epa.gov/superfund/programs/lead/
Wasserman et al. (1994)	Consequences of lead exposure and iron supplementation on childhood development at age 4 years. *Neurotox. Terat.* 16, 233-240, 1994.
WHO (2005)	Tobacco Information and Prevention Source (TIPS), Country profiles by region, all EU-15 countries, available at: (http://www.cdc.gov/tobacco/who)

Abbreviations

BAF	bioaccumulation factor
BCF	bioconcentration factor
CMR	carcinogenic, mutagenic and reprotoxic (substances)
ESR	Existing Substances Regulation, EC Council Regulation (EEC) No 793/93 of 23 March 1993 on the evaluation and control of the risks of existing substances.
GI	gastrointestinal
LOAEL	lowest observed adverse effect concentration
PET	Poly-Ethyleneterephthalate
RAR	Risk Assessment Report
RIVM	National Institute for Public Health and the Environment (Rijksinstituut voor Volksgezondheid en Milieu), The Netherlands.
RWC	reasonable/realistic worst case
TDI	tolerable daily intake
TGD	Technical Guidance Document (see also reference TGD, 2003)
VRA	Voluntary Risk Assessment (Report)
WHO	World Health Organization

A 1: Background information on "Indirect exposure via the environment"

In the following subchapter the way in which indirect exposure via the environment was/is assessed in previous or ongoing risk assessments for metals is presented. These sections vary in extent and detail according to the input received from industry and to the state of the respective RAR or VRAR, some of which were in an early draft stage at the time of compilation of this fact sheet.

A 1.1: Copper

In the VRA on copper, the following sources of exposure were evaluated at a local scale: inhalation and diet.

Inhalation exposure - local scale

Limited measured data of relevance are available for airborne copper levels as measurements need to be made outside the borders of a production site, but within a distance of 1 km from the boundary of the company premises.

In general, no details were available on methods of sampling and analysis, nor on quality control procedures employed.

Airborne levels were calculated with EUSES, and based on copper levels measured in stack emissions.[5] In case of outliers, the use of air pollution control equipment (APCE) was checked.
Older data was treated with care, since emissions have been reduced considerably over the last 20 years due to stricter environmental regulations.

Calculated values do not include fugitive emissions. With highly efficient air pollution control equipment, fugitive emissions may form the major component of total emissions and calculated values could be underestimates.

In case of copper, the limited comparison possible for measured and calculated data did however not reflect this hypothesis with for a certain case a median calculated value up to approximately four times the measured concentrations. However, this might also have reflected the uncertainties in the measured data.
Typical and RWC inhalation exposures are calculated using the added copper concentrations with the addition of the regional background and the default inhalation volume of 20 m^3/day. These values do not incorporate the effect of fugitive emissions. In order to make an allowance for fugitive emissions, the two highest maximum values were taken forward to risk characterisation as reasonably conservative estimates of RWC and typical inhalation exposure respectively.
No measured data for particle size distribution are available. It was assumed from the use of fabric filters that only very fine particulate matter is emitted by stacks from these plants. It was however noted that the particle size distribution in the vicinity of the plants may be heavily dependent on the fugitive emissions component.

5 The exposure concentration in air is performed ? on the basis of the available dataset i.e. the emission data presently available from industry. Generic scenarios -based on calculated emissions from sector-specific maximum emission factors and production data- were applied to the sites for which no emissions are available.

Concentrations in air were calculated as the ‘added’ yearly average air concentration at 100 m from point source. Concentrations were assessed from reported data on total point emission to air, number of days that emission takes place, total fugitive emission to air, daily point emission to air

Dietary exposure – local scale

Estimates of dietary exposure are based on air to soil and soil to plant transfer. This procedure was conducted according to the EUSES methodology, whereby 10 years continuous deposition at current emission levels is added to the natural background level in soil. Where relevant measured deposition data were available these were used. Otherwise calculated deposition values were used.

Where measured deposition data were available, these were also compared with calculated deposition data to verify whether measured values –including fugitive emissions- would be higher than calculated values (based on stack emissions).
Local monitoring data of copper in soils were not used as they are influenced by historical pollution. Historical pollution is outside the scope of the RA.
Typical and RWC soil concentrations are calculated using the added copper concentrations with the addition of the background for "natural" soil to give a total local soil concentration. This forms the basis of the estimation of soil to plant transfer.
Estimates for indirect exposure are restricted to the consumption of fruit and vegetables grown in the vicinity of the industry sectors considered. Other foodstuffs are assumed to be sourced regionally. This is consistent with the approach adopted in other risk assessments (e.g. Zinc, Cadmium) but not consistent with the TGD.
To assess the accumulation of copper in food the BCF & BAF method as defined in the TGD was not used as it is not applicable to metals & inorganic metalcompounds. (This is well explained in MERAG. This issue should just be flagged in HERAG and reference should then be made to MERAG for more information.).
A key point for the estimation of copper accumulation in plants is the fact that copper in the edible parts of plants is relatively insensitive to soil copper, due to the homeostatic regulation mechanism. Deposition on the other hand could contribute substantially to copper in the edible parts of plants, especially leafy vegetables.
Experimental data indicated that lettuce is the plant accumulating most copper in the edible parts. Lettuce was therefore taken as a conservative surrogate for all fruit and vegetable consumption.
Typical and RWC values for added copper in lettuce were derived from experimental data at deposition rates representative of current local deposition rates.

Drinking water and ingestion of dust

Typical and RWC estimates were made for the regional scale, since these estimates were considered to be more conservative with respect to risk.

Regional scale assessment

The following sources were evaluated at regional scale: inhalation, dust (children), diet, drinking water (acute and chronic):

The methodology for the regional exposure assessment uses a database compiled from the published literature. There are three reasons for this approach. Firstly, the TGD describes methods exclusively applicable to organic compounds. Secondly, these methods apply to xenobiotic substances and are not applicable to essential trace elements without modification, specifically in respect of essentiality. Thirdly, as an essential trace element, there is clearly more substantial literature on exposure to copper than envisaged in the TGD.

Three databases were searched at the British Library in London: 1) the Library's Environmental Abstracts database, 2) Medline and 3) Aqualine, a database dealing exclusively with water-related research. Other authoritative sources such as the ICA database and key publications by the WHO (IPCS) were also consulted.
The typical case for both deficiency and excess is defined as the median exposure and the reasonable worst case as the 10^{th} (10P-RWC) and 90^{th} percentile (90P-RWC) respectively. This convention is followed as far as possible.

Exposure via air

Monitoring data for copper in air is rather sparse. Some outdoor data were available for urban and rural areas in the UK. Also indoor data were available for the UK.
Inhalation exposure however constitutes a quantitatively insignificant route, and the overall regional exposure assessment is considered to be insensitive to error in inhalation exposure. Exposure through inhalation was calculated using a default inhalation volume of 20 m3/day and a conservative exposure level of 100 ng/m3. This approach yielded an inhalatory intake some 600 times lower than from dietary exposure.

Ingestion of dust by children

Young children also incur exposure through ingestion of dust by hand to mouth contact. This exposure may consist of a mixture of house dust and garden dust.
Since copper levels in house dust have been shown to be consistently higher than those in garden dust, house dust levels are used for the exposure estimate. Data were available from several cities in the UK and rural and urban areas in Germany. Typical and RWC values were derived from urban household dusts where metal concentrations in dust are considered to be higher. Exposure estimates are calculated for children of age 0-1 yrs, 1-4 yrs, 6-7 yrs, 7-12 yrs, 12 yr old and using the IEUBK dust ingestion data for the different age categories.

Exposure via food

This assessment is based on a large amount of publicly available data, including duplicate diet studies and market basket studies. The TGD approach assessing exposure through modelling was not used. The food basket in the published version of the TGD for example, comprises only “raw materials” (meat, fish, dairy produce and fruit and vegetables). It does not include grain products, prepared foods, snacks or confectionary which are increasingly important components of diets in developed countries and some of which are significant sources of copper.
Because of the large amount of data, it was also possible to make estimates for different age categories.

Consumption of alcoholic beverages

As copper is used as a fungicide in vineyards an assessment was also made of copper exposure through consumption of alcoholic beverages. A limited amount of data was collected on Cu in wines in Italy and Greece.
Average annual alcohol consumption for the EU in 2001 is given in the WHO alcohol control database as 9.2 litres of pure alcohol per person (www.euro.who.int/alcoholdrugs/policy/20020611). Information on the distribution is not available. Given an alcohol content of 12% for wine, this corresponds to an annual wine consumption of approximately 77 litres or 0.21 litres/day, assuming that alcohol intake is entirely from wine.
Typical daily copper intake from alcoholic beverages (calculated as wine) was found to be minor.

Exposure via drinking water

A separate analysis was made of exposure to copper through drinking water as copper pipes are widely used in residential and commercial water distribution systems and the diet studies do not always specify whether beverages are included.
Concentrations should be measured at the tap to include both components of exposure, i.e. input and leaching.
Acute effects have been linked to short term exposures to corrosive drinking water highly enriched in copper. Estimates were therefore made for both acute and chronic exposure but different methods were used. For acute exposure, the highest copper concentrations are found in first-draw water following overnight standing in the pipes. It was therefore assumed that consumption of first-draw water represents the most significant exposure in respect of acute effects. Acute exposure is

estimated by taking a realistic sample volume of 200-400 mls corresponding to 1-2 mugs to represent a typical early morning beverage intake. This consumption is treated as a bolus dose.
In contrast, composite sampling representative of daily consumption is the most appropriate indicator in respect of chronic effects.
On the basis of the available relevant data, four categories of exposure from drinking water were identified: consumption of bottled water -which effectively contributes minimal added exposure; consumption of "typical" tap water, applicable to most EU citizens, in which low levels of copper are present, and consumption of "moderately corrosive" or "corrosive" water applicable to certain geographical sub-populations.

A 1.2: Nickel

A 1.2 Nickel

During the finalisation of this fact sheet in August 2007, the indirect exposures assessment part of the EU RAR nickel was available as a draft (dated 2007-06-28), and it was kindly made available to the authors of this HERAG fact sheet. The rapporteur member state (Denmark), together with industry, circulated this draft to the other member states' competent authorities and further discussion is foreseen at TCNES III in September 2007. Therefore, the reader should be aware of the fact that some of the issues presented here might change in the later and final versions of the RAR. Because of the draft status, only in some cases individual calculations and results are presented here.

In contrast, the following is a combination of direct extracts from the draft RAR (status 2007-06-28) with comments by the authors of this HERAG fact sheet. This form of presentation has been chosen to facilitate the use of this fact sheet annex as guidance for further assessments by better understanding the reasoning and argumentation used in this assessment.

The indirect exposure assessment in the nickel RAR is conducted in a very systematic, well-structure way. The assessment is basically divided into two parts: (i) assessment of external exposure and (ii) assessment of the resulting internal exposure using absorption factors. It is currently foreseen to include the latter in the risk characterisation section, which is not the key topic of this fact sheet. Therefore this issue is only briefly considered below.

The assessment of external exposures if further subdivided as follows and this synopsis adapts the original structure of the current draft RAR documents:

Assessment of external exposure
- Local Scale – Ni concentrations in environmental compartments
- Local Scale – calculation of external doses
- Regional Scale – Ni concentrations in environmental compartments
- Regional Scale – calculation of external doses

Assessment of internal exposure (later to be included in the risk characterisation)

Assessment of external exposure

Introduction

Indirect exposure of humans to nickel may principally occur via the typical exposure pathways
1) inhalation of ambient or indoor air,
2) ingestion of food and water,
3) ingestion of dust/soil, and
4) dermal contact with subsequent percutaneous absorption

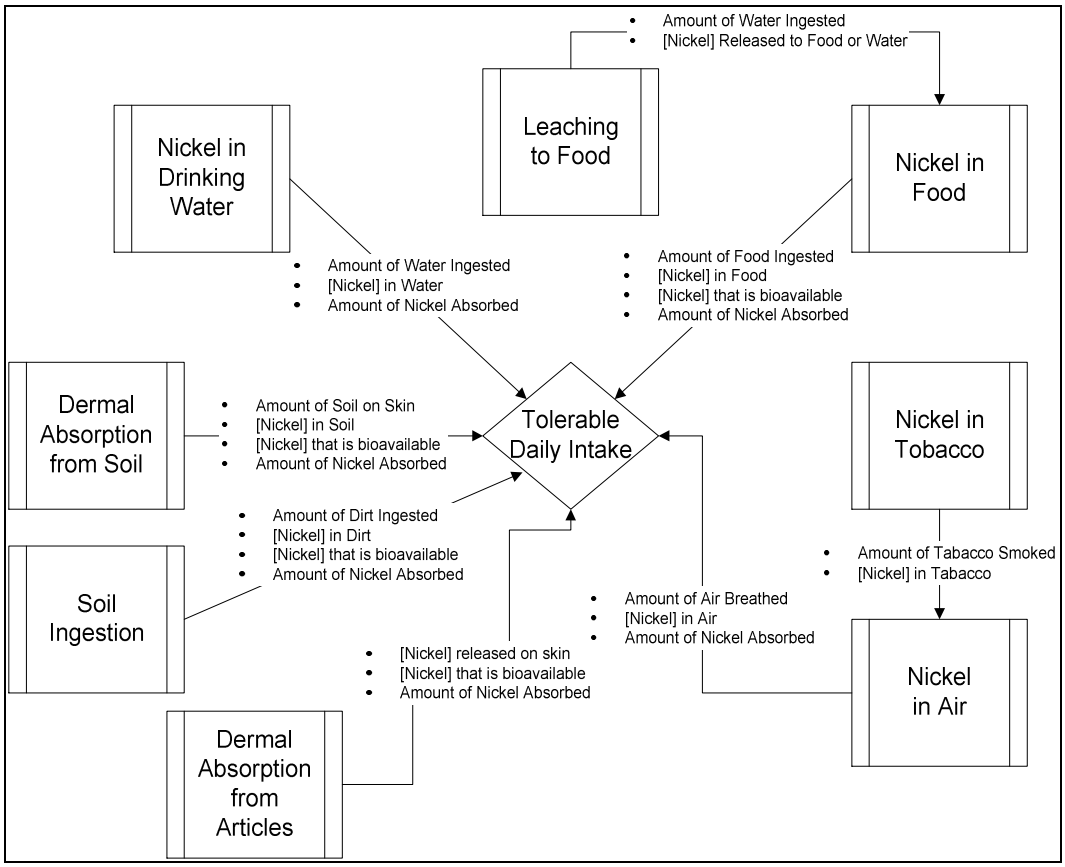


Figure: Schematic overview of different Ni exposure pathways to humans

Consequently, frequent reference is made to the Ni RAR environmental exposure section, since human exposure to nickel is directly related to the concentration in the environmental compartments air, soil and water; intake via food is in turn related to Ni in locally grown vegetables and fruits. Uptake of nickel from soil and deposition of Ni from ambient air contribute to the Ni content of locally grown vegetables and fruits. The assessment of indirect exposure for nickel is carried out separately for the local scale and the regional scale. The local risk assessment addresses exposure for populations in close proximity to production sites for Ni (producing sectors) and Ni using sectors, while the regional scale addresses the exposure for the remainder 'background' of the EU.

For each route of intake, a typical scenario and a reasonable worst case scenario (RWC) are assessed. The typical scenario reflects the median, or in other words the 'typical' exposure, while the RWC scenario represents the exposure at the higher end (95th percentile) of the population distribution. Median (50th percentile) concentrations of nickel in the environmental compartments, and median dietary intake data serve as input for the typical scenario, while P95 concentrations in soil, water, air and P95 dietary intake are used as inputs for the RWC scenario. For soil and deposition concentrations of the local scales, the input from the environmental exposure assessment is in terms of 90th percentile and not 95th percentile. In these cases, the 90th concentrations are used for the calculation of indirect exposure for the RWC scenario. These P90 values are used for situations pertaining to environmental exposure (e.g., deposition). For quantifying the exposure to humans, emphasis is placed on the use of P95 exposures for the RWC scenario.

Exposure is generally calculated for the adult population. However, soil ingestion is generally considered negligible for adults and is thus discounted, whereas it is a potentially significant pathway for children. For that reason, different exposure levels are calculated for both adults and children. Intake rates for children are – as a starting point - taken from the US-EPA exposure factors handbooks.

Dermal contact to environmental Ni is not considered to be critical at relevant environmental concentrations covered by this report, since the high concentrations needed to elicit a dermal allergic reaction would only be relevant in extreme cases (e.g., severe soil pollution in connection with point

sources). Nickel exposure by dermal absorption from articles is part of the consumer exposure section of the RAR.

Smoking: Exposure via smoking is considered an unintentional route of exposure to nickel. Because exposures to cigarette smoke contain a great variety of different substances, the risks associated with nickel content or cigarettes are better assessed by considering the process of smoking. Nevertheless, for information purposes, the possible contribution of Ni from smoking is included as an annex to the indirect exposure assessment. Because of the draft status of the indirect exposure section of the RAR, details are not further presented here.

Local scale - concentrations in environmental compartments

Local scale – nickel deposition and air concentrations

Ni air concentrations (typical = median value and RWC = P90 value) for the different Ni producing sectors (smelters and refineries) and Ni using sectors (stainless steel, multiple steel, Ni alloy, steel production, chemicals, catalysts, plating, metal products, batteries, powder metallurgy and recycling) were modelled in the ENV section of the RAR based on site-specific information of the different sites of the sectors. It is noted that P90 and not P95 concentrations were used for RWC scenarios in the environmental exposure section. These P90 data are also used here as input for situations pertaining to environmental exposure.

Local air and deposition data were modelled with the aid of the EUSES model. The EUSES model was also applied to calculate the additional Ni soil concentration due to the local Ni activities (PEC local soil). The site-specific additional Ni soil concentration was added to the (country–specific) regional natural soil concentration, and the resulting sum represents the local total soil concentrations (PEC total local soil).

The typical (=median) and RWC (=90th percentile) local deposition data and Ni concentrations in air by sector were then taken forward to the indirect local exposure assessment through locally grown vegetables and fruits.

Local scale – street and house dust concentrations

The inventory of nickel in street and house serves as input for the soil/dust ingestion of nickel. Especially for small children, this pathway is relevant.

Nickel concentrations in street and house dust are not available from the EUSES calculations. In order to obtain data for Ni dust in the local assessment, a literature search was performed. No information for Ni in house dust for dwellings in the vicinity of Ni producing/consuming industries was found. Moreover, the suitability of measured data in house dust would also be questionable because of the impact of historical emissions on house dust concentrations. Instead, data from the regional assessment are used (see below). The typical and RWC Ni concentrations for the regional assessment are already rather conservative, and can therefore most likely be used for the local assessment. This is a more conservative approach than substituting local dust concentrations by local soil concentrations since the latter are lower. In the study of Berghmans et al. (2006), house dust Ni concentrations and soil concentrations were unrelated. Therefore applying a house dust/soil concentration ratio factor to extrapolate the local soil Ni concentrations to local indoor dust Ni concentrations is not considered an option.

Local scale – nickel drinking water concentrations

It is considered unlikely that people living in the neighbourhood of a Ni production or Ni use plant (i.e. in local scale range) consume other drinking water than water from the regional supply, if they are connected to a public drinking water system (in analogy to the copper EU RA). Therefore, the Ni concentrations from the regional assessment are used. For the regional scale, a general EU wide

scenario for tap water, and an additional scenario for groundwater is elaborated based on Danish groundwater data on nickel (see below).

Local scale – nickel concentrations in food

This chapter in the Ni RAR is again inter-related with the regional assessment. For the local assessment, it is reasonable to assume that there are no farms in the near vicinity of Ni industries, and that people in the vicinity of Ni industries only grow and eat local fruits and vegetables. In a first approach, it is assumed that 100 % of fruit and vegetables consumed by the people living in the local area are locally produced. For the estimation of total nickel intake with food, including other, not-locally produced food items concentration in these other food items are taken from the dietary intake of the regional scale (see below). In this part of the assessment only the additional contribution of the local environment to concentrations in fruits and vegetables were calculated (e.g. based on only the additional Ni found in local soils as compared to the general Ni concentration in soils).

Given the lack of measured data on nickel concentrations in food items at a local scale, Ni concentrations in fruits and vegetables were modelled. The Ni concentrations in plant tissues and on plant surfaces result from three plant exposure routes: (1) soil to crop transfer via root uptake, (2) soil particles that are attached to the crop, either to the roots, or, via soil re-suspension, to the above-ground crop and (3) atmospheric deposition.

The Ni RAR addresses these three routes as follows:

(1) **soil to plant transfer:** Since current TDG methodology for the prediction of soil-to-plant transfer is inapplicable to metals, literature was evaluated for the existence of transfer models for nickel. Two reports presenting soil-plant transfer models were found and are discussed in detail in the RAR, but both reports were assessed as having limitations rendering them inapplicable for the situation assessed in the RAR. As an alternative, a literature survey was conducted on coupled soil and plant Ni data for edible crops and this literature was subjected to a quality and reliability evaluation considering the applicability of the data (e.g. field or lab/pot trial, soil and plant types, etc.). Only field trial data at relevant Ni soil concentrations were retained for the calculation of a median and a RWC nickel soil-to-plant transfer factor (termed BCF). The validity of the median BCF for predicting Ni concentrations in crops in this context was further verified by a validation exercise for regional soil and crop data.

(2) **soil particles attached to the crop:** conceptually, the contribution of Ni in soil particles attached to crops is a different pathway than Ni uptake from soil via the roots. However, due to the nature of the field experiments on which the soil-plant transfer models are based, the contribution of attached soil particles also falls in many cases under the soil-plant transfer model. In addition, the vegetables and fruits aimed for human consumption are (considering normal household practices) also rinsed before consumption, thus there is no need to incorporate the contribution of attached soil particles explicitly. However, the statement that fruits and vegetables are cleaned before consumption can be debatable. For example, eating new potatoes with peel is a common practice. Since this scenario is in parallel considered in the regional scale assessment, and accounting for the differential soil concentration between local and regional, this would lead to very small extra contributions in the local scale. Therefore, this route is not further taken into account in the calculations below.

(3) **direct atmospheric deposition on plants:** The only publication related to the influence of Ni deposition on Ni concentrations in edible crops was that of Ylaranta (1996). Nickel concentrations in soil and plants, including wheat, rye grass and lettuce, were measured in 1985, 1986 and 1988 in the vicinity of a copper-nickel smelter in Finland. The study even included a reference site 100km away from the plant and the influence of historically contaminated soil in the vicinity of the smelter-plant on the deposition measurements was evaluated by setting up addition pot-experiments with uncontaminated soils. However, the Finnish climatic conditions of the Ylaranta study are not representative for local scales elsewhere in Europe and the crops studied in the Ylaranta study (lettuce, wheat grain and rye grass) do not cover all main classes of vegetables and fruits typically eaten by humans. Therefore, a more general (not specific for nickel) US-EPA deposition model was used, and the dataset of Ylaranta was used to verify if the model is applicable to nickel (in addition the validation was done with Pb and Cd data).

In summary, for the local assessment of intake of nickel with food, the plant concentrations in fruit and vegetable were modelled by combining the uptake via the roots with the atmospheric deposition.

Local Scale - calculation of external doses

Exposure via inhalation

Ni air concentrations (typical = median value and RWC = P90 value) as modelled for the different Ni producing and using sectors were combined with conservative default inhalation rates of 20 m³/day for adults (TGD default) and 10 m³/day for children (1-2 years old).

Exposure via ingestion of soil and dust

For this scenario, the exposure is assessed separately for adults and children, for which this route is specifically relevant because of the significant mouthing behaviour in small children. Furthermore, it is distinguished between chronic and acute exposure via this route. For the chronic exposure, the Ni RAR rather closely follows the Pb VRA in applying default ingestion rates (135 µg/day soil and dust combined for children, 50 µg/day for adults) and the ratio of 55/45 % of dust/soil ingestion, with both parameters deducted from the IEUBK model. As described above, local scale Ni concentrations in soil are derived from EUSES modelling and region-specific background soil concentrations. The Ni concentration in dust is adapted from the regional assessment (see below). As acute exposure, the Ni RAR considers pica behaviour of children[6], i.e. the deliberate ingestion of soil. A soil intake rate of 10 g/day, as recommended by the US-EPA Exposure Factors Handbook for this scenario is used and combined with the typical and RWC soil concentrations for the local scale.

Exposure via drinking water

Since it is considered unlikely that people living in the neighbourhood of a Ni production or Ni use plant (i.e. in local scale range) consume other drinking water than water from the regional scale, the Ni concentrations from the regional assessment are used. These concentrations are combined with default water ingestion rates (see regional assessment below).

Exposure via dietary intake

For all food products except vegetables and fruit, dietary intake of the regional scale can be applied for the local assessments. Rationale for this is that only vegetables and fruits are expected to be produced locally in gardens of people living in the neighbourhood of Ni using/producing industries. In addition, this should also be considered as a conservative approach since this approach assumes that all eaten vegetables and fruits are locally produced.

The input for the additional Ni intake by humans in local settings are the additional Ni plant concentrations (due to soil→ plant transfer and deposition) and the default daily intake of vegetables and fruit of 563 g/day for adults (TGD default for the EU). This leads to additional Ni intakes for the local assessment from 0.01 µg Ni/day (typical scenario, sector metal products) to 46 µg Ni/day (RWC scenario, stainless steel sector). The US-EPA average consumptions of fruit and vegetables for children are used for exposure to children (28.2 g/day/kg body weight) of 1-2 years, corresponding to fruit and vegetable intakes of ± 300 g/day.

[6] consideration of PICA is not a standard approach in EU risk assessments, since this may be considered aberrant child behaviour

Regional Scale – Ni concentrations in environmental compartments

Regional scale – nickel air concentrations

The 'typical' Ni ambient air concentration is calculated as the EU-average of country-specific (year mean) mean measured ambient Ni concentrations for 2005 for Austria, Belgium, the Czech Republic, Denmark, Spain, the U.K., Latvia, and Slovakia (AirBase, the public air quality database system of the European Environment Agency) and further but slightly older data for the Netherlands, Germany, Italy and France (obtained from a DG ENT position paper). The typical Ni ambient air concentration derived in this way is 4.5 ng Ni/m³. It is noted that this average is rather largely influenced by the Belgian (17 ng/m³) and Italian data (8.5 ng/m³), for which is know that sample locations are close to metal industry operations or predominantly in the rural Rome area, respectively. By omitting the Belgian and Italian data, the average would be lowered to 2.9 ng Ni/m³. However, although the latter figure matches the EU modelled (EUSES) regional concentration of 2.69 ng/m slightly better, the 4.5 ng Ni/m³ is used as a conservative estimate for the typical value in a first approach.

The RWC Ni air concentrations (which serves as the basis for regional RWC scenario) in Europe was assessed by the 95th percentile of mean year average concentrations of basically the same underlying dataset as for the typical scenario above. A P95 value of 14.6 ng Ni/m3 is taken forward to the RWC scenario for the regional scale.

Regional scale – soil and house dust concentrations

The EUSES modelled soil Ni concentrations for the regional scale (TGD default region, 10 %, PEC_{total} regional) were 19.2 mg Ni/kg dm for agricultural soil, 16.9 mg Ni/kg dm for natural soil and 17.6 mg Ni/kg dm for industrial soil, respectively. The median value of 17.6 mg Ni/kg dm is used as input for the typical scenarios in the regional assessment. Because of comparability of data, no differentiation between natural, agricultural and industrial soil was made for the typical situation. These modelled values matched with the 50th percentile value of 14 mg Ni/kg dm for measured soil concentrations in Europe. However, the slightly higher modelled values are preferred over the measured data because measured data are only available for a few countries, which may not be representative for all of Europe.

The EUSES modelling section does not report P95 or P90 values for use as RWC concentrations. Instead, P90 values of measured Ni soil concentrations are used as input for the RWC scenario for the regional assessment. Based on a data set of nickel P90-values in European soil, a RWC Ni soil concentration of 26.2 mg Ni/kg dm for agricultural soils, 27.8 mg Ni/kg dm for forest soils land 35.8 mg Ni/kg dm for grassland soils was derived. The median value of these three P90 values, i.e. 27.8 mg Ni/kg is taken forward to the exposure calculations. It is noted that P90 instead of P95 Ni soil concentrations are entered here for Ni soil concentrations. This is done because of pragmatic reasons: 1) data of the Environmental Exposure section are expressed in this form, and 2) also for the local scale P90 instead of P95 soil concentrations are taken forward for the indirect human exposure section. Should soil ingestion become an important pathway, this deviation from the P95 concept as to be used in the RWC scenarios should be evaluated in a next step.

Ni concentrations in dust were derived separately for house dust and street dust:

House dust: Average measured concentrations for nickel in house dust were found in a variety of publications; however limited data were available for Europe. Therefore, data was also included from Australia and Canada, for which general indoor house conditions can be assumed to be similar. Averaged over the studies which reported median values, a typical (i.e. average of the medians of the different studies) Ni dust concentration of 48.7 mg Ni/kg dm was derived. Attic dust (mean 343-888 mg Ni/kg, data from Albania) is not considered relevant for chronic exposure to indoor dust by children, and is therefore excluded. Two publications provided distributions on Ni concentrations in terms of 95th percentiles (respectively 116 mg Ni/kg and 394 mg Ni/kg). The average of the two P95 values is used for the RWC scenario (i.e. 255 mg Ni/kg).

Street dust: Eight references were identified providing measured concentration of Ni in street dusts. Most results for cities in Europe are in close agreement, with the exception of the results from Sicily possibly due to entirely different climatic conditions. Furthermore, data from Albania were excluded because of their origin (close to Cr- extraction industry). The most recent study for W. Europe is that of Charlesworth et al. (2003). The authors compare metal loading on street dust samples from a large city (Birmingham) with a smaller city (Coventry). The authors hypothesize that results would show that in a larger city higher concentrations of metals would be found. This is, in fact, the case for all metals except for Ni (median value for Birmingham and Coventry were 16.6 and 141.5 mg/kg dm, respectively). Finally, the average of the 6 median values of Ni street dust concentrations (115 mg Ni/kg) is used as input for the typical regional scenario. This is probably a rather conservative approach because this average is largely influenced by 2 rather high concentrations (218 and 248 mg Ni/kg) of studies based on a limited number of samples (8 and 2), compared to the lower values (16.6 and 14.6 mg Ni/kg dm) of the UK (for large city dust) and Canadian study which are based on a larger dataset (respectively 100 and 45 samples). Given the broad range (> 10-fold) of median Ni concentrations in street dust, and the limited number of studies for which Ni ranges (in terms of P95) were reported, it is preferred here to use the 95th percentile of the median concentrations of separate studies as input value for the RWC scenario, i.e. 240 mg Ni/kg dm (this value is even larger than the average of the 2 reported P90/P95 values).

Regional scale – nickel concentrations in drinking water

Nickel exposure to humans via drinking water exists of (1) an environmental contribution, namely the Ni present in ground or surface water (indirect exposure via the environment), and (2) a contribution from articles such as tubings, taps and fittings, which serve as transfer media in the water supply chain. The latter is not included in the environmental exposure, but instead, in the consumers exposure section. This separation between environmental (ground water contribution) and consumer exposure (contribution from pipes and fittings) is however necessary in order to identify from which source there is a significant Ni contribution to the drinking water which may help in the decision making process for potential risk management measures. The focus of this section is thus on the environmental 'background' nickel levels in drinking water, not influenced by nickel release from pipes and fittings.

A recent EU-wide database on nickel concentration in drinking water should be available, since Member States are obliged to report nickel drinking water quality to the European Commission. However, the European Commission has a serious delay in processing the data and cannot report recent data. Currently, the most recent report refers to the data of the period 1996-1998. The report of data 1998-2001 was due end of 2006; however, at this moment, it is not yet published. Furthermore, the summary reports published in this context by the EU Commission do not contain individual measured concentrations, but only a county-specific figure giving the percentage of data points over the limit value (50 µg/L until 1998, 20 µg/l since 1998; the EU drinking water guideline is based on a WHO provisional guideline, which was revised in 2005 to 70 µg Ni/l).

Since this non-compliance percentage is of little use for the risk assessment, a comprehensive literature survey was conducted to establish an inventory of Ni concentrations in EU drinking water from published literature. Data of water sampled at private houses are only further considered if sampled after flushing the pipes and tubings, to minimize the contribution of Ni release from tubings en pipes (which is part of the consumer exposure assessment). Using this household data and samples from water-work (inlet to public water supply system), a typical (1.5 µg/L) and a RWC value (3.7 µg/L) are derived.

Further to this general EU-wide scenarios, a second scenario was considered: In Denmark, drinking water supply is based almost solely on groundwater. A large monitoring campaign on groundwater quality is running in Denmark, by the Geological Survey of Denmark and Greenland (GEUS). Data from this water works groundwater monitoring says that in a period from 1993 - 2002. 6972 wells have been analysed for nickel and in 3362 nickel was detected. In 221 wells the drinking water limit of 20 µg/l was exceeded. The median value was 2 µg/l and the 95th percentile was 8.8 µg/l. These figures are taken forward to the exposure calculation for this extra scenario.

Regional scale – nickel concentrations in food

The common approach in the above sections (air, soil, dust, water) is the assessment of nickel concentration in the different media to further combine these with consumption/inhalation rates. However, this approach was not followed for the assessment of Ni intake with food, since several dietary studies exist, the use of which considerably reduces uncertainty in the assessment (see also section 2.2 of this HERAG fact sheet). The approach used in the Ni RAR is described further below.

Nevertheless, the RAR in its appendix presents Ni concentrations monitoring data for a wide range of food items, as obtained in a comprehensive literature search and evaluation.

Regional Scale – calculation of external doses

Inhalation exposure

For the calculation of inhalation exposure, the typical and RWC nickel concentrations in air, as derived from EU-wide monitoring date (AirBase database, see above), was combined with default inhalation rates of 20 m^3/day for adults and 10 m^3/day for children (1-2 years).

Exposure via soil and dust ingestion

Nickel exposure via ingestion is assessed separately for children and adults. The distinction between these 2 groups is especially relevant for exposure via soil and dust ingestion given the significant mouthing behaviour in small children. The default IEUBK soil/dust ingestion rates are used for children: 0.135 g/day for children 1-2 years. For adults, 50 mg soil/dust per day is used (US-EPA Exposure Factors Handbook, 1997). According to the assessment reported in the Pb RA, the IEUBK default soil/ingestion rates are considered rather conservative for the general European situation. Nevertheless, the defaults are used as a first approach and can be refined if the risk characterization provides a justification to do so. The IEUBK ratio of 55/45 % of dust/soil ingestion is applied. Thus, of the 135 mg soil/dust ingestion, 74 mg is dust and 61 mg is soil.

Nickel concentrations in two forms of dust, i.e. indoor dust and street dust were assessed separately (see above). House dust concentrations are used in all scenarios for the assessment of Ni exposure via dust ingestion. The street dust concentrations are relevant in urban areas (without gardens, no soil ingestion) where children play outside on streets, and, for this additional scenario, the street dust ingestion can be considered as a substitute for soil ingestion by children. In combining the soil and dust concentrations with the ingestion rates, a breakdown was made for persons living in urban areas with no gardens (indoor dust + street dust ingestion) and persons living in areas where soil ingestion is more likely than street dust ingestion (general scenario). It is noted that children living in a place without a garden probably ingest less soil than others, and, in these cases, the soil/dust ingestion rates are probably too conservative. Nevertheless, in this first approach, no refining for soil/dust ingestion rates is made.

In addition, as an acute exposure scenario, a soil intake rate of 10 g/day was considered for children with pica behaviour and combined with the typical and RWC Ni soil concentrations.

Exposure via drinking water

Drinking water concentrations for Ni were assessed (i) for a general EU scenario using monitoring data from households (flushed samples) and water works (see above) and (ii) a additional “Ni-rich-ground-water“ scenario (see above). The derived typical and worst case figures were combined with the following default water consumption rates:

	water intake (l/day)	
	typical	RWC
children 1-2 years	0.50[a]	0.63[c]
children 3- 5 years	0.54[a]	0.76[c]
Adults	1.3[b]	2.3[b]

[a] source: HERAG
[b] source: US EPA exposure handbook (1997)
[c] source: US EPA children's' exposure handbook (2006)

Furthermore, an acute scenario was assessed, assuming a consumption of 400 mL of water all at once.

Exposure via the diet

On the regional scale, EU dietary studies are used to estimate Ni exposure via food intake. In the scientific literature, three methods are used to assess the daily intake of trace elements such as nickel: (1) market basket studies, (2) total diet studies and (3) duplicate meal studies. In market basket studies, individual food items are sampled from retail outlets and analyzed. The daily Ni intake is then calculated as based on these Ni concentrations in food samples and the estimated consumption. In total diet studies, food items are processed for consumption (e.g. cooked if appropriate) and are then analyzed individually or in food groups. Nickel intake is calculated as the product of the Ni concentration in the food and the amount consumed. In duplicate meal studies, duplicate samples of whole meals, snacks and beverages are collected, mixed and analyzed.

A Ni dietary intake survey was already partly performed in the consumer exposure section of the RAR. However, this data from the consumer exposure section is considered not representative for the current situation in the entire EU. Firstly, data are mainly from studies performed in the U.K. and Denmark, or from non EU Member States (Canada). Eating patterns, habits and nickel content in foodstuff might differ between regions in the EU. In addition, the only southern Europe Ni exposure study which was included (Italy) dates from 1982, and might therefore be outdated.

To obtain a better EU geographical coverage and to get insight in time trends in Ni dietary exposure in the EU, the literature regarding Ni dietary exposure is reviewed as assessed for relevance and reliability. Special attention was paid to the methodology that was applied in the dietary exposure studies. Data from non EU Member States were excluded from further evaluation.

Based on the most recent and relevant dietary intake studies in the EU, a typical dietary intake value (117 µg/day for adults) was derived. In contrast to the approach for the other exposure pathways, the typical intake is based on average instead of median intakes. This approach has mainly practical reasons (in the source reports, mean and not median intake data were reported). Nevertheless, the daily *average* intakes are expected to be representative for the 'typical' intake because the average Ni dietary intake is not very likely biased by extreme low or high Ni intakes (this is reflected in the rather small difference between average and P95 consumption). The few publications that reported P95 or P97.5 Ni dietary intake were converted to P95 Ni dietary intakes (under the assumption of normally distributed data). The average of the resulting P95 values (UK, 1997: 210 µg Ni/day; France: 149 µg Ni/day; Denmark, 1985: 252 µg Ni/Day: Denmark, 1990: 281 µg Ni/day; Denmark, 1995: 252 µg Ni/day- leads to the RWC dietary intake of 234 µg Ni/day (adults).

In the context of dietary exposure to, two further scenarios are currently included in the draft RAR: A high nickel diet scenario for sensitized individuals (concern: elicitation of dermatitis), and a nickel dietary intake scenario for children. However, since both these scenarios are still under extensive discussion and not fully developed yet, they are not further addressed here.

Assessment of internal exposure

The step between external exposure via inhalation and ingestion of food, soil/dust and water and the resulting systemic dose involves the assessment of the fractions of nickel in these compartments that are bioavailable and that are absorbed after intake. Nickel absorption factors for these different media were retrieved from the human health effects assessment parts of the Ni RA reports.

In this synopsis, the Ni absorption factors for each of the pathways will be only briefly discussed. Cross reference may also be made to the separate HERAG fact sheets on inhalation and gastrointestinal absorption.

Nickel absorption factor for inhalation

Based on the toxicological information gathered in the Ni RA reports on health for the various Ni compounds, it was advised to use a value of 100 % for the absorbed fraction of nickel from the respiratory tract following exposure by inhalation of soluble nickel compounds (nickel sulphate, nickel chloride, nickel nitrate) for particulates with an aerodynamic diameter below 5 µm (respirable particles). For particulates with aerodynamic diameter above 5 µm (non-respirable fraction), the absorption of nickel from the respiratory tract is considered to be negligible as these particles predominantly will be cleared from the respiratory tract by mucociliary action and translocated into the gastrointestinal tract and absorbed. For nickel metal, an absorption factor of 6% was derived based on animal studies (Ni RA reports on health). No information was available for nickel carbonate.

This data shows the need to assess (i) speciation and (ii) the particle size distribution of Ni in ambient air to be able to assess the absorbed dose.

The EC DG.ENV position paper on ambient air pollution by As, Cd and Ni compounds (EC, DG ENV, 2000) has actually already addressed exactly this issue. The speciation of nickel compounds in ambient air (i) in urban background air samples (n = 4) and (ii) in air samples near a an industrial site (steel mill, n =8) have been reported. The average nickel speciation in the urban air samples was: 22.4% soluble Ni, 8.3 % sulfidic Ni, 18.6 % metallic Ni and 50.7 % oxidic Ni. In the industrial air samples, the Ni speciation was on average: 42.1 % soluble Ni, 4.5 % sulfidic Ni, 7.4 % metallic Ni, 46.0 % oxidic Ni.

Furthermore, the EC DG. ENV position paper reports 4 modes of the size distribution for nickel which points at different source categories.

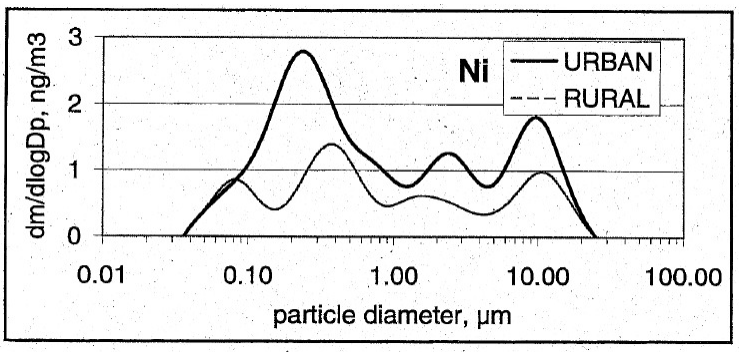


Figure: Average of size distribution at two sites near Helsinki (1996 – 1997) (source: EC, DG ENV position paper)

Apparently, the shapes of the size distributions from urban and rural sites do not differ much. Nickel compounds often show significant shares also in the coarse particle mode with diameters around 10 µm. The figure also indicates that a significant portion of the particles are larger than 5 µm and therefore will not have 100 % absorption from the lungs.

The EC DG.ENV position paper on ambient air pollution by As, Cd and Ni compounds (EC, DG ENV, 2000) considers that the absorption of nickel from ambient air should not be higher than 50% because only half of the nickel compounds in ambient air are highly water soluble and the water insoluble nickel compounds (mostly oxidic nickel) will have much lower absorption factors (perhaps even lower than the 6% calculated for nickel metal).

Based on the recommendations in the EU position paper which consider speciation and particle size of Ni in ambient air, it is decided to apply the 50 % inhalation absorption factor to convert external inhalation doses to internal doses.

Nickel absorption factors for ingestion of water, soil/dust and diet

The extent of absorption from the gastrointestinal tract strongly depends on the solubility of the nickel compound ingested and is influenced by whether the nickel compound is administered in drinking water, to fasting subjects, or together with food. Under consideration of these aspects, available literature/studies have been thoroughly evaluated for the HH section of the RAR.

In conclusion, for the purpose of risk characterisation, a value of 30% is used for the absorbed fraction of nickel from the gastrointestinal tract following oral exposure to soluble compounds (nickel sulphate, nickel chloride, nickel nitrate, and nickel carbonate) in the exposure scenarios where fasting individuals might be exposed to these nickel compounds. In all the other exposure scenarios, a value of 5% is used for the absorbed fraction of nickel from the gastrointestinal tract.

For the calculation if internal dose from ingestion of water, soil/dust and diet in the context of the indirect exposure assessment, the factor of 30 % is adapted for drinking water and the factor of 5% is applied for the ingestion of soil/dust as well as for the intake of Ni with food.

The subsequent calculations and results, which are basically the multiplication of the absorption factors with the external exposures derived as summarised above, are not reproduced here.

A 1.3: Zinc

General aspects

The most important exposure to Zinc for the general population is by the ingestion of foods. Especially meat and meat products, milk and milk products, bread and starchy foods contribute to the dietary Zinc intake. The average dietary intake of Zinc by adults in nine European countries was reported to be 9.1-12.3 mg/day. Only for adult males in Germany and Italy a higher daily dietary intake of 14-15 mg/day was reported (Van Dokkum, 1995). These figures are confirmed for the Netherlands in a recent report on the food intake of the general population (Hulshof et al., 1998): the average daily intake of Zinc is 9.4 mg with a minimum of 0.6 mg and a maximum of 39 mg. The 95-percentile value is 15.4 mg (P_5=4.7, P_{10}=5.5, median=9.0, P_{90}=13.8). The intake figures are based on a random group of 6,250 persons. The differences in Zinc intake vary due to source and variety of the food.

An epidemiological study has been carried out by Kreis (1992) in which the health effects of Cadmium and Zinc were investigated in a contaminated area in the southern part of the Netherlands (Kempenland). A population sample aged 30-69, with a residence of at least 15 years in a rural village in Kempenland (NL) was compared with a control population of an unpolluted area. About 75% of the inhabitants of both areas consumed at least half of their vegetables from local gardens. The plasma concentration of Zinc did not differ between the exposed (n=299) and the reference population (n=295) after adjustment for age and gender. The author concluded that, in contrast to Cadmium, Zinc exposure probably did not differ between the two villages.

The Zinc metal RAR cites NL national monitoring data for groundwater, surface water and air. However, it also states that under normal conditions, drinking water and ambient air are minor sources of Zinc intake (<0.01 mg/day). Interestingly, monitoring of drinking water for Zinc has ceased in the Netherlands at water companies.

In conclusion, the most recent average dietary intake of Zinc is around 10 mg/day. Compared to this intake via food, intake via drinking water and ambient air is considered negligible.

Local exposure

Based on TGD analogous calculation procedures, surface water maximum local Zinc concentrations (PEC_{add}s) of 543 µg/l and 783 µg/l (total Zinc) were estimated for the production and processing of Zinc metal, respectively. Maximum atmospheric Zinc concentrations (PEC_{add}s) were 36.9 µg/m^3 and 5.82 µg/m^3, for production and processing, respectively.

Note: Apart from Zinc metal, several other Zinc industries have reported zero emissions to water, and in some cases also zero emissions to air, which is why in these cases, indirect exposure via the environment originating from industrial emissions can safely assumed to be zero.

A 1.4: Lead

Summary of the indirect exposure assessment in the VLRA

As a result of both natural and anthropogenic sources, lead is present in multiple environmental media that can result in human exposure. For adults, drinking water, beverages and food are the principle determinants of exposure. The exposure of children, while influenced by the above, is predominantly determined by the ingestion of lead-containing soil and dust. Lead in air is currently only a significant source of exposure in the vicinity of point sources. Assessment of human exposures thus requires evaluation of multi-media inputs. For children, integrated exposure estimates and resulting blood lead levels were estimated by the Integrated Exposure Uptake Biokinetic (IEUBK) model. Adult exposures were evaluated using a physiologically based pharmacokinetic model developed by Ellen O'Flaherty specifically for lead exposure assessment in adults.

Extensive data bases existed for the presence of lead in most environmental compartments of concern and permitted generation of typical and worst case exposure estimates. Model predictions could in turn be compared to the results of numerous blood lead surveys that had been conducted in most (but not all) EU member states.

Estimating current levels of exposure, and alignment of model results with those of blood lead surveys, emerged to be a challenge. Blood lead levels within the EU had been influenced by a number of historical applications (lead in gasoline, water pipes, soldered food cans etc). Although these dispersive applications of lead had been phased out in all countries evaluated, the timing and speed with which phase-out policies had been implemented varied from country to country. Levels of lead exposure in each country were thus "moving targets" and had to be interpreted with respect to progress made in the implementation of phase-out policy.

Initial modelling efforts constructed a generic exposure scenario representative of continental exposure. As expected, this generic exposure scenario appeared to over-estimate exposure in many countries. Better alignment of model predictions and blood lead survey data was achieved when two different regional exposure scenarios were developed. The first estimated exposure in countries that had implemented early and aggressive phase-out policies for dispersive applications. The second estimated exposure in countries somewhat slower to implement phase-outs (e.g. phase out of lead in gasoline was still being complete in 2000, the latest year of comprehensive data collection for the Risk Assessment). It was further necessary to subdivide exposure scenarios into urban an rural exposure environments – monitoring data indicated that lead in soil levels were significantly higher in urban settings. The resulting four different exposure scenarios were then compared to blood lead survey data. Predictions of blood lead levels in adults and children were reasonably concordant with observed blood lead data in rural settings. However, predictions for urban environments significantly exceeded the results of blood lead surveys.

Progressive adaptation of model parameters was then undertaken, altering model assumptions documented by other studies to result in erroneous model predictions. IEUBK makes several assumptions that, while often applicable to the United States, can be erroneous. The model assumes a strong relationship between exterior soil lead levels and the lead content of household dust. However, the amount of bare soil in most urban EU cities is limited. Although comprehensive dust lead monitoring programs were limited in number, the existing data indicated that IEUBK estimates of household dust levels were up to two orders of magnitude higher than those actually present. Replacement of default model predictions with observed household dust levels resulted in lower blood lead levels that, while still high, were more concordant with the results of blood lead surveys. Model assumptions regarding soil ingestion rates were next examined. Urban environments with relatively little bare soil pose limited opportunities for soil ingestion – over-prediction of blood lead levels is suspected to result as a consequence of this in some US cities. Default IEUBK soil and dust ingestion rates where thus reduced (e.g. from 135 to 40 mg). This level of reduction was selected based upon recent studies suggesting that these lower levels were in fact more realistic then the higher default values used by IEUBK. Following this reduction, close concordance between model predictions and blood lead survey data was obtained.

Final modelling, supported by blood lead data, indicated that blood lead levels for adults and children in countries that had implemented early phase-out policy were low under typical and worst case conditions. Elevated blood lead levels of potential concern were predicted for countries late to phase

out dispersive lead applications. However, very recent data indicated that rapid declines in blood and environmental lead had occurred in these countries and that acceptable levels of lead in blood characterised > 98% of adults and children within the EU.

Deviations from the TGD

The preceding assessment differed from TGD procedures in several ways. lead-specific exposure models were available to estimate exposure and blood lead variability in the EU. These models utilise default estimates of soil ingestion lower than those recommended by some EU models. Calibration of the models further reduced the estimate of soil ingestion rates. This adjustment may be the source of some discussion during review of the Risk Assessment by the Member States.

Assumptions of reduced soil lead bioavailability (compared to lead in food) are also made by exposure assessment models. Reductions in soil lead bioavailability as a function of being incorporated into a soil matrix are well documented and routinely applied in lead exposure assessment, but do not appear to be explicitly recommended by TGD procedures. Chemical speciation can further reduce lead in soil bioavailability and procedures exist for demonstrating this reduction for use in lead exposure assessment models. Such studies were not undertaken in the present assessment, although it should be recognised that reductions in soil lead bioavailability may have contributed to the initial lack of concordance between model predictions and observed blood lead survey data.

Local Sources

Somewhat different approaches were used to predict the impact of local sources upon blood lead. International consensus “slope factor” estimates were used to calculate the amount of lead in air what would directly produce a 1 µg/dL increase in blood lead. Facility emissions (most often calculated estimates) were then compared to this benchmark air lead concentration of 500 ng/m^3. Facilities with air emissions lower than this benchmark were assumed to pose insignificant exposure risk and excluded from further consideration. Facilities in excess of this benchmark were subjected to further evaluation. These evaluations took several forms:

a. Blood lead survey data were sought for populations in the immediate vicinity of local sources of concern. This survey data was able to confirm the presence or absence of exposure risk at several local sources.
b. Emissions calculations were based upon production levels and practices in the year 2000. More recent data were sought from facilities of concern and, in several instances, significant emission reductions had occurred that removed concerns over potential exposure excess.
c. Air lead levels calculated using TGD models represent levels 100 m from the local source emission stack. In a number of instances, the nearest residential populations are located at significantly greater distances. More sophisticated plume modelling is ongoing to evaluate air lead levels in such residential areas.

Local source evaluation also included estimation of soil lead elevations that would result from current airborne emissions. In all instances, these emissions were low and would not produce significant elevations of blood lead. The impact of historical emissions was not included in this analysis.

Fugitive emissions

Estimates of fugitive emissions were not available for the majority of the facilities examined and were not explicitly factored into the analysis. In several instances, measured air lead data were available and could be compared to calculated emission estimates. In general, calculated and measured air lead levels were similar, suggesting that fugitive emissions were not significant. However, this conclusion must be tempered with the caveat that the potential for fugitive emissions can vary dramatically as a function of the nature of the local sources and emission control technologies employed. Significant levels of fugitive emissions could exist at some local sites.

A 1.5: Cadmium

For the general population living in uncontaminated areas, smoking and diet are the main sources of Cd exposure. Other sources of exposure are soil and dust ingestion, drinking water and inhalation of air. The daily Cd uptake through environmental exposure was estimated as being the sum of all exposure routes, based on average values for ambient environmental Cd levels, and for three groups of the general population: children, adults with sufficient body iron stores and adults with depleted body iron stores. An additional scenario was included representing a local scenario where Cd concentrations in soil, air and diet are all elevated. A tabular summary of these scenarios is given further down below.

The figures used for the daily Cd uptake via the different routes of exposure were derived as follows:

Air:
Average Cd concentrations in EU countries are found in the range <1-5 ng/m^3 in rural areas, 5-15 ng/m^3 in urban areas and 15-50 ng/m^3 in industrial areas. This Cadmium is associated with particles in the respirable range, and for adults it is estimated that about 25 % of the daily Cd intake from the atmosphere is absorbed (IPCS, 1992). At a total daily air intake of 20 m^3, this would lead to 0.025 µg Cd uptake at a Cd concentration in air of 5 ng/m^3, and 0.075 µg Cd uptake at a Cd concentration in air of 15 ng/m^3. This daily uptake is small compared to that from food or from smoking: based on measurements carried out in Stockholm and assuming a daily respiration volume of 13 m^3, Vahter et al. (1991) estimated that airborne Cd accounted for only about 1 % of the total daily absorbed amount of Cd. In houses of smokers, significantly higher Cd air levels are observed as compared to houses of non-smokers (IPCS, 1992).

Soil and dust:
Ingestion of dust and/or soil by young children is known to be an important source of exposure for elements such as lead. However, this pathway is most probably not a dominating exposure route for Cd. The estimated average intake of household dust by children is 100 mg/day (IPCS, 1992). Based on data of Cd in household dusts in UK (mean 7 mg/kg, n = 4500), it was concluded that the average daily intake of 0.7 µg is much smaller than food intake (IPCS, 1992). At an absorption rate of 0.05, this would lead to a daily uptake of 0.035 µg/day, which is less than 10% of the total daily uptake. Consequently, in the risk characterisation, the exposure via uptake of soil and dust was considered but it did not largely affected the outcome.

Smoking:
Tobacco plants naturally contain high Cd concentrations in leaves and cigarettes contain 1-2 µg Cd per cigarette, the amount varying considerably with the origin of the tobacco (IPCS, 1992). About 10 % of this Cd is inhaled and it is estimated that 25-50 % of the inhaled Cd is absorbed. As a result, smoking a pack of 20 cigarettes daily results in a net uptake of 0.5-2 µg. This value is large compared to the daily Cd uptake from air (0.02 µg) and in the same range of the daily Cd uptake from food Cd (0.35-1.6 µg). Cd intake through smoking 20 cigarettes per day increases the Cd systemic dose 1.2 to 7 fold above that in non-smoking individuals with equivalent Cd intake through other sources.

Drinking water:
Drinking water usually contains low Cadmium levels (<1 µg/l) and, consequently, Cd exposure from the intake of drinking water or water-based beverages (~2 L) is relatively unimportant compared to dietary intake (IPCS, 1992).

Dietary intake:
It is generally acknowledged that dietary intake is the major source of Cd exposure for the non-smoking general population. Levels of Cd in food items are typically high in offal, organs, equine products, shellfish, crustacean, cocoa, mushrooms and some seeds. The proportion of these products in the average dietary Cd intake is low because of their low average consumption. There may be certain parts of the population, however, that have elevated intake of Cd from such food. Typical groups with high dietary Cd intake are those with preference for shellfish or mushrooms.

Dietary Cd intake is generally estimated based on market basket or on total diet studies. These studies calculate the average dietary Cd intake using an average diet for the selected population. There are, however, variations in Cd concentrations in various foods and in the consumption of the

various foods between individuals and population groups. Thus, there are large individual variations in the dietary intake due to differences in dietary habits. Only few of the market basket or total diet studies include the variability of individual diets so that e.g. the frequency of groups with high Cd intake in a population could be calculated. Duplicate meal or faecal output studies offer the advantage that variability in Cd intake between individuals can be assessed. The available data from duplicate meal studies are, however, limited. It has been reported that duplicate meal studies underestimate true intake by 15-20 % (Johansson et al., 1998).

Reviews of dietary Cd intakes show that the average Cd dietary intake in European countries range between 5 and 90 µg day^{-1}, but with most values ranging between 10 and 35 µg day^{-1}. As a result of improved detection limits, early data on dietary Cd intake are usually higher than more recently obtained data. As an example, the best US dietary Cd data indicated 26-51 µg Cd day^{-1} in the early 1970s.

The ambient scenarios are furthermore subdivided into smokers and non-smokers. Individuals with low iron stores may absorb much more Cd via the GI route, on average 2 times more (Berglund et al., 1994). Furthermore, children may absorb relatively more Cd than adults because of increased absorption from the gastro-intestinal tract, a higher food intake per kg body weight and a diet of high milk and cereal contents. No data were found, however, on the relative GI absorption rate in children compared with adults. The proposed absorption rate is 0.03 for both adults with sufficient body iron stores and for children.

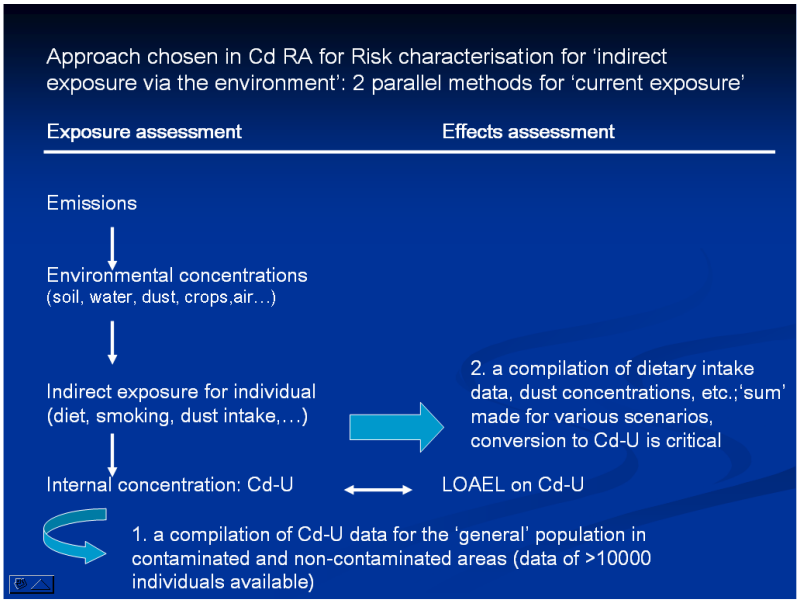
Table: Summary of scenarios for the indirect exposure to Cd via the environment

scenario 0: children (4-7 years old)		
source	Cd uptake (µg day^{-1})	assumptions
air	0.012 -0.037	air Cd 5-15 ng/m³; daily inhalation 10 m^3; absorption rate = 0.25
soil and dust	0.04	dust or soil Cd 7 mg/kg;100 mg intake absorption rate = 0.05
drinking water	<0.05	Cd water<1 µg/l; absorption rate = 0.05; 1l/day consumption
dietary intake	0.4	dietary Cd 8 µg/day absorption/intake ratio = 0.05
sum	**0.5 µg/day (0.025 µg/kg_{bw}/day)**	
scenario 1: adults with sufficient body iron stores		
source	Cd uptake (µg day^{-1})	assumptions
air	0.025 -0.075	air Cd 5-15 ng/m; daily inhalation 20 m^3; absorption rate = 0.25
soil and dust	0.02	dust or soil Cd 7 mg/kg; absorption rate = 0.03
smoking	0.5-2.0	smoking of 20 cigarettes; 1-2 µg Cd per cigarette; absorbed fraction 0.025-0.05
drinking water	<0.06	Cd water<1 µg/l; absorption rate = 0.03 2l/day consumption
dietary intake	0.21-0.96	dietary Cd 7-32 µg/day, absorption rate = 0.03
sum	**non smokers: 0.33-1.12** **smokers: 0.82-3.12**	
scenario 2: adults with depleted body iron stores		
source	Cd uptake (µg/day)	assumptions
		as above, but absorption rate of 0.06 for dietary Cd, soil/dust/water Cd
sum	**non smokers: 0.53-2.08** **smokers: 1.03-4.08**	
scenario 3: near point sources (adults with sufficient body iron stores)		
source	Cd uptake (µg day^{-1})	assumptions
		as scenario 2 but air Cd is 22-1000 ng/m, soil Cd 70 mg/kgand dietary Cd 17-34 µg/day
sum	**non-smokers : 0.89 – 1.40 (22 ng/m^3)** **non smokers: 5.9-6.4 (1000 ng/m^3)**	

Local scenario (scenario 3):

The Cd uptake near point sources is dominated by inhalation with given assumptions of estimating dietary Cd. The contribution of air Cd to dietary Cd has neglected the Cd deposition on locally produced food. There is indirect evidence that this might largely contribute to crop Cd concentrations but there are no data to estimate this contribution correctly. On the other hand, restrictions on food production near point sources are often in place but there is no information to generalise the current situation in EU. Therefore, scenario 4 should be considered as indicative only. Soil/dust ingestion may be an important additional source of exposure in contaminated areas. Soil or household dust Cd concentrations exceeding 100 mg/kg have been reported around former refineries or mining areas (IPCS, 1992). The availability of soil Cd is, however, probably smaller than food Cd or Cd salts. Cd concentrations in local water pits can be elevated in areas with historical Cd pollution (Lauwerys et al., 1990). Ground water Cd may also be a significant contribution to Cd exposure in areas with acid soils (Bensryd et al., 1994).

In the Ca RA daily uptake values were converted in internal concentrations (Cd-U) and combined with measured Cd-U data to be able to compare exposure values directly with the N(L)OAELs in the risk characterisation, based on toxicokinetics models.



References for this section on Cadmium:

- Bensryd I, Rylander L, Högstedt B, Aprea P, Bratt I, Fåhraéus C, Hólmen A, Karlosson A, Nilsson A, Svensson BL, Schütz A, Thomassen Y and Skerfving S (1994). Effect of acid precipitation on retention and excretion of elements in man. Science of the Total Environment, **145**, 81-102
- Berglund M, Åkesson A, Nermell B and Vahter M (1994). Intestinal absorption of dietary Cadmium in women depends on body iron stores and fiber intake. Environmental Health Perspectives, **102**, 1058-1066
- IPCS, International Programme on Chemical Safety (1992). Cadmium. Environmental Health Criteria; 134. World Health Organization, Geneva
- Johansson G, Åkesson A, Berglund M, Nermell B and Vahter M (1998). Validation with biological markers for food intake of a dietary assessment method used by Swedish women with three different dietary preferences. Public Health Nutrition, **1**, 199-206
- Lauwerys R, Amery A, Bernard A, Bruaux P, Buchet JP, Claeys F, De Plaen P, Ducoffre G, Fagard R, Lijnen P, Nick L, Roels H, Rondia D, Saint-Remy A, Sartor F and Staessen J (1990). Health effects of environmental exposure to Cadmium. Objectives, design and organization of the Cadmibel study: a cross sectional morbidity study carried out in Belgium from 1985-1989. Environmental Health Perspectives, **87**, 283-289
- Vahter M, Berglund M, Lind B, Jorhem L, Slorach S, and Friberg L.(1991). Personal monitoring of Lead and Cadmium exposure-a Swedish study with special reference to methodological aspects. Scand J Work Environ Health, **17**, 65-74

A 1.6: Aluminium

The following preliminary written comment was provided by industry on this topic:

There is some indirect exposure to aluminium from the environment through water. This is a minor component of total daily intake and has been thoroughly evaluated through analytical data for water.

In the EU there is also a monitoring value for aluminium in drinking water of 0.2 mg/l, to be monitored where aluminium salts are used for water cleaning purposes.

Several studies of general population exposure to aluminium are available, and the key routes are known (via food).

A 2: Background information on "Consumer Exposure"

This chapter on background information contains extracts from various summaries of previous and current risk assessments conducted for metals. The presentation may vary in extent and detail as a reflection of the way in which it is either presented in the respective RA report, or the way it was made available to the authors of this fact sheet, or even according to the state of the RA report, some of which were in an early draft stage at the time of compilation of this fact sheet. **Note: For the currently ongoing risk assessments, the sections below may be amended in the future. This explicitly includes figures, model parameters, results in calculations and conclusions in the risk characterisation. Please refer to the latest versions for up-to-date information.**

Nevertheless, the sections below (presented metal by metal) were considered to represent descriptions of examples of metal-specific consumer issues, which are summarised and categorised in abbreviated fashion in the body of this fact sheet.

Note: References for citations contained in the following sections are not given in this fact sheet but may be obtained from the original risk assessment reports.

A 2.1: Antimony

Antimony trioxide (Sb_2O_3, ATO), which is the most extensively used Antimony compound, is currently undergoing an EU risk assessment. Although there is no known direct consumer use of the substance ATO as such, it is used in several products, mainly as a flame-retardant in plastics (including PVC) and in textiles. Significant quantities of ATO are also used in PET production (as a catalyst), as additives in glass manufacture and as pigments in paints and ceramics.

Four scenarios for consumer exposure to ATO are presented in the current (first) draft RAR: “drinking from a PET-bottle” and “sucking on upholstery fabric” (both resulting in oral exposure), “indoor air” (inhalation) and “sitting on upholstery fabric” (dermal exposure). The following “reasonable worst-case exposure estimates” were derived (note: the ATO RAR assigns conclusion (ii) to all four consumer exposure scenarios):

(i) For drinking from a PET bottle, the reasonable worst-case oral exposure to Antimony trioxide is 0.6 µg/kg bw/day for a child (corresponding to 0.5 µg Antimony/kg bw/day), and 0.17 µg/kg bw/day for an adult (corresponding to 0.14 µg Antimony/kg bw/day). This is based on measured values and supported by calculations.

Comment: The concentration of Antimony in the soft-drink or water (C) is set by the Swedish rapporteur at the leaching limit value of 5 µg/l based on Council directive 98/83/EC, and it is assumed that all Antimony is present as Antimony trioxide. However, a UK Food Standards Agency survey has yielded a measured (N=161 samples) RWC of 0.5 µg/L, and the bulk of the Sb in the PET matrix is considered to be chemically bonded as the glycolate within the PET polymer, so that this estimation is overly conservative.

(ii) For sucking on upholstery fabric, the following assumptions were used in an equation for oral exposure: oral absorption is assumed to be 100%, a child (Bw 10 kg) sucks on a 50 cm^2 (A_f) of a fabric back-coated with Antimony trioxide daily, one hour per day (f_{cc}) for two years. The area density (S_a) for Antimony trioxide is about 2.5 mg/cm^2 (S_a). The fractional release rate of Antimony trioxide is estimated as 0.001/d (μ_a) based on leaching of Antimony from polyvinylchloride cot mattresses.

$$Exposure = \frac{S_a \cdot A_f \cdot \mu_a \cdot f_{cc}}{BW}$$

Using the equation and the assumptions above the average oral dose rate is estimated to be 0.52 µg kg/d.

(iii) For dermal exposure from sitting on flame-retardant containing upholstery, the extreme worst-case dermal exposure is calculated to 0.002 mg/kg bw/day for an adult. The corresponding dermal exposure for a child is 0.004 mg/kg bw/day.

(iv) For exposure via indoor air, the reasonable exposure of Antimony trioxide is calculated to 0.24 $\mu g/m^3$. This corresponds to 0.07 and 0.2 µg/kg/d for an adult and a child, respectively.

Comment: the model prediction of the RAR yields an "estimated time-average concentration for Sb2O3 of 0.24 µg/m³. The UK survey of Antimony in house dust (median value: 13 µg/g, corresponding to 15.6 µg Sb_2O_3/g, based on a rel. content of Antimony in Sb_2O_3 of 84%) presented in the same section of the RAR indicates that 1.56% of house dust may represent diantimony trioxide. We note that this attractive model computes a time-average house dust value in British homes of >15 mg/m³, which is not exactly pleasantly descriptive of what otherwise is known as a nation fond of cleanliness.

A 2.2: Cadmium

The consumer exposure section in the RAR on Cadmium metal addresses five scenarios, which include the potential exposure to Cadmium compounds in general.

Scenario 1: active electrode material in Nickel-Cadmium batteries
Scenario 2: pigments used mainly in plastics, glasses and ceramics, enamels and artist's paints
Scenario 3: use of Cadmium as stabilisers for plastics or polymers
Scenario 4: metal plating (steel and some non-ferrous metals)
Scenario 5: component of alloys

Few additional product types containing Cadmium are known, but data is considered too fragmentary to discuss it and/or to derive potential exposure estimates. According to industry, Cadmium compounds in consumer products are either present in a stable matrix or in massive metallic form, and as such not available for exposure.

Of the above mentioned five scenarios, three scenarios were judged in the RAR as potentially relevant for consumer exposure (Ni-Cd batteries, metal plating and alloys). In both batteries and plating scenarios, consumer exposure is considered to be very low, and conclusion (ii) is assigned in the RAR.

In the scenario involving consumer uses of alloys containing Cadmium metal, two types of products are distinguished:

(i) For brazing material, a conservative estimate is proposed by cross-reading from the corresponding occupational scenario, and this value is taken forward to the risk characterisation section. Therefore, conclusion iii is reached for acute respiratory effects.

Brief description of the occupational exposure scenario used for this cross-reading: Possible sources of occupational exposure to Cadmium oxide are brazing, soldering and welding, with a solder

containing Cadmium or when welders are operating on material containing Cadmium or plated with Cadmium. A possible dermal exposure is concluded to be very low, because at the high temperature used during these tasks, unprotected work is not possible. Workers are exposed by inhalation to the solder fume. Historically, the alloys used for silver solder (also named hard solder) contained up to 25 % of Cadmium and since the melting point and boiling points of Cd are lower than those of other metals within the alloy, silver solder fumes contained very high proportions of CdO (up to 85 %). Cadmium metal was mainly used to allow fragile metallic structures to be soldered at lower temperature (e.g. jig soldering). Although this use of Cd has been abandoned in the last 10 years (Cd-free solder), data provided by several member states indicate that Cd soldering or soldering with Cd containing material was still in use at some workplaces during the nineties and some measured data for Cadmium concentrations in air are available.

Accidental cases of significant external exposure to Cd (with atmospheric Cd concentrations reaching up to several mg/m³) have also been reported during soldering while the presence of Cd in the soldering material was unexpected. This latter pattern of exposure was however not considered as being representative for typical working conditions during welding/soldering but rather as “accidental”, unintentional “misuse”. From additional information provided by UK, it appears that Cadmium does not play a part in soldering but that it is commonly present in hand brazing consumables, in concentrations up to approximately 25%.

Cadmium is used to control the melting temperature and flow properties of the alloy. Because of its high volatility, Cd is thought to concentrate into the fumes. When used at the correct temperature the fume from brazing is quite limited but often workers overheat the parent metal to encourage the braze to run and wet more easily, although this should not be necessary. This overheating results in the generation of copious fumes. Cd is not thought to be still used in any welding consumables. The only instance would be if Cadmium plated materials were welded e.g. resistance welding of studs.

In the Cadmium RAR, data provided by the member states is presented, resulting in the following conclusion: Only limited data have been reported for this scenario and probably relate to different processes and/or working conditions as values for Cd in air are in a wide range (from 0.2 to1800 µg Cd/m³). No more information on job description and exposure pattern is available. Elevated Cd-air values have also been reported in accidental cases where exposure to Cd was not expected to occur. From the provided data, a reasonable worst case estimate for inhalation exposure may be derived (280 µg Cd/m³, 95th percentile of the “Berufsgenossenschaftlicher Arbeitskreis Altstoffe Bundesrepublik Deutschland“ data from 1998).

(ii) For jewels (bracelet) containing Cadmium alloys, a conservative exposure estimate of 60 g/cm²/week (dermal) was derived. Assuming that the bracelet is worn continuously, that the corresponding skin surface is 10 cm² and that dermal absorption is 1 %, a daily uptake of 1 µg Cadmium was taken across to the risk characterisation as a conservative estimate. Conclusion (iii) is reached in the RAR because of the potential for genotoxicity and carcinogenicity.

It is mentioned in the RA that although in the other scenarios a significant consumer exposure is probably limited to very limited, it must be recognised that quantitative data to document exposure of the consumer is scarce.

Note: The exposure of consumers to Cadmium from cigarette smoke and other tobacco products is addressed in the “indirect exposure” section of the RAR. Extract of this scenario:

Tobacco plants naturally contain high Cd concentrations in leaves and cigarettes contain 1-2 µg Cd per cigarette, the amount varying considerably with the origin of the tobacco (IPCS, 1992a). About 10 % of this Cd is inhaled and it is estimated that 25-50 % of the inhaled Cd is absorbed. As a result, smoking a pack of 20 cigarettes daily results in a net uptake of 0.5-2 µg. This value is large compared to the daily Cd uptake from air (0.02 µg) and in the same range of Cd uptake from food Cd (0.35-1.6 µg). The mean blood Cd (Cd-B) in active smokers is significantly higher than in unexposed non-smokers, and it is very close to the mean Cd levels in passive smokers (Shaham et al., 1996). Smoking is directly associated with increased Cd-B.

A 2.3: Copper

The data on copper in consumer products is sparse relative to that for indirect exposure. Trade Organisations and the Medline database were consulted. Where assumptions are introduced in the VRA in order to estimate exposure (e.g. from handling coins), these are judged as conservative with respect to risk. As for other modes of exposure, all exposure data refer to external exposure as indicated in the TGD. Consumer products are generally not considered a significant source of exposure. The scenarios assessed in the Copper VRA are as follows:

(i) Inhalation exposure from smoking:

Smoking rates among adults in most EU countries range from 25-35% with a minimum of 14% in Portugal and a maximum of 38% in Greece (www.who.int/tobacco/en/). Detailed data for age and sex specific smoking rates for the UK indicate that smoking rates are highest among young men aged 20-34 years (www.heartstats.org). Differences in smoking rates as a function of sex are modest with a maximum difference of approximately 6% in the 20-34 year age group. Social class is the major determinant of smoking rates but, even among manual workers, smokers are still a minority (33% for 2001, the last year for which data are available).
Cigarette smoke contains traces of several heavy metals including copper. Available data for copper indicate that as little as 0.2% of copper in cigarettes is available in cigarette smoke. Since exposure is proportional to consumption, a RWC of 0.0005 mg/day for the general population was estimated based on the mean value for consumption of 20 cigarettes per day.
Typical exposure for the general population is taken as zero.

(ii) Handling of coins:

Coins in circulation in the EU may contain significant amounts of copper. In a study to investigate leaching of copper from coins, patch tests were conducted in which a range of coins were taped to skin for periods of 24-72 hours and copper released was recovered both from the skin and coin surface. In a further series of experiments, manipulation tests were performed to determine the transfer of metals to the fingers following successive manipulation of coins. 58 coins were individually manipulated by volunteers counting them from one container to another. Each manipulation took an average of 2.6 secs to perform. Wipe samples were taken from the three fingers used before and after the experiments to determine the transfer of metals and to correct for background skin levels. The tests were performed with and without pre-washing in demineralised water and also with prior polishing of the coins. The results were expressed as the amount of copper transferred to the fingers after the manipulation of a single coin. Pre-washings made negligible impact on contamination while polishing had a major impact by removing transferable metal from the coins' surface.
In the absence of frequency/contact parameters for coins in the TGD, it is assumed that a single coin is manipulated continuously for periods of 5 or 10 minutes. Given an average manipulation time of 2.6 secs, this represents a total of approximately 115 or 230 manipulations, respectively. It is further assumed that the coins handled are a mixture of polished and unwashed coins representing the different stages of the coins' lifecycle. Applying a multiplier of 1.6 to the transfer rate of 0.10 µg per manipulation for polished coins and 1.4 µg per manipulation for unwashed coins, gives an estimated transfer of copper to the fingers of 0.14 and 0.28 mg/day respectively.

Note: a distinction between typical and RWC exposure is not made.

(iii) Jewellery:

Dermal exposure to copper jewellery may occur particularly to jewellery with a high copper content such as bracelets marketed for their supposed anti-rheumatic properties. No data on the frequency of such exposure or the contact parameters are available and an accurate quantitative assessment was therefore not possible. As a worst case approximation, the data cited above for coins are used where coins were taped to skin and leaching determined by patch tests. Since coins typically are comprised of 60-90% copper, this analogy was considered reasonable. The internal dimensions of a bracelet are based on literature data (24cm^2). It is assumed that this surface area is in continuous daily contact

with the skin but it was noted that this probably overstates the actual level of contact. Dermal exposure was assessed by multiplying the rate of leaching of copper from coins with the surface area of the bracelet in daily contact with the skin. It is assumed that jewellery of this nature is worn by relatively few individuals.
The internal dimensions of a bracelet are taken as 1.2 x 20cm = 24cm². It is assumed that this surface area is in continuous daily contact with the skin although this probably overstates the actual level of contact. Based on the rate of the leaching of copper from coins, given as 60-120 µg/cm²/week, this amounts to a dermal exposure of 60/120 x 40 = 1.4-2.9 mg/week or 0.2-0.41 mg/day. It is assumed that jewellery of this nature is worn by relatively few individuals. An exposure of 0.41 mg/day is therefore taken as a RWC estimate.

Typical exposure is taken as zero.

(iv) Cosmetics, toiletries and hair care products:

(not assessed here since covered by TGD approach)

(v) Intra-uterine devices and copper dietary supplements:

(not assessed here, since both uses may be considered a pharmaceutical application)

A 2.4: Lead

The exposure of consumers to lead is extensively assessed in the currently ongoing Voluntary lead Risk Assessment (VLRA). Some historical applications that have been or shortly will be phased out were listed, but were not extensively reviewed in the consumer exposure section. Also not explicitly assessed was the exposure to lead from “illicit products” except in instances (e.g. ceramics) where these products might be confused with products that are commercially available within the EU. Both historical and illicit applications are assumed, for the purposes of this risk assessment, to pose an unacceptable risk of consumer exposure.

From current applications, some were initially judged as posing negligible exposure, whereas an extensive list of further current applications was assessed in detail. For all these applications typical exposure was found to be negligible, while for some uses a reasonable worst case exposure estimate could be derived.

Historical applications

Lead use in a variety of historical applications has impacted, and to a certain extent continues to impact, upon general population blood lead levels in the EU. Inasmuch as these applications have been, or shortly will be, phased out they are not extensively reviewed in the VLRA. The potential impact of such applications is reflected in assessments of “indirect exposure via the environment”. Such applications include: lead in gasoline, lead in water pipes, lead-soldered food cans.

Current applications for which negligible exposure is presently assumed

Lead use in a number of consumer products is not expected to result in exposures of concern as a result of consumer use, provided product use is in the fashion for which it is intended. For the purpose of consumer exposure assessment, negligible exposure is presently assumed for: lead acid batteries (all types), automotive structural and electronic applications, consumer electronics and underground cable sheathing.

Current applications assessed in detail: typical and worst case exposure negligible

Numerous other consumer applications were assessed in detail. Each application is described, giving the type of lead containing material used, the route of exposure, typical frequencies and typical

durations of use. In several cases where the reasonable and customary use of a product does not pose a realistic route of exposure, negligible exposure is assumed for both typical and worst-case consumer exposures. The following table summarises these scenarios:

Table: Current consumer application assessed in detail in the Voluntary Lead Risk Assessment, for which both typical and worst-case consumer exposure were found to be negligible. Designation of "exposure route" in the following table as "none" identifies products where the reasonable and customary use of a product does not pose a realistic route of exposure.

Scenario	Exposure route
hunting bullets	oral
weapon reloading	inh. / ing.
weights/sinkers	dermal
artistic materials*	dermal
electric solder	inh. / dermal
plastics (wire cable)	none
plastics (exterior PVC)	dermal
plastics (water pipes)	oral

Scenario	Exposure route
plastics (other interior)	oral
paint (commercial use)	none
radiation shielding	none
Lead sheets (maintenance)	dermal
decorative ceramics	dermal
decorative crystal glass	dermal
crystal glass flatware (dish)	oral

* Compliance with consumer labelling instructions presumed.

Current applications assessed in detail: worst case exposure estimate given

For a further set of scenarios, consumer exposure is relevant and reasonable worst case estimates were derived (typical exposure negligible in all instances). Worst-case exposure scenarios were ideally defined as the upper 90th percentile of the consumer exposure distribution. However, data are usually inadequate to derive an exposure estimate. Expert judgment was then applied to derive worst-case exposure estimates based upon assumptions of a high frequency of consumer use. A specification of "inadequate" in the following tables is an indication that insufficient information was available to make a quantitative evaluation of exposure potential but that some level of exposure might plausibly be postulated to occur during consumer use.

Scenario	Exposure route	Worst case exposure
Lead water pipes. This application is assumed to be discontinued and thus pose negligible exposure to the typical consumer. In recognition that lead water pipes remain in service in some countries, the impact of their use was estimated as a worst-case exposure scenario, based on available data.	oral	35 µg/L
Candles with lead wick core	inhalation	16 µg/m³ (chronic)
Hair dye	dermal (scalp)	1000 µg/day (chronic)
Lead shot (residue in hunted game)	oral	inadequate (chronic)
Lead shot (firing range)	inhalation	112- 238 µg/m³ (acute)
Lead sheet, external run-off	oral	inadequate (chronic)
Lead sheet, child contact	dermal / oral (hand)	inadequate (100-1000 µg per contact event?)
Lead sheet ornaments, child contact	dermal / oral (hand)	inadequate (100-1000 µg per contact event?)
Ceramics*, current EU	oral	4.3 – 49.8 µg/day
Ceramics, current ISO	oral	4.8 – 13.7 µg/day
Ceramics, import (non-ISO)	oral	> 1 mg Pb / L can result in beverages or acidic sauces (chronic)
Crystal glass, stemware	oral	6 µg/day (chronic)
Crystal glass decanter, short term storage	oral	2 µg/day (chronic)
Crystal glass decanter, long term storage	oral	up to, or greater than 1 mg Pb /L in beverages (acute)

*Ceramics are distinguished by their functional nature and/or the leachate limits to which the products comply. Decorative ceramics (vases, art objects) are distinguished from articles used in the preparation and serving of food and thus offering the possibility of oral lead exposure. "Current EU" refers to food contact products manufactured to be in compliance with EU leachate test standards and "Current ISO" refers to food contact products in compliance with more stringent ISO test standards. "Import (non-ISO)" refers to generally illegally imported food contact items that may have improperly fired glazes and very poor leach test properties.

A 2.5: Nickel

According to the RARs for Nickel and its compounds, consumer exposure to Nickel was assessed in detail in the EU RAR, and typical and worst-case exposure estimates were taken forward to the risk characterisation, based on the following scenarios:

(i) Dermal exposure: Nickel metal-items in direct and prolonged contact with the skin (*e.g.,* jewellery, eye glasses, belt buckles, watches) and items in direct and frequent contact with the skin (*e.g.,* coins, keys). The toxicological endpoint of concern for consumer exposure via the dermal route is dermatitis. Therefore, exposure is expressed as external exposure. Due to a lack of data for consumer exposure, cross-reading from professional dermal exposure is used. 0.04 mg Ni /day (typical) and 0.120 mg Ni/day (worst-case) for coins and tools are used as the basis for the evaluation of dermal consumer exposure. The exposure is for both hands (surface area 840 cm^2).

(ii) Oral exposure to Nickel from food and drinking water: The total amounts of Nickel intake from dietary food and water have been estimated by Council of Europe (2001) and UK EGVM (2003). The UK EGVM have calculated the daily intake from food and water as 0.21 mg/day for the food intake (the 95.5^{th} percentile of available data) and the water intake as 0.04 mg/day, calculated on the basis of a Nickel concentration of 20 µg/l and a drinking water intake of 2 litres/day. This gives a total intake of 0.25 mg/day. The Council of Europe figure is 0.4 mg/day (Council of Europe, 2001).
Estimates for children are very much more uncertain. … The total intake in children is expected to be similar to the estimated adult intake. The average weight for a 3 – 12 year old boy or girl is 28 kg and for an adult is 72 kg (US EPA, 1997). This would mean that the Nickel intake from food and water expressed as mg/kg for a child would be roughly twice that of an adult.

(iii) Oral exposure to Nickel released from food contact materials (e.g. pots, utensils), drinking water supply (*e.g.,* faucets, pipes), and orthodontic materials. To construct a reasonable worst-case scenario for Nickel exposure from drinking water supplies, it is assumed, that a 60 kg person drinks a total of 2 L of water from a double grip mixer tap made of Chromium plated brass. Three times daily this person drinks one glass of water (0.2 L which approximately corresponds to the volume of water standing in the tap, fittings and valves) which has been standing in the mixer for 8 hours. The remaining 1.4 L of water is assumed to have been standing in the mixer for only 30 minutes. Based on experimental studies supporting this scenario, a reasonable worst-case exposure to Nickel from water installations of 3 µg Ni /kg bw/day is derived and taken forward to the risk characterisation. Taking into consideration the very high concentrations measured in two test-rig experiments, i.e. the fact that newer installations release more Nickel than older once, it cannot be excluded that consumers for certain periods of their lives may be exposed to even higher levels of Nickel arising form drinking water installations.

(iv) Oral exposure to Nickel sulphate (hexahydrate) in vitamin supplements. The RAR on Nickel sulphate summaries the availability of Nickel via multi-vitamin/mineral dietary supplements. In some EU member states Nickel containing supplements are available (with up to 5 µg as a recommended daily dose). In other countries Nickel does not seem to be included in dietary supplement products. An estimation of a typical or worst-case exposure via this route is not conducted. Note: the current WHO TDI for Nickel is 11 µg Ni/kg/day.

(v) Local and systemic exposure to Nickel release from surgical implants. In the EU, the materials used as surgical implants are regulated by the three EC Directives on active implantable medical devices, (EEC, 1990a), general medical devices (EEC 1993d) and *in vitro* diagnostic medical devices (EC 1998c). These are so-called “New Approach” Directives, in which technical standards have been developed. Iatrogenic exposure was not considered further in the Risk Characterisation.

The following two tables summarise the consumer exposure risk characterisation for Nickel Metal and Nickel Sulphate.

Nickel Metal – consumer exposure

	Dermal Exposure: Nickel metal- items in direct and prolonged contact with the skin (*e.g.,* jewelry, eye glasses, belt buckles, watches), items in direct and frequent contact with the skin (*e.g.,* coins, keys)	**Oral Exposure**: Nickel released from food contact materials (*e.g.,* pots, utensils) and drinking water supply (*e.g.,* faucets, pipes)	**Local and Systemic Exposure (Iatrogenic Exposure):** Nickel released from surgical implants, non dental medical devices, and orthodontic and dental materials.
Sensitization	➢ Scenario "Direct and prolonged skin contact" (with nickel containing objects, e.g. jewellery): (conclusion i) ➢ Scenario "Metal piercing posts": (conclusion i) ➢ For coins and other nickel releasing objects (e.g. tools): (conclusion ii)	For release of nickel from food contact materials to food and drinking water or release of nickel from taps and fittings to drinking water (conclusion ii) for nickel-sensitized individuals and (conclusion iii) for severely sensitized individuals was reached.	Risk characterization was not done because these scenarios / this route of exposure is already regulated. Such materials are regulated under 3 directives: 1. Active Implantable Medical Devices 2. General Medical Devices 3. In vitro Diagnostic Medical Devices
Dermal Irritation,	(conclusion ii)	NA	NA
Acute and repeated toxicity	(conclusion ii)	(conclusion ii)	
Mutagenicity	(conclusion i on hold)	(conclusion i on hold)	
Carcinogenicity	(conclusion ii)	Old (conclusion i). Results now available for conclusion ii	
Reproductive Effects	(conclusion ii)	(conclusion ii)	

Nickel Sulphate Hexahydrate - – consumer exposure

	Oral Exposure. Nickel sulphate in vitamin supplements [5 µg/day]
Sensitisation	Oral elicitation of dermal response: ➢ Nickel sensitized population (conclusion ii) ➢ Population with severe Nickel sensitization (conclusion iii)
Acute and repeated toxicity	(conclusion ii)
Mutagenicity	(conclusion i on hold)
Carcinogenicity	Old (conclusion i). Results now available for conclusion ii
Reproductive effects	(conclusion ii)

A 2.6: Zinc

Consumer exposure to Zinc metal is only qualitatively assessed in the EU RAR, and is considered to occur through dermal contact with watering-cans, buckets, nails, gutters etc., for which however the exposure is considered negligible because of short contact time and minimal contact with the metal.

However, from a small selection of uses such as paints, impregnating agents, cosmetics and drugstore products, specific exposure estimates for Zinc compounds have been derived, based upon which for the different Zinc compounds a cumulative uptake of approximately 1.6 mg Zn^{2+}/day was taken across to the risk characterisation.

Considering that the daily dietary requirement of Zinc for an average adult varies in a range of 12-17 mg/day, and considering an NOAEL of 50 mg/day, then the cumulative level of consumer exposure to Zinc and Zinc compounds can be considered negligible.

In contrast, Zinc (as gluconate) is taken by a small section of the population as a dietary supplement, partly in daily doses up to 50 mg/day. However, it may safely be argued that this is not a consumer use, but in fact a pharmaceutical one, and therefore does not fall under the category “consumer exposure”.

A 2.7: Tin

Though Tin has so far not been subjected to a ESR RAR or VRA, the following information was made available by industry:

Industry (ITRI Ltd. and APEAL, the Association of European Producers Of Steel for Packaging) has conducted a single centre, double-blind controlled study to assess the tolerability of orally administered canned foodstuffs (tomato soup) containing Tin migrating from the packaging. As test item, tomato soup (250 ml) containing Tin at concentrations of 201 and 267 mg/kg was fed to a group of 23 volunteers. The results of this clinical tolerance study showed that Tin levels up to 267 mg/kg in canned food caused no adverse effects in healthy adult subject.

Reference: Boogaard P.J., Boisset M., Blunden S., Davies S., Teng Jin Ong, Taverne P.: Comparative assessment of gastrointestinal irritant potency in man of Tin(II) chloride and Tin migrated from packaging. *Food and Chemical Toxicology* 41 (2003) 1663-1670.