European Union Risk Assessment Report

DIANTIMONY TRIOXIDE

CAS No: 1309-64-4

EINECS No: 215-175-0

RISK ASSESSMENT

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RISK ASSESSMENT

November 2008

Sweden

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Final report: **2008**

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Foreword

This Draft Risk assessment Report is carried out in accordance with Council Regulation (EEC) 793/93[1] on the evaluation and control of the risks of "existing" substances. "Existing" substances are chemical substances in use within the European Community before September 1981 and listed in the European Inventory of Existing Commercial Chemical Substances. Regulation 793/93 provides a systematic framework for the evaluation of the risks to human health and the environment of these substances if they are produced or imported into the Community in volumes above 10 tonnes per year.

There are four overall stages in the Regulation for reducing the risks: data collection, priority setting, risk assessment and risk reduction. Data provided by Industry are used by Member States and the Commission services to determine the priority of the substances which need to be assessed. For each substance on a priority list, a Member State volunteers to act as "Rapporteur", undertaking the in-depth Risk Assessment and recommending a strategy to limit the risks of exposure to the substance, if necessary.

The methods for carrying out an in-depth Risk Assessment at Community level are laid down in Commission Regulation (EC) 1488/94[2], which is supported by a technical guidance document[3]. Normally, the "Rapporteur" and individual companies producing, importing and/or using the chemicals work closely together to develop a draft Risk Assessment Report, which is then presented at a Meeting of Member State technical experts for endorsement. The Risk Assessment Report is then peer-reviewed by the Scientific Committee on Toxicity, Ecotoxicity and the Environment (CSTEE), now renamed Scientific Committee on Health and Environmental Risks (SCHER) which gives its opinion to the European Commission on the quality of the risk assessment.

This Draft Risk Assessment Report has undergone a discussion in the Competent Group of Member State experts with the aim of reaching consensus by interpreting the underlying scientific information, or including more data. The Competent Group of Member State experts seek as wide a distribution of these drafts as possible, in order to assure as complete and accurate an information basis as possible. The information contained in this Draft Risk Assessment Report does not, therefore, necessarily provide a sufficient basis for decision making regarding the hazards, exposures or the risks associated with the priority substance.

This Draft Risk Assessment Report is the responsibility of the Member State rapporteur. In order to avoid possible misinterpretations or misuse of the findings in this draft, anyone wishing to cite or quote this report is advised to contact the Member State rapporteur beforehand.

[1] O.J. No L 084, 05/04/199 p.0001 – 0075

[2] O.J. No L 161, 29/06/1994 p. 0003 – 0011

[3] Technical Guidance Document, Part I – V, ISBN 92-827-801 [1234]

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OVERALL RESULTS OF THE RISK ASSESSMENT[4]

CAS Number: 1309-64-4
EINECS Number: 215-175-0
IUPAC Name: Diantimony trioxide

Environment

Aquatic compartment

Surface water

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Sediment

Conclusion (iii) There is a need for limiting the risks; risk reduction measures which are already being applied shall be taken into account.

This conclusion applies to the generic scenarios for formulation and application of flameretardant textile back-coating and to one production site (site P1).

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all other scenarios, including nineteen sites using diantimony trioxide in textile applications and three production sites, that all report releases.

Waste water treatment plants

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

4 Conclusion (i) There is a need for further information and/or testing.
Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.
Conclusion (iii) There is a need for limiting the risks; risk reduction measures which are already being applied shall be taken into account.

Terrestrial compartment

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Atmosphere

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Secondary poisoning

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Marine compartment

PBT-assessment

There is currently no agreed approach to perform a PBT-assessment of a metal, therefore a PBT-assessment will not be performed.

Marine water

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Despite having RCR >1 conclusion iii) is not drawn for application of flame-retardant back-coating. The reason for this is that, according to information from IAOIA, none of the sites covered by the survey IAOIA performed to collect exposure data from all their customers is located by the sea. However, it has to be pointed out that the coverage of this survey regarding textile backcoating sites was rather low. Therefore, it cannot be ruled out that textile backcoating sites located at the sea having emissions to the marine environment may exist.

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Marine sediment

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Despite having RCR >1 conclusion iii) is not drawn for formulation and application of flame-retardant back-coating. The reason for this is that, according to information from IAOIA, none of the sites covered by the survey IAOIA performed to collect exposure data from all their customers is located by the sea. For the formulation of flame-retardant in textiles, the coverage of this survey is high and there is a high probability that for this use area the marine scenario may not be relevant. For application of textile back-coating on the other hand the coverage of the survey is lower and it cannot be ruled out that sites located at the sea having emissions to the marine environment may exist.

Secondary poisoning in the marine environment

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Human health

Human health (toxicity)

Workers

Conclusion (iii) There is a need for limiting the risks; risk reduction measures which are already being applied shall be taken into account.

Conclusion (iii) applies to skin irritation for all scenarios to indicate the need for classification. Once classified the conclusion (iii) will be changed to conclusion (ii).

Conclusion (iii) also applies to repeated dose toxicity (local pulmonary toxicity after inhalation) and carcinogenicity (pulmonary carcinogenicity) for the following scenarios: **Production of Diantimony Trioxide:** Conversion, Refuming and Final handling with and without RPE, **Use as a catalyst in production of PET**: Powder handling, **Use as flame-retardant in production of plastics**: Raw material handling, **Use as flame-retardant in treated textiles**: Formulation, **Use in pigments, paints, coatings and ceramics:** Loading and mixing, **Use as flame-retardant in production of rubber:** Formulation and Processing.

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

Conclusion (ii) applies to all other endpoints.

Consumers

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

Conclusion (ii) applies to all endpoints.

Humans exposed via the environment

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

Conclusion (ii) applies to all endpoints.

Combined exposure

The most important sources of human exposure to diantimony trioxide are probably identified. Additions of individual scenarios are not considered to change any of the conclusions.

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Human health (physico-chemical properties)

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

Conclusion (ii) applies to flammability and explosive and oxidising properties. These properties are not considered to form a hazard hence further characterisation is not undertaken in this report. In addition, there is no need for further information and/or testing with regard to physico-chemical properties.

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EUSES Calculations if available can be viewed as part of the report at the website of the European Chemicals Bureau: http://ecb.jrc.ec.europa.eu/

TABLES

FIGURES

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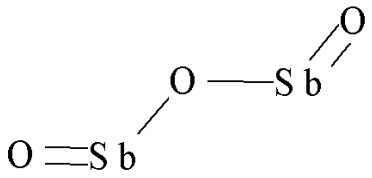
1 GENERAL SUBSTANCE INFORMATION

1.1 IDENTIFICATION OF THE SUBSTANCE

Antimony is a metalloid that belongs to group 15, period 5 of the periodic table of the elements. Oxidation states of antimony include –3, 0, +3, and +5, where the two latter, i.e. +3 and +5, are the two predominant environmental ones.

CAS No: 1309-64-4
EINECS No: 215-175-0
IUPAC Name: Diantimony Trioxide
Molecular formula: Sb_2O_3
Structural formula:



Molecular weight: 291.52
Synonyms: Antimony (III) oxide
Antimony (3+) oxide
Antimony oxide (Sb2O3)
Antimony peroxide
Antimony trioxide
Antimony oxide
Antimony sesquioxide
Antimony white
Flowers of antimony
Senarmontite
Valentinite
Sesquioxide
C.I. Pigment White 11
C.I. 77052

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1.2 PURITY/IMPURITIES, ADDITIVES

A survey on the purity of commercial grades of diantimony trioxide on the EU-market has been made by a consultant on behalf of the International Antimony Oxide Industry Association, IAOIA (EBRC, 2006k). The purity for diantimony trioxide was given as 99.3 to 99.5 % (with the exception of wetted forms, for which a lower specification limit of 95 % was given). The only two relevant impurities are arsenic and lead. As of June 2006 all diantimony trioxide used within the EU will contain less than 0.1 % As (before this date approximately 3.6 % of the diantimony trioxide used in the EU contained between 0.1 and < 0.2 % As, the

rest < 0.1 % As). The range given by EU producers is 0.0040 to 0.0860 % As. The content of lead in diantimony trioxide used in the EU is less than 0.25 % Pb.

Other impurities occurring in trace amounts: Cu, Fe, Ni, SO_4^{2-}, Si, Mn, Mg, Sn, Al, Ag, Cd, Bi, V, and Se.
The impurities present primarily depend on the geographical mineralogy from which the raw material is derived.

1.2.1 Additives

The commercially available form of diantimony trioxide has no stated additives.

1.3 PHYSICO-CHEMICAL PROPERTIES

The physico-chemical properties of diantimony trioxide are summarised in Table1-1.

Table1-1 Summary of physico-chemical properties

Property	Value	Comment	Reliability score
Physical state	Solid	The commercial product is a white, odourless, crystalline powder	
Melting point	655°C,	Budavari, 1996	4*
Boiling point	1550°C (1013 hPa), 1425°C (1013 hPa)	Gangolli, 1999, Budavari, 1996	4** 4*
Specific density	5.9 g/cm^3 (at 24ºC)	Smeykal, 2005 Density differs from crystalline structure.	1
Vapour pressure	1 mmHg (~133 Pa) at 574°C	Budavari, 1996	4*
Water solubility			2
Distilled water	pH 5: 19.7 mg Sb_2O_3 / l pH 7: 25.6 mg Sb_2O_3 / l pH 9: 28.7 mg Sb_2O_3 / l (at 20ºC)	UMWELTANALYTIK GMBH, 1993)	
Reconstituted standard water, 7 days	pH 8: 2.76 mg Sb/l (at 22.2ºC)	LISEC WE-14-018 Loading 100 mg Sb_2O_3/l	
Partition coefficient	Not relevant		
Granulometry	0.2-13.89 µm (particle size) 0.92-5.96 µm (D50)	Weidenfeller, 2005 Franke, 2005	1 1
Flash point	No data		
Autoflammability	No data		
Flammability	No data		
Explosive properties	No data		
Oxidizing properties	No data		
Heat of Vaporization	17.82 kcal/mol	Budavari, 1996	4*

Property	Value	Comment	Reliability score
Index of Refraction	2.087 - Senarmontite 2.18, 2.35 - Valentinite	Budavari, 1996	4*

* This reference refers to "The Merck Index" which is a peer-reviewed handbook of collected physico-chemical data.

** This reference refers to " The dictionary of substances and their effects" which is a peer-reviewed handbook of for instance collected physico-chemical data.

1.3.1 Physical state

The commercial product is a white, odourless, crystalline powder.
Diantimony trioxide has two molecular arrangement (Grund and Hanusch, 2000; Budavari, 1996; Kirk-Othmer, 1992a):

- Senarmontite [CAS No. 12412-52-1] below 570°C - colourless cubic crystals (Figure 1-1).

- Valentinite [CAS No 1317-98-2] above 570°C – white orthorhombic crystals which becomes yellow when heated but turns white again on cooling (Figure 1-2).

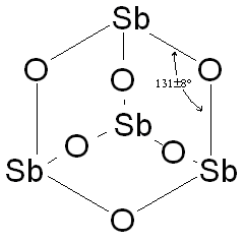

Figure 1-1 Senarmontite

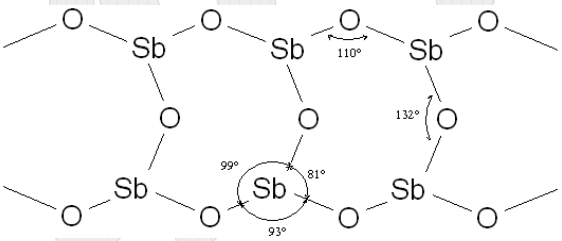

Figure 1-2 Valentinite

At higher temperatures, the stable form is the orthorhombic valentinite, which consists of infinite double chains. The orthorhombic modification is metastable below 570°C; however it is sufficiently stable to exist as a mineral.

Since diantimony trioxide can and will exist in both these modifications at environmental conditions, and no data are available to differentiate between the two as regards exposure and effects, the intention of the RAR will be to cover both with the CAS Number for diantimony trioxide, i.e. 1309-64-4.

1.3.2 Melting point

A melting point of 655°C is reported (Budavari, 1996).

1.3.3 Boiling point

The boiling point of diantimony trioxide is quoted as 1550°C (Gangolli, 1999; IUCLID), however there is also a reported boiling point of 1425°C (Budavari, 1996; Kirk-Othmer, 1992a).

1.3.4 Density

The specific density of diantimony trioxide has been reported to be 5.9 g/cm^3 at 24ºC (Smeykal, 2005).

Density differs from crystalline structure:

– *Senarmontite:* 5.2 g/cm^3 (Budavari, 1996; Kirk-Othmer, 1992a) 5.252 g/cm^3 (Grund and Hanusch, 2000)

– *Valentinite:* 5.67 g/cm^3 (Kirk-Othmer, 1992a), 5.72 g/cm^3 (Grund and Hanusch, 2000).

1.3.5 Vapour pressure

The vapour pressure of diantimony trioxide has been reported to be 1 mm Hg at 574°C (Budavari, 1996) which approximately corresponds to 133 Pa..

1.3.6 Solubility

1.3.6.1 Water solubility

Information on water solubility of Sb_2O_3 is limited. There are no studies performed according to the OECD guidelines. The available data contain one study that has measured solubility of the substance in distilled water and ten studies that have investigated solubility of diantimony trioxide in reconstituted standard water. Overall, the temperature in these studies varied between 17.8 – 23.6ºC.

1.3.6.1.1 Solubility of Sb_2O_3 in distilled water

The solubility of Sb_2O_3 was determined at pH 5.0, 7.0 and 9.0. Ten grams of the substance was mixed with 100 ml distilled water. The pH was adjusted and the solution agitated for 24 hr at 20°C. The resulting solid matter was filtered off and the Sb content of the filtrate was determined by the hydride-AA technique. The values reported are calculated as Sb_2O_3. The solubility of Sb_2O_3 at the different pH-values was: 19.7 mg Sb_2O_3/l at pH 5, 25.6 mg Sb_2O_3/l at pH 7, and 28.7 mg Sb_2O_3/l at pH 9 (UMWELTANALYTIK GMBH, 1993) performed at 20°C.

1.3.6.1.2 Solubility of Sb_2O_3 in reconstituted standard water (ISO 6341)[5]

The solubility of Sb_2O_3 has been measured in: (i) five 24-hr Screening tests transfomation/dissolution, i.e. WE-14-012e at 22ºC (LISEC, 2000), WE-14-030 at 20.6-20.65ºC (LISEC, 2002b), WE-14-021 at 20.7-20.8ºC (LISEC, 2002d), WE-14-018 at 23.6 ºC (LISEC, 2002e), and WE-14-020 at 17.8ºC (2002f); (ii) in three 7 days full tests WE-14-018 at 22.2ºC, (LISEC, 2002e) WE-14-020 at 17.8ºC (LISEC, 2002f) and CanMET, 2004 at 22ºC (Skeaff and Hardy, 2004) and in (iii) a 28 days full test WE-14-020 at 17.8ºC (LISEC, 2002f) (following the Transformation/Dissolution Protocol (GHS, 2005) In these tests the solubility is determined by measuring total dissolved concentration.

1.3.6.1.3 pH dependent dissolution pattern of Sb_2O_3

The test WE-14-021 (2002d) was performed in the ISO 6341 medium at seven different pH levels (i.e. from pH 1 to pH 10) performed at 20.7-20.8ºC. The loading of Sb_2O_3 was 100 mg l^{-1}. The solutions were agitated for 24 hr at 100 rpm. This does not fulfil the protocol, which states that the solutions should be agitated rapidly and vigorously. However, the speed of agitation applied in the tests seemed to have little influence on the results. Sampling and Sb analyses were carried out after 24 hr. The dissolved Sb concentration after 24 hr is given in Table1-2, where also the actual pH values (for pH 7 and pH 8) that were measured during the test are presented

Table1-2 Concentration of dissolved Sb (mg/l)) in reconstituted standard water (ISO 6341) after 24 hr (WE-14-021 (LISEC, 2002), WE-14-018 (LISEC, 2000) with a loading of 100 mg Sb_2O_3/l.

Test	pH	pH measured (mean)	Concentration of dissolved Sb (mg/l)
WE-14-021	1		4.37
	3		2.18
	5		1.11
	6		0.858
	7	0 hr = 7.01 24 hr = 6.98	0.618
WE-14-018	8	0 hr = 8.38 24 hr = 8.27	1.86
WE-14-021	10		2.16

The results from the study reveal the following dissolution pattern of Sb_2O_3 (see Figure 1-3).

[5] Reconstituted standard water (i.e. ISO 6341 medium) is used in the Transformation/Dissolution Protocol that aims to determine the rate and extent to which metals and sparingly soluble metal compounds can produce soluble available ionic and other metal-bearing species in aqueous media at the rate of concern under environmental conditions. This medium is sterilised by filtration (0.2 µm) before use in the test.

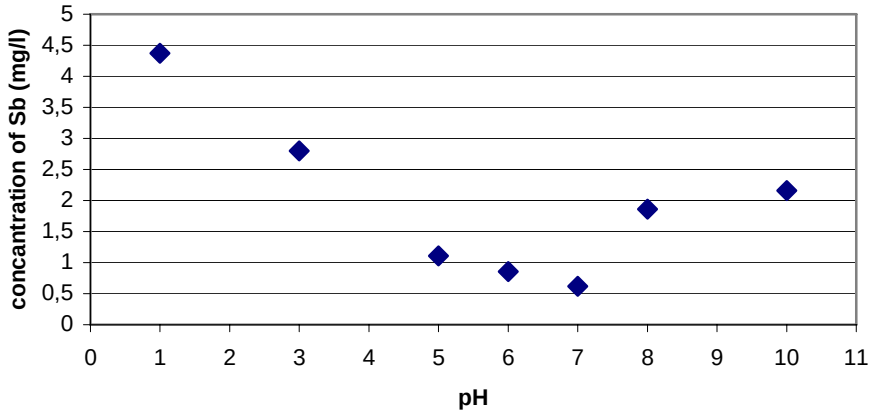

Figure 1-3 Mean dissolved Sb concentration after 24 hr in function of the pH.

According to the results from the test, dissolution of the substance in the test medium decreases constantly from pH 1 to pH 7. Above pH 7 the trend changes and the solubility of Sb_2O_3 increases rapidly to pH 8, where a new equilibrium is established and the increase in solubility becomes much slower.

1.3.6.1.4 Dissolution of Sb_2O_3 between pH 7 and pH 8

Ten studies were performed in reconstituted standard water (ISO 6341) for 24 hr, 7 days and 28 days with high, medium and low loadings to investigate water solubility of the substance in the pH range 7 to 8. The results and pH conditions during these tests are reported in Table1-3.

Table1-3 Dissolved Sb concentration in reconstituted standard water (ISO 6341) after 24 hr , 7 days and 28 days with high, medium and low loadings

Test	Time	Loading (mg Sb_2O_3/l)	rpm	pH (measured, mean)	Dissolved Sb concentration [mg Sb/l]
WE-14-030 (LISEC, 2002b)	24 hr	100	200	0 hr = 7.15 24 hr = 7.16	0.812
	24 hr	100	200	0 hr = 7.52 24 hr = 7.55	1.06
	24 hr	100	200	0 hr = 7.87 24 hr = 7.86	1.54
WE-14-012e (LISEC, 2000)	24 hr	100	200	0 hr = 8.01 24 hr = 8.07	1.86
WE-14-018 (LISEC, 2002e)	24 hr	100	100	0 h = 8.4 24 h = 8.3	**1.86**
	7 d	100	100	0 h = 8.52	**2.76**

				7 days = 8.11	
CanMET (Skeaff and Hardy, 2004)	7 d	10	100	0 h = 7.90 7 d =7.95	0.370
WE-14-020 (LISEC, 2002f)	24 hr	1	100	0 h = 8.06 24 h = 8.06	0.016
	7 d	1	100	0 h = 8.06 7 d = 7.90	0.058
	28 d	1	100	0 h = 8.06 28 d = 7.90	0.118

These results confirm the solubility pattern of Sb_2O_3 that has been revealed by the study WE-14-021 (LISEC, 2002) performed at a broad range of pH-values (i.e. the increase of solubility of diantimony trioxide from pH 7 - the lowest concentration of free Sb ions- with increasing pH).
Besides that, graphical plotting of the actual pH conditions recorded during the 24-hr screening tests against concentrations of free Sb ions achieved at corresponding pH levels (see Figure 1-4) produces a line showing, in more detail, dissolution pattern of the substance between pH 7 and 8.

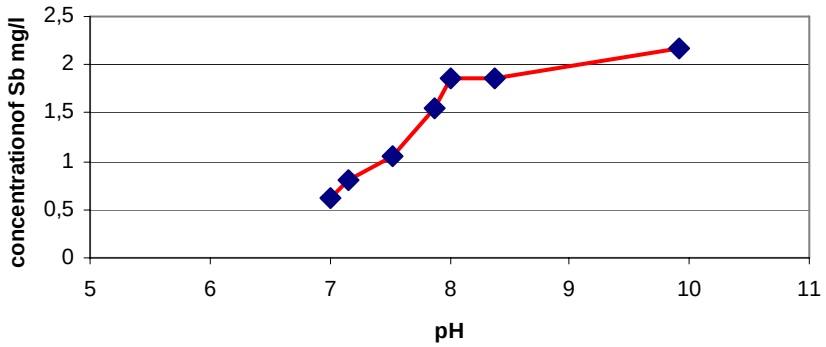

Figure 1-4 Mean dissolved Sb concentration after 24 hr at pH interval 7-8 (10)

Again, in correspondence with Figure 1-3 the results presented in Figure 1-4 show changes in dissolution trends around pH 7 (i.e. rapid increase of solubility) and pH 8 (i.e. reduction of the increase rate of solubility). The speed of agitation applied in the tests seemed to have little influence on the results.

The results from the WE-14-018 test (7-days Full test; loading of 100 mg Sb_2O_3/l) reveal the following dissolution pattern of Sb_2O_3 (see Figure 1-5).

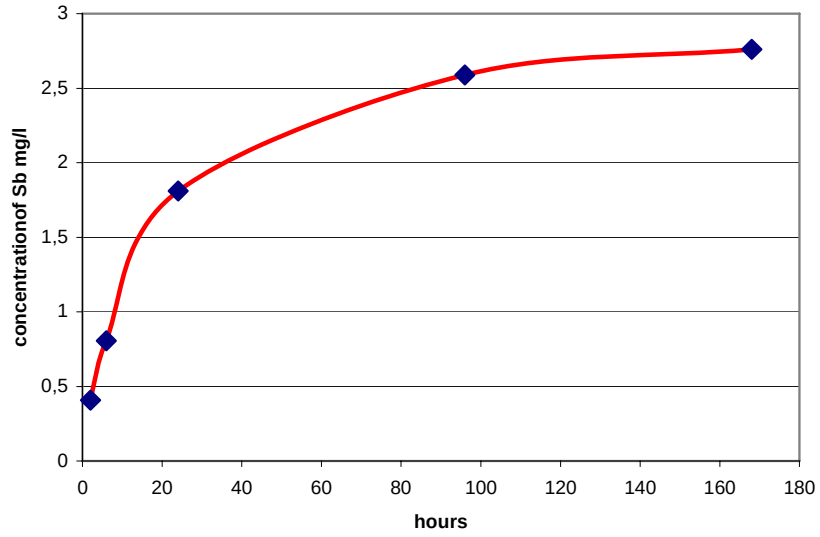

Figure 1-5 Transformation/dissolution curve for the 7-days test in reconstituted standard water (ISO 6341).

The results from the 28-days full test; loading of 1 mg Sb_2O_3/l) reveal the following dissolution pattern of Sb_2O_3 (see Figure 1-6).

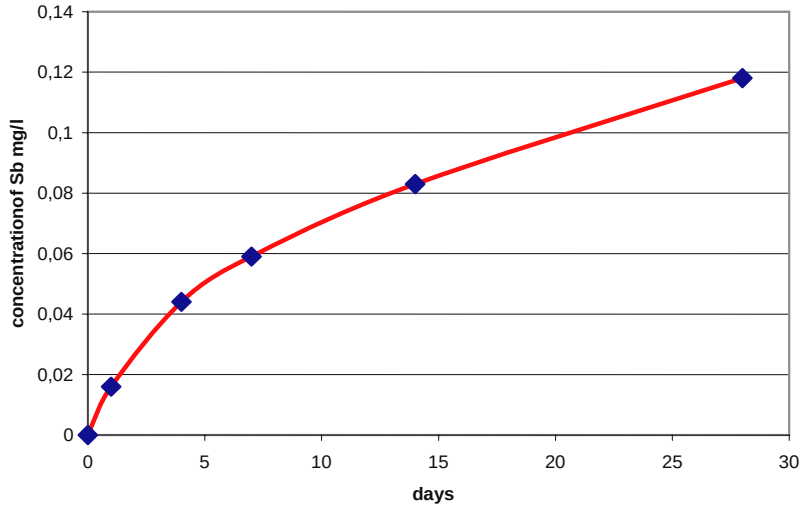

Figure 1-6 Transformation/dissolution curve for the 28-d test in reconstituted standard water (ISO 6341).

1.3.6.1.5 Selection of water solubility for modelling purposes

The results from the water solubility study in distilled water do not correspond to the results with the studies that measured water solubility of the substance in reconstituted standard water (ISO 6341). The study performed in distilled water suggests that the water solubility of Sb_2O_3 increases with the increasing pH over the range of 5 to 7 (and 9). The opposite picture

was revealed by the tests in the ISO medium. In addition it is difficult to conclude from the test report why the study performed in distilled water resulted in so much higher water solubility of Sb_2O_3.

The discrepancy between water solubility studied in distilled water and in reconstituted standard water (ISO 6341) might be explained by the larger Ca concentration (2mM) in the reconstituted standard water and precipitation of $Ca[Sb(OH)_6]_2$. Upon dissolution in oxic systems Sb(III) is easily oxidized to Sb(V), which easily hydrolyse and form the anion $Sb(OH)_6^-$ (see also section 3.1.3.1.2, aquatic transformation). (Johnson et al. 2005) present a solubility product (Kso = $[Ca^{2+}][Sb(OH)_6]_2$) of 10-12.55, which predicts a maximal Sb concentration of 0.012 mM or 1.44 mg Sb/l at 2 mM Ca. This corresponds to 1.73 mg Sb_2O_3 /l which is remarkably close to the maximum solubility observed in the ISO 6341 medium. Since the ISO medium is considered more relevant for natural conditions, the value for water solubility resulting from the tests conducted in ISO medium will be used for modelling purposes.

For modelling purposes a water solubility of 2.76 mg Sb/l will be used in the risk assessment.

1.3.6.2 Solubility in other solvents

Diantimony trioxide is insoluble in organic solvent (Kirk-Othmer, 1992a).

1.3.7 Granulometry

Weidenfeller reported particle sizes from 0.2 to 13.89 µm. Franke reported D50s of 0.92 to 5.96 µm. An assessment of the dustiness and particle size distribution of diantimony trioxide relevant for the potential inhalation toxicity has been made. (EBRC, 2006k) and the study and the results are presented and discussed in Chapter 4.

1.3.8 Autoflammability

An entry of autoflammability (year 1987) is included in IUCLID, but no value is reported. (IUCLID)

1.4 CLASSIFICATION

1.4.1 Current classification

Diantimony trioxide is classified as a dangerous substance within the meaning of Directive 67/548/EEC and is listed in Annex 1 of this Directive (21st ATP), being assigned the following risk and safety phrases:

Category 3 carcinogen
Xn Harmful

R40	Limited evidence of a carcinogenic effect
S36/37	Wear suitable protective clothing and gloves
S2	Keep out of reach of children
S22	Do not breathe dust

1.4.2 Proposed classification

In addition to the current classification the rapporteur proposes:

Xi; R38 (Irritating to skin)

Rationale for the proposed classification:

The classification proposal is based on practical experience in humans.

The proposed classification for the environment is:

It is proposed by the TC C&L (agreed by the RMS) not to classify as dangerous for the environment.

Rationale for the proposal

This proposal not to classify diantimony trioxide follows the classification strategies for metal compounds (para 9.7.5.3. (GHS, A9, 2003) and ECBI/61/95 Add.51 Rev.4).

1. Lowest effect values for aquatic species

Table1-4 Lowest effect values for aquatic species

Trophic level	Species	Acute toxicity mg Sb/l	Chronic toxicity mg Sb/l
Fish	*Pargus major* *Pimephales promelas*	LC_{50} (96 hr) = 6.9 (measured total)	NOEC = **1.13** (measured total)
Invertebrates	*Daphnia magna* *Hydra (Chlorohydra viridissima - Hydra oligactis*)	LC_{50} (48 hr) = 12.2 (measured total) LC_{50} (72 hr) = **1.77 – 1.95** (measured filtered)	NOEC = **1.74** (measured total)

Algae and aquatic plants	*Raphidocelis subcapitata* *Lemna minor*	EC_{50} (72 hr) > 36.6 (measured total) EC_{50} (96 hr) > 25.5 (measured dissolved)	NOEC (72 hr) = **2.1** (measured total) NOEC (96 hr) = **12.5** (measured dissolved)

The effect assessment has found a 72 hr LC_{50} of 1.77 – 1.95 mgSb/l performed on the invertebrates *Chlorohydra viridissma and Hydra oligactis* as the lowest valid value for acute toxicity. This value is thus the acute reference value used in environmental classification of Sb_2O_3.

2. Water solubility of Sb_2O_3– dissolution pattern

Water solubility of Sb_2O_3 has been measured in three 24-hour screening tests that follow the Transformation/Dissolution protocol (Annex 10, (GHS, 2005). The test WE-14-021 has investigated the dissolution of the substance in a broad range of pH (i.e. between 1 and 10), see

Table1-5.

Table1-5: Concentration of dissolved Sb (mg/l) after 24 hr (LISEC, WE-14-021, WE-14-018).

pH	**Concentration of dissolved Sb (mg/l)**	**pH adjustment/ buffering**
1	4.37	HCl
3	2.18	HCl
5	1.11	HCl
6	0.858	5% CO_2
7	0.618	HCl
8	1.86	Air-buffering
10	2.16	NaOH

The results from the test show the following dissolution pattern of Sb_2O_3 (see Figure 1-7 below):

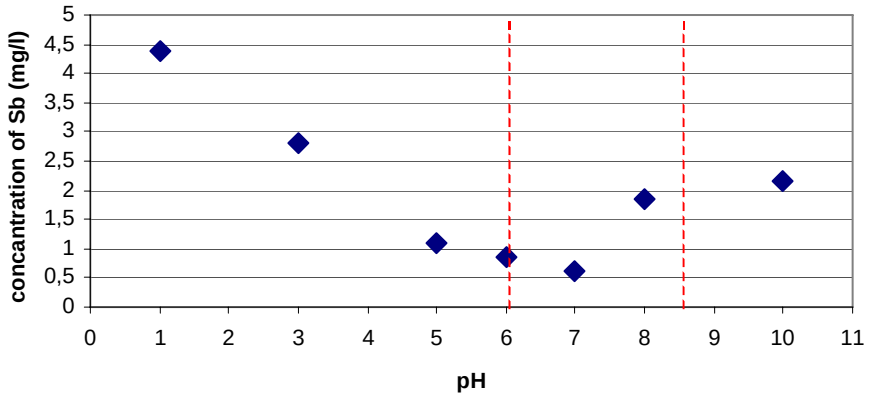

Figure 1-7 Mean dissolved Sb concentration after 24 hr in function of the pH. The red lines represent pH interval relevant for environmental classification.

According to the results from the test, dissolution of the substance in the test medium decreases constantly from pH 1 to pH 7. Above pH 7 the trend changes and the solubility of Sb_2O_3 increases with growing pH. The most dramatic change is observed between pH 7 and 8.

From Figure 1-7 it can be concluded that within the relevant pH range for the 24 hr-screening test (i.e. between 6.0-8.5) the highest concentration of Sb ions would be reached in the upper end (i.e. pH 8-8.5).

3. Water solubility of Sb_2O_3 between pH 7 and 8 (10).

   Except the dissolution tests described above there are additional studies that have investigated water solubility of Sb_2O_3 at pH interval from pH 7 to pH 8 and with a 100 mg/l loading; the results are presented in Table1-6 and Figure 1-8 (below). In order to see the dissolution trend after pH 8 also Sb ions concentration at approximately pH 10 is shown.

Table1-6 Dissolution of diantimony trioxide at pH 7 - pH 8, with a 100 mg/l loading

Test	mg Sb/L	rpm	Measured pH (mean)
WE-14-030	0.812	200	0 hr = 7.15 24 hr = 7.16
	1.06	200	0 hr = 7.52 24 hr = 7.55
	1.54	200	0 hr = 7.87 24 hr = 7.86
WE-14-012e	**1.86**	200	0 hr = 8.01 24 hr = 8.07
WE-14-018 (WE-14-021)	**1.86**	100	0 hr = 8.38 24 hr = 8.27

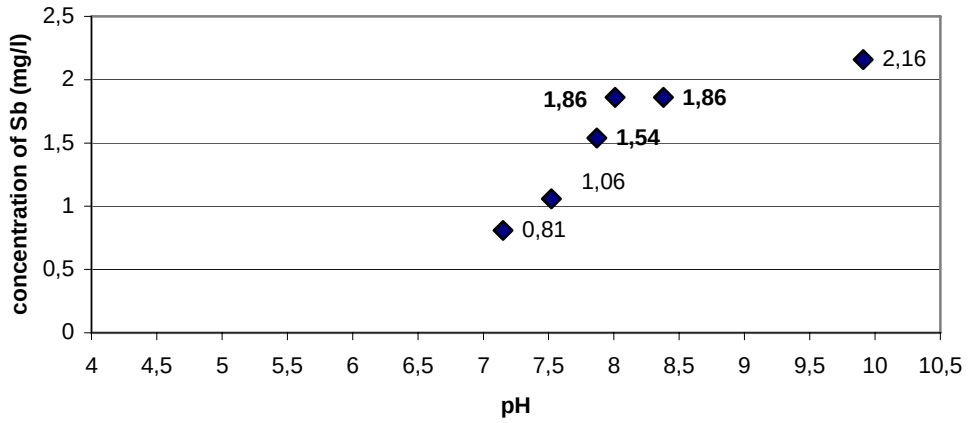

Figure 1-8 Mean dissolved Sb concentration after 24 hr at pH 7-8 (10) interval, with a 100 mg/l loading

The actual/measured pH values were used as the input data for the diagram. The three points (indicated with bold values on the diagram) present results from the tests that were performed as representative for pH 8 (see also table 2 for pH values measured during the tests). According to a common procedure an average of these three results could be taken and used as representative for water solubility of Sb_2O_3 at pH 8.

However, the water solubility of the substance is pH dependent. According to the data there is a rapid increase in solubility between pH 7 and 8. Somewhere at pH 8 this equilibrium changes and the increase in solubility becomes much slower. It is thus reasonable to assume that these points are there not as a result of a laboratory variation from the tests performed at values around pH 8, but they indicate true dissolution trend(s) of the substance (which may also be concluded from Figure 1-7.

4. Comparison of aquatic toxicity data and solubility data

   As mentioned above the highest concentration of Sb ions within the relevant pH interval will be achieved between pH 8 – 8.5 and lies above but close to 1.86 mg/l (see also Figure 1-9).

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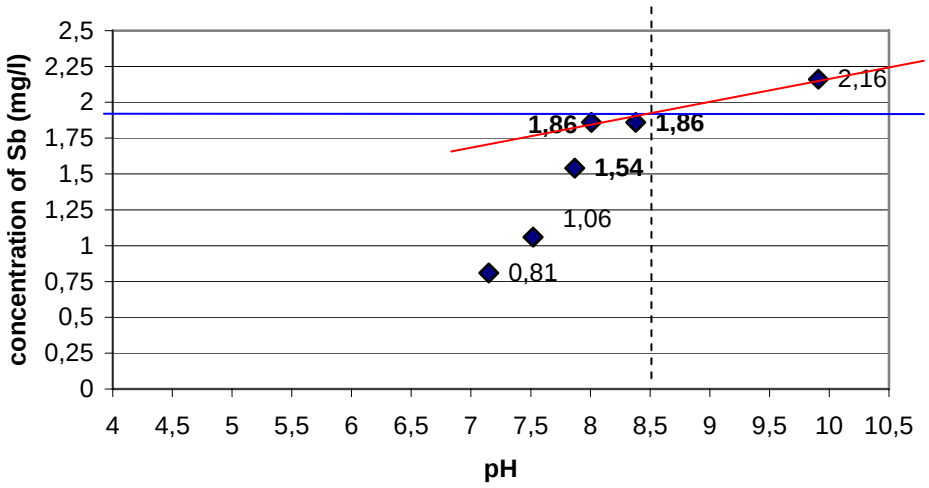

Figure 1-9: Approximate derivation of the highest Sb concentration at pH 8.5 during 24-hr screening test with a loading of 100 mg Sb_2O_3/l

However, for the purpose of classification the value of 1.86 mg Sb/l (achieved at both pH 8 and 8.38) will be used in the comparison of 24-hr Screening test data with short-term toxicity reference data.

As mentioned above the solubility of Sb_2O_3 has also been measured in a 7-days and a 28-days full transfomation/dissolution test (CanMET, 2004, LISEC, WE-14-018 and WE-14-020) following the Transformation/Dissolution protocol (GHS, 2005) is presented in Table1-7.

Table1-7 Concentration of dissolved Sb (mg/l) after 7 days (Lisec, WE-14-018 and CanMET, 2004) and 28 days (Lisec, WE-14-020)

pH	pH measured (mean)	Duration	Loading (mg Sb_2O_3/l)	Concentration of dissolved Sb (mg/l)
8	0 hr = 8.52 7 days = 8.11	7 days	100	2.76
8	7.90 – 7.95	7 days	10	0.370
8	0 hr = 8.06 28 days = 7.90	28 days	1	0.118

For the purpose of classification the values of 2.76, 0.370 and 0.118 mg Sb/l will be used in the comparison of 7-days and 28-days full test data with short-term and long-term toxicity reference data, respectively.

The total dissolved concentration, which contains metal ions + other species as well, of 1.86 mg/l measured at pH 8 is close to or equal to the short-term ecotoxicity reference value, i.e. 1.77 – 1.95 mg Sb/l. It is therefore reasonable to consider dissolution data from the 7- day full test in order to gain more information and better understand the dissolution of the substance in the ISO medium.

A 7-day dissolution study with a loading of 10 mg Sb_2O_3 /l at pH 8 was performed by CanMET (2004). The pH conditions were stable throughout the test (i.e. the measured levels lay in the range 7.90 to 7.95). Its results showed that after 7 days 370 µg Sb/l was measured in the solution. This concentration of free metal ions does not exceed the short-term ecotoxicity reference value. Another 7-day dissolution study (LISEC, WE-14-018) was performed at pH 8 (the initial pH at 0 hours was 8.52 decreasing to 8.11 in the course of 7 days) with the loading of 100 mg Sb_2O_3/l. After 7 days 2.76 mg Sb/l were found in the solution in a concentration of free metal ions that exceeds the short-term ecotoxicity reference value of 1.77 – 1.95 mg Sb/l. This could therefore (in absence of rapid partitioning from the water column) lead to a classification of R52-53. 53 (Harmful to aquatic organisms and may cause long-term adverce effect in the aquatic environment).

5. Use of the 'escape clause' and proposal for not classifying diantimony trioxide

In case of an R52-53 classification an escape clause (leading to the removal of the classification) may be used if the dissolved metal ion concentration at the low loading rate after a total period of 28 days is less than or equal to the long-term NOECs.

A 28-day test (LISEC, WE-14-020) was performed at pH 8 (the initial pH at 0 hours was 8.06 decreasing to 7.91 after 28 days) with loading of 1 mg Sb_2O_3 /l. After 28 days 0.118 mg Sb/l was measured in the solution. This value should be compared to the NOEC value from the organism that has lead to classification, i.e. the *Hydra spp*. This is however very difficult as no such information is neither available nor any standard test may be employed to investigate this.

For this particular case the TC C&L agreed, therefore, to look at the acute to chronic toxicity ratio (ACR) for available data among the three trophic levels. The ACR are for fish approximately equal to 6; for Daphnia approximately equal to 7 and for Lemna above 2. Theoretically, the ACR for Hydra spp. needs to be around 16 or greater if the classification of R52-53 would not be removed due to the escape clause. This is thought to be most unlikely and for this particular case the TC C&L agreed to propose not to classify Sb_2O_3 as dangerous for the environment.

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2 GENERAL INFORMATION ON EXPOSURE

According to the Goldschmidt classification, antimony is classified as a strong chalcophile, i.e. has a strong affinity for sulphur and therefore concentrates in sulphides. The most important antimony ore minerals are the sulphides Stibnite (Sb_2S_3) and Jamesonite ($Pb_2Sb_2S_5$); they are followed by the oxides Valentinite (Sb_2O_3) and Senarmontite (Sb_2O_3). Antimony compounds are often found in ores of metals such as copper, lead, silver, gold, and arsenic, and are a common component in coal and oil. It occurs sparingly as the free metal, usually in association with arsenic, bismuth or silver. (Butterman and Carlin, 2004)

The average concentration of antimony in the earth's crust has been suggested to be approximately 0.2-0.3 mg/kg (Lisk, 1972; Bowen, 1979; Wedepohl, 1995), with much higher concentrations in rocks containing antimony minerals, such as stibnite. The average antimony concentration in soils is about 0.5 mg/kg (Reiman and Caritat, 1998) to 1 mg/kg (Bowen, 1979), but wide ranges have been reported.

Antimony is produced from minerals. The most important mineral source of the metal is Stibnite (Antimony trisulfide, Sb_2S_3), but it is also found in trace amounts in silver, copper and lead ores. China has the largest reserves of Stibnite in the world and accounts for some 80% of the world's 110 200 tonnes mine production (1999). (Roskill Information Services, 2001). Antimony is also present in trace amounts (typically 1 mg Sb/kg) in coal, primarily with the organic matter. (Kirk-Othmer, 1992a)

2.1 LIFE-CYCLE OF DIANTIMONY TRIOXIDE

An overview of the flow of diantimony trioxide in EU is shown in Table 2-1. Quantities used in the figure below are mainly from the industry (EURAS bvba, 2003; IAOIA, 2006b; Docherty, 2001), with information on consumer use from a report by the University of Surrey (University of Surrey, polymer research centre, 1999). The report on consumer use is from 1999, but the total numbers are based on 2002 and 2005 respectively, this is not likely to be of any real significance for this risk assessment. The exact split between different uses is very difficult to establish, hence the numbers in Table 2-1 should be interpreted with caution.

Information on the use and production of diantimony trioxide was received from Cyprus, the Czech Republic, Estonia, Hungary, Latvia, Lithuania, Poland and the Slovak Republic. No production was reported and total imports were less than 1 000 tonnes/year. No uses beside those already found in EU15 were reported (for year 2005). This information is relevant to determine that the situation concerning diantimony trioxide in the new EU MS is very similar to that in EU15. The numbers have not been included in this risk assessment, as it has been deemed to have little impact on the end-result.

Table 2-1 Overview of the flow of diantimony trioxide in EU15, numbers in tonnes/year

	Year 2005	Year 2002		
Import	-	9000		
		⇓		
Production	-	21425	⇒	**Export** 5425

⇓

Formulation to			
Flame-retardant			
PVC, foam, coating	8 800	9 000	
HDPE/LDPE	-	500	
PP	-	1 800	
ABS	-	1 900	
unsaturated polyester resins	-	750	
PBT	-	1 800	
HIPS	-	1 100	
Polyamides	-	1 650	
not specified non-PVC plastic	9 200	-	
rubber	2 200	2 250	
epoxies	-	200	
adhesives, paints, coatings	-	400	
textile back-coatings	1 750	1 800	
other	-	450	
total flame-retardants	**21 950**	**23 600**	
PET resins and fibres	950	650	
Optical, art and other special glass	250	250	
Pigments, paints and ceramics	1 100	500	
Other	450	-	
Total:	**24 700**	**25 000**	

Processing to:			
Flame-retardant textiles	1 750	1 800	
Cables	-	3 250	
Other flame-retarded polymer articles (E & E articles etc)	-	18 150	
Application of adhesives, paints & coatings	-	900	no longer in Sb_2O_3 form when in pigment
Glass screens, optical, art etc	250	250	no longer in Sb_2O_3 form
PET packaging and other articles	550	-	no longer in Sb_2O_3 form
PET fibres	400	-	no longer in Sb_2O_3 form
Total:	**2 950**	**24 350**	

Consumer use:	Year1999
Upholstered furniture	600
Television back-casings	800
Television circuit boards	80

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Business machines in homes		105
Other consumer E&E products		105
Other		5
	Total:	**1 695**

Consumer and/or industrial use:	
Flame-retardant textiles	1 200
PET textiles etc	65
Other flame-retarded polymer articles (E & E articles etc)	17 055
Items coated with flame-retardant adhesives, paints and coatings	900
Optical and art glass articles	250
Cables	3 250
Total:	**22 720**

Waste disposal:	
(See discussion in chapter on environmental exposure)	
Recycling	-
Incineration	-
Landfill	-
Total:	**-**

2.2 PRODUCTION

2.2.1 Introduction to Production

Import of diantimony trioxide into the EU is mainly from China (more than 90 % of imported quantity in 2000) and USA. The quantity of diantimony trioxide imported/exported as a component of finished products, e. g. electrical and electronic articles, is not known.

Global diantimony trioxide production in 2005 was 120 000 tonnes (IAOIA, 2006b), increasing from 112 600 tonnes in 2002, with China producing the largest part (47 %) followed by US/Mexico (22 %), Europe (17 %), Japan (10 %) and South Africa (2 %) and other countries (2 %) (EURAS bvba, 2003).

2.2.2 Production processes

Diantimony trioxide is currently (2006) being produced at four sites in EU15. Two sites ceased production in recent years.

Diantimony trioxide is produced via two routes:

a) Re-volatilizing of crude diantimony trioxide

b) Oxidation of antimony metal

Oxidation of antimony metal dominates in EU. Diantimony trioxide manufacturers typically buy antimony metal on the open market.

There are several processes for the production of crude diantimony trioxide or metallic antimony from virgin material. The choice of process depends on the composition of the ore and other factors. Typical steps include mining, crushing and grinding of ore, sometimes followed by flotation and separation of the metal using pyrometallurgical processes (smelting or roasting) or in a few cases (e.g. when the ore is rich in precious metals) by hydrometallurgical processes. These steps do not take place in EU but closer to the mining location.

2.2.2.1 Re-volatilizing of crude diantimony trioxide

1) Crude stibnite is oxidised to crude diantimony trioxide using furnaces operating at approximately 850 to 1 000°C according to the following reaction:

$$2\ Sb_2S_3 + 9\ O_2 \rightarrow 2\ Sb_2O_3 + 6\ SO_2$$

2) The crude diantimony trioxide from the first step is vaporised and condensed according to the following schematic formula (detailed process conditions are considered confidential):

$$Sb_2O_3\ (solid) + heat \rightarrow Sb_2O_3\ (gas)$$

2.2.2.2 Oxidation of antimony metal

Antimony metal is oxidized to diantimony trioxide in furnaces. The reaction is exothermic. Diantimony trioxide is formed through sublimation and recovered in bag filters (bag house). The size of the formed particles is controlled by process conditions (in furnace and gas flow). The reaction can be schematically described by:

$$4\ Sb + 3\ O_2 \rightarrow 2\ Sb_2O_3$$

(Grund and Hanusch, 2000; EURAS bvba, 2003; Swedish National Testing and Research Institute, SP, 2000).

2.2.2.3 Steps in the production

1) Reduction of oxide or sulphide to antimony metal (not always done on site) if metal is used as raw material.

2) If metal is used it is sometimes, but not always, crushed.

3) Loading of furnace with antimony metal or crude diantimony trioxide.

4) Oxidation of metal to diantimony trioxide in furnace or re-volatization of crude diantimony trioxide.

5) Mixing/blending of master-batches or premixes. These may be dry or in wetted forms (e. g. with ethylene glycol or plasticizers). This step is not done on all production-

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sites. Not all the diantimony trioxide is mixed/blended with other components at the production site.

6) Packaging, typical packaging is big-bags (inner plastic bag typically containing 500 to 1200 kg of product with outer supporting bag) or 25-kg paper or plastic sacks. (EURAS bvba, 2003)

2.3 USES

2.3.1 Introduction

An overview of the main uses of diantimony trioxide in the EU is summarised in Table 2-2 below.

Table 2-2 Use of diantimony trioxide in EU15

Use	Quantity (tonnes/yr) and percentage of total quantity	
	Year 2005 (EURAS, 2006a)	Year 2000 (Docherty, 2001)
Flame-retardant in plastics (except PVC)	9 200 (38 %)	12 800 (51 %)
Flame-retardant in PVC	8 800 (36 %)	9 000 (36 %)
Flame-retardant in rubber	2 200 (9 %)	not specified
Flame-retardant in textiles	1 750 (7 %)	1 800 (7 %)
Catalyst in PET production	950 (4 %)	650 (3 %)
Additive in glass manufacture	250 (1 %)	250 (1 %)
In pigments, paint and ceramics	1 100 (5 %)	500 (2 %)
Total:	24 250	25 000

2.3.2 Use as flame-retardant

Combustion occurs through a chemical reaction that is sustained by free radicals. It is characterised by two principal stages:

1. A heat source initiates degradation of the polymer to yield volatile products of low mass, which migrate to the polymer surface and enter the gas phase.
2. The volatile products are oxidised by free radical reactions (burning), which evolves further heat to produce more volatile components from the polymer

Halogenated (based upon bromine or chlorine) flame retardants break down when heated. This leads to the formation of chlorine and bromine free radicals. These free radicals react with the free radicals formed in the combustion, yielding stable products and thereby terminating the combustion process. Diantimony trioxide acts synergistically through the formation of an antimony halide that scavenges free radicals. The exact mechanism of this synergistic action is not known. (Risk and policy analysts Limited, 2000)

Diantimony trioxide has limited fire-retardant properties of its own, but is an effective synergist for halogenated compounds such as halogenated flame-retardants or polymers containing halogens such as PVC. The addition of diantimony trioxide makes it possible to reduce the amount of halogenated flame-retardant that is added. Antimony pentoxide (Sb_2O_5) and sodium antimonite ($Na[Sb(OH)_6]$) are also used as synergistic flame-retardants. The synergistic effect of diantimony trioxide with organic halogen compounds was discovered in the 1930's. (Kirk-Othmer, 1992b, 1993a; Slooff et al., 1992)

The consumption of diantimony trioxide in flame-retardant applications per country in EU is shown in Table 2-3 below (IAOIA, 2006b).

Table 2-3 Consumption of diantimony trioxide in flame retardant application per country in EU Europe (2005)

Country	Consumption (%)
Germany	16
Belgium	6
Switzerland	2
Spain	5
France	22
Finland	2
Italy	22
Netherlands	4
Sweden	4
UK	12
Other	3

2.3.2.1 Use as flame-retardant in plastics and rubber

According to EuPC, the European Plastics Converters, there are more than 27 000 companies in EU specialised in processing plastics. The processing of plastics and rubber containing diantimony trioxide does not require specialised equipment. Diantimony trioxide is used as flame-retardant in rubber by approximately 40 companies, in non-PVC plastics by approximately 220 companies and in PVC by approximately 200 companies (EBRC, 2006c; EBRC, 2006g; EBRC, 2006e).

Flame-retarded plastics are typically used in electrical and electronic equipment, in cables, in automotive parts, in some building materials and in some packaging. The industry estimates that 12 % (210 000 tonnes) (APME, Association of Plastic Manufacturers in Europe, 2001b) or in another source 30 % (450 000 tonnes) (APME, Association of Plastic Manufacturers in Europe, 2002) of the plastics used in the electrical and electronic sector contains flame-retardants, mainly in PC´s, monitors, printers, copiers, TV's and small and large household appliances. The latter source estimates that some 59 % of the plastics use non-halogenated flame-retardants and 41 % use halogenated. The flame-retardant system used depends on a number of considerations. Diantimony trioxide is only a synergist with halogenated flame-

retardants; consequently it cannot be assumed that all flame-retardant systems in use contain diantimony trioxide.

Flame-retardants are used in cables and paints to prevent the conversion of a spark into a fire and subsequently to prevent the spread of a fire throughout a structure along the wiring or paint.

PVC is flame-retardant in itself, addition of diantimony trioxide improves the flame-retardant properties. The flame-retardant properties are reduced as non-flame retardant plasticizers are added.

Flame-retarded rubber is used in some automotive items (e. g. matting) electrical equipment and in some special industrial rubber applications where fire risk reduction is critical, e. g. conveyor belts in coalmines. Flame-retardant systems without diantimony trioxide (e. g. aluminium trihydrate) are typically used.

The amount of diantimony trioxide added depends on several factors such as type of halogenated compound and polymer, required physical properties of the final polymer, flame retarding requirement, cost considerations etc. Typical content in the final polymer is up to 8 % but percentages up to 25 % are mentioned. Diantimony trioxide is incorporated into the polymer but is present as a separate phase. The potential for migration of diantimony trioxide out of the polymer is discussed in the exposure part of the chapter on human health. (Kirk-Othmer, 1993b; Kirk-Othmer, 1992b; Slooff et al., 1992). Where tetrabromobisphenol A is used as an additive flame retardant, it is generally used with diantimony oxide for maximum performance. Diantimony trioxide is generally not used in conjunction with tetrabromobisphenol A in reactive flame retardant applications (where tetrabromobisphenol A becomes covalently bound to epoxy or polycarbonate resins), e. g. in printed circuit boards. (RAR, 2003)

Manufacturing of cables and other plastic products principally involve two main steps, mixing and forming. The mixing is either of granules/powder or powder additives to e g rubber slabs. The forming involves heating and a forming operation that depends on polymer and article produced. In practice diantimony trioxide is normally mixed into the polymer compounds as powder additives, either directly or by first making a master-batch of this and other components. The powder is sometimes supplied in pre-dosed bags that are added without opening to the mixing process. During forming e g extrusion, the temperature of the polymer may be up to 300 °C, but may be much lower depending on type of polymer. The forming is done in partially closed systems. The time at which the polymer is subject to high temperatures is kept as short as possible to avoid degradation of the material.

The life expectancy for plastic articles varies widely depending on application and user. Because of the scarce information, no attempt has been made to estimate the service life for the various products containing plastic with diantimony trioxide. For the emission assessment it has been assumed that the amount of diantimony trioxide in use is constant over time (see section 3.1.2.2.3).

Standard grades of diantimony trioxide for this application have particle sizes of 1 to 2 μm, but most producers have finer grades of around 0.5 μm. The latter can give benefits in engineering polymers in terms of impact strength. Coarser grades of 2.5 to 10 μm have benefits mainly in darker, highly coloured or pigmented PVC formulations because of their lower tinting properties (Docherty, 2001).

2.3.2.2 Use as flame-retardant in textiles

Approximately 30 companies in EU15 use diantimony trioxide to produce flame-retarded textiles (EBRC, 2006f).

Flame-retarded textiles are used in textiles that are used in vehicles, in protective clothing, mats, curtains, upholstered furniture, tents, canvas, straps etc. Requirements vary depending on national legislation. Flame-retardant systems without diantimony trioxide are also used in textiles. (Kirk-Othmer, 1996)

Textiles are given flame-retardant properties through a number of different approaches. Diantimony trioxide is used in back-coating, where a fire-resistant layer is attached to one side of the finished textile. This textile is then typically used in textile-covered articles (e g furniture, mattresses). Flame-retarded textiles typically contain 4 to 6 percent of diantimony trioxide, with content in the (dry) back-coating of up to 24 % (when a flame-retardant system including diantimony trioxide is used) (European IPPC bureau, 2002; EBRC, 2006f).

2.3.3 Use in glass

Five glass manufacturers in EU25 use diantimony trioxide. Sodium antimonate is replacing diantimony trioxide in this sector and is more commonly used than diantimony trioxide (EBRC, 2006b).

Diantimony trioxide is used as fining agent or as a degasser in the manufacture of art, optical and fluorescent light bulb glass, and in glass for screens for television and computers etc. It is added to the glass during manufacture and helps the removal of bubbles from the glass. The mechanism involves several steps. In the final step (part of) the trioxide is oxidised to the pentoxide, it is therefore likely that the antimony in the final glass is primarily in the pentoxide form. Total content of Sb in the finished glass is typically around 0.8%. (Sternbeck et al.., 2002a) It is usually mixed with the glass and other additives in a dry form. To reduce exposure to man and environment at many smaller sites, the mixing and sometimes also pelletizing may be done at one site and the mixture then sent to the various glass manufacturing sites.

The powder containing diantimony trioxide is melted in furnaces and formed either in machine processes or in the case of art glass sometimes by manual glass blowing. The glass melt is heated to a temperature around 1400 °C. Production is traditionally in batches, the furnace is typically loaded in the evening and the melt then processed the following day. Modern large facilities have 24-hour continuous tank smelters, balancing the take-off of crystal at one end with the raw-material charging at the other to maintain a constant level of molten glass within the system.

2.3.4 Use in pigments

Diantimony trioxide is used in the manufacturing of “Complex Inorganic Coloured Pigments” (CICP), which are further used in subsequent industries such as plastics (50 %), coatings (35 %), enamels and ceramics (10 %) and building materials (5 %). The pigment production process involves chemical transformation of the input materials into a crystal (rutile) host matrix in which various metals (e.g., Ti, Ni, Cr) apart from Sb are incorporated. Antimony is chemically bonded (as Sb(V) in the rutile lattice), taking the place of some of the Ti-ions.

Once incorporated into these rutile structures, antimony is no longer present as diantimony trioxide. For this reason, this report is restricted to the relevant process stages involving releases of diantimony trioxide itself. Apart from the use in pigment production, only two other uses have been reported, namely as pigment in the ceramics industry and as flame retardant in special paints. No exact split of tonnage per use is available, these uses will therefore be handled together (EBRC, 2006i).

Various antimony compounds are part of pigments used in ceramics decoration colours together with e g lead-, cadmium-, zinc- and chromium-compounds. They are applied dry and "fired" at temperatures up to 1250 °C.

2.3.5 Catalyst in PET-manufacture

There are 22 production sites (year 2005) for PET container resin in the EU, with a total production capacity of 2.94 million tonnes/year. Capacities range from 30 to 345 thousand tonnes/year. Annual EU25 consumption was estimated to 2.8 million tonnes for year 2005. Diantimony trioxide is used at 11 sites (involving 13 plants). (EURAS, 2006a)

In Europe, PET container resin is mainly (approximately 82 %) used in bottles and containers for (in order of quantity) bottled waters, soft-drink bottles, edible oils and pharmaceuticals. Other uses include containers for household cleaners and other non-foods, APET and CPET sheet and industrial strapping. About 200 kt of post-consumer recyclate are expected to be used in these applications. On top of the PET container resin production, EU25 has production capacities for 412 kt of Polyester Staple Fibre, 437 kt Polyester Filament Fibre and 296 kt of Polyester Film. This gives a total demand of 4.1 million tonnes of virgin and recycled PET. (IAOIA, 2006a)

PET resins are produced commercially by two similar processes, using ethylene glycol (EG) and either (i) dimethyl terephthalate (DMT) or (ii) terephthalic acid (TPA). In both cases, the bis-(2-hydroxyethyl)- terephthalate (BHET) monomer is first produced as an intermediate, yielding either methanol (DMT process) or water (TPA process) as by-products. The BHET monomer is then polymerised at low pressure under heating and the presence of the antimony catalyst to the PET resin. (EBRC, 2006h). Diantimony trioxide is used as polymerisation catalyst. Other catalysts are also in use, but diantimony trioxide dominates.

The final concentration of diantimony trioxide in PET is typically around 180 to 220 ppm, but can be up to 550 ppm.

The diantimony trioxide used in this application must have:

- Low impurity levels of lead and arsenic (around max 100 ppm of each element) and other impurities, typically Fe < 100 ppm; Cl < 100 ppm; Cu < 0.01 %; Zn 0.002 %; Pb 0.001 %; Ni < 0.1 %; SO_4 < 0.01 %. With a total impurity specification of max 0.05 %.

- Good solubility in MEG, i.e. low levels of insolubles such as higher oxidation steps of antimony oxide.

- Small fluctuations over time in physical properties.

Diantimony trioxide was, and in some cases still is chemically purified in order to reduce the level of impurities. It is supplied as powder or powder wetted with some 3 % MEG.

Diantimony trioxide is handled when preparing the catalyst solution. The preparation consists of the following activities: diantimony trioxide is dissolved in monoethylene glycol in a heated, stirred feed vessel and forms antimony glycolate. The antimony glycolate solution is then injected into the process vessel and the PET polymerisation reaction begins. In the initial stages of the reaction, excess water is removed and then as the reaction progresses and the polymer viscosity increases, excess monoethylene glycol is removed. When a specified viscosity is achieved the amorphous polymer is cooled and pelletised. Following this first process a secondary process is then employed. This process takes amorphous PET pellets and whilst in the solid phase crystalises the polymer. This is then transferred to a solid phase polymerisation process where more monoethylene glycol and other volatile species are removed and the viscosity of the polymer increases to a required level. The PET pellets are then formed, using a range of standard polymeric processing techniques into an article or precursor article. In the case of PET bottles a precursor article called a preform and is made on an injection moulding machine. This preform is then reheated and blown into a bottle shape in specially designed processing equipment which is called a blow moulding machine.

Bottles of PET are used as either single trip or multitrip bottles. In a few countries a proportion of bottles are used for multitrip purposes. These bottles are filled, consumed, collected washed and refilled (can be done in 7 to 15 cycles). Single trip bottles are collected and recycled in various ways. A rough estimate based on statistics from Petcore (PET container recycling Europe, 2003) indicates that around a third of the PET used in containers is collected and recycled. This proportion is growing. Currently the biggest outlet for recovered PET is the fibre market (70 %). The fibre market uses recovered PET in a number of different applications, e g padding to stuff anoraks, sleeping bags and soft toys, fleece fabric for sweatshirts, jackets and scarves, geotextiles, non-woven fabrics used in shoes, backpacks and umbrellas. PET fibres are used in car carpets and upholstery, and carpets and rugs. Other uses of recovered PET are for extruded sheets (both for food and non-food packaging) and for strapping tape (industrial use). Less than 10 percent is used to manufacture PET bottles, both for food and non-food applications, but that share is increasing. A proportion of the PET bottles will end up in household waste or as litter, when municipal solid waste is incinerated with energy recovery, PET bottles will significantly contribute to the calorific value of the waste. This fraction will decrease as recovery of packaging waste is improved. This improvement should also be driven by EU's packaging waste directive, 2004/12/EC. (Anonymous, 2006; Anonymous, 2000; PET container recycling Europe, 2003; IAOIA, written communication, 2000).

2.4 DISPOSAL

Because of the widespread final use of diantimony trioxide it is not possible to determine how it is disposed of. Most of it is likely to be incinerated or used as landfill (APME, Association of Plastic Manufacturers in Europe, 2001a). Directive 2002/96/EC on waste electrical and electronic equipment (WEEE) and, to a lesser extent, directive 2000/53/EC on end-of life vehicles, will have an impact on the disposal of a large proportion of articles containing diantimony trioxide. The WEEE directive should be in force in all member states in August 2004. The disposal is further discussed in the chapter on environmental exposure.

The WEEE directive requires that plastics used in EEE and containing brominated flame-retardants should be separated, since diantimony trioxide is used together with brominated flame retardants in this application it will be in this separated part. The influence of diantimony trioxide on the possibility to recycle WEEE in thermal processes has been

discussed in a recent article by Tange et al (2005). The discussion indicates that it might be possible to recover antimony in one type of thermal process, but that too much antimony might cause process problems in other types of thermal processes.

2.5 ANTIMONY EXPOSURE APART FROM DIANTIMONY TRIOXIDE

When determining levels of antimony compounds the actual chemical analysis is almost universally made on the element antimony (Sb) rather than the actual compound. It should therefore be noted that there are a number of sources of the element antimony outside the scope of this report. The following list of sources of antimony is not complete, and should serve as examples only. A further discussion can be found in the chapter on environmental exposure where some approximate estimates of contribution of antimony from different sources have been made.

Antimony pentoxide and sodium antimonite are used as synergistic flame-retardants. Antimony is present in trace amounts (typically 1 mg Sb/kg) in coal, primarily with the organic matter (Kirk-Othmer, 1992a) and in cigarettes (the tobacco on average contained 0,1 mg Sb/kg dry weight of which 20 % was estimated to be inhaled when smoking) (Nadkarni and Ehman, 1970; Wyttenbach et al., 1976). Other sources of antimony include the metallurgical industry (e.g. secondary lead extraction), ore-processing (many ores contain trace amounts of antimony), pigments (other antimony compounds than diantimony trioxide are also used), lead shot used in hunting etc, lead alloys used in fishing, fireworks, corrosion of antimony-containing alloys, vehicle tyres (antimony pentasulphide is employed in the vulcanising of rubber, synthetic rubber etc), brake linings (antimony trisulphide) (Slooff et al., 1992; UK Department of Health, 1988).

2.6 TRENDS

2.6.1 General trends

Little or no growth is forecasted for antimony metal in the metallurgical markets. Lead-acid batteries will increasingly use lead-calcium rather than lead-antimony alloys.

2.6.2 Trends in flame-retardant use

The industry estimates the growth of diantimony trioxide use as flame retardant to 2 % per year in the EU. Changes in regulations concerning flame retardancy will have a big impact on demand. (EURAS bvba, 2003)

There is a trend in the electrical and electronic equipment industry towards use of reactive as opposed to additive brominated flame retardants, i.e. the flame retardant form a part of the polymer structure, e. g. tetrabromobisphenol A. Although diantimony trioxide have a synergistic effect on both reactive and additive halogenated flame retardants, it is not generally used with the latter type (Risk and policy analysts Limited, 2000).

2.6.3 Trends in use as catalyst for PET production

Global demand for PET container resin is predicted to grow at some 8 % per year. Applications that are predicted to grow in significance are drinks and food, such as preserves that are hot-filled and bottles for beer (with barrier polymers, scavengers or physical barrier construction to improve the gas-barrier properties). Import/export ratio of PET to/from the EU depends on economical factors, including such factors as anti-dumping measures (Anonymous, 2000; Kirk-Othmer, 1996).

2.6.4 Trends in use in glass production

Diantimony trioxide is being replaced by sodium antimonate as a fining agent or as a degasser in glass production, in particular in lead crystal glass (EBRC, 2006b).

2.7 LEGISLATIVE CONTROLS

Antimony trioxide is currently classified under Directive 67/548/EEC and adaptions.

Commission decision 2000/532/EC of 3 May 2000 stipulates that Antimony Trioxide shall be classified as “heavy metal” in the classification of hazardous waste in that decision.

Directive 2000/76/EC of the European parliament and of the council on the incineration of waste gives maximum air emission limit values. The value relevant for antimony and its compounds refers to the total emission of antimony and its compounds together with a number of other elements and their compounds.

Council decision 2003/33/EC establishing criteria and procedures for the acceptance of waste at landfills pursuant to articles 16 of and Annex II to directive 1999/31/EC, gives leaching limit values for antimony from waste acceptable at landfills for inert waste.

Commission Directive 2002/72/EC of 6 August 2002 relating to plastic materials and articles intended to come into contact with foodstuffs, stipulates a maximum specific migration limit of Antimony Trioxide from plastics used in contact with foodstuffs.

Council Directive 98/83/EC on the quality of water for human consumption, give a maximum level of the element antimony in water intended for human consumption.

Council directive 88/378/EEC of 3 May 1988 on the approximation of the laws of the member states concerning the safety of toys gives a maximum level of bioavailability resulting from the use of toys for antimony and other elements. The definitions and the standard, EN 71, which accompanies this directive, are under discussion.

Antimony trioxide is classified as hazardous goods according to ADR with UN number 1549 (solid form) and UN hazard class 6.1 Toxic substances, but special provision 45 applies and stipulates that antimony oxides that contain not more than 0,5% of arsenic calculated on total mass are not subject to the requirements of ADR. Antimony trioxide on the European market has arsenic content below this limit (see section 2.1) and is therefore not classified as dangerous goods according to ADR/RID.

Several countries in EU have occupational exposure limits for Antimony Trioxide. Examples are presented in Table2-4 below. A full investigation of current levels has not been done.

Table2-4 Permitted or recommended occupational exposure levels of antimony

Country	mg/m³	comments	year	reference
Austria	0.5	daily mean, inhalable fraction	2001	Grenzwerteverordnung 2001 (BGBl. II Nr 253/2001) and Änderung der
Austria	5	short term, inhalable fraction	2001	Grenzwerteverordnung 2001 (BGBl. II Nr 184/2003)
Denmark	0.5		1994	Berg and Skyberg, 1998
Finland	0.5		1996	Berg and Skyberg, 1998
Norway	0.5		1995	Berg and Skyberg, 1998
Iceland	0.5		1989	Berg and Skyberg, 1998
Sweden	0.5	total Sb and SbO_x dust	2000	Swedish Work Environment Authority, 2000
Germany	0.5		1997	Berg and Skyberg, 1998
The Netherlands	0.5		1997-1998	Berg and Skyberg, 1998
USA (ACGIH)	0.5		1997	Berg and Skyberg, 1998
USA (NIOSH)	0.5	antimony	1997	Berg and Skyberg, 1998
USA (OSHA)	0.5	Sb and compounds as Sb	1997	Berg and Skyberg, 1998

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3 ENVIRONMENT

3.1 ENVIRONMENTAL EXPOSURE

3.1.1 General discussion

Releases of the element antimony to the environment occurs from natural processes (e.g. the weathering of rocks and soil runoff), mining and processing of ores, industry (e.g. glass, dyestuff, ceramics, and fire retardants), combustion of coal, metal production (e.g. Cu) and refuse incineration (Slooff et al. 1992; ATSDR, 1992; Pacyna, 1984). Several studies have also found a positive correlation between locally increased concentrations of antimony in air, soil and water and road traffic (de Miguel et al., 1997; Dietl et al., 1997; Hares and Ward, 1999; Pierson and Brachaczek, 1983; Stechmann and Dannecker, 1990; Sternbeck et al., 2001). The main source of this antimony has been suggested to be abrasions of brake linings, which contains up to 5-7% by weight Sb_2S_3 used as friction reducers (Sternbeck et al., 2001). Other possible sources of antimony are tires, motor bearings with antimony alloys, and balance weights of lead containing antimony.

Investigations by Meharg (1994) showed that considerable amounts of heavy metals, including antimony, were released into the aquatic and terrestrial environments following large-scale plastic fires. A fire in a warehouse containing Clingfilm (a highly plasticized formulation of PVC) resulted in a release of 2 – 40 tonnes of antimony to the environment. The vectors for emissions included the fire fighting water run-off, molten plastic run-off and the smoke plume.

Using a peat core from a Swiss bog, Shotyk et al. (1996) calculated the present day enrichment factor for antimony, which was about 70, and thereby showing that anthropogenic activities in Europe significantly has affected the atmospheric flux of antimony throughout the past 2000 years. Barbante and colleagues (2004) determined the enrichment factor for the last 350 years (from the 1650s to the mid 1990s) for a number of elements, including antimony, using snow/ice cores drilled from a glacier near the Monte Rosa massif, located in the Swiss-Italian Alps. The enrichment factor for antimony for that time period was determined to be 2.5.

A comparison, performed in Pacyna and Pacyna (2001), of global anthropogenic emission estimates of antimony from 1995 with the global natural emission estimates by Nriagu (1989) from 1983, suggest that the two are comparable in size, 1.6×10^3 t/y and 2.4×10^3 t/y, respectively. The identified major sources of anthropogenic emissions were stationary fossil fuel combustion (47%), nonferrous metal production (mostly Cu and Ni) (35%), and refuse incineration (17%), and the major identified sources of natural emissions were wind-borne soil particles (33%), volcanoes (30%), and sea salt spray (23%). Other assessments have indicated that the importance of the natural sources was considerably minor, as compared to the anthropogenic. Both Lantzy and Mackenzie (1979) and Galloway (1982) estimated that the anthropogenic input was approximately 39 times higher than the natural. Recent results by Shotyk and colleagues (2005) indicate that the ratio of anthropogenic to natural Sb emissions may be about a factor of ten or more.
EURAS (2003) performed a study, commissioned by the International Antimony Oxide Industry Association (IAOIA), in which a questionnaire was sent to producers and

downstream users of Sb_2O_3. The outcome of this study was a compilation and review of the local exposure data (both human and environment) for Sb_2O_3 as provided by Industry. The collected data were transformed into a tabular RAR format and presented to the rapporteur for direct input in the exposure models. Further information gathering resulted in two updated reports (EURAS, 2006a; EURAS, 2006b). The information in the following sections on environmental releases is from these updated reports unless otherwise stated.

The use of diantimony trioxide has in chapter 2 in this RAR been divided into use as flame-retardant, as catalyst in Poly Ethylene Terepthalate (PET)-manufacture, as finisher (in glass), and in pigment, based on information from industry (Docherty, 2001; EURAS bvba, 2003). Based on that, the following use categories will be used in this chapter when estimating release of diantimony trioxide/antimony to the environment:

- Use as flame-retardant in plastics and rubber
- Use as flame-retardant in textiles
- Use as catalyst in PET-manufacture
- Use in paints, pigments, adhesives, and coatings (as flame-retardant or pigment)
- Use in glass (as finisher)

The use of diantimony trioxide in ceramics (as pigments) is mentioned, but appear to be relatively minor and no volumes related to it have been found. The use in ceramics is covered together with paint and pigments. The volumes for the different categories, defined above, will be followed separately through formulation, processing stage, professional/consumer use and disposal.

3.1.2 Environmental releases

Emissions, concentrations and effect levels are originally reported sometimes as antimony (Sb) and sometimes as diantimony trioxide (Sb_2O_3). In this chapter values are expressed both as Sb and Sb_2O_3 in the section on emissions, and only as Sb in the risk characterisation. Values in EUSES are of Sb. Sb_2O_3 consists of 83.5678% Sb, and this percentage has been used for conversion.

During 2006 IAOIA has, in cooperation with consultants, gathered data on environmental exposure. The results have been presented in a report (EURAS, 2006a). This shows that there are around 550 industrial users of diantimony trioxide in EU15. A distribution of users based on their annual use of diantimony trioxide is presented in table (Table3-1). It can be seen that relatively few large users account for more than 60% of the total use.

Table3-1 Diantimony trioxide use per site, distribution (data from 2003, 2004-2005).

Annual diantimony trioxide use [t/y]	Percentage of	
	total number of companies [%]	total tonnage [%]
< 10	44	8
10 to 125	47	29
> 125	9	63

Local exposure information from 2003, 2004 and 2005 has been collected. The focus for the data collection has been on sites with larger consumption of diantimony trioxide, but data for smaller sites have also been included where submitted.

When information on release from one source is available from several years, the value from the latest year has been chosen if: there is a clear trend over time, and information is available to support that the change is not random, e.g. information on installation of emission abatement equipment or changes in procedures or processes. If there is no clear trend or no supporting information, the 90th percentile of reported values has been used.

3.1.2.1 Diantimony trioxide production

Information on releases have been reported from all four sites currently (2006) producing diantimony trioxide in EU15. An overview of the available information on emissions to air and water is presented in Table3-2 (EURAS, 2006a).

Table3-2 Information on emission data to air and water from producers of Sb_2O_3 in the EU.

	Available information and measured release values to	
Site	**Air (stack emissions)**	**Water**
P-1	for year 2001 to 2005	for year 2001 to 2005
P-3	for year 2001 to 2004	no emissions to water (stated 2005)
P-4	for year 2001 to 2004	no emissions to water (stated 2005)
P-5	below detection limit (measured 2003)	for year 2002 to 2005

The measured data is representative and the quality of data is good and in general the information on changes in processes affecting releases is good. The release value chosen for calculating the PEC is:

- Value from the latest year if there is a clear trend and information on measures taken that affect releases (e.g. re-cycling of wastewater, procedures for assuring filter-function);
- The 90 percentile value of reported values for the years after measures have been taken if there is no clear trend in the years after measures have been taken.

Local emissions

Reported local emissions are summarised in Table3-3.

Table3-3 Local emissions from diantimony trioxide producing plants.

	Air stack emissions		**Water**		**Notes**
	kg Sb_2O_3/y	kg Sb/day	kg Sb_2O_3/y	kg Sb/day	
P-1	1 520	3.7	5.6 to surface water	0.06	2005 values
P-3	67 (min 64, max 67)	0.17	0	0	Air 90P of values for 2001 – 2004; Water stated 2005
P-4	40 (min 20, max 42)	0.17	0	0	Air 90P of values for 2002 – 2004; Water stated 2005
P-5	0	0	1.7	0.01	Air measured 2003; Water 90P of values for

			(min 1.2, max 2.1) to municipal STP		2003 – 2005

No information is available on the handling of sludge from the municipal STP receiving emissions from site P5. It is however assumed that the sewage sludge will be spread on agricultural soil.

Total emissions

The sum of the total reported release to air is the total release, with the highest release from a single site taken as the regional release.

The sum of the total reported release to water (assuming a 50 % removal in STP where relevant) is the total release, with the highest release from a single site taken as the regional release.

The flow rate of receiving waters was not reported, default values will be used in the modelling.

Summary of regional and continental emissions

Local scenarios will be site specific based on reported releases. The regional emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-4 Distribution of environmental emissions due to production of diantimony trioxide.

Distribution	Total (kg Sb_2O_3/y)	Continental (kg Sb_2O_3/y)	Regional (kg Sb_2O_3/y)
Air	1 627	107	1 520
Surface water	6	0	6
Waste water	1	0	1

3.1.2.2 Use as flame-retardant in plastics and rubber

The split of diantimony trioxide used as flame-retardant in polymers into different application/industry sectors is not always straightforward. Many sectors overlap and it is not always clear where the volumes belong. This has very little impact on the determination of the total environmental emissions however and will therefore not be discussed further in this section. Environmental emissions of diantimony trioxide from the following industry/application sectors are handled under the heading of flame-retardants in plastics and rubber: Compounding (formulation) and industrial use of PVC and other plastics, manufacture of wire and cables, manufacture of electrical and electronic equipment, compounding and industrial use of rubber and other uses of diantimony trioxide as flame-retardant in plastics and rubber.

There are various stages in polymer processing, such as compounding (blending of the polymer with various additives) and conversion (production of the finished articles). The different processes are not necessarily carried out at the same site, i.e. there are companies that specialise in compounding to produce master-batches. Master-batches are plastic compounds that contain high concentrations of additives that are subsequently mixed in the main polymer matrix.

A proportion of the diantimony trioxide used as flame-retardant in plastics and rubber is converted into low dusting forms (e.g. master-batch granules or damped powder) or is pre-dosed and packed into plastic bags that are loaded without opening into a closed system (e.g. a mixer). Emissions to the environment from handling such material will be much lower than from handling of powder. In the non-PVC plastics sector approximately half of the diantimony trioxide is consumed as dry powder, the rest being mainly in the form of non-dusting master-batches and a small part as wetted powders (EBRC, 2006c). In the PVC sector at sites using less than 125 t Sb_2O_3/y slightly less than half of the diantimony trioxide is consumed as dry powder, some 80% at sites using more. The rest is consumed mainly in the form of wetted material or non-dusting master-batches, with smaller amounts being consumed in pre-dosed form or as an aqueous dispersion (EBRC, 2006d). In the rubber industry at sites using less than 10 t Sb_2O_3/y, approximately half of the diantimony trioxide is consumed as dry powder, around 15% is consumed as dry powder at sites using more than 125 t/y, and less than a third is consumed as dry powder at sites using between 10 and 125 t/y. The rest is consumed mainly in pre-dosed form, with minor quantities being consumed in the form of non-dusting master-batches or in wetted form (EBRC, 2006e).

The range of tonnage diantimony trioxide used per site is shown in Table 3-5.

Table 3-5 Diantimony trioxide use per site, flame-retarded plastics and rubber.

Sector	Number of sites in EU15 using diantimony trioxide	Diantimony trioxide use per site [t/y]	Sites using > 125 t Sb_2O_3/y	Ref
Flame-retarded PVC	200	< 1 to 650	17	EBRC, 2006d;,IAOIA, 2006b
Flame-retarded non PVC plastics	224	1 to 500	17	EBRC, 2006c; IAOIA, 2006b
Flame-retarded rubber	41	10 to 500	3	EBRC, 2006e; IAOIA, 2006b

Releases are likely to be significantly higher in an initial stage where diantimony trioxide is handled as powder, compared to later stages. Information on releases has been collected through a questionnaire. The coverage, in percentage of number of sites and total tonnage based on general information received is shown in Table3-6.

Table3-6 Overview of EU coverage of diantimony trioxide processing sites in the EU based on general information submitted

Sector	Coverage	
	number of sites [%]	tonnage [%]
Flame-retarded PVC	8	26
Flame-retarded non PVC plastics	5	23
Flame-retarded rubber	7	32

As can be seen from this overview the coverage in terms of number of sites is rather low, but fairly good in terms of tonnage. The data-gathering has focussed on sites using larger amounts of diantimony trioxide (> 125 t/y), and the sites responding can be said to be fairly typical of sites in their respective sector (good geographical coverage). The initial approach taken will therefore be to use site-specific scenarios using reported release values to cover larger sites, and to construct a hypothetical generic site based on a yearly consumption of 125 t diantimony trioxide and default emission factors to cover sites with small and medium use of diantimony trioxide.

3.1.2.2.1 Releases from formulation (raw materials handling, compounding etc)

The handling of raw materials from their arrival on site to their addition to polymers is undertaken by a variety of means. These include manual handling of bags and sacks, conveyor belts and pneumatic or pumped transfer from bulk storage vessels.

The process of compounding during which additives are incorporated into plastics can take place either during the polymer production, the final conversion process or as a secondary process with the specific function of compounding. The latter of these dominates the compounding activity in the plastics industry, and is often undertaken by specialist companies. Most of the compounding processes are closed, but may still produce some particulate emissions. Losses mainly occur early in the mixing cycle and localised containment may be used to recover the material for recycling. The emissions from the compounding are therefore estimated to be far lower than in the previous raw materials handling step.

Emission factors for the generic scenario handling of raw materials and compounding has been taken from OECD Emission Scenario Document (ESD) on Plastics additives (OECD ESD Plastics additives, 2003). Initially, some emissions will be to atmosphere, but ultimately all particulates will be removed or settled and losses will be to solid waste or wastewater as a result of wash down. Some diantimony trioxide will remain in the bag or sack due to agglomeration effects.

Emissions to air

Reported data for emissions to air range from zero to 10 kg Sb/y and are presented in Table3-8. The emission scenario document on plastics additives (OECD ESD Plastics additives, 2003) gives an emission factor of zero to air (inorganic flame-retardant, particle size < 40 μm). This is based on the assumption that all particulate losses will eventually be to wastewater, and is included in the emission factor to water. This assumption has also been used in this risk assessment, the emission to air has therefore been set to zero for one generic scenario. However, since the reported releases are mainly to air, while the ESD gives releases

only to water, a generic scenario directing all releases to air has also been developed. A further investigation of where the release should be directed will be done if either of these generic scenarios causes concern.

Emissions to wastewater and surface water and solid waste

Reported data for emissions to water range from 0 to 1 kg/y and are presented in Table3-8. The emission scenario document on plastics additives (OECD ESD Plastics additives, 2003) gives an emission factor of 0.006 to wastewater and/or solid waste from raw materials handling (not including residue in packaging) and of 0.0005 from compounding (inorganic flame-retardant, particle size < 40 μm). The processes where diantimony trioxide is used are dry processes (with the possible exception of the last step, cooling, which may be in a water bath) where water generally will have a detrimental effect. Washing down the equipment with water is therefore generally not done; "dry" cleaning methods are preferred. The losses will therefore mainly be to waste. It is possible that a small amount of the diantimony trioxide lost in raw materials handling may enter the wastewater streams as a result of cleaning down floors and equipment. It is assumed that such releases to wastewater are accounted for in the emission factor for compounding, and the emission factor for raw materials handling will be used to estimate losses to waste. This assumption is supported by reported information on work processes etc. (EBRC, 2006c; EBRC, 2006d; EBRC, 2006e).

The information on type of wastewater treatment plants in this sector is limited, some but far from all have on-site treatment of the wastewater. For the generic scenario it has been assumed that the wastewater goes to a municipal STP.

Total, regional and continental emissions

The quantity used per year in PVC is 8 784, in non-PVC 9 222 and in rubber 2 196, which gives a total of 20 202 t of Sb_2O_3. No complete information on the geographic distribution of compounding with diantimony trioxide is available. There is a large number of sites where compounding takes place, with a reasonable distribution over Europe. The distribution of handled volume in a region has therefore been set to the default value of 10% of the total, or the highest emission reported for a single site if this is higher (for air). As a first approach the total releases have been calculated as follows: The sum of reported releases is assumed to cover 25% of the total tonnage in this sector; 25% of the total tonnage is assumed to be in non-dusting forms giving negligible releases compared to dry powder; Releases from handling the remaining tonnage is calculated using the default emission factors from the OECD ESD (see above). Based on the reported values this is likely to over-estimate the releases. Total releases to wastewater is then:

tonnage except non-dusting forms used at non-reporting sites	50 % * 20 202 * 10^3 kg
multiplied by OECD ESD emission factor	* 0.0005
plus reported releases	+ 0.7 kg
equals	= 5 051 kg Sb_2O_3/y

Reported emissions to air are assumed to end up in wastewater in accordance with the reasoning in the OECD ESD, only reported emissions to air are included below. Where total

dust emissions have been reported (sites AMI-7U, MN14-G) it has been assumed that the percentage of antimony trioxide in the particulate emission is 8 % (typical content in final polymer article, see section 2.3.2.1). This assumption has only been made to calculate total emissions here. For the calculation of PECs, monitoring data has been used to determine the regional PEC and default emission factors have been used to predict the local PECs. The total release to water is presented in Table3-7. Number of release days has been set to 300.

Very limited information on waste has been reported, therefore the emission factor for raw materials handling (including material remaining in bags) 0.016 to solid waste, will be used for the total volume. The release to solid waste is not discussed further, but is handled together with the waste from all other uses and life-cycle stages.

The resulting environmental emissions are presented in Table3-7 below. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-7 Distribution of environmental emissions due to formulation of flame-retardants for the use in plastics and rubber.

Distribution	Total (kg Sb_2O_3/y)	Continental (kg Sb_2O_3/y)	Regional (kg Sb_2O_3/y)
Air	33	21	12
Wastewater	5 051	4 546	505
Solid waste	323 232	290 909	32 323

Local emissions

As explained in section 3.1.2.2, a generic scenario will be used to cover sites where the extent of reported values is deemed not to be sufficient. This scenario will be a site using 125 t Sb_2O_3/y in the form of dry powder, default number of emission days 300 (default from TGD Table B2.3), default emission factor from OECD ESD and wastewater to municipal STP. This gives a release to wastewater of 62.5 kg Sb_2O_3/y (0.0005 * 125 t/y) and zero to air. Since the reported releases are mainly to air, while the ESD gives releases only to water, a generic scenario directing all releases to air has been developed. A further investigation of where the release should be directed will be done if either of these generic scenarios causes concern. The reported emissions and the local emissions from the generic scenario are presented in Table3-8.

Table3-8 Local emissions due to formulation of flame-retardants for the use in plastics and rubber.

	Sub-sector	Stage	Form used	Air reported data are stack emissions unless stated		Wastewater	
				kg Sb_2O_3/y	kg Sb/day	kg Sb_2O_3/y	kg Sb/day
Generic		formulation	powder	0	0	62.5	0.17
Generic air		formulation	powder	62.5	0.17	0	0
RC47I	PVC	formulation	powder	0 fugitive emission: 0.06	0	0	0
RC35I	PVC	processing	powder	no data	no data	no data	no data
MN24Ga	PVC	form + proc	wetted	0	0	0	0

	Sub-sector	Stage	Form used	Air reported data are stack emissions unless stated		Wastewater	
MN24Gb	PVC	form + proc	powder	0	0	0	0
AMI5S	PVC	processing	powder	no data	no data	0	0
RC63S	PVC	form + proc	powd + masterb	0 fugitive emission : 0	0	0	no data
MN44I	PVC	processing	wetted	no data	no data	no data	no data
RC43I	PVC	formulation	pre-dosed	0 fugitive emission : 0	0	no data	0
AMI-9U	PVC	form + proc	powder	no data	no data	0	0
AMI-10B	PVC	formulation	powder	1	0.005	0	0
AMI-12B	PVC	form + proc	powder	no data	no data	no data	no data
FP-1	PVC	formulation	wetted + dispersed	1	0	0	no data
ProcPVC-1	PVC	processing	dispersed + masterb	0	0	0	0
ProcPVC-2	PVC	form + proc	wetted + pre-dosed	0	0	0	0
ProcPVC-4	PVC	formulation	powder	no data	no data	no data	no data
ProcPVC-5	PVC	form + proc	wetted	no data	no data	0	0
ProcWC-1	PVC	form + proc	wetted	no data	no data	0	0
AMI7U	non-PVC	processing	powder	10 (note 1) fugitive emission: 48 (note 1)	0.12	0	0
MN69U	non-PVC	formulation	powder	no data	no data	no data	no data
MN32I	non-PVC	formulation	masterbatch	0	0	0	0
RC31Ia	non-PVC	formulation	masterbatch	9.5	0.02	< 1	0
RC31Ib	non-PVC	formulation	masterbatch	no data	no data	no data	no data
AMI-2-NL	non-PVC	formulation	masterbatch	0	0	0	0
AMI-4NL	non-PVC	formulation	masterbatch	no data	no data	no data	no data
AMI-17B	non-PVC	formulation	masterbatch	0	0	no data	no data
AMI-6B	non-PVC	formulation	powder	0	0	0	0
MN14G	non-PVC	formulation	powder	60 (note 1)	0.25	no data	no data
AMI-15F	non-PVC	processing	masterbatch	no data	no data	no data	no data

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	Sub-sector	Stage	Form used	Air reported data are stack emissions unless stated		Wastewater	
AMI18S	non-PVC	form + proc	powder	no data	no data	< 1	0
FP-4	non-PVC	formulation	masterbatch	no data	no data	no data	no data
AMI-14S	rubber	form + proc	masterbatch	no data	no data	no data	no data
RC52-I	rubber	form + proc	powder	0.1	0	no data	no data
RC25G	rubber	form + proc	pre-dosed	< 12	0.03	0	0

Note 1: these values represent total emissions of particulates, the composition has not been determined, the emissions of antimony is therefore lower than these values, but not known

The generic scenario gives releases higher than the highest reported numbers. The reported numbers are mainly from sites using more than 125 t/y, thus the generic scenario is a conservative reasonable worst case and will be taken forward to the risk characterisation.

3.1.2.2.2 Releases from industrial use (conversion)

There are some significant differences between the processes used to convert thermosetting and thermoplastic materials. Diantimony trioxide is mainly used as a flame-retardant in thermoplastics and rubber and only to a limited extent in thermosetting resins. Most of the conversion takes place in closed processes with a smaller fraction being converted in partially closed processes. Since diantimony trioxide is essentially non-volatile at the temperatures used in these processes, the split of volume used in closed or partially closed processes is not critical for the estimation of emissions.

Reported releases are included in Table3-8 above. The only processing site with significant reported releases is AMI7U. This has been handled under formulation as it uses dry powder without any earlier formulation step. For the industrial use scenarios default values will be used for the entire sector. In view of reported data from formulation sites using master-batches, this is likely to over-estimate the releases.

Emissions to air

The emission scenario document on plastics additives (OECD ESD Plastics additives, 2003) gives an emission factor of zero to air (inorganic flame-retardant, all other operations than machining/grinding). The emission to air has therefore been set to zero for this use pattern.

Emissions to wastewater and surface water

The emission scenario document on plastics additives (OECD ESD Plastics additives, 2003) gives an emission factor of 0.0001 to wastewater and/or solid waste (inorganic flame-retardant, all other operations than machining/grinding).

The sites where conversion is done do not normally use large amounts of water in their processes. It has therefore been assumed that most sites do not have any wastewater treatment plant of their own and that the wastewater goes to a municipal STP.

Total, regional and continental emissions

The quantity used in this application is 20 202 t of Sb_2O_3 (see above). No complete information on the geographic distribution of conversion of polymers containing diantimony trioxide is available. There are a large number of sites where conversion takes place, with a reasonable distribution over Europe. The distribution of handled volume in a region has therefore been set to 10% of the total. The environmental emissions generated by EUSES are presented in Table3-9 below. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-9 Distribution of environmental emissions due to industrial use of diantimony trioxide as flame-retardant for the use in plastics and rubber.

Distribution	Total (kg Sb_2O_3 / y)	Continental (kg Sb_2O_3 / y)	Regional (kg Sb_2O_3 / y)
Air	0	0	0
Wastewater	2 020	1 818	202

Local emissions

For the evaluation of generic local emissions, the fraction of the total volume released that can be assumed to be released through a single point source (fraction of main source) is set to 0.1 and the number of emission days to 300 (default from TGD Table B3.9). Reported site-specific emissions and the values for local emissions generated by EUSES are shown in Table3-10 below.

Table3-10 Local emissions from industrial use of diantimony trioxide as flame-retardant in plastics and rubber.

	Air		Wastewater	
	kg Sb_2O_3 /year	kg Sb/day	kg Sb_2O_3 / year	kg Sb/day
Generic	0	0	20	0.06

3.1.2.2.3 Releases during service life (including "waste remaining in the environment")

Most of the articles in which diantimony trioxide is used as flame-retardant are not normally subject to much wear and tear or aggressive liquids (e.g. water with extreme pH levels). Furthermore diantimony trioxide has a low volatility and low water solubility. It is therefore unlikely that volatilisation and/or leaching from articles in use are a major source of diantimony trioxide emissions into the environment. The contribution to the total emissions from waste remaining in the environment will therefore be very small compared to substances that are used in articles subject to much wear, e.g. tyres, shoe-soles. Some emissions will occur due to aging and abrasion, and the emission factor for this has been set to 0.0001 to water and zero to other compartments in accordance with emission scenario document (OECD ESD Plastics additives, 2003). It has been assumed that the amount of diantimony

trioxide as flame-retardant in plastic and rubber articles in use is more or less constant over time, i.e. that it is a "steady-state" situation. This is a simplification, but an acceptable one in view of the limited data available to estimate emissions.

The quantity used in this application is 20 202 t of Sb_2O_3 (see above). Default values from TGD are used for input of the regional volume (10%), and for emission days (365).

The environmental emissions generated by EUSES are presented in Table3-11. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-11 Distribution of environmental emissions due to service-life of flame-retardants for the use in plastics and rubber.

Distribution	Total (kg Sb_2O_3 / y)	Continental (kg Sb_2O_3 /y)	Regional (kg Sb_2O_3 / y)
Air	0	0	0
Surface water	2 020	1 818	202

3.1.2.2.4 Releases during waste treatment

Emissions during this stage are included in section 3.1.2.7 (Release from disposal) below.

3.1.2.3 Use as flame-retardant in textiles

Reported values from the textile industry are presented in Table 3-13. The information is from formulators, processing sites and in some cases from sites where both processing and formulation takes place. As the processing step is generally the step giving the highest emissions, the sites with both formulation and processing will be handled under processing.

Twenty-nine sites using diantimony trioxide has been identified in this sector, with an annual total use of 1 757 t Sb_2O_3/y. The sites use between 2 and 326 t Sb_2O_3/y with an average of 83. Sixteen sites have reported information on release to water, covering 80 % of the tonnage used, 10 sites have reported information on release to air covering 38 % of the tonnage used. The number of production days reported range from 10 to 355. Twelve sites have reported that they have a wastewater treatment on site before releasing to surface water. Various processes (physico-chemical, biological or combinations) are reported. (EURAS, 2007).

3.1.2.3.1 Releases from formulation

The total usage of Sb_2O_3 for flame-retardants backcoating formulations in the textile industry in the EU15 is reported to be 1 757 t/y (EURAS, 2007).

Emissions to air

The reported emissions cover a fairly large part of the use in this sector (38 % of tonnage and 34% of sites). The reported values are deemed to be representative for the industry sector based on this coverage together with information on processes collected and presented by industry and the regional distribution (replies from 7 countries in EU15) (EBRC, 2006f). All

sites reporting emissions to air have no or very low emissions (< 2kg Sb_2O_3/y). Any release to air will be particles that are likely to settle fairly close to the plant. The highest of the reported emission factors (20 g/t for site AMI-8B) have been used to calculate releases for the sites not reporting emission data.

The total releases to air are then:

tonnage used at sites not reporting emission data	62 % * 1 757 * 10^3 kg
multiplied by highest reported emission factor	* 20*10^{-6}
plus reported releases	+ 1.9 kg
equals	= **23.7 kg Sb_2O_3/y**

Emissions to wastewater and surface water

The reported emissions cover a large part of the use in this sector (80 % of tonnage and 55 % of sites). The reported values are deemed to be representative for the industry sector based on this coverage together with information on processes collected and presented by industry and the regional distribution (replies from 7 countries in EU15) (EURAS, 2007).

The derivation of a 90-percentile on the available dataset of emission factors is questionable, and the emission factors of RC74U and AMI20B will be excluded because the former is also a processing site and will be dealt with under that life-cycle-stage and the latter had a non-functioning filter during the measurements. Therefore the highest of the other reported emission factors (0.001064 for site FT-2) have been used to calculate releases for the sites not reporting emission data.

The total releases to water are then:

tonnage used at sites not reporting emission data	20 % * 1 757 * 10^3 kg
multiplied by highest reported emission factor	* 0.001064
plus reported releases	+ 1 085 kg
equals	= **1 459 kg Sb_2O_3/y**

The total emissions are presented in Table 3-12

Summary of regional and continental emissions during formulation for textiles

The geographical distribution of textile using diantimony trioxide as flame-retardant is concentrated to Belgium (43 % of the total diantimony trioxide use in this sector), Germany (30 %) and the UK (19 %).

The distribution of emissions will, based on the discussion above, be 30% regional and 70% continental (see Table 3-12 below) or the release from the site reporting the highest release if this is higher. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table 3-12 Distribution of environmental emissions due to formulation of diantimony trioxide used as flame-retardant in textiles.

Distribution	Total (kg Sb_2O_3/y)	Continental (kg Sb_2O_3/y)	Regional (kg Sb_2O_3/y)
Air	24	17	7

Wastewater	1 459	1 021	438

Local emissions

The reporting sites are deemed to be representative for sites in this use pattern and therefore default values for emissions will not be used. The generic formulation scenario will be based on the highest reported release factor, which is 20 g/t to air (from site AMI-8B) and 1 064 g/t to surface water (from site FT-2). The reported emission factor to water is after waste water treatment on site. It has been assumed, as a realistic worst case, that waste water will be treated either on site or in an off-site (municipal) STP, not both. Sewage sludge has been assumed to be used on agricultural soil, except for those sites reporting that this is not done. Some reporting sites have on-site WWTP whose effluent goes to a municipal STP, others have either only on-site WWTP or no on-site WWTP but release via municipal STP. Industry has given the 90-percentile of tonnage used per formulation site, for the sites not reporting emission data. (International Antimony Oxide Industry Association, 2007). This tonnage, 192 t/year, has been used for the generic site. Reported site-specific and generic emissions are shown in Table 3-13 below.

Table 3-13 Local emissions and emission factors from the use of diantimony trioxide as flame-retardant in textiles.

	Stage	**Form**	**Air[2]**			**Water**			**Note**
			kg Sb_2O_3 / y	g Sb / day	Emission factor [g/t]	kg Sb_2O_3 / y	kg Sb / day	Emission factor [g/t]	
Generic formulation	formulation	powder	3.8	11	20	201	0.59	1064	
Generic industrial use	processing	powder	0	0	0	1050	2.92	10 000	
RC74U	both	powder	0 fugitive emission 0.06	0.14	0	346	1.61	7 689	
AMI-8B	both	powder	1.2	20	20	0	0	0	
AMI-11B	formulation	dispersion	0	0	0	0	0	0	
AMI-16B	formulation	powder	0	0	0	0	0	0	
AMI-19U	formulation	powder	0.7	2.0	3	108	0.36	448	
AMI-20B	formulation	powder	no data	no data	no data	936	3.55	6 610	1
FT-2	formulation	no data	0	0	0	19	0.06	1 064	
FT-3	formulation	disp + powd	no data	no data	no data	no data	no data	no data	
FT-5	formulation	powder	no data	no data	no data	0	0	0	
FT-6	formulation	powder	no data	no data	no data	no data	no data	no data	
AMI-21B	formulation	powder	no data	no data	no data	0	0	0	
Site 543	formulation	powder	no data	no	no data	0	0	0	

	Stage	Form	Air[2]			Water			Note
			kg Sb_2O_3 / y	g Sb / day	Emission factor [g/t]	kg Sb_2O_3 / y	kg Sb / day	Emission factor [g/t]	
				data					
Site 542	formulation	powder	0	0	0	0	0	0	
Site 58	formulation	powder	no data	no data	no data	0	0	0	
Site 534	formulation	powder	no data	no data	no data	no data	no data	no data	
Site 530	processing	wetted	0	0	0	note 3	note 3	note 3	
Site 535	formulation	dispersion	0	0	0	no data	no data	no data	
Site 133	processing	dispersion	no data	no data	no data	0	0	0	
Site 72	processing	wetted	no data	no data	no data	no data	no data	no data	
Site 132	formulation	powder	no data	no data	no data	no data	no data	no data	
Site 99	processing	wetted	0	0	0	0	0	0	
Site 533	formulation	powder	no data	no data	no data	0	0	0	
Site 96	formulation	powder	no data	no data	no data	22	0.10	759	
Site 531	both	powder	no data	no data	no data	no data	no data	no data	

[1]Filter not functioning during measurements, has now been repaired.
[2]Reported air release data are stack emissions unless stated.
[3] Concentration in effluent is lower than detection limit (20 µg/l) (EURAS, 2008)

3.1.2.3.2 Releases from application to textiles (industrial use)

Reported information on releases from processing is presented in Table 3-13. The annual amount of diantimony trioxide used for application to textiles is assumed to be the same as for the formulation step i.e. 1 757 t/y.

Emissions to air

Since diantimony trioxide is a non-volatile compound, and in addition is in a water-dispersion in this step, it is assumed that the emission to air is zero.

Emissions to wastewater and surface water

The total emissions are only used for comparing the different sources of antimony in the environment. It is not used to predict the regional PEC which is based on measured data. The information on releases from industrial use is limited, and the proportion of total tonnage used in the reporting sites is not known. The total releases have therefore been predicted using the total volume used in this use and the emission factor from the OECD Emission Scenario Document on Textile Finishing. For coating the default emission factor is 0.01, this factor is

in the same range as the highest reported emission factor for a site with formulation and processing (7 689 g/t). Although the reporting site's releases are diluted into a municipal STP with a high dilution and is then released into a river with a high water flow, it is not unreasonable to assume that this emission factor might also apply to other sites releasing to municipal STPs with less dilution in the STP and receiving water.

Summary of regional and continental emissions during processing

The distribution of emissions will, based on the discussion above for the formulation, be 30% regional and 70% continental. Values for emissions generated by EUSES are shown in Table3-14 below. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-14 Distribution of environmental emissions due to processing of diantimony trioxide used as flame-retardant in textiles.

Distribution	Total (kg Sb_2O_3 / y)	Continental (kg Sb_2O_3 / y)	Regional (kg Sb_2O_3 / y)
Air	0	0	0
Wastewater	17 570	12 299	5 271

Local emissions

Because of the limited information on releases from industrial use a generic scenario has been included. The emission factor from the OECD Emission Scenario Document on Textile Finishing will be used to calculate the total releases as well as to develop a generic local scenario. For coating the default emission factor is 0.01, this factor is in the same range as the highest reported emission factor for a site with formulation and processing (7 689 g/t).
For the evaluation of releases from the generic local scenario the fraction of the regional release that is assumed to be released from the main local source (fraction of main source) is set to 0.2 and the number of emission days to 300 (default from TGD Table B3.12). Releases are assumed to be to a STP, and sewage sludge is assumed to be used on agricultural soil. The emissions in the generic local scenario generated by EUSES are shown together with the site-specific data from formulation, processing and combined sites in Table 3-13.

3.1.2.3.3 Releases during service life

Washing

A possible source of release of Sb_2O_3 from fabrics is during the washing of the fabric itself. This release is however expected to be low since some of the fabrics are not removable, and those that are, are only expected to be washed infrequently, perhaps in the order of once a year. Also washing tests done on back-coated fabrics indicate that the concentration of diantimony trioxide, bromine and decabromodiphenyl ether present in the samples tested remained essentially constant during the washing procedure in both the tested fabrics. (Anonymous, 2004). Releases from washing are therefore likely to be in the form of particles that are assumed to be covered by waste remaining in the environment.

"Waste remaining in the environment" (Particulate losses) from disposal

"Waste in the environment" can be considered to be particles (or dust) of polymer products that contain Sb_2O_3. These particles are primarily released to the urban/industrial soil compartment, but may also end up in sediment or air. End-products with outdoor uses are most likely to be sources of this type of waste, where releases can occur over the lifetime of the product due to weathering and wear. For Sb_2O_3, most of the coated textiles will be used in indoor applications (e.g. furniture), which may not lead directly to release to the external environment. Possible particulate losses from laundering etc. are considered in the previous section. In addition, releases of this type can occur from disposal processes, particularly where articles are dismantled or subject to other mechanical processes. Since in textiles, Sb_2O_3 is backcoated onto textiles in a polymer matrix, this also has the potential to generate that "waste remaining in the environment" over the service life of the textile.

A similar approach to that used for textiles in the draft risk assessment of decabromodiphenyl ether (RAR, 2004) will be used here. This method assumed a 2% loss from all textiles at disposal. The emissions are likely to mainly be to soil. In the draft decabromodiphenyl ether risk assessment it was assumed that 75% of the emissions would be to industrial/urban soil and 0.1% to air, with the remainder occurring to surface water (and hence sediment). In this scenario no emission is expected to air, therefore 75% is expected occur to industrial/urban soil and 25% to surface water.

Using these assumptions with the current usage figure for Sb_2O_3 in textiles (1 757 t/y), gives a total EU emission of 35 t/y (this assumes that the amount of textiles disposed of each year is equivalent to the amount of new textiles produced each year). Assuming that 30% of this release occurs in a region (see above) the total regional and continental releases from this source can be estimated as below (see Table3-15). It should be noted that there is a very large uncertainty in these estimates.

Summary of regional and continental emissions during service life of textiles flame-retarded with diantimony trioxide

A table showing the total environmental emissions during service life of textiles flame-retarded with diantimony trioxide is presented in Table3-15 below. These data are not used to calculate the regional PEC as this is based on measured data (see section 3.1.4).

Table3-15 Summary of the distribution of environmental emissions of diantimony trioxide during service life of textiles flame-retarded with diantimony trioxide.

Distribution	Total (kg Sb_2O_3 / y)	Continental (kg Sb_2O_3 / y)	Regional (kg Sb_2O_3 / y)
Industrial/urban soil	26 355	18 450	7 908
Surface water	8 785	6 150	2 636

3.1.2.3.4 Releases during waste treatment

Emissions during this stage are included in section 3.1.2.7 (Release from disposal) below.

3.1.2.4 Use as catalyst (PET industry)

The diantimony trioxide used as catalyst in the PET-industry is sometimes supplied wetted with monoethyleneglycol (MEG) to reduce dusting. This wetting is normally done at the diantimony trioxide production site and hence allows safer handling at PET production sites. There are two industrial use (conversion or processing) steps, first the production of the PET-polymer (polymerisation) and secondly the production of articles of PET (e.g. bottles). Due to technical constraints in the EUSES model, the first step has been entered as formulation, which is formally an incorrect classification, and the second as industrial use. This has no bearing on the results and is the best way to handle this in the model. This will *only* be done in EUSES, in the text the steps will be called production of PET polymer and production of articles of PET respectively.

3.1.2.4.1 Releases from production of PET resin (polymerisation)

Eleven sites in EU15 producing PET resin use diantimony trioxide. All of them answered the questionnaire sent. All reported data for water emissions, and approximately half reported data for air emissions. The emissions to air are very low (< 2 kg Sb_2O_3/y). The information available is deemed to be reliable and relevant for the purpose of characterising the releases, hence no default emission values will be used. No information on number of emission days were disclosed, therefore a default value of 300 days will be used.

Local emissions

The reported site-specific emissions are shown in Table3-16. The values from the site with the highest emissions will be taken forward to the risk characterisation.

Table3-16 Local emissions from the use of diantimony trioxide as catalyst in PET polymerisation (resin production).

	Air stack emissions		Wastewater		WWTP
	kg Sb_2O_3 / y	kg Sb/day	kg Sb_2O_3 / y	kg Sb/day	
PET-2	reported to be very low	-	6.0	0.017	no
PET-3	no data	no data	1.3	$3.6 * 10^{-3}$	off-site
PET-4	no data	no data	1.2	$3.3 * 10^{-3}$	off-site
PET-5	0	0	13	0.036	on-site
PET-7	no data	no data	0.4	$1.0 * 10^{-3}$	on-site
PET-8	< 0.08	$0.2 * 10^{-3}$	0.4	$1.0 * 10^{-3}$	on-site and off-site
PET-9	no data	no data	50	0.14	off-site
PET-10	$3.4 * 10^{-7}$	$0.9 * 10^{-9}$	20	0.056	on-site
PET-13	no data	no data	0.4	$1.2 * 10^{-3}$	on-site
PET-16	2.0	$7 * 10^{-3}$	20	0.057	on-site

PET-17	1.8	$6 * 10^{-3}$	0	0	no water emissions

Total, regional and continental emissions from formulation of diantimony trioxide in PET

The total emissions have been set to the sum of the reported site-specific emissions. The regional distribution of PET-production has been shown to be fairly evenly spread over EU (IAOIA, 2004). The regional emission is therefore set to the emission from the site with the highest value. The environmental emissions are presented in Table3-17. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-17 Distribution of environmental emissions due to use of diantimony trioxide in polymerisation of PET.

Distribution	Total (kg Sb_2O_3 / y)	Continental (kg Sb_2O_3 / y)	Regional (kg Sb_2O_3 / y)
Air	5	3	2
Wastewater	113	63	50

3.1.2.4.2 Releases from production of articles and fibres of PET (industrial use)

No information on emissions was reported. The production of articles of PET is (from an emission perspective) very similar to the conversion of plastics containing diantimony trioxide as flame-retardant. Therefore the same source, OECD emission scenario document on Plastic additives (OECD ESD Plastics additives, 2003) has been used for this scenario. There is no specific heading for catalysts, but fillers or inorganic flame-retardants have been chosen as representative for diantimony trioxide in this case. The result in this scenario will be the same irrespective of whether emission factors for fillers or inorganic flame-retardants are used.

Emissions to air

The emission scenario document on plastics additives (OECD ESD Plastics additives, 2003) gives an emission factor of zero to air (inorganic flame-retardant, all other operations than machining/grinding). The emission to air has therefore been set to zero.

Emissions to wastewater and surface water

The emission scenario document on plastics additives (OECD ESD Plastics additives, 2003) gives an emission factor of 0.0001 to wastewater and/or solid waste (inorganic flame-retardant, all other operations than machining/grinding). In the absence of other information this emission factor is used for the estimation of emissions to water.

The sites where production of PET articles is done do not normally use large amounts of water in their processes. It has therefore been assumed that most sites do not have any wastewater treatment plant of their own and that the wastewater goes to a municipal STP.

Summary of regional and continental emissions processing diantimony trioxide in PET

The quantity of diantimony trioxide used in this application is 950 t (PET resin 541, PET fibres 409). This multiplied with the emission factor from the OECD ESD (see above) gives a total release of 95 kg/year. No detailed information on the geographic distribution of production of articles from PET is available. There are a large number of sites where production takes place, with a reasonable distribution over Europe. The distribution of handled volume in a region has therefore been set to 10% of the total. The results are presented in Table3-18. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-18 Distribution of environmental emissions due to industrial use (shaping) of diantimony trioxide in PET.

Distribution	Total (kg Sb_2O_3/y)	Continental (kg Sb_2O_3/y)	Regional (kg Sb_2O_3/y)
Air	0	0	0
Wastewater	95	85	10

Local emissions

For the evaluation of generic local emissions the fraction of the regional release that is assumed to be released from the main local source (fraction of main source) is set to 0.05 and the number of emission days to 300 (default from TGD Table B3.9). No site-specific emissions were reported. The emissions in the generic local scenario generated by EUSES are shown in Table3-19.

Table3-19 Local emissions due to industrial use (shaping) of diantimony trioxide in PET.

	Air		Wastewater		Notes
	kg Sb_2O_3/year	kg Sb/day	kg Sb_2O_3/year	kg Sb/day	
Generic (from EUSES)	0	0	0.5	$1.3 * 10^{-3}$	

3.1.2.4.3 Releases during service life

PET is mainly used for packaging for foodstuff. This is discussed in section 4.1.1.3 Consumer exposure. A proportion of this PET is later recycled. Recycled PET is mainly used in the fibre market, mainly in various clothing and geo-textiles (for backpacks etc). For the non-packaging use of PET, emissions have been assumed to be mainly from washing of PET-fibres. An emission factor of 0.0001 has been chosen based on the extraction studies made on PET in contact with foodstuff[6] (Fordham et al., 1995). The fraction of PET recycled is around 30% (with a high degree of uncertainty in this number due to different geographical and time splits in the statistics) and growing, based on statistics from Petcore (2003).

In addition, textiles containing polyester also contain antimony since Sb_2O_3 is the major catalyst in production of polyester plastics, such as PET, and has so been for the last 50 years

[6] a sample of 3 g PET containing 160 µg/kg of diantimony trioxide in 20 ml (approximated in this calculation to 20 mg) of food simulant gave (after leaching at 40°C, 10 days) a concentration in the simulant of 2.7 µg/kg giving an emission factor of (0.020 * 2.7)/(3 * 160) = 0.0001

e.g. (Otto et al., 2001). The concentration of Sb in polyester plastic normally ranges 150-300 ppm (Otto et al., 2001), which is in concordance with the concentration of antimony in textiles using polyester, which have been measured to be 300 ppm (IFP Research AB, 1999). Antimony is partially leached when the textiles are being processed, and antimony concentrations of 2 mg/l have been measured in dye bath after dying polyester. Antimony may also to some extent leach at washing and rinsing. More than 90% of all polyester for textiles (mainly PET) being produced uses antimony based catalysts (IFP Research AB, 1999). It was noticed in two municipal wastewater treatment plants in Borås, Sweden, to which textile industry are connected, that the concentration of antimony in sewage sludge increased during recent years. This increase coincided in time with import of polyester from countries outside of Europe, and it was suspected that the concentration increase in sewage sludge was due to contamination of antimony in this polyester (Sternbeck et al., 2002a). In addition, Eriksson (2001) measured the concentration of antimony in sludge from WWTP, and the highest concentration was measured in the city of Borås, which is situated in a region in Sweden with a large textile industry.

All emissions from service life are assumed to be to water.

Emissions to wastewater

Of the 950 t of diantimony trioxide used annually as catalyst in PET manufacture, 409 t are used in fibres. Using the emission factor from the previous section, the diffuse emissions from the use of this quantity are therefore 0.0001 * 409 = 0.041 t Sb_2O_3/y. This is assumed to go to municipal STPs. PET resin is mainly used in packaging and releases to the environment are assumed to be negligible compared to other releases.

Total, regional and continental emissions service life diantimony trioxide in PET

The emissions from service-life are diffuse and assumed to be equally distributed throughout Europe i.e. 10% on the regional scale and 90% on the continental scale. The results are presented in Table3-20 below. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Default values from TGD are used for input of emission days (365).

Table3-20 Distribution of environmental emissions due to service life of diantimony trioxide in PET.

Distribution	Total (kg Sb_2O_3/y)	Continental (kg Sb_2O_3/y)	Regional (kg Sb_2O_3/y)
Air	0	0	0
Wastewater	41	37	4

3.1.2.4.4 Releases during waste treatment

Emissions during this stage are included in section 3.1.2.7 (Release from disposal) below.

3.1.2.5 Releases from use of diantimony trioxide in paint, pigments and ceramics

Diantimony trioxide is used in the manufacturing of "Complex Inorganic Coloured Pigments" (CICP), which are further used in subsequent industries such as plastics (50%), coatings (35%), enamels and ceramics (10%) and building materials (5%). The pigment production process involves chemical transformation of the input materials into a crystal (rutile) host matrix in which various metal cations (e.g., Ti, Ni, Cr) apart from Sb are incorporated. Antimony is chemically bonded (as Sb (V) in the rutile lattice, taking the place of some of the Ti-ions.

Once incorporated into these rutile structures, antimony is no longer present as diantimony trioxide. For this reason, this report is restricted to the relevant process stages involving releases of diantimony trioxide itself. Apart from the use in pigment production diantimony trioxide is used as flame-retardant in special paints. The former use consumes some 500 t Sb_2O_3/y and the latter 400 t Sb_2O_3/y. The use in Ceramics is covered by these scenarios, and is not detailed further.

Diantimony trioxide is consumed primarily as dry powder in this sector, with smaller amounts being used in wetted form or in an aqueous dispersion.

Twenty-nine sites using diantimony trioxide has been identified in this sector, with an annual total use of 1 123 t/y. Five sites use more than 125 t/y, and the largest one use 240 t/y. Eight sites have reported release data to water, covering 79 % of the tonnage used, five sites have reported release to air covering 50 % of the tonnage used. The number of production days reported range from 30 to 365. Eight sites have reported that they have a wastewater treatment on site before releasing to surface water. In the six cases where specified it is a physico-chemical process. The sludge is disposed to landfill (EURAS, 2007; EBRC, 2006i).

3.1.2.5.1 Releases from formulation (paint and pigments manufacture)

Emissions to air

The reported emissions cover a fairly large part of the use in this sector (50 % of tonnage and 17 % of sites). The reported values are deemed to be representative for the industry sector based on this coverage together with information on processes collected and presented by industry and the regional distribution (replies from 10 countries in EU15) (EURAS, 2007).

The derivation of a 90-percentile on the available dataset of emission factors is questionable, therefore the highest reported emission factor (0.00011) have been used to calculate releases for the sites not reporting.
The total releases to air are then:

tonnage used at non-reporting sites	50 % * 1 123 * 10^3 kg
multiplied by highest reported emission factor	* 0.00011
plus reported releases	+ 31.5 kg
equals	= **93 kg Sb_2O_3/y**

Emissions to wastewater and surface water

The reported emissions cover a large part of the use in this sector (79 % of tonnage and 28 % of sites). The reported values are deemed to be representative for the industry sector based on this coverage together with information on processes collected and presented by industry and the regional distribution (replies from 10 countries in EU15). (EURAS, 2007)

The derivation of a 90-percentile on the available dataset of emission factors is questionable, therefore the highest reported emission factor (0.00011) have been used to calculate releases for the sites not reporting.
The total releases to wastewater are then:

tonnage used at non-reporting sites	21 % * 1 123 * 10^3 kg
multiplied by highest reported emission factor	* 0.00011
plus reported releases	+ 38 kg
equals	= **64 kg Sb_2O_3/y**

Total, regional and continental emissions diantimony trioxide used in paint

To calculate the total releases, reported values have been used for the tonnage where these are available, with highest reported emission factor used for the remaining tonnage. The default value (10%) for distribution between regional and continental has been used.

The results are presented in Table3-21 below. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-21 Distribution of environmental emissions due to formulation of paint containing diantimony trioxide.

Distribution	Total (kg Sb_2O_3/y)	Continental (kg Sb_2O_3/y)	Regional (kg Sb_2O_3/y)
Air	93	84	9
Wastewater	64	57	6

Local emissions

For the local scenarios, two realistic worst case generic scenarios with the highest reported emission factors and values for emission days generated from TGD B tables have been developed, one for pigment production (using 50 t Sb_2O_3/y), the other for production of paint with diantimony trioxide as flame-retardant (using 40 t Sb_2O_3/y). These sites release waste water to a municipal STP, and the number of emission days is 300 (default from TGD Table B3.13).

The reported local emissions and the modelled for the default generic sites are presented in Table3-22. Release per day has been calculated using reported site-specific number of emission days and bypass STP has been used in modelling because the emission factor used is after on-site WWTP. Sewage sludge is assumed (as a realistic worst case) to be used on agricultural soil and this is included in the modelling.

Table3-22 Local emissions due to the formulation of paints and pigments containing diantimony trioxide.

	Stage	Form	Air *	Water

			kg Sb_2O_3/y	kg Sb/day	emission factor	kg Sb_2O_3/y	kg Sb/day	emission factor
Generic, paint	formulation	powder	4.3	0.013	0.0001	4.6	0.013	0.0001
Generic, pigment	formulation	powder	5.4	0.016	0.0001	5.7	0.016	0.0001
MN61S	form + proc	powder	0.1	0.002	$1 * 10^{-6}$	0	0	no data
AMI-3G	form + proc	powder	31.4	0.075	0.0001	34	0.080	0.0001
AMI-13S	processing	powder	no data	no data	no data	no data	no data	no data
ProcPaint-1	formulation	powder	no data	no data	no data	1.4	0.004	$6 * 10^{-6}$
Site 136	form + proc	disp+pow	no data	no data	no data	0	0	0
Site 34	form + proc	powder	no data	no data	no data	0	0	0
Site 578	formulation	powder	no data	no data	no data	no data	no data	no data
Site 575	formulation	powder	$2 * 10^{-6}$	0	0	no data	no data	no data
Site 581	form + proc	powder	0	0	0	2.4	0.006	0
Site 135	formulation	dispersion	0	0	0	0	0	0
Site 40	form + proc	powder	no data	no data	no data	0	0	0
Site 564	processing	powder	no data	no data	no data	no data	no data	no data
Site 580	formulation	powder	no data	no data	no data	no data	no data	no data
Site 568	form + proc	powder	no data	no data	no data	no data	no data	no data
Site 569	processing	powder	no data	no data	no data	no data	no data	no data

* stack emissions

3.1.2.5.2 Releases from industrial use (application of paint)

When diantimony trioxide is used in pigment production the diantimony is included into a crystal host matrix and will not leach out of this structure under foreseeable use. What is covered in this scenario is when diantimony trioxide is added to paint as a synergistic (see section 2.3.2) flame-retardant.

Not much information is available on the application of paint containing diantimony trioxide and no data for emissions were reported. The application of paint is largely done in the same way as ordinary decorative paints, i.e. by roller, brush or (less often) spraying. The painters often, but not always, have some extra training. The painting is largely done in temporary workplaces, i.e. in the building where the object (wall etc.) to be flame-retarded is located (Personal communication with industry, 2003). Default tables from TGD part II was used to ascertain emission factors. There is an emission scenario document for the paints, lacquers and varnishes industry, but this particular application field is not covered, and it is therefore recommended in the ESD that the default tables of TGD part II be used. Flame-retardants are not included as use category in the applicable table. Use category fillers have been used as diantimony trioxide fits that category (for emission purposes).

Emission sources will be relevant only for a very limited time, i.e. while the painting is in progress, which is likely to be days rather than weeks, and the time between paintings of a certain object is likely to be several years. Furthermore, the quantity of diantimony trioxide

used at a specific paint object is limited as are the emissions. Therefore no local scenarios have been developed for this scenario. It has instead been handled as diffuse emissions.

The quantity used in this scenario is 400 t (see chapter 2).

Emissions to air

There were no reported data for emissions to air. The emission factor given is zero (IC=14 Paints, lacquer and varnishes industry, Table A3.15: solvent based, use category 20 fillers).

Emissions to wastewater and surface water

There were no reported data for emissions to water. The emission factor from TGD part II is 0.001 (IC=14 Paints, lacquer and varnishes industry, Table A3.15: solvent based, use category 20 fillers) and will be used for the whole volume. The location of the emissions varies from time to time, because of the way the paint is used. Releases are assumed to go to a STP.

Total, regional and continental emissions diantimony trioxide used in industrial use of paint

The emissions are assumed to be equally distributed throughout Europe, and the default distribution factors have been used, i.e. 10% on the regional scale and 90% on the continental scale. Values of emissions generated by EUSES are shown in Table3-23 below. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-23 Distribution of environmental emissions due to industrial use of diantimony trioxide in paint.

Distribution	Total (kg Sb_2O_3/y)	Continental (kg Sb_2O_3/y)	Regional (kg Sb_2O_3/y)
Air	0	0	0
Wastewater	400	360	40

3.1.2.5.3 Releases during service life

The type of paint where diantimony trioxide is used is generally subject to limited abrasion and aggressive liquids. It has been formulated to retain its flame-retarding properties, which means that it must not leach out the compounds that give these properties. Any emissions of diantimony trioxide are likely to be in the form of part of flakes or particles of paint. The dominating use is indoors. Paint that is removed or flake will therefore be either handled as industrial waste or as a worst case as household waste. In either case the contribution of environmental emissions from this source will be covered by the waste treatment scenarios.

3.1.2.5.4 Releases during waste treatment

Emissions during this stage are included in section 3.1.2.7 (Release from disposal) below.

3.1.2.6 Use in glass

Diantimony trioxide is being used at five sites in the glass sector. It is being rapidly and continuously replaced by sodium antimonate in this use. Diantimony trioxide is used together with lead oxides to produce crystal glass (EBRC, 2006b).

3.1.2.6.1 Releases from formulation

Reported release values are available for one site. This site covers 36 % (90 t/y) of the total usage of Sb_2O_3 in glass formulation in the EU, which is expected to be about 244 t/y (EURAS, 2006a) The site has provided information on emissions to air and water for the year 2001 and 2004.

Emissions to air

The reported emission factor, which is based on measured data, was 256 * 10^{-6} (2001 measurements) and 3.3 * 10^{-6} (2004 measurements). The higher (2001) emission factor will be used as a realistic worst case also for the remaining 64% of the volume because of the very limited amount of reported data. Even though this is a limited fraction of the total glass formulation sites using Sb_2O_3, it is considered that the emission figures from this site is more representative for the unknown fraction than the TGD defaults for the industry and use category "Other". Importantly, this assumption is not considered to be of any major importance for the diantimony trioxide emissions as a whole, bearing in mind the relatively limited and diminishing fraction of Sb_2O_3 associated with this use.

Emissions to wastewater and surface water

The glass-formulating site stated no emission to water. Based on general process information from two more sites (EBRC, 2006b), it is assumed that this is relevant also for the remaining 64% of the volume.

Total, regional and continental emissions during glass formulation using Sb_2O_3

The generic site developed (see below) is placed in the region, the result is presented in Table3-24. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4).

Table3-24 Distribution of environmental emissions due to use of diantimony trioxide in formulation of glass.

Distribution	Total (kg Sb_2O_3/y)	Continental (kg Sb_2O_3/y)	Regional (kg Sb_2O_3/y)
Air	63	40	23
Water	0	0	0

Local emissions

The latest (2004) reported local emissions from the reporting glass-formulating site, MN62S are presented below, reported number of emission days is 225. A generic scenario based on default values from EUSES with a total use of 90 t Sb_2O_3/y based on the use at the reporting

site, and a reported historical emission factor (from 2001) from MN62S, has been taken as a realistic worst-case, and is presented in Table3-25. No information on other sites is available.

Table3-25 Local emissions from the formulation of diantimony trioxide in glass.

	Form	Air stack emission		Water	
		kg Sb_2O_3/y	kg Sb/day	kg Sb_2O_3/y	kg Sb/day
Generic	powder	23	0.154	0	0
MN62S	powder	0.2	0.0007	0	0

3.1.2.6.2 Releases from processing

The formulation of a diantimony trioxide containing product, see above, will be followed by a processing stage, in which the diantimony trioxide containing product will be processed. The total volume used will be 244 t/y, which is the same as for the preceding formulation scenario.

Information from one glass producer, identified as ProcGlass-2 in this RAR, using Sb_2O_3, is available (Carlsson, 2002). This site covers 10% (25 t/y) of the total usage of Sb_2O_3 in glass formulation in the EU. Information from another glass producing site, identified as ProcGlass-3 in this RAR, using less Sb_2O_3 (< 10 t/y) is also available. (Davidsson, 2007). Even though this is only a relatively small fraction of the total glass processing sites using Sb_2O_3, it is considered that the emission figures from these sites are more representative for the unknown fraction (85%) than the TGD defaults for the industry and use category "Other". Importantly, this assumption is not considered to be of any major importance for the diantimony trioxide emissions as a whole, bearing in mind the relatively limited and diminishing fraction of Sb_2O_3 associated with this use.

Emissions to air

Data for ProcGlass-2 on emissions to air were available from 1990 to 2001 (fairly stable use and no change in process during this period, air filter installed 1984). The 90P-value of the emission factors was 1.3×10^{-3}. This emission factor will be used in the risk assessment. The more recent value (for 2006) for ProcGlass-3 gives an emission factor less than 0.15×10^{-3}. The waste-gas from the handling of raw-materials is released via a textile filter.

Emissions to wastewater and surface water

The glass-manufacturing sites have reported emission to water, which is assumed to be relevant also for the unknown processors. Measured elevated values of antimony near glass producers (see Table3-68) are probably caused partly by aerial deposition and/or historical pollution. ProcGlass-2 reports releases and use that gives an emission factor to water of 0.3×10^{-3}, for 2000, whilst ProcGlass-3 have an emission factor of 0.09×10^{-3} for 2006. Both sites have on-site WWTP with sludge being handled as hazardous waste. Due to the limited coverage, the higher emission factor will be used for calculating releases for sites without information on releases.

Total, regional and continental emissions during glass processing using Sb_2O_3

The known glass-processing site covers 15 % (35 t/y) of the total usage of Sb_2O_3 in glass processing in the EU. It is assumed that the EU-market will contain several small glass manufacturers, rather evenly distributed, therefore the standard 10 % regional, 90% continental distribution will be used. Values of emissions generated by EUSES are shown in Table3-26 below. These emissions are not used to calculate the regional PECs, as these are based on monitored data (see section 3.1.4). Releases are assumed to be to surface water rather than a STP as the sites have on-site WWTP.

The total releases to air are then:	
tonnage used at sites not reporting emission data	85 % * 244 * 10^3 kg
multiplied by highest reported emission factor	* $1.3*10^{-3}$
plus reported releases	+ 31 kg
equals	= **301 kg Sb_2O_3/y**

The total releases to surface water are then:	
tonnage used at sites not reporting emission data	85 % * 244 * 10^3 kg
multiplied by highest reported emission factor	* $0.3*10^{-3}$
plus reported releases	+ 13 kg
equals	= **75 kg Sb_2O_3/y**

Table3-26 Distribution of environmental emissions due to use of diantimony trioxide when manufacturing glass.

Distribution	Total (kg Sb_2O_3/y)	Continental (kg Sb_2O_3/y)	Regional (kg Sb_2O_3/y)
Air	301	271	30
Surface Water	75	67	8

Local emissions

For the evaluation of local emissions, the reported value for site ProcGlass-2 has been taken as a realistic worst case, the releases are presented in Table3-27. Exact information on number of emission days is not available. Default values from EUSES (120 and 150 emission days respectively) have been used. Based on available general information on production pattern, this is likely to be an underestimation.

Table3-27 Local emissions due to the use of diantimony trioxide when manufacturing glass.

	Air		Surface Water	
	kg Sb_2O_3/y	kg Sb/day	kg Sb_2O_3/y	kg Sb/day
ProcGlass-3 (Davidsson, 2007)	1.2	0.007	0.7	0.004
ProcGlass-2 (Carlsson, 2002)	32	0.22	7.5	0,05

3.1.2.6.3 Releases during service life

This life-cycle stage is not considered relevant for Sb_2O_3 used in glass.

3.1.2.6.4 Releases during waste treatment

Emissions during this stage are included in section 3.1.2.7 (Release from disposal) below.

3.1.2.7 Release from disposal

Antimony concentration in municipal solid waste has been estimated to be about 10-60 ppm (IAWG, 1995), even though large variation can be expected between different waste fractions. Concentrations of several hundreds up to several thousands ppm have been reported from electrical and electronic appliances (Vehlow and Mark, 1997; Wagner et al., 1992). Different composition of the various input waste streams is most probably the reason for discrepancies between different studies even though they all appear to be within the same magnitude. (Van Velzen et al., 1998) concluded that the average antimony concentration of the feed stream to waste incinerators approximately is 42 mg Sb/kg waste. (Sternbeck and Munthe, 2001) analysed ashes from four Swedish incinerators for household and industry refuse in which the antimony concentration was determined to be 40 ± 27 mg Sb/kg refuse. The concentration value of antimony in the Municipal Solid Waste (MSW) taken forward in this RAR will be 40 mg Sb/kg waste. This estimate is supported by calculations[7], using the selected concentration

[7] Calculations performed by EURAS (e-mail to rapporteur: 12/10-2004):
Assumption by the rapporteur (1 % emission for air and 0.3 % emission to water but corrected for ww/dw conversion)

Sb content in MSW = 40 g/t

MSW incinerated = 160,058 kt ww. Then the total amount of waste incinerated expressed as wet weight should be first translated to dry weight. With a typical moisture content of 30 %.

160,058 * 0.7 = 112,041 kt dw

Sb in MSW (kg) = 4,481,624 kg

Air emissions = 0.01* 4,481,624 = 44,816 kg
Water emissions = 0.003*4,481,624 *0.4 (60 % removal in STP) = 5,378 kg
Going to sludge = 0.003*4,481,624 *0.6 (60 % removal in STP) = 8,067 kg

Antimony present in incineration residues =

4,481,624 – 44,816 – 5, 378 -8,067 = 4,423,363 kg

50 % goes to bottom ash = 4,423,363 * 0.5 = 2,211,681 kg
50 % goes to fly ash = 4,423,363 * 0.5 = 2,211,681 kg
On average bottom ash constitutes 21.3 % by weight of the waste input (expressed as wet weight) and fly ash 2.8 % (see TRAR, 2003).

Amount of bottom ash produced = 160,058 kt ww * 0.213 = 34,092 kt dw

Amount of fly ash produced = 160,058 kt ww * 0.028 = 4,482 kt dw

Sb concentration in bottom ash = 2,211,681 kg Sb/34,082 kt bottom ash = 65.9 mg/kg dw

Sb concentration in fly ash = 2,211,681 kg Sb/4,482 kt fly ash = 493 mg/kg dry wt

of 40 mg Sb/kg dw, which results in concentrations of 64 mg Sb/kg dw and 493 mg Sb/kg dw in fly ash and bottom ash, respectively. These values are in the range of reported concentrations measured, which are 10-432 mg Sb/kg dw and 260-1,100 mg Sb/kg dw for fly ash and bottom ash, respectively (DTU, 2001).

(van Velzen et al., 1998) made an extensive review of antimony applications and flow paths in waste, and estimated that over 60% of the total industrial consumption is in the production of flame-retardants, and the remaining 40% was estimated to be equally divided between metal and non-metal products. (Paoletti et al., 2001) compared the industrial application of antimony compounds with the Sb contributions of solid waste fractions, and speculated that antimony entering a waste combustion plant mainly will be present in the oxidic form, that the metallic form should be of lower importance and that an unknown fraction may be present as antimonates. Sb_2O_3 is the absolute dominating oxic form of antimony presently used and also constitutes the largest fraction of all antimony compounds used. The major use of Sb_2O_3 is in products with limited possibilities of recycling (e.g. flame-retarded textiles and plastics), why much of the total antimony measured in MSW probably has Sb_2O_3 origin. Additionally, a considerable amount of imported products produced outside of the EU, containing Sb_2O_3, are likely to be disposed of within the EU and end up in the MSW.

Waste management practise varies considerably among different countries and regions in the EU, so in determining the realistic worst case, due consideration need to be given whether incineration, direct landfilling or recovery should be considered as the realistic worst case. According to Table3-29 the current situation in the EU is that about one fourth of the MSW is incinerated, and the remaining amount is landfilled (see (TRAR302+303_env.doc, 2003) for more information). However, according to the TGD, unless more detailed information is available on waste management of specific product types containing the substance in concern, two alternative scenarios may be explored as a first approach assuming either 100% incineration or 100% landfilling. Both these conditions are relevant for different parts of the EU. As a consequence of this sensitivity analysis any risk indication may refer to a certain waste management practice rather than an average waste management in a generic EU region.

Table3-28 Landfilling and incineration of MSW (in kt ww) in Europe for the period 1995-2001 (from TRAR302+303_env.doc , 2003).

Country	Year	MSW landfilled (kt ww)	MSW incinerated (kt ww)
Austria	1999	1,099	479
Belgium	1998	1,473	1,369
Denmark	1999	361	1,730
Finland	1997	1,610	80
France	1998	23,352	10,781
Germany	2001	16,000	12,000
Greece	1997	3,561	0
Ireland	1995	1,432	0
Italy	1998	20,768	1,949
Luxembourg	1998	62	123
Netherlands	1999	1,136	3,859

Norway	1998	1,843	374
Portugal	1999-2002	2,603	1,060
Spain	1999	17,477	1,327
Sweden	1998	1,300	1,400
UK	1999	26,860	2,590
Total EU-16	160,058	120,937	39,121

3.1.2.7.1 Incineration

Antimony entering into standard MSW incineration will be distributed among various output fractions, such as stack emissions (flue gas), wastewater, fly ash, bottom ash and slag. The distribution pattern of antimony over these incineration residues is depending on the physical-chemical properties, the gas cleaning technology and the operation and maintenance conditions. While the flue gas and wastewater emissions are immediate, emissions of the incineration residues (via disposal and/or re-use) are delayed. The approach taken in the TRAR302+305 (2003) will be followed, when possible, in this RAR. However, only the present time scenario will be used.

Emissions to air

The results of mass balance studies from some refuse incinerators indicate that the emission of antimony to air is lower, or substantially lower than 1% (Table3-29). As can also be seen from this compilation, incinerators that do not use wet scrubbers have higher emissions to air.

Table3-29 Distribution in percentage of antimony at waste incineration (Sternbeck et al., 2002).

Reference	Wet scrubber	Bottom ash	Fly ash	Emission to air	Untreated water	Emission to water
van Velzen et al. ,1998	Yes	48.3	50.9	0.07	0.7	< 0.3
Nakamura et al., 1996	Yes	53.7	45	0.4	0.9	
Nakamura et al., 1996	No	35.2	63.6	1.2	-	
Paleotti et al., 2001	Yes	45	~55	< 1	< 1	-

Sternbeck and Östlund (1999) concluded, based on literature data, that 0.1-1% of the antimony in MSW that was incinerated is emitted to the atmospheric compartment. This interval is in agreement with the figures presented in Table3-29.

Measured antimony concentrations are very scarce since antimony is only a minor constituent in flue gas.

According to the Swedish Association of Waste Management, RVF (Hedenstedt, 2005), emission of Sb to air due to MSW incineration in Sweden range between 10-20 kg/y. Using the higher end of this interval, the amount of Sb being emitted to air in Sweden, at a 100% incineration scenario, would be 38.6 kg Sb (= 20 kg/0.52; according to Table3-28 above, the

proportion MSW being incinerated in Sweden is about 52%). The total amount of MSW that would be incinerated at a 100% incineration scenario would be 2700 t/y, and the corresponding amount of MSW that would be incinerated in the EU would equal 160,058 t/y. Assuming the same efficiency of emission control systems in the entire EU, the total amount of antimony being emitted to air due to incineration of MSW would be about 2.3 t/y (= 38.57 kg Sb/y×160058/2700). However, according to RVF (Hedenstedt, 2005) the emission control systems in use in Sweden are probably among the more efficient in use in the EU, why this figure therefore not may be consider a realistic worst case.

Measurements in Germany for the MSW in Bamberg show a typical concentration of 0.0016 mg Sb/Nm^3, with a maximum value of 0.0018 mg Sb/Nm^3 (EURAS comments on section 3.1.2.7 releases from disposal *Draft RAR, June, 2004*, 2004). Using this latter value results in an air emission in the EU of 1.6 t Sb/y (=160,058 kt ww * 5,500 Nm^3 flue gas/t * 0.0018 mg Sb/Nm^3)/1000 = 1,584 kg Sb).

Based on the above, the lower value in the range proposed by Sternbeck and Östlund (1999), i.e. 0.1%, will be used.

Combining this emission figure with the average antimony concentration (40 ppm) in MSW (see section 3.1.2.7), a typical moisture content in MSW of 30%, and 100% incineration results in an emission of 4.5 t/y to the atmospheric compartment (see calculations below).

Emission to air: 160,058 * 1.0 * 0.7 * 40 * 0.001 = 4.48 ~ 4.5 t Sb/y
Ashes: 160,058 * 1.0 * 0.7 * 40 * 0.999 = 4477 t Sb/y

The emission figures calculated above are presented in Table3-30 below.

Table3-30 Emissions of antimony to air including distribution to ashes due to 100% incineration of MSW within the EU.

Conc. of Sb in MSW	Emission factor at incineration	Emission to air (t Sb/y)	Ashes (t Sb/y)
40 ppm	0.1%	4.5	4477

Emissions to water

Emissions to water results essentially from discharge of wastewater from incineration plants with wet flue gas cleaning systems. The flue gas from waste incinerators is usually passed through a series of wet scrubbers to remove gaseous pollutants like SO_2, HCl, HF, etc. (van Velzen et al., 1998). During this operation, a certain amount of heavy metals (Hg, As, Sb, etc.) is also dissolved in the washing liquid. Before disposal, the scrubbing liquid is treated for purification. This process includes precipitation of heavy metals in the form of hydroxides, however (Jekel et al., 1990 and Enders et al., 1990) reported that only 60% of the antimony is removed by precipitation. For the calculation of this RAR it will be assumed that 60% of the antimony in the influent incinerator wastewater is removed as sludge and that 40% remains in the wastewater and will be released to WWTP. The sludge generated goes to a hazardous waste landfill.

Based on the figures in Table3-29 <0.3% will be released to water. An emission figure of 0.3% will be used as a realistic worst case in this RAR.

Combining this emission figure with the average antimony concentration (40 ppm) in MSW (see section 3.1.2.7), typical moisture in MSW of 30%, and 100% incineration of MSW within the EU results in the emissions presented in Table3-31 below.

Emission to wastewater: 160,058 * 1.0 * 0.7 * 40 * 0.003 * 0.4 = 5.4 t Sb/y

Emission to sludge: 160,058 * 1.0 * 0.7 * 40 * 0.003 * 0.6 = 8.1 t Sb/y

These emission estimates are supported by calculations[8], performed on reported effluent concentrations for the Bamberg incinerator, which resulted in emissions to wastewater and sludge of 6.4 t Sb/y and 8.1 t Sb/y, respectively.

Table3-31 Emissions of antimony to water and sludge due to 100% incineration of MSW within the EU.

Emission to wastewater (t Sb/y)	Emission to sludge (t Sb/y)
5.4	8.1

Delayed emissions from incinerator residues

While the emissions with flue gas are immediate, emissions from the residual fractions will be delayed and may result in a long-term diffuse emission potentially contaminating groundwater, surface water and soil. In this RAR a distinction is only made between bottom ash and fly ash. Other flue gas cleaning products generated in the process of removing acid gases are not specifically addressed.

Contradictory to the thermodynamic models, which predict a high volatile behaviour of antimony compounds under the conditions of solid waste combustion (with a major fraction leaving the combustion chamber with the flue gas), experimental data show that about 50% of the antimony input remains in the bottom ashes (Paoletti et al., 2001). This has been explained by the formation of thermally stable antimonates due to interactions with other chemical compounds such as calcium oxide in the fuel bed. Such species would counterbalance the volatilizing effect resulting from the presence and/or formation of antimony oxides and chlorides. Antimony showed a pH-dependent leaching behaviour in the bottom ashes with a maximum leachibility for pH-values about 6-8, which is typical for anionic species, and therefore supports the thesis that antimony is present in the grate ashes in an anionic speciation (Paoletti et al., 2001).

A practical consequence of these anionic properties of antimony is the delayed release after aging when carbonation and hydration processes have lowered the pH value, from originally 11-13, to about 8-10 (Meima and Comans, 1998). This could possibly lead to an underestimation of the leaching behaviour if the leaching tests were carried out on fresh ashes.

[8] Calculations performed by EURAS (2004):

If 13.4 t Sb to wastewater is used, an effluent concentration of 0.03 mg/l can be calculated.

For the Bamberg incinerator a concentration of 0.04 mg/l is reported.

(160,058 kt ww * 2,500 m3/kt * 0.04 mg Sb/)/1000 = 16, 006 kg Sb = 16 t Sb/y

16,006 * 0.4 = 6,402 kg = 6.4 t Sb remaining in the wastewater
16,006* 0.6 = 9,604 kg = 9.6 t Sb going into sludge

The incineration of MSW in the EU results in considerable amounts of ashes, which results in an interest of finding possible use of the ashes.

Most of the fly ash generated by incinerators in the EU is landfilled with or without prior treatment. For fly ash it is general practice that it is placed in hazardous waste landfills or used for reclamation of old mine shafts or quarries. Fly ash has been re-used, with the largest application being as filling materials in asphalt (TRAR302+303_env.doc, 2003, and references therein). However, in the framework of the upcoming legislation it is unlikely that the use of fly ashes in asphalt will be a viable option for the future.

The use of processed bottom ashes in engineering applications just started in some countries like the UK whereas its use in the Netherlands in civil engineering started in the 1980´s. The ashes are used unbound as a bulk fill, e.g. to construct embankments, as substitute aggregate or for bound uses through incorporation into road paving (tarmac, asphalt) or construction blocs.

However, whether or not the ashes can be used depends on the outcome of specific leaching tests. If the results of the leaching tests are exceeding an imposed limit the bottom- or fly ash is classified as hazardous waste and should be landfilled in a hazardous waste landfill.

Burning of wastes containing antimony (certain plastics and textiles) in fluid bed boilers gives high leakage of antimony from the bottom ashes (Pettersson et al., 2003). The leakage can exceed the EU criteria for deposition of waste at landfills (2003/33/EC, 2003), which may result in requirements for handling as hazardous wastes. The leaching of antimony was not alarmingly high in the fresh ashes (Pettersson et al., 2003). However, leaching increased considerably after storage of the ashes for three months, and the antimony leaching from aged bottom ashes was high (compared to EU´s criteria for inert landfill (2003/33/EC, 2003). Storing of ashes does not reduce the leaching of antimony, as it normally does for metals. Instead the leaching increases, possibly due to dissolvment of minerals such as ettringit ($Ca_6Al_2(SO_4)_3(OH)_{12} \cdot 26\ H_2O$), when pH in the ashes drops from 11-12 to 8-10 (Meima and Comans, 1998). Ettringit have been shown to act as host and enclose certain metals, particularly oxyanions such as antimony, and have been found in grate ashes from refuse incinerators (Paoletti et al., 2001). The leaching of antimony from the grate ashes has proven difficult to reduce. Different methods have been tried, but the only conclusion that could be drawn was that the amount of antimony ending up in the bottom ashes had to be reduced (Suér and Lyth, 2003). Presently, antimony is considered as the most problematic substance in ashes from municipal incinerators as regards possible reuse of bottom ashes (Suér 2003, personal communication).

The emissions associated with landfilling of incineration products have not been assessed.

Summary of the overall antimony emissions due to incineration of MSW

As performed in the (TRAR302+303_env.doc, 2003), the 10% rule will be used to derive a reasonable worst-case emission estimate of the regional emission since incineration activities (large number of sites) are reasonably spread over the EU territory. The total annual amount of antimony emissions to the different compartments, due to incineration of MSW, is summarized in Table3-32.

Table3-32 Total annual amount of antimony emission to air, water and landfill due to 100% incineration of MSW within the EU.

Compartment	Released amount of antimony (t/y)	Continental (90%; t Sb/y)	Regional (10%; t Sb/y)
Air	4.5	4.	0.4
Wastewater	5.4	4.9	0.5
Landfill (ash + sludge)	4485	4037	448

3.1.2.7.2 Landfill

Release of pollutants from a landfill can occur over an indefinite period. Hence, the daily or annual release may result in a very small PEC and does not reflect the long-term emissions of a landfill. The guidance presented in the TGD on how to quantify the current and future landfill emissions, are more of a qualitative than quantitative nature, why the method outlined in (TRAR302+303_env.doc, 2003) will be followed in this RAR. The scenario used will be 100% landfill (see discussion above in Release from disposal).

Emissions of landfills can occur primarily by generation of landfill gasses and leaching of contaminants. Feldmann and Hirner (1995) analyzed landfill gases from a domestic waste deposit located in central Germany. They measured volatile Sb-gases ($Sb(CH_3)_3$) in the concentration range of 24-72 µg Sb/m^3 being released to the atmosphere, which can be compared with the concentrations of antimony in air of 0.003 and 0.014 mg/m3 which have been measured at low and high traffic intensity (Dietl et al., 1997), respectively. This indicates that landfill gases from domestic waste deposit, at least locally, may be important sources of antimony emissions to the atmosphere. However, in the long run, pollution via leachate is considered as being more important since production of landfill gas only lasts about one to two decades (Baccini et al., 1987; Flyhammar, 1995).

Leachate is generated as a result of the expulsion of liquid from the waste due to its own weight or compaction loading (termed primary leachate) and the percolation of water through a landfill (termed secondary leachate). The source of percolating water could be precipitation, irrigation, groundwater or leachate recirculated through the landfill.

Precipitation represents the largest single contribution to the production of leachate. There is some variation in the potential generation of leachate within the EU because precipitation and evapotranspiration depends on geographical location. In general, leachate quantities tend to be higher in the North of the EU than the South. But equally large variations can be found from east to west and over relatively short distances within Member States (Hjelmar et al., 1994). Reported leachate volumes vary from 25 m^3 to 3,000 m^3 per hectare (Flyhammar, 1995; Qian et al., 2002).

Looser and co-workers (1999), analysed leachates from 41 landfills, of which 6 were municipal solid waste landfills, in (mainly) Switzerland, but also in Italy and France. The resulting 50^{th} and 90^{th} percentiles of antimony concentrations measured were 0.4 µg Sb/l and 1.5 µg Sb/l, respectively. (Öman et al., 2000) measured antimony concentrations in leachate from municipal landfills in Sweden. The leachate concentrations from the three different landfills were 0.82, 1.1 and 1.7 µg Sb/l, respectively (detection limit 0.01 µg/l). The

concentrations measured are notably similar in size. The value of 1.5 µg/l will be used as a realistic worst case in this RAR.

It is assumed in this RAR that each waste component has the same likelihood of leaching out one gram of antimony (e.g. one gram of antimony in a certain amount of flame-retardant textile is assumed to have the same likelihood of leaching out as one gram of antimony in flame-retardant plastics).

Regional emissions of landfilling MSW

The regional emissions of antimony per year from MSW landfills in the EU can be calculated with the following formula.

Antimony flux (kg/y) = Landfill surface (ha) x leachate generation (m^3/ha y) x antimony concentration in the leachate (1.5 10^{-9} kg/l).

In the TRAR302+303_env (2003) a maximum leachate volume of 2,500 m^3/ha.y was calculated for an average rainfall of 7,999 m^3/ha.y. An average landfill surface area of 14.7 ha and an approximate volume of 18.7 ktonne MSW per landfill were also calculated in the TRAR302+303_env (2003). Assuming that each landfill has a surface area of 14.7 ha the total landfill surface can be calculated. Finally the antimony flux is calculated with the equation mentioned above. The calculation was thus made as follows:

The number of landfill sites in the EU was calculated from total quantity of MSW per year in the EU *divided by* average quantity per landfill site and year.

160 058 [kt/y] / 18.7 [kt/y] = 8 559

The total surface areas of all landfill sites in the EU was calculated from number of landfill sites *times* the average surface area for a landfill site.

8 559 * 14.7 [ha] = 125 821 [ha]

The total (from all landfill sites) antimony flux was calculated from the total surface area of all landfill sites *times* leachate volume per surface area in landfill *times* concentration of antimony in leachate.

125 821 [ha] * 2.5 * 10^6 [l/(ha y)] * 1.5 * 10^{-9} [kg/l] = 472 [kg/y]

The generated flux (leachate) may either be discharged to an off-site municipal sewage plant, discharged directly to surface water or enter into the groundwater compartment. Collection and discharge to a Sewage Treatment Plant (STP) is by far the most common discharge route for leachates from municipal waste landfills. A smaller proportion of leachate is discharged directly to surface waters. The latter is only allowed if the leachate quality fulfils certain requirements (sometimes pre-treatment, e.g. aerated lagoons, is needed). Most often this quality is governed by the presence of increased levels of BOD, COD and ammonium. Since at present no such information on legislations in the different MS is available, it is as a realistic worst-case scenario assumed that an antimony concentration of 1.5 µg Sb/l is of

suitable quality, and a discharge therefore is possible. Therefore the scenario of direct discharge to surface water is included.

Metals have been regarded only as a minor problem in the waste management of leachates and only rarely posed a significant problem in leachates from domestic waste landfills (Robinson, 1995).

Discharge criteria to surface water vary from one Member State to another. The highest proportion of landfills discharging directly to surface water is 23 % (Germany) with less than 10 % in other Member States (Hjelmar et al., 1994).

Permitted discharge to groundwater is uncommon for modern MSW landfills but may occur by old landfills or in the framework of an engineered leachate attenuation site (Robinson, 1995).

Since the number of sites designed with bottom liners and on-site leachate treatment plants is currently increasing it was proposed in the TRAR302+303_env (2003) to use the following regional allocation key for assessing the current emissions of landfills operational to date:

- 10 % direct discharge to groundwater (attenuation/dilution sites)
- 10 % direct discharge to surface water (sometimes an on site pre-treatment step is included)
- 80% to wastewater

Summary of the overall antimony emissions due to landfilling of MSW

An overview of the overall antimony emissions to groundwater/surface water and sludge (in kg/y) in Europe due to landfilling of MSW is presented in Table3-33. The overall antimony flux was calculated with the methodology described in previous paragraphs.

Table3-33 Total annual amount of antimony emission to surface water and groundwater due to 100% landfilling of MSW within the EU.

	MSW (kt ww)	**Total antimony flux (kg/y)**	**Fugitive emissions to surface water (kg/y)**	**Fugitive emissions to groundwater (kg/y)**	**Emissions to wastewater (kg/y)**
Allocation key			10 %	10 %	80 %
Total EU-16	160 058	472	47	47	378

The total amount of MSW in 1995-2001 for the EU-16 was 160,058 kt ww (112,041 kt dw). Using the methodology described in previous paragraphs, a total yearly antimony flux of 472 kg has been calculated, assuming 100% landfilling. Based on the calculations above the antimony emission to the groundwater compartment due to landfilling MSW is 47 kg Sb/y. An additional 47 kg is emitted to surface water and 378 kg of antimony is emitted in wastewater.

As performed in the (TRAR302+303_env.doc, 2003), the allocation of the total landfill emissions to the regional scale has been performed with the 10% rule (Table3-34). The emissions to ground water have not been taken further in the risk assessment.

Table3-34 Total annual amount of Sb emissions to groundwater/surface water and sludge within the EU due to 100% landfilling of MSW.

Compartment	Released amount of antimony (kg Sb/y)	Continental (90%; kg Sb/y)	Regional (10%; kg Sb/y)
Surface water	47	42	5
Wastewater	378	340	38
Groundwater *	47	42	5

* emissions to groundwater are included for information only, they are not used further in the risk assessment

3.1.2.8 Unintentional sources

Many of the important sources of anthropogenic emissions of antimony such as mining- and processing of ores, metal production, combustion of coal, and road traffic can be considered as unintentional sources. Emissions due to production, processing, and industrial use of antimony compounds besides Sb_2O_3 are also to be considered as unintentional sources. It is however decided not to include those as information is scarce and Sb_2O_3 constitutes the largest fraction of antimony used, 60% of total use of antimony (Roskill Information Services, 2001).

3.1.2.8.1 Production of non-ferrous metals

Since antimony compounds often are found in ores of metals such as copper, lead, silver, gold, and arsenic, mining and processing of these ores, and the subsequent production of these metals may thus also result in antimony emissions.

Skeaff and Dubreuil (1997) calculated atmospheric emission factors of antimony based on data from 1993 from Canadian smelters producing Pb, Cu, Ni and Zn. The emission factor to the atmospheric compartment from Pb-production was 20 g Sb/t, from Cu + Ni production 1.5 g Sb/t, and for Zn production 13 g Sb/t.

A recent estimate of emission to air from secondary copper smelting and refining in the EU was an annual average <3 g Sb/t of copper anode produced (EUROMETAUX Copper Industry, 1999). This emission estimate is very close to the one calculated by Skeaff and Dubreuil (1997), based on 10-year-old Canadian data. Since no emission figure is available for primary smelting, it is assumed that the emission figures for primary and secondary copper smelting are comparable in size, why the same figure is used for both processes. The figure of 3 g Sb/t will be used as an upper estimate, and the emission figure of 1.5 g Sb/t as a lower estimate of antimony emitted to the atmosphere due to production of copper in the EU.

The total copper smelting (primary and secondary) was 1391 kt in the year 2002 in the EU (Lindahl, 2003).

The release to the atmospheric compartment due to production of copper thus ranges from 2.1 t Sb/y, at the lowest, to 4.17 t Sb/y, at the highest. The higher of these two rather similar values, i.e. 4.17 t Sb/y, will be used in the RAR. The emissions are diffuse and the distribution will be 10% regional and 90% continental.

3.1.2.8.2 Coal combustion

The concentration of antimony in coal normally varies between 1-4 mg Sb/kg (Klein et al., 1975; Statens Naturvårdsverk, 1976; IEA, 1987). The mean value of antimony in USGS large database (2600 samples) of coal is 1.07 mg/kg (USGS, 1997). Emission factors for release to the atmosphere for coal combustion were presented by Nriagu and Pacyna (Nriagu and Pacyna, 1988) to be 0.2-1.5 g Sb/ton. The size of the range is determined by the concentration of Sb in the coal, and the type and efficiency of the pollution control installation. However, these emission figures appear to be too high considering the average Sb-concentration in coal. Efficiency of emission control devices in coal-fired power plants has been estimated to be 95-99.9% (DHV, 1985).

The consumption of coal in the EU during year 2002 was 233.1 million tonnes (EUROSTAT, 2003).

Using the consumption of coal combusted within the EU (233.1 million tonnes), the mean value of antimony concentration in coal (1.07 mg Sb/kg coal), and the range of emission control devices (95-99.9%) results in the following two alternative calculations:

95% efficiency of emission control devices: 233,100,000 * 1.07 * 0.05 = 12.47 t Sb/y
99.9% efficiency of emission control devices: 233,100,000 * 1.07 * 0.001 = 0.249 t Sb/y

The releases to the atmospheric compartment due to combustion of coal thus range from 0.25 t Sb/y, at the lowest, to 12.5 t Sb/y, as the highest. The higher of these values, i.e. 12.5 t Sb/y, will be used in the RAR. This number is probably an overestimation of the true emission from this source. The emissions are diffuse and the distribution will be 10% regional and 90% continental.

3.1.2.8.3 Road traffic

Several studies have found strong correlation between the concentrations of copper and antimony in areas affected by road traffic, suggesting a common source (Stechmann and Dannecker, 1990; Pakkanen et al., 2001). (Sternbeck et al., 2002b) found an average Cu:Sb emission factor ratio of 4.6:1 in two Swedish road traffic tunnels, which is close to the value found in an German tunnel (Stechmann and Dannecker, 1990), and suggested that the common source was the abrasion of brake linings. (Dietl et al., 1997), who found a maximum enrichment of antimony in the size interval 2.5-10 μm, which are particle sizes not characteristic of exhaust emissions, suggested an important source to be abrasions of tyre and brake linings. Furuta and co-workers (2005) analysed airborne particulate matter (APM) for a number of major elements and trace elements in air samples from Tokyo, and derived enrichment factors (EFs) for each element in each set of APM (d < 2 μm, 2-11 μm and > 11 μm). Antimony had the highest EFs of all elements analysed, and the results suggested that the antimony level in APM was extremely high. Each set of APM (diam. < 2 μm, 2-11 μm

and > 11 µm) was classified by shape, and the shape dependent constituents of single APM particles were quantitatively measured by SEM-EDX. High concentrations of antimony was found in APM< 2 µm and square particles. Particles less than 2 µm and square shaped particles were major particles produced by actual car braking experiments. Based on these experimental results the authors concluded that the source of antimony in squared APM < 2 µm is brake pad wear.

Use of antimony in brake friction materials have been reported for antimony sulphate as filler (Garg et al., 2000) and for antimony sulphide as frictional additives function as lubricants (Chan and Stachowiak, 2004; Melcher and Faullant, 2000). At a temperature of 380 ºC, which is a temperature easily reached in brakes, Sb_2S_3 have during experimental conditions been found to oxidise to Sb_2O_3 (Melcher and Faullant, 2000). Using the same type of experiment Jang and Kim (2000) observed thermal decomposition of Sb_2S_3 at temperatures above 300ºC. In braking simulations they noted a change in the friction coefficients of the Sb_2S_3-containing brake pads which they attributed to the formation of Sb oxides at elevated temperatures. (Uexküll et al., 2005) analysed brake pads and brake dust. In the braking and stopping simulations >90% (mass) of the dust was inhalable (PM10). Using differentiation via selective solubility the authors suggested that a considerable amount of Sb_2S_3 oxidised during the braking process, likely leading to the formation of Sb_2O_3 in the braking process.

(Sternbeck et al., 2002b) studied metal emissions from road traffic in two heavily trafficked tunnels in Gothenburg (Sweden), and derived emission factors, mainly representing vehicle emissions, for some metals, such as antimony. The vehicle derived metals (Cu, Zn, Cd, Sb, Ba, and Pb) mainly derive from wear rather then combustion; Cu, Ba, and Sb, probably dominated by brake wear. There are also indications that heavy-duty vehicles are stronger emitters of Ba and Sb, but not of Cu, than light duty vehicles. The average emission factors for the two tunnels were 31.6 ± 11.6 µg Sb/v km (Tingstad) and 50.8 ± 14.2 µg Sb/v km (Lundby). Both emission figures are based on TSP (Total Suspended Particles). The difference in size of these emission factors was suggested to be differences in fleet composition or in driving pattern.

These emission factors are considered fairly representative for urban driving, but possibly to high for highway driving. Ongoing studies by (Sternbeck, 2003) on measurements of PM10-fraction in central Stockholm (urban driving) and adjacent to the motorway E4 south of Stockholm (highway driving) resulted in the emission factors 20 ± 12 µg Sb/v km and 1.82 ± 1.28 µg Sb/v km, respectively.

Since the emission figure from central Stockholm only include PM10-fraction this probably underestimate the total emission of antimony, why instead the highest of the two emission figures from the two tunnels, i.e. 50.8 ± 14.2 µg Sb/v km, will be used as a worst case. As a lower estimate of antimony emitted due to traffic the emission figure for highway (PM10) will be used.

The total amount of vehicle-kilometres in 2001 was 103,577,000,000; this figure contains laden kilometres, reported by all EU member states except Greece, and does not include Own Account-transport performed by Portuguese hauliers (Smihily, 2003).

The highest estimate of antimony emitted due to road traffic is 5.26 t Sb/y, which is an overestimation since it was the highest of the two emission figures from the study by Sternbeck et al. (2002b) and not all kilometres driven in the EU was the high emitting urban

driving. The lowest estimate of antimony emitted due to road traffic is 0.19 t Sb/y, which is an underestimation since it only estimates the PM10-fraction and not all kilometres driven in the EU was the low emitting highway driving.

Thus, the amount of antimony released in the EU due to road traffic ranges between 0.19 to 5.3 t Sb/y. However, since not all kilometres driven in the EU are intense high emitting urban traffic, it is decided to use the lower of the two tunnel emission factors, i.e. 31.6 ± 11.6 µg Sb/v km, as a more reasonable worst case in the RAR. This results in an emission of 3.3 t Sb/y.

The emissions are likely to mainly be to soil. In the draft DEHP risk assessment it was assumed that 75% of the emissions would be to industrial/urban soil and 0.1% to air, with the remainder occurring to surface water (and hence sediment). In this scenario no emission is expected to air, why 75% is expected occur to industrial/urban soil and 25% to surface water.

The allocation of emissions to the regional scale has been performed with the 10% rule (see Table3-35)

Table3-35 Total annual amount of Sb emissions to industrial/urban soil and surface water due to road traffic within the EU.

Compartment	Released amount of antimony (t Sb/y)	Continental (90%; t Sb/y)	Regional (10%; t Sb/y)
Industrial/urban soil (75%)	2.5	2.25	0.25
Surface water (25%)	0.8	0.72	0.08

3.1.2.8.4 Hunting, shooting and sport fishing

The report by Sloof and co-workers (Sloof et al., 1992) identified the lead containing antimony, spread through hunting, shooting and sport fishing as one of the most important sources of antimony pollution in soil and water in the Netherlands. An early emission of 16,000 kg Sb and 4,000 kg Sb was estimated to soil and water in the Netherlands, respectively. Assuming that the total amount of antimony released to the EU via this route can be estimated based on the proportion of the EU population situated in the Netherlands (population in the Netherlands = 15,100,000 (Anonymous, 1994); population in the EU = 370,000,000 (EUSES 2, 2004), the total amount emitted to soil and water would be 392,000 kg Sb/y and 98,000 kg Sb/y, respectively. However, the Dutch estimate is uncertain since it is based on figures that are over 15 years old, and that the use of lead containing antimony may have change considerably since then. In addition, a difficulty arises when to approximate the actual emissions of antimony that should be entered in to EUSES. A more proper estimate of this source should for instance consider i) an accumulation has occurred over a number of years, ii) that the lead containing antimony spread to soil includes shooting-ranges as well as other types of soils, iii) bioavailability of the antimony varies with a number of factors such as time, compartment (type of soil or water) it is situated in, etc. As there presently are no suitable methods available for doing such an estimate, combined with the uncertainty associated with these estimates, the rapporteur considered not to use these emissions in the EUSES calculations.

However, in order to be able to get an approximation of the possible importance of this source, it is assumed that 100% of the antimony spread during one year leaches to the environment. This simplified scenario is considered to include the emissions from the accumulation of all previous years. In this scenario, the releases to water have been entered into natural waters in the EUSES-model. The releases to soil have been entered into agricultural soil even though this may not be the most relevant compartment for emissions of antimony from this source. A comparison of the regional PECs obtained with and without this contribution is performed in section 3.1.4 and in the risk characterization. The considerable uncertainty associated with this estimate is recognized.

3.1.2.8.5 Summary of emissions due to unintentional sources

The emissions estimated above are summarized in Table3-36 below.

Table3-36. Summary of emissions due to unintentional sources. Values within brackets represent unintentional sources including the emissions from "Hunting, shooting and sport fishing".

Compartment	Released amount of antimony (t/y)	Continental (t Sb/y)	Regional (t Sb/y)
Air	16.67 (16.67)	15 (15)	1.67 (1.67)
Industrial/urban soil	2.5 (2.5)	2.25 (2.25)	0.25 (0.25)
Surface water	0.8 (98.8)	0.72 (88.9)	0.08 (9.88)
Agricultural soil	(392)	(353)	(39)

3.1.2.9 Summary and conclusion of release estimates

A summary of the estimated release estimates is presented in Table3-37 (100% incineration assumed) and (100% landfilling assumed). Releases to industrial urban soil have been predicted using default EUSES values unless specified otherwise.

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Table3-37 Summary of release estimates to be used in EUSES. Emissions are given in Sb.

Scenario	Life cycle step	Regional release kg Sb/y	Continental release kg Sb/y
Production of diantimony trioxide	Production	1 270 to air 5 to surface water 877 to industrial urban soil	89 to air 1 to surface water 913 to industrial urban soil
Use as flame-retardant in plastics and rubber	Formulation	10 to air 422 to wastewater 169 to industrial urban soil	18 to air 3 799 to wastewater 1 520 to industrial urban soil
	Industrial use	0 to air 169 to wastewater 169 to industrial urban soil	0 to air 1 519 to wastewater 1 520 to industrial urban soil

Scenario	Life cycle step	Regional release kg Sb/y	Continental release kg Sb/y
	Service life	0 to air 169 to surface water 0 to industrial urban soil	0 to air 1 519 to surface water 0 to industrial urban soil
Use as flame-retardant in textiles	Formulation	6 to air 366 to wastewater 44 to industrial urban soil	14 to air 853 to wastewater 103 to industrial urban soil
	Industrial use (Application to textiles)	0 to air 4405 to wastewater 2 200 to industrial urban soil	0 to air 10 278 to wastewater 5 130 to industrial urban soil
	Service life (Washing and loss at disposal)	0 to air 2203 to wastewater 6 609 to industrial urban soil	0 to air 5 139 to wastewater 15 418 to industrial urban soil
Use as catalyst (PET industry)	Polymerisation of PET	2 to air 42 to wastewater 8 to industrial urban soil	3 to air 53 to wastewater 71 to industrial urban soil
	Production of articles of PET	0 to air 8 to wastewater 40 to industrial urban soil	0 to air 71 to wastewater 355 to industrial urban soil
	Service life	0 to air 3 to wastewater 0 to industrial urban soil	0 to air 31 to wastewater 0 to industrial urban soil
Use in paint	Formulation	8 kg/y to air 5 to wastewater 3 to industrial urban soil	70 kg/y to air 48 to wastewater 90 to industrial urban soil
	Industrial use	0 to air 33 to wastewater 0 to industrial urban soil	0 to air 301 to wastewater 0 to industrial urban soil
Use in glass	Formulation	19 to air 0 to water 7 to industrial urban soil	33 to air 0 to water 13 to industrial urban soil
	Industrial use	25 to air 7 to surface water 102 to industrial urban soil	226 to air 56 to surface water 916 to industrial urban soil
Incineration (100%)	Disposal	450 to air 500 to wastewater	4 050 to air 4 900 to wastewater
Landfill (100%)	Disposal	5 to surface water 38 to wastewater	42 to surface water 340 to wastewater

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Scenario	Life cycle step	Regional release kg Sb/y	Continental release kg Sb/y
Unintentional sources*		1 670 to air 250 to industrial urban soil 80 (9 880) to surface water 0 (39 000) to agricultural soil	15 000 to air 2 250 to industrial urban soil 720 (88 920) to surface water 0 (352 000) to agricultural soil
Total emission			
Incineration scenario: (100% incineration)*		3 460 (3 460) to air 2 463 (12 263) to surface water 5 954 (5 954) to wastewater 10478 (10 478) to industrial urban soil 0 (39 000) to agricultural soil	19 504 (19 504) to air 7 436 (95 616) to surface water 21 853 (21 853) to wastewater 28 300 (28 300) to industrial urban soil 0 (353 000) to agricultural soil
Landfill scenario: (100% landfill)*		3 010 (3 010) to air 2 468 (12 268) to surface water 5 492 (5 492) to wastewater 10478 (10 478) to industrial urban soil 0 (39 000) to agricultural soil	15 450 (15 450) to air 7 478 (95 658) to surface water 17 293 (17 293) to wastewater 28 300 (28 300) to industrial urban soil 0 (353 000) to agricultural soil

*Values within brackets also include the scenario for "Hunting, shooting and sport fishing".

As can be seen Table3-37 above, the two scenarios, 100% incineration or 100% landfill, results in similar emission levels to the different compartments, with the former having higher emissions for air and somewhat higher for waste-water. The regional PECs will be based on measured data (see section 3.1.4); hence this table is included as background information only.

An inclusion of the "Hunting, shooting, and sport fishing"-scenario would dominate the emissions to surface water and agricultural soil.

3.1.3 Environmental fate

Diantimony trioxide will dissolve and thereby generate antimony ions (e.g. Vangheluwe et al., 2001), and the environmental fate section will therefore discuss the fate of antimony in general.

The speciation and physico-chemical state of antimony are important for its behaviour in the environment and availability to biota. For example, antimony incorporated in mineral lattices is inert and unlikely to be bioavailable. Most analytical methods for antimony do not distinguish between the various forms of antimony. While the total amount of antimony may

be known, the nature of the antimony compounds, the importance of adsorption, etc. are not. This information, which is critical in determining the availability of antimony, is apt to be site-specific (ATSDR, 1992).

There are uncertainties surrounding the thermodynamic data for Sb compounds, and as a consequence, the Eh-pH diagrams differ between different sources. Earlier diagrams suggest that antimony is immobile under oxidizing conditions, occurring as solid oxides (e.g. (Brookins, 1988), but more recent diagrams show that in oxidizing conditions, $Sb(OH)_6^-$ is the most important species, confirming the relatively high mobility of Sb under oxidizing conditions (e.g. Filella et al., 2002a; Filella et al., 2002b).

3.1.3.1 Transformations in the environment

Antimony, being a natural element, cannot by definition be degraded. However, it can be transformed between different binding/speciation forms and oxidation states. Therefore, the headings including degradation normally found in RARs from Existing substances have been changed to transformation.

3.1.3.1.1 Atmospheric transformation

Conclusions that can be drawn from this section:

i) anthropogenic activities may result in long-range transport of diantimony trioxide far from its source

ii) combustion/incineration processes transform antimony compounds to diantimony trioxide regardless of the pre-incinerated form of antimony

iii) there are indications that diantimony trioxide may dissolve in the atmosphere and that the trivalent form will oxidise to the pentavalent form

Most of the antimony that is released to the atmosphere from anthropogenic sources results from metal smelting and refinement, combustion of coal, refuse and sludge incineration, and road traffic.

When released into the atmosphere as an aerosol, antimony is believed to be oxidised to diantimony trioxide by reactions with atmospheric oxidants (ATSDR, 1992). According to Ainsworth (1988), diantimony trioxide particles do not undergo changes in chemical composition, particle size, or morphology after emission, even though a surface coating of sulphate may form. A study of the wet and dry deposition over an 8-week period on an island in the German Bight, which was presumably far from sources, found 87% of the deposited antimony dissolved in rain, 11% in particulate matter in rain, and only 2% as dry deposition (Stössel and Michaelis, 1986). Thermodynamic calculations show that dissolved antimony predominately should exist as Sb(V). Antimony found in rain and snow has been reported to predominantly exist in the +5 oxidation state (Metzger and Braun, 1986). The above indicate that at least some of the trivalent antimony, released into the atmosphere in the form of diantimony trioxide, through dissolution followed by oxidative processes may transform to pentavalent antimony. Dissolved concentrations of antimony have been measured in both rain

and snow (see Table3-62). Concentrations measured ranged from tens of ng Sb/l to tens of µg Sb/l, depending on the location of the measurements made.

Monitoring data indicate that antimony can be transported far from its source (Dutkiewicz et al., 1987; Hvatum et al., 1983; Hillamo et al., 1988; Naturvårdsverket, 1999c). The result of long-range transport of antimony has been shown in Norwegian bogs and sediments, where the concentrations decreased from south to north (Steinnes et al., 2001). The same pattern has also been observed in a Swedish study on metals in the moss *Pleurozium schreberi* and moor layer (Naturvårdsverket, 1999c). A model that relates particle size to volatility estimates average atmospheric half-lives of 4.6 days for diantimony trioxide (Müller, 1985).

Another example of potential of antimony for long-range transport was presented in the article by Landsberger and co-workers (1992). They found that the concentration of antimony increases about five-fold during the winter period in the arctic atmosphere. This increase has been attributed to increased stability of the arctic air mass during the winter, its extension to regions with industrial activities in Eurasia and Western Europe, and a scarcity of precipitation.

Average concentrations of 0.89 µg Sb/l in rainwater have been reported from coastal regions in the Netherlands (Stuyfzand, 1991). According to Stuyzand (Stuyfzand, 1992) rainwater (contaminated with antimony) probably is the most important antimony source for shallow groundwater.

3.1.3.1.2 Aquatic transformation

Conclusions that can be drawn from this section regarding the fate of antimony in water:

i) in natural waters antimony exists almost exclusively in the dissolved phase in the two valency states +3 and +5. Both Sb(III) and Sb(V) ions hydrolyse easily, and Sb(III) is present as the neutral species $Sb(OH)_3$, and Sb(V) as the anion, $Sb(OH)_6^-$. According to thermodynamic calculations, antimony should almost exclusively be present as Sb(V) in oxic systems, and as Sb(III) in anoxic systems. Even though the dominant species in oxic waters is Sb(V), Sb(III) has been detected in concentrations much above what is predicted, and the reverse is true for Sb(V) in anoxic systems. The presence of these thermodynamically unfavourable species (i.e. Sb(III) in oxic waters, and Sb(V) in anoxic) requires a mechanisms for the production and slow rates of conversion (i.e. kinetic stabilization), which however not yet are fully understood,

ii) reports exist on both conservative (i.e. the concentration only changes with dilution or evaporation) behaviour, or a behaviour corresponding to mildly scavenged element with surface (atmospheric) input,

iii) in addition to the inorganic forms of antimony, there also exist methylated forms of trivalent and pentavalent antimony,

iv) interactions between the antimony species (anionic $Sb(OH)_6^-$ or the neutral $Sb(OH)_3$) present in natural waters and the predominately negatively charged

natural organic matter may occur, but any firm conclusion on its importance is presently hard to make,

v) solubility of the diantimony trioxide is dependent on the conditions (Eh/pH), and the time factor. Studies on deposited antimony particles (most probably antimony oxides) in seawater indicated order of days to obtain complete dissolution.

Conclusions that can be drawn regarding the fate of antimony in sediment:

i) the adsorption of antimony in oxic sediments has been correlated with the presence of iron-, manganese-, and aluminium oxides,

ii) the decrease in bioavailable antimony in water by oxic sediments is not a permanent decrease, as the adsorption on the hydrous oxides is dependent on both pH and oxic condition (which may change). In addition, antimony may become bioavailable to organisms inhabiting the sediment through ingestion of the sediment,

iii) in anoxic systems, and in the presence of sulphur, depending on pH, antimony forms soluble or insoluble stibnite, SbS_2^- and $Sb_2S_3(s)$, respectively. This may result in a larger decrease in bioavailable antimony, as compared to the oxic part of the sediment.

Water

As a natural constituent of soil, antimony will be transported into streams and waterways due to weathering and run-off from soils. Much of this Sb is associated with particulate matter, and it is transported to and settles out in areas of active sedimentation, such as where a river empties into a lake or a bay (Beijer and Jernelöv, 1986). Antimony will also enter the aquatic compartment via effluents from mining and manufacturing, municipal and industrial discharges, atmospheric deposition, sewage sludge, and activities such as hunting, shooting and sport fishing (in the latter three cases as a constituent in lead alloys) (Slooff et al., 1992).

In an extensive review series by Filella and co-workers (Filella et al., 2002a; Filella et al., 2002b) that focused on antimony in natural waters, updated information is given on antimony occurrence and speciation. At environmental concentration levels, at the pH range commonly found in natural waters and in the absence of sulphur, antimony exists in the dissolved phase regardless of its oxidation state (Filella et al., 2002b). Both Sb(III) and Sb(V) ions hydrolyse easily, and antimony(III) is present as the neutral species $Sb(OH)_3$ (also quoted as $HSbO_2$), and Sb(V) as the anion, $Sb(OH)_6^-$ (also quoted as SbO_3^-). In addition to the inorganic forms of antimony, there also exist methylated forms of antimony (e.g. the non-volatile methylstibonic acid $CH_3SbO(OH)_2$).

The two forms of antimony in which antimony is considered to exist in natural waters, i.e. $Sb(OH)_3$ and $Sb(OH)_6^-$, have low charge density and the behaviour in open oceans is not considered highly reactive. Reports exist on both conservative (i.e. the concentration only changes with dilution or evaporation) behaviour, or a behaviour corresponding to mildly scavenged element with surface (atmospheric) input (Filella et al., 2002a). Comparison of

antimony contents in the deep waters of different oceans indicates that no antimony accumulation is produced during deep-water oceanic circulation (Cutter and Cutter, 1998).

Similarly as for soil solutions (see section 3.1.3.1.3), precipitation of $Ca[Sb(OH)_6]_2$ may limit the concentration of Sb in the higher concentration range (mg/l), since the concentration of Ca in natural waters generally is in the range of a few up to 100 mg/l (FOREGS, http://www.gtk.fi/publ/foregsatlas/article.php?id=15: range 0.23-592 mg/l median: 40 mg/l, mean: 55 mg/l, 90P: 119 mg/l).

Thermodynamic

According to thermodynamic data, antimony should be present as Sb(V) in oxic systems, and as Sb(III) in anoxic systems. However, Sb(III) has been detected in oxic waters (Bertine and Lee, 1983; Kantin, 1983; Andreae and Froelich, 1984; Takayanagi and Cossa, 1997; Cutter et al., 2001), and Sb(V) has been detected in anoxic waters (Bertine and Lee, 1983; Andreae and Froelich, 1984; Helz et al., 2002). For thermodynamically unfavourable species to exist in water, not only mechanisms for their production must exist (see below), but also mechanisms for their slow rates of conversion (i.e. kinetic stabilization).

Although, according to Latimer (1952), the Sb(III)/Sb(V) couple is capable of attaining rapid equilibrium in both acid and basic solutions; when redox process is accompanied by hydrolysis reactions, the overall process can be much slower (Brown and Swift, 1949; Bonner, 1949; Neumann and Brown, 1956; Bonner and Goishi, 1961; Cheek et al., 1961). No systematic studies exist on Sb(III) oxidation kinetics under natural water conditions. Published observations seem to indicate that, at natural antimony levels, Sb(III) is gently oxidised to Sb(V): (Sun et al., 1982) studied Sb(III) and Sb(V) stability in lake and wastewaters by adding known amounts of both species (2 μg/l). Sb(III) in lake water appeared to be unstable, since none could be detected after standing for 6 h, when no stabilizing agent was added and that the oxidation process was prevented for a period of six days when tartaric acid was added to a final concentration of 1% (w/v) The authors considered that the oxidation of Sb(III) to Sb(V) was due to the effect of other substances present in the lake water. Cutter (1992) estimated the Sb(III) oxidation rate by using the depth profiles in the upper 100 m of the Black sea. He calculated a pseudo first order rate constant of 0.008 day^{-1} (residence time = 125 days). This rate includes all form of removal since Sb(III) may also be scavenged by particles. The same author (Cutter et al., 2001) considered that Sb(III) probably has longer residence times in seawater because this value was calculated at the suboxic-oxic interface of the Black Sea where the presence of manganese and iron oxides is likely to increase the oxidation rate. However, the levels of Sb(III) detected in oxic waters also require the existence of a continuous source (biota, atmospheric Sb_2O_3 deposition, Sb(V) reduction). The rate of conversion of Sb(III) into Sb(V) has been observed to decrease with increasing acidity of the water samples (Sun et al., 1982; Sherman et al., 2000). This effect may be related to complex hydrolytic processes taking place simultaneously with the redox reaction.

An Eh-pH diagram (Figure 3-1) can be used to illustrate the redox behaviour of elements like antimony. This diagram (taken from Filella et al., 2002b) is constructed using environmentally realistic concentrations of antimony and dissolved sulphur. From this diagram it can be seen that antimony is present as soluble $Sb(OH)_6^-$ in oxic systems, and as soluble $Sb(OH)_3$ in anoxic systems at natural pH values. During reducing conditions, and in the presence of sulphur, stibnite, $SbS_3(s)$, will be formed during low to intermediate pH

values. At higher pH values, the soluble SbS_2^- species will replace stibnite (Filella et al., 2002b).

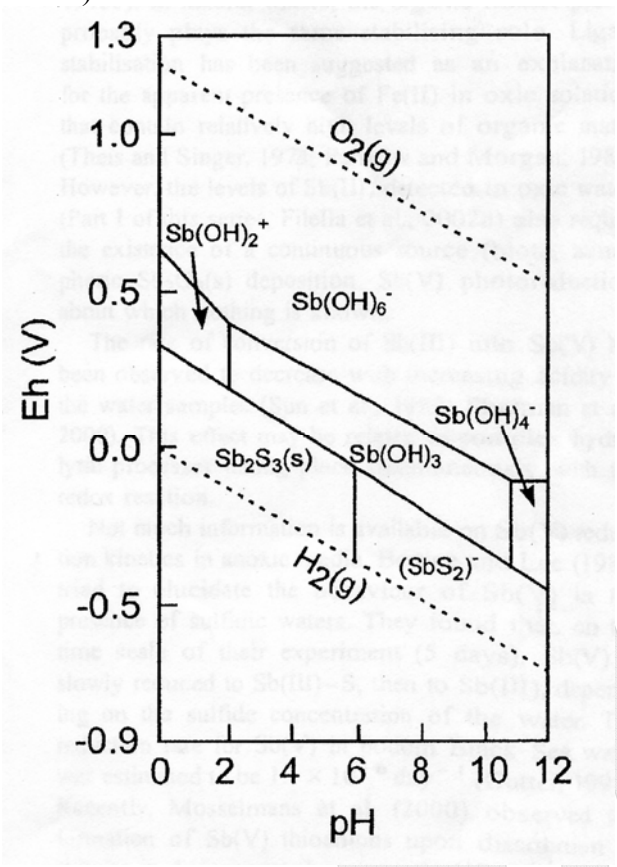

Figure 3-1 Eh-pH diagram of antimony in the Sb-S-H_2O system at a dissolved antimony concentration of 10^{-8} mol/l and a dissolved sulphur concentration of 10^{-3} mol/l. From Filella et al. (2002b).

Presence of Sb(III) in oxic waters

The presence of Sb(III) in oxic surface waters has been attributed to phytoplanktonic activity. However, relatively little is known about algae uptake and detoxifying mechanisms, even though they may be important due to their possible roll in antimony redox speciation. An alternative to the phytoplanktonic source has been suggested to be a photochemical reduction of Sb(V) to Sb(III).

Bertine and Lee (1983) detected a few percent of Sb(III) of the total Sb in the water column of Saanich Inlet, Canada. It was suggested that this was the result of a production of trivalent antimony in the photic zone.

Kantin (1983) collected marine algae (*Ulva sp*., *Enteromorpha sp*., and *Saragassum sp*.) near San Diego Bay, USA, and found that Sb(V) was the dominant specie in the algae. However, the *Saragassum sp.* contained up to 30% of Sb(III), thus demonstrating the possibility to form reduced antimony species.

Andreae and Froelich (1984) observed a surface maximum of Sb(III) in the Baltic Sea, and attributed it to the presence of biological activity in this layer. They also analysed the

chemical speciation of antimony in phytoplankton (largely diatoms) from the eastern North Pacific. The trivalent species were found to make up an important fraction of the inorganic antimony in phytoplankton.

Benson (1998) incubated the diatom *Thalassiosira nana* with $^{125}SbCl_3$, which resulted in protein-bound antimony, a stibnolipid, and a group of water-soluble radioactive products. The algal metabolism of antimony species appears to parallel that of arsenate, where the reduction, methylation, and adenosylation of antimony serve to mediate its excretion.

(Maeda et al., 1997) cultured freshwater algae *Chlorella vulgaris*, which had been isolated from an arsenic-polluted environment, in medium containing antimony (derived from trivalent antimony potassium tartrate) for a period of six days. After exposure, the algae were transferred to antimony-free medium, where the excreted antimony consisted of 60% Sb(III) and 40% Sb(V). The antimony accumulated within the algae was combined with proteins of a molecular weight of 40 kDa. According to the authors the relative proportions of Sb(III) and Sb(V) excreted indicated that the algae were able to convert Sb(III) to the less toxic form Sb(V), since the Sb(III) had been the original source of antimony. An additional interpretation of these results could be that some of the trivalent antimony in the medium had oxidized to pentavalent antimony which had been taken up and either remained in a pentavalent form and/or been reduced to trivalent antimony. Reduction of pentavalent antimony has previously been described by Kantin (1983), (see above).

Takayanagi and Cossa (1997) studied the vertical profiles of antimony in Pavin Lake, France. A slight depletion of Sb(V) was observed and an enrichment of Sb(III) in the surface water, which was suggested to be due to biological activity. Sb(III) and Sb(V) coexisted at similar concentrations in the surface water, while at depths Sb(V) was the only detectable species, where only the latter is in accordance with thermodynamic predictions.

(Cutter et al., 2001) found no correlation between the Sb(III) detected in surface water and biotic tracers, such as nutrients and chlorophyll *a*, in a study covering subtropical to equatorial Atlantic ocean. As an alternative to the phytoplanktonic source of Sb(III), the authors therefore suggested photochemical reduction of Sb(V) to Sb(III).

Cutter (1992) calculated a pseudo first order rate constant of 0.008 day^{-1}, for Sb(III) removal in deep oxic waters of the Black Sea, which equals a residence time of 125 days (1/k), This rate constant includes all forms of removal (i.e. not only oxidation) since Sb(III) may also be scavenged by adsorption to particles. Using amorphous forms of natural and synthetic Fe oxyhydroxides and synthetic Mn oxyhydroxides, (Belzile et al., 2001) calculated pseudo first order rate constants of 0.57, 0.89, and 2.35 day^{-1}, respectively. These estimates are two orders of magnitude higher than the value presented by (Cutter 1992) for deep waters in the Black Sea. This difference emphasizes the major role played by the Fe and Mn oxyhydroxides in the oxidation of Sb(III).

Presence of Sb(V) in anoxic waters

The true speciation of antimony in anoxic waters is unclear. According to thermodynamic calculations antimony should exclusively be present as Sb(III) in absence of oxygen. However, pentavalent forms of antimony have been reported from different anoxic systems.

Bertine and Lee (1983) found no major reduction of Sb(V) to either Sb(III) or the Sb(III)-sulphur complex in anoxic waters of the Saanich Inlet. It was assumed that Sb(V) was present as thioantimonates, since the analytical method used did not differentiate between antimonate and thioantimonate species. An additional experiment, performed by the authors, sulphide was added to Sb(V) spiked seawater, and the results supported the assumption of Sb(V) being present as thioantimonate.

Andreae and Froelich (1984) observed only 44% Sb(III) of total amount of inorganic antimony in anoxic waters of the Baltic Sea. Only in the deepest sample did this percentage of Sb(III) increase to 93%, mostly as a result of a decrease in the concentration of total inorganic antimony rather than an increase of Sb(III). The authors suggested that the Sb(V) may be present as thioantimonate, which forms in the presence of antimonate and sulphide ions (Cotton and Wilkinson, 1972). Another mechanism suggested was the delivery of Sb(V) on detritus sinking from oxic waters.

Cutter (1991) reported the occurrence of Sb(V) in anoxic waters of the Black Sea, and suggested different mechanisms to explain the results. Sinking detritus from the oxic zone containing Sb(V), and advection of surface water with high concentrations of Sb(V) was the proposed mechanisms. The study also resulted in a pseudo first order rate constant (0.0004 y^{-1}) for the reduction of Sb(V) in the Black Sea. This rate includes all forms of removal since Sb(V) may also be scavenged by adsorption to particles.

Cutter (1992) presented results showing decrease with depths of concentrations of Sb(V) in surficial sediments underlying the anoxic water column, and decrease with depths of the fraction of sedimentary Sb(III + V) that could be recovered (using the 2 M HCl leach technique given by Andreae and Froelich (1984) of the total amount of Sb, which suggests that antimony is being incorporated into a more refractory phase such as pyrite (Huerta-Diaz and Morse, 1990).

In conclusion, the presence of Sb(V) in anoxic waters has been attributed to i) sinking detritus containing Sb(V), ii) advection from surface water, and iii) formation of thioantimonate species. All of these mechanisms must be coupled with relatively slow rates of reduction, as suggested by Peterson and Carpenter (1983) for arsenate.

Methylated antimony species

In addition to the inorganic forms of antimony in the aquatic environment, there also exist methylated antimony species, which account for about 10% or less of the total dissolved antimony and are present throughout the water column, with a tendency of higher values in the surface layer. More information on biological methylation of antimony can be found in the section “Biomethylation”.

Natural organic material

There is some experimental evidence for complexation of Sb with natural organic material (NOM), but results do not allow any firm conclusions to be drawn regarding its role in antimony fate in natural aquatic systems.

The dominant form of antimony in oxic environments is the anionic $Sb(OH)_6^-$ which probably is less likely to interact with the organic material due to the predominantly negative charge on the NOM at the pH of natural waters, as compared to the neutral species $Sb(OH)_3$. It is therefore probable that the level of interaction with NOM may differ between the two antimony species.

Addition of organic substances such as tartaric acid (Sun et al., 1982), lactic, citric, ascorbic acids (de la Calle Guntinas et al., 1992) to natural or synthetic solutions has been shown to have a stabilising effect on Sb(III). Solutions of Sb(III) prepared from potassium antimony tartrate have been shown to be stable for long periods (Al-Sibaai and Fogg, 1973; Andreae et al., 1981; de la Calle Guntinas et al., 1992). Thus, in natural waters, NOM may play the same stabilising role for the thermodynamically unfavourable trivalent antimony specie (Filella et al., 2002b). However, results presented by Buschmann and Sigg (2004) indicate that humic acids catalyses Sb(III) oxidation to Sb(V), which is assumed to be released as $Sb(OH)_6^-$.

There are a few studies that provide experimental evidence for complexation of Sb with NOM in natural waters. In a study by Gillian and Brihaye (1985) involving voltametric measurements, a significant fraction (20-60%) of Sb (mostly present as Sb(V)) released after UV-irradiation of seawater taken from the Belgian coast was attributed to NOM complexes. By using XAD resins, a stable association between aquatic humics and Sb was identified in the bottom waters of Lake Pavin in France (Albéric et al., 2000). (Deng et al., 2001) identified a fraction of Sb associated with NOM, ranging from 35 to 67% of total Sb, in three Canadian lakes receiving atmospheric loading from the Sudbury smelters. No information was given on the oxidation state, but since the measurements were made in surficial oxic waters, Sb(V) was presumably dominant. The same analytical procedure, as in (Deng et al., 2001), was also used in a study by (Chen et al., 2003) in two additional lakes from the same district. It was found that 35-85% (average 61%) of the Sb in porewaters of one of the lakes, and 35-55% (average 50%) in those of the other lake was present in a refractory fraction, which could partially or entirely be made of NOM. Buschmann and Sigg (2004) determined conditional distribution coefficients (D_{om}) for Sb(III) binding to three commercial humic acids that differed in carbon content and number of functional groups (terrestrial, coal, and aquatic) at environmentally relevant Sb(III)/DOC ratios and as a function of pH using an equilibrium dialysis method. Maximum binding of Sb(III) was observed around pH 6 for two of the humic acids, with the third showing constant D_{om} values up to pH 6 and decreasing values above pH 6. The D_{om} was observed to be inversely related with the Sb(III)/DOC ratio since a 60 times lower Sb(III)/DOC ratio resulted in a 20 times higher D_{om}. In addition, the D_{om} depends on the individual humic acids, indicating that different functional groups are involved and/or different degrees of stabilisation by chelation or H-bridges. Chemical modelling of the Sb(III)-humic binding at different pH values is consistent with two binding sites involving i) a phenolic entity forming a neutral complex and ii) a carboxylic entity forming a negatively charged complex. The authors proposed that over 30% of the total Sb(III) content may be bound to NOM under environmentally relevant conditions.

The fact that the presence of refractory or NOM-bound Sb often not is included in the literature could be due to the analytical protocol used. Most protocols are usually designed to measure Sb(III) and total Sb separately, then the difference between the two is considered to be Sb(V), but they do not include any treatment for destruction of NOM.

Solubility of antimony containing aerosol particles in marine waters

Crecelius (1980) determined antimony solubility from marine aerosol particles and found that 40% had dissolved after 24 h. The samples were neutron-activated, added to filtered seawater and after 24 h of gentle and continuous mixing at temperatures of 19-20 ºC, aliquots were removed, filtered and gamma counted on a Ge(Li) diode.

Austin and Millward (1985) reported in seawater 27% solubility of antimony, contained in urban particles after a leaching period of 2 h with stirring in room temperature, followed by filtration and analysis using AAS with hydride generation and a heated quartz tube.

(Kersten et al., 1991) performed a 100 h leaching study of antimony present in polluted North Sea coastal aerosol, which resulted in a solubility of 83%, using filtered and purified seawater, stirring at temperatures of 20-25ºC, filtered aliquots analysed using graphite furnace atomic absorption spectrophotometry.
These data indicate that the antimony in these aerosol particles (probably present as antimony oxides) was dissolved in a relatively short period of time (order of days to obtain complete dissolution).

Sediment

Oxic sediments

Several studies have correlated the adsorption of antimony in soil and oxic sediment with the presence of hydrous iron-, manganese-, and aluminium oxides (Crecelius, 1975; Brannon and Patrick, 1985; Mok and Wai, 1990; Thanabalasingam and Pickering, 1990). The hydrous oxides posses pH-dependent surface charge, with a zpc (zero point of charge) that is the point where the net charge is zero. At pH-values below the zpc, the net charge is positive, and above it is negative. The oxides of iron and aluminium are positively charged at the pH values commonly encountered in the environment, and adsorb anionic compounds, such as pentavalent antimony. The manganese oxides, which in the study by Thanabalasingam and Pickering (1990), also were shown to adsorb antimony have a charge that depends on the form of manganese oxide, with zpc ranging from about 2 to 7. The sorption on the manganese oxides was in fact higher than that on the aluminium and iron oxides. The effect of pH on adsorbed antimony pointed to a number of competing influences, where the effect probably is more related to the chemical form of the adsorbate species than on the net surface charge. The Sb(III) derived from the Sb_2O_3 dissolution will in an oxidizing environment, unless there are kinetic limitations, oxidise to Sb(V) and thereby result in the anionic $Sb(OH)_6^-$ with affinity for positively charged surfaces.

(Belzile et al., 2001) showed that Sb(III) strongly adsorbed to amorphous forms of iron and manganese, and that the trivalent antimony within a few days effectively was oxidised to pentavalent antimony. A potential mechanism suggested, as regards the iron oxyhydroxides, involved (i) adsorption of $Sb(OH)_3$ and the formation of a surface complex with the Fe(III) oxyhydroxide (one Sb(III) on two Fe(III) sites), (ii) the transfer of two electrons from antimony to two iron atoms, (iii) the release of oxidised antimony (one Sb(V) for two Fe(II), and (iv) the release of the reduced Fe (II). A similar mechanism was proposed for the manganese oxyhydroxides. The oxidizing capacity of the manganese oxides was about twice

as efficient, measured as time for complete oxidation, when compared to the iron oxides. A slightly lower oxidation rate was observed at lower pH (< 6 - 6.5) as compared to the oxidation rates observed at pH 7.2 and 9.0 (natural Fe oxyhydroxides), pH 7.1 – 10.2 (synthetic Fe oxyhydroxides) and pH 7.5 and 8.6 (synthetic Mn oxyhydroxides). This difference was attributed to the lower stability of the oxyhydroxide compounds under mildly acidic conditions.

In a recent study by (Chen et al., 2003) different chemical forms of antimony were measured in porewaters and sediment of two lakes in Sudbury (Canada) characterized by contrasting redox conditions at the sediment-water interface. One of the two lakes was a clearwater lake, with a pH of approximately 6.3, and was well oxygenated all year, while the other lake was a well buffered, slightly alkaline lake at pH 7.5, which comes close to anoxia during the summer. In porewaters, Sb(III) was present under reducing conditions where it existed as SbS_2^-, which agrees with thermodynamic calculations. Sb(V) was detected mainly under oxic conditions. However, it was also found under reducing conditions (in the presence of sulphides), where its presence was attributed to the oxidizing effect of Fe and Mn oxyhydroxides or the slow kinetics of reduction of Sb(V). An additional third form of Sb identified as refractory was obtained after UV irradiation of the water samples, suggesting an association of Sb to low molecular weight organic matter. The distribution of Sb in sediments of the two lakes revealed (through the comparison of profiles and statistical correlations) the importance of Fe and Mn oxyhydroxides in controlling the behaviour of Sb, particularly in the lake where the interface was clearly oxic. Porewater profiles indicate that the dissolution of Fe and Mn oxyhydroxides under anoxic conditions leads to the simultaneous release of dissolved Sb previously sorbed onto these compounds.

Anoxic sediments

In reducing sediments, the control of the solubility of Sb by iron sulphides is suggested. The antimony that is released when the hydrous oxides dissolve will in the presence of sulphide, depending on pH, form soluble or insoluble stibnite, SbS_2^- and $Sb_2S_3(s)$, respectively (Filella et al., 2002b). Formation of Sb(V) thioanions have been observed upon dissolution of stibnite in deoxygenated NaHS solutions (Mosselmans et al., 2000). The authors proposed oxidation of Sb(III) to Sb(V) to explain the results. As support of this theory they cited the fact that Cutter (1991) detected more Sb(V) in Black Marine waters where higher HS^- prevailed, as compared to the Sb(III) predominance at the top of the sulfidic zone.

Effect of pH

Leaching experiments on sediments by Mok and Wai (1990) showed an increasing release of antimony from the sediments with decreasing pH, but also a sharply increased release at high pH. Therefore a near-neutral pH condition was suggested to favour long-term stability of mine wastes with respect to antimony. However, their data show the lowest mobility of antimony at pH 4.3 (in total 16.7 ng Sb/kg sediment released at pH 4.3 during leaching under aerobic conditions, compared to ~100 ng/kg at pH 6.3), which is in agreement with data for soils (see below, strongest sorption at pH about 4). The authors also found that the form of antimony released at the leaching experiments was dependent on the pH used. At pH 2.7, the bulk of the antimony released was trivalent, at pH 4.3 the concentrations of tri- and pentavalent antimony were comparable, and at pH 6.3 and above, the pentavalent form was the predominant species. However, it is rare that the pH in sediments in the environment is below 5.5-6, except when sulphide rich sediments are oxidised (Parkmann, 2004).

Uptake of metals through ingestion of sediments

Experiments with four types of benthic invertebrates (*Potamocorbula amurensis*, *Macoma balthica*, *Neanthes arenaceodentata*, and *Heteromastus filiformis*) performed by Lee and co-worker (2000) showed that feeding behaviour and dietary uptake controlled bioaccumulation of the metals cadmium, silver, nickel, and zinc. Metal concentration in animal tissue correlated with metal concentrations extracted from sediments, but not with metal in porewater, across a range of reactive sulphide concentrations, from 0.5 to 30 µmol/g. Uptake of antimony through ingestion of antimony containing sediments is therefore also considered a possible route of exposure.

3.1.3.1.3 Transformation in soil

In general, the knowledge on weathering reactions, mobility and adsorptive behaviour of antimony, its compounds and ions is relatively limited.

However, the following conclusions can be drawn from the literature regarding the fate of antimony in soil:

i) the sorption and precipitation of $Ca[Sb(OH)_6]_2$ seem to be more important than the dissolution processes of Sb_2O_3 as regards the fate of antimony.

ii) the solubility of antimony compounds depends on the soil conditions (Eh/pH) and the time given to dissolve.

iii) the most important soil characteristic as regards the mobility of antimony in soil (and sediments), appears to be the presence of hydrous oxides of iron, manganese, and aluminium, to which antimony may adsorb. In addition, these hydrous oxides seem to oxidise dissolved trivalent antimonite ($Sb(OH)_3$) to the pentavalent antimonate ($Sb(OH)_6^-$).

iv) the largest effect of pH on sorption seems to be around 3 - 4, with decreasing sorption at higher pH-values. The effect of pH as such is probably less important as compared to the effect of the hydrous oxides. The effect of pH on antimony mobility seems to be via the hydrous oxides, via the influence on valence of antimony and the solubility of antimony compound, and via the increasing negative charge of the soil at increasing pH (and hence, weaker sorption of the negatively charged $Sb(OH)_6^-$).

v) due to the anionic character of the dissolved species ($Sb(OH)_6^-$), antimony is expected to have a low affinity for organic carbon. However, there exist results that indicate that the sorption of Sb(V) by humic acid in acid soils with high proportions of organic matter may be more important than previously suspected, although the strong Sb(V) scavenging potential of $Fe(OH)_3$ probably results in diminished role of organic matter binding in soils with high amounts of non-crystalline hydroxides.

vi) the cationic exchange reactions, which are the main sorption reactions on clay minerals, are not expected to be important for the anionic antimony.

vii) initial differences on sorption depending on type of antimony compound used diminish with time.

viii) the influence of the concentration of added antimony on sorption appears to be small

ix) a higher Sb porewater concentration can be achieved in transformation studies when using Sb_2O_3, as compared to when using $SbCl_3$. The limiting factor appears to be precipitation of $Ca[Sb(OH)_6]_2$.

Mobility

The mobility of antimony in soils depends on the form of antimony, the nature of the soil, and the environmental conditions in the soil.

Most studies (Ragaini et al., 1977; King, 1988; Trnovsky et al., 1988; Springborn Life Sciences, 1988; Ainsworth et al., 1990a) indicate that antimony has a low mobility in soil, but there are also studies indicating the opposite (Gerritse et al., 1982; Coughtrey et al., 1983). These differences may depend on the experimental set-up used (which form of antimony, etc.).

When studied in areas not affected by atmospheric deposition of Sb_2O_3, antimony has a moderate mobility in soil and probably occurs as antimonate ($Sb(OH)_6^-$) in soluble form (Kabata-Pendias and Pendias, 1992).

Springborn (1986) investigated the mobility of antimony in four different soils (clay, sandy loam, silt loam, and sand soils) using thin-layer chromatography. It was concluded that there was no systematic or even significant movement of antimony in any of the soil types examined. However, the concentration used was above solubility, there was no measurement of the dissolved antimony concentration applied on the plates, and small amounts of antimony were detected in all zones.

(Trnovsky et al., 1988) found, at a battery reclamation plant, that despite high levels of antimony in soil and sediments, the antimony concentration in an aquifer 3-9 m below the surface was 0.1 mg Sb/l (a high concentration, only found in heavily polluted areas), and no antimony was detected in two deeper aquifers (9-15 m below, and 33-46 m below, respectively). The other contaminants (Al, As, Cd, Mn, Ni, Pb, and sulphate) measured in this study could all be detected, not only in the most surficial aquifer, but also in the two deeper aquifers.

(Both Ragaini et al., 1977), who studied soil cores near a Pb-smelting complex, and (Ainsworth et al., 1990), who studied soil cores near an Sb-smelter, described antimony accumulation in the surface layer, and decreasing concentrations with increasing depths. In both studies the soil was aerially contaminated with Sb_2O_3.

(Hammel et al., 2000) studied the mobility of antimony in agricultural soils, where residual material, contaminated due to historical mining activities, had been filled. The concentration

of a mobile antimony fraction, extracted with NH_4NO_3, ranged between 0.02-0.29 mg Sb/kg dw, corresponding to a mobility of 0.06-0.59%. Studies performed in other soils (loamy sand or silt loam) experimentally contaminated with KSbO-tartrate, Sb_2S_3, and Sb_2S_5, after six months of aging, resulted in significantly higher mobility, as compared to the soils from mining areas. The highest mobility, 7.8-8.9%, was observed when antimony was applied as KSbO-tartrate. Addition of Sb_2S_3, and Sb_2S_5, resulted in mobilities of 2.9-6.8% and 1.5-3.9%, respectively.

Prüeß (1994) reported that the mobility of antimony increased with decreasing acidity of the soils. The mobility was calculated as the ratio between the amount of Sb extracted with NH_4NO_3 and the amount extracted with *aqua regia* for 333 topsoil samples from SW Germany. The importance of pH on antimony mobility in these soils was however low (r^2 = 12%, $p<0.0001$).

Ainsworth and co-workers (1991) compared the distribution of antimony in two soils sampled from a grassland nearby an Sb smelter (100 and 250 m away) located on Tyneside in north-east England, with the distribution of antimony in a control soil sampled from a rural grassland in Northumberland (Ainsworth et al., 1991; Ainsworth et al., 1990a). Even though the soils used were all taken from grasslands, no further information on similarities or dissimilarities in soil characteristica were presented, which needs to be kept in mind when comparing the results from the different soils. The concentration of antimony was highest at the surface (0-5 cm), and decreased with depth (5-10 cm, 10-15 cm). The concentrations measured in the control soil were lower than the corresponding depths at the smelter site. The authors also performed a sequential fractionation procedure to determine the form of antimony. The totally extracted amount (mg Sb/kg) and the percentage of antimony in the most mobile fractions (see footnote in Table3-38 for more info) from the different sites and depths are presented in Table3-38.

Table3-38 Totally extracted concentrations of Sb in different soil samples (performed on 2 g samples of dried soil).

Soil	Concentration (mg Sb/kg)	%. in most mobile fractions*
100 m from Sb smelter < 5 cm 15-20 cm	 288 178	 13 21
250 m from Sb smelter < 5 cm 15-20 cm	 163 77	 12 21
Control	8	69
Control + Sb_2O_3 added to increase the the concentration of Sb by approximately 200 mg/kg	212	8

*Extractants used and type of fraction within parenthesis: ammonium acetate pH 7 (soluble or on ion exchange sites), sodium acetate pH 5 (bound to carbonates) and 0.1 M hydroxylammoniumchloride /0.01 M HNO_3 (possibly bound to manganese oxides).

When Sb_2O_3 was added to the control soil, the extraction pattern resembled that of the contaminated site, with only proportionally small amount extracted in the more easily mobilized fractions (as compared to the residual fraction). A percentual comparison between the different soil samples showed a much higher mobility of antimony in the control soil as compared to the contaminated soils. But when also considering the total amount of antimony

(see Table3-38) it becomes clear that the contaminated soils contain more mobile antimony *per se*, as compared to the control soil. The concentration of antimony in the most mobile fractions in the most contaminated soil (100 m from Sb smelter) was similar at the two depths, 37 mg Sb/kg (< 5 cm) and 37 mg Sb/kg (15-20 cm), respectively, as compared to 5.5 mg Sb/kg in the control soil.

The levels and chemical form of antimony in the soil around a smelter was determined by Takaoka and colleagues (Takaoka et al., 2005), using X-ray absorption fine structure (XAFS) spectra. In this facility antimony metals and oxides were smelted from antimony ores between 1967 and 1981. At present, only oxidation of antimony to Sb_2O_3 is conducted. Soil samples were collected at several locations around the smelter. The levels of antimony at the 34 sites were measured at different depths of soil. The antimony concentration in soil was determined using instrumental neutron activation analysis (INAA). In the surface layer (top 1 cm of soil), antimony concentrations ranged from 3.7 to 2100 mg Sb/kg. In almost all of the soil samples, except from the highly contaminated soils, the concentrations decreased to background levels at depths of 1-2 m. The soil close to the facility was very contaminated, and the concentration of antimony did not at these sites decrease with increasing depth. The highest concentration of antimony (2900 mg Sb/kg) was detected in the soil at a depth of 20 cm. In some of the depths profiles from highly contaminated soils, the antimony concentration was slightly higher at depths of 10-20 cm than in surface soil. Conversely, in less contaminated soils, the concentration of antimony decreased with increasing depth. These results indicate that antimony is transferred in a vertical direction. According to the Sb-K edge X-ray absorption edge (XANE) spectra, the antimony in the soil was in the form of Sb(V) compounds. The similarity of extended XAFS (EXAFS) spectra suggested that the speciation of antimony was independent of the sampling site, which indicates that Sb_2O_3 emitted from the smelter was converted into Sb(V) compounds in the soil, however not to orderly structures like Sb_2O_5.

Johnson and co-workers (2005) studied bank materials behind the targets at seven Swiss shooting ranges in order to determine whether or not Sb and other elements (Ni, Cu, Bi, Tl and Hg) originated from Pb alloy bullets become more soluble as a result of weathering and what mechanisms possibly controlling their solubility. In the leaching experiments, Sb was almost exclusively present in solution as Sb(V) in concentrations up to 5 mg Sb/l.The Ca mineral $Ca[Sb(OH)_6]_2$ was suggested to control dissolved Sb(V) concentrations in soil at high concentrations. It was concluded that at the prevailing pH (7.8-8.6) of the soils studied, the release of Sb was significant and considerably higher than the other elements studied.

Transformation experiments

A transformation/dissolution test with Sb_2O_3 in ecotox medium (ISO 6341 medium) at different pH performed by Lisec (2002b) showed that the lowest dissolved Sb concentration was obtained at neutral pH (pH 7), with higher concentrations obtained at both lower and higher pH. The mean dissolved concentrations obtained after 24 h for pH 5, 6, 7, and 8 were 1.1, 0.86, 0.62, and 1.9 mg Sb/l, respectively. This means that when comparing the amount dissolved antimony at neutral pH with the amount dissolved at pH 5, 6, and 8, the solubility's at these lower/higher pH-values differ by approximately a factor of 2, 1.4, and 3, respectively.

Diantimony trioxide has a low water solubility, but as previously was described for sediments, it will dissolve given due time, as also was demonstrated by Vangheluwe and co-workers (2001)

Antimony has a strong affinity for sulphur and therefore concentrates in sulphides, with the most important antimony mineral being stibnite (Sb_2S_3). However, the sulphide form of antimony is not stable under oxidizing conditions and the oxidation of the stibnite to its transformation product valentinite (Sb_2O_3), is a relatively quick process (days/weeks) (Ashley et al., 2003), as was also suggested by the data of (Pauwels1985). (Vangheluwe et al., 2001) performed transformation experiments in two different soils (one sandy soil and one loamy soil) with the antimony compounds $SbCl_3$ and Sb_2O_3, both at the concentration 50 mg Sb/kg dry matter. The sandy soil had a pH of 5.6, and the loamy soil (loam) had a pH of 7.3. The soils were either immediately incubated at 20 ºC or submitted to two drying/re-wetting cycles (DW) first and then incubated. After two, four, twelve, and twenty-four weeks, the pore water was filtered (0.45 μm) and analyzed with ICP-OES (see Table3-39).

Table3-39 Sb concentration (mg Sb/l) in pore water of a sandy and a loamy soil with 50 mg Sb/kg of dry substance applied as salt or oxide, after an incubation of 2, 4, 12, and 24 weeks (% CV in italics).

Soil type	Concentration (mg Sb/l)	Continuously wet				With 2 drying/rewetting cycles			
		2 w	4 w	12 w	24 w	2 w	4 w	12 w	24 w
Sand	Sb_2O_3	0.91	0.83	1.45	1.77	0.43	0.54	1.44	1.71
			0.5	*0.8*	*1.4*	*1.1*	*0.6*	*1.2*	*0.8*
	$SbCl_3$	5.71	3.34	3.14	2.58	2.71	1.87	2.41	1.66
			5.6	*3.5*	*0.9*	*1.4*	*0.2*	*7.9*	*3.8*
Loam	Sb_2O_3	0.19	0.15	0.44	0.51	0.12	0.11	0.44	
			5.2	*21.9*		*4.0*	*0.9*		
	$SbCl_3$	1.61	0.92	1.16	0.98	1.37	0.86	0.94	0.67
			4.0	*6.3*	*11.7*	*1.2*	*1.6*	*5.7*	*1.6*

As can be seen from the table, the pore water concentration in the $SbCl_3$-treated soils decreased over time, indicating that Sb is fixed stronger with time. However, the pore water concentration in the Sb_2O_3-treated soils increased over time, and was at the longest test interval, i.e. 24 w, in the sandy soil with DW even slightly higher than for the $SbCl_3$-treated soil. It is however not unlikely that the same would have been seen, i.e. about equal pore water concentration, in the loamy soil with DW if the 24 w incubation period would have been performed (was not performed due to insufficient amount of soil). However, based on the data available for 24 w in the continuously wet soil, it appears that the transformation in the loamy soil is a bit slower as compared to the sandy soil.

A comparison of the pore water concentration after applying the element as an oxide and as a salt allows estimating to what extent the oxide was transformed to a soluble form.

$$\% \text{ transformed to a soluble form} = (c_{oxide} / c_{salt}) \times 100$$

where c_{oxide} is the pore water concentration when applying the element as an oxide, and c_{salt} is the pore water concentration when applying it as salt.

Table3-40 Percentage of Sb_2O_3 transformed to a soluble form after incubation of 2, 4, 12, and 24 weeks.

Incubation period / Soil type	Continuously wet				With 2 drying/rewetting cycles			
	2 w	4 w	12 w	24 w	2 w	4 w	12 w	24 w
Sand	16	25	46	69	16	29	60	104
Loam	12	16	38	52	8	13	47	

The oxide transformation could quite well be described by first order kinetics. After two weeks, only 10 to 15 % of the Sb_2O_3 was transformed to a soluble form, and after 12 weeks almost half was transformed, which shows that the oxide dissolves and results in soluble forms of Sb (see Table3-40 above). It is thus likely that the difference in amount of measured soluble antimony between the salt and the oxide will disappear in the long run.

Similar results were found by Pauwels (1985) using Sb_2O_3 (amorphous), Sb_2O_3 (senarmontite), and Sb_2S_3. The difference in Kd values between Sb_2O_3 (amorphous) and K(SbO)-tartrate was rather small (Table3-41). Larger differences were initially found for Sb_2O_3 (senarmontite) and Sb_2S_3, compared with K(SbO)-tartrate, but this difference diminished with time for Sb_2S_3 (not tested for senarmontite).

Table3-41 Kd values (in l/kg) measured before (one month of incubation after antimony amendment, Kd (b) or after plant growth (for some months, Kd (a). Solution concentrations of antimony were measured in water extracts (S:L=1/5 kg/l) (Pauwels, 1985).

Sb compound	Conc. (mg Sb/kg)	sandy loam (pH = 5.7)		heavy clay (pH = 7.8)	
		Kd (b)	Kd (a)	Kd (b)	Kd (a)
Sb_2O_3 (amorph.)	32	60		94	
	1000	128	100	161	111
Sb_2O_3 (senarmotite)	1000	1316		962	
Sb_2S_3	32	160		178	
	1000	649	208	544	109
K(SbO)-tartrate	1000	79	78	111	93

Smolders and Mertens (2007) spiked soil samples, identical to the soil used by (Oorts et al., 2005) in terrestrial toxicity studies, with Sb_2O_3 or $SbCl_3$ at 500, 1000, 2000 or 10000 mg Sb/kg. Soil solution (pore water) was extracted at two days and at one, two, three, four and five weeks after Sb spiking by centrifugation (3000g, 15 min). The resulting mean porewater concentrations can be seen in Figure 3-2 and Figure 3-3.

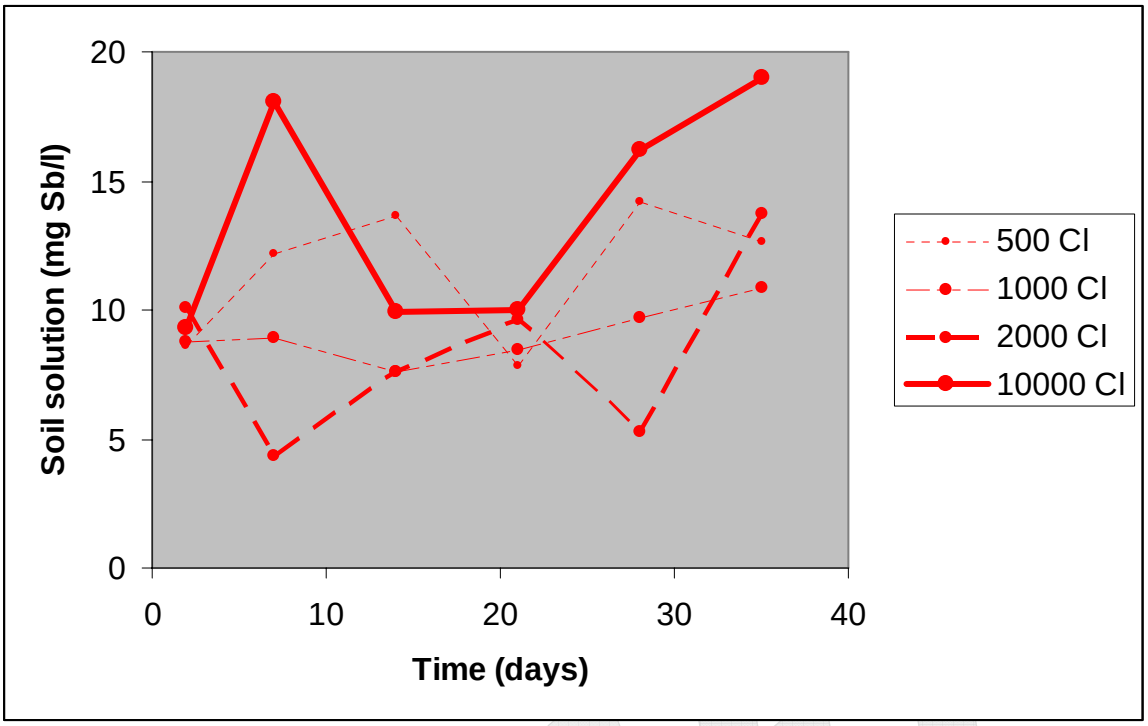

Figure 3-2 Mean concentration of Sb in pore water of soils spiked with $SbCl_3$ at 500, 1000, 2000 or 10000 mg Sb/kg. The soil solution was extracted in triplicate per soil sample at two days and at one, two, three, four and five weeks after Sb spiking by centrifugation (3000g, 15 min).

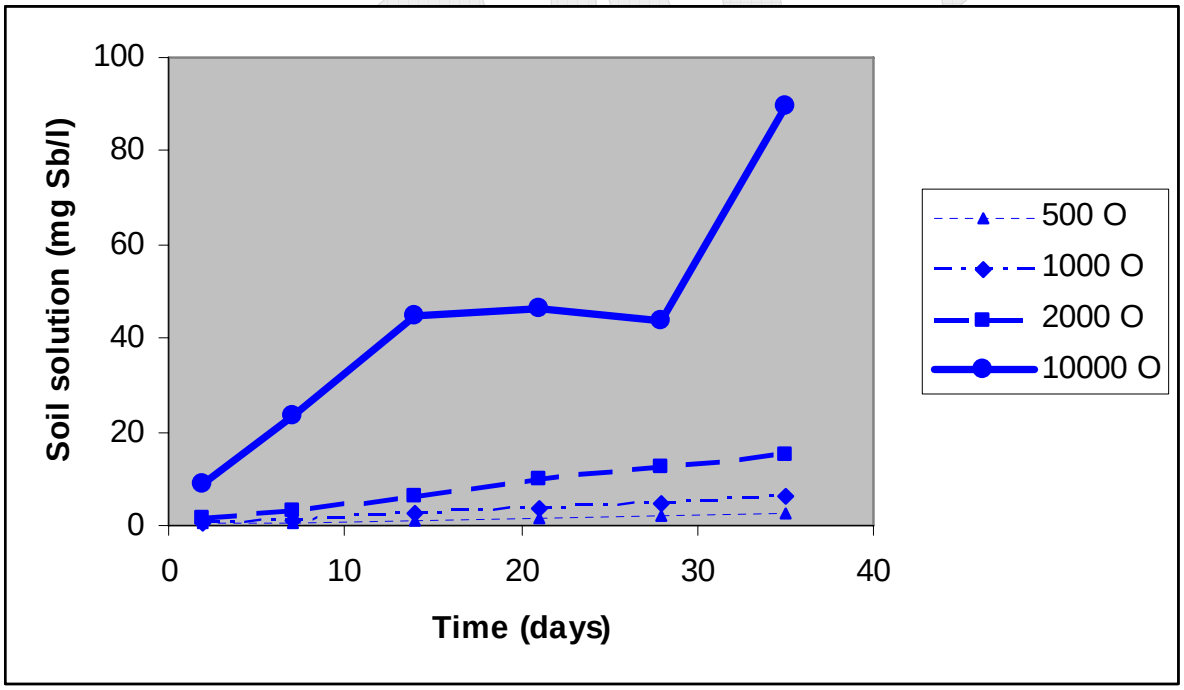

Figure 3-3 Mean concentration of Sb in pore water of soils spiked with Sb_2O_3 at 500, 1000, 2000 or 10000 mg Sb/kg. The soil solution was extracted in triplicate per soil sample at two days and at one, two, three, four and five weeks after Sb spiking by centrifugation (3000g, 15 min).

The authors compared the pore water concentrations in the $SbCl_3$ and the Sb_2O_3 spiked soils at 2000 mg Sb/kg, which was the highest concentration used in the study by (Oorts et al., 2005) and for which no toxicity was observed in the Sb_2O_3 spiked soils. Since the measured pore water concentration after one week was 3.2 and 4.4 mg Sb/l for the Sb_2O_3 and $SbCl_3$

spiked soils, respectively, and the concentrations measured at all later samplings not were considered to be "significantly lower" in the Sb_2O_3 as compared to the $SbCl_3$ spiked soils at 2000 mg Sb/kg, the pre-equilibrium of one week used in the toxicity study by (Oorts et al., 2005) before testing was considered sufficient.

However, this conclusion is not considered possible to draw when also including the results from the soils spiked with $SbCl_3$ at the other doses. For instance, the pore water concentration measured after one week at 2000 mg Sb/kg, i.e. the 4.4 mg Sb/l (which was used by the authors in the line of evidence when arguing that 1 week was a sufficient pre-equilibrium period) is most likely to low. This since the mean concentrations for the soils spiked with 500, 1000 and 10000 mg Sb/kg after one week were 12.2, 8.9 and 18.1 mg Sb/l, respectively. Furthermore, when comparing the soil solution concentration of Sb measured in the soil spiked with $SbCl_3$ in the 2000 mg Sb/kg dose group, with the concentrations measured in pore water of the other doses spiked with $SbCl_3$ at the various time intervals, conclusions based on comparisons with the 2000 mg Sb/kg dose appears most uncertain.

To conclude, the data presented by Smolders and Mertens (2007) does not support that the one week pre-equilibrium period used in the terrestrial toxicity study by Oorts et al. (2005) was sufficient and resulted in equal doses of Sb to biota in the $SbCl_3$ and Sb_2O_3 spiked groups at 2000 mg Sb/kg.

The authors hypothesised that their observations were related to the different chemical behaviour of the added Sb_2O_3 and $SbCl_3$ in soil. When $SbCl_3$ is added it hydrolyses in soil solution to $Sb(OH)_3$ and releases two or three protons depending on the pH (two protons at pH <3).

$SbCl_3 + 2\ H_2O \rightarrow Sb(OH)_3 + 3H^+$ at $pH \geq 3$

$\rightarrow Sb(OH)_2^+ + 2H^+$ at $pH < 3$

Besides lowering the soil pH, the released protons will also compete with other cations for binding at soil sites, which release e.g. Ca^{2+} or Mg^{2+}, resulting in increased ionic strength (as also was observed in the $SbCl_3$-spiked samples). Similarly Sb_2O_3 is also hydrolysed to $Sb(OH)_3$, but this reaction does not generate protons and therefore does not change the soil pH with increasing doses of Sb_2O_3.

$Sb_2O_3 + 3\ H_2O \rightarrow 2\ Sb(OH)_3$

In oxic environments the pentavalent $Sb(OH)_6^-$ is the dominant form and oxidation of $Sb(OH)_3$ to $Sb(OH)_6^-$ results in the release of one proton.

$Sb(OH)_3 + 2\ H_2O + 0.5\ O_2 \rightarrow Sb(OH)_6^- + H^+$

The pentavalent $Sb(OH)_6^-$ may react with Ca ions to form Ca-antimonate ($Ca[Sb(OH)_3]_2$), which precipitate.

$2\ Sb(OH)_6^- + Ca^{2+} \rightarrow Ca[SbO_3]_2$ (s)

This precipitation might explain the observed saturated Sb soil solution concentration (roughly around 10 mg Sb/l in the $SbCl_3$-spiked soils) at increasing Sb doses. The authors modelled the Sb solubility data and predicted that Sb added as $SbCl_3$ precipitates as $Ca[Sb(OH)_6]_2$ from doses about 400 mg Sb/kg with an equilibrium Sb soil solution concentrations of about 12 mg Sb/l using a solubility product of log Ksp = -9.6 for Ca-

antimonate, based on own solubility data. When instead adding the Sb as Sb_2O_3, the model predicts an equilibrium soil solution concentration of about 45 mg Sb/l.

Sorption

Results by Pauwels (1985) indicate that the strongest sorption of antimony in soil occur at pH values around 3 to 4 (Figure 3-4). The decrease in sorption at higher pH values can be attributed to the increasing negative charge of the soil surface, and hence, weaker sorption of the anionic $Sb(OH)_6^-$.

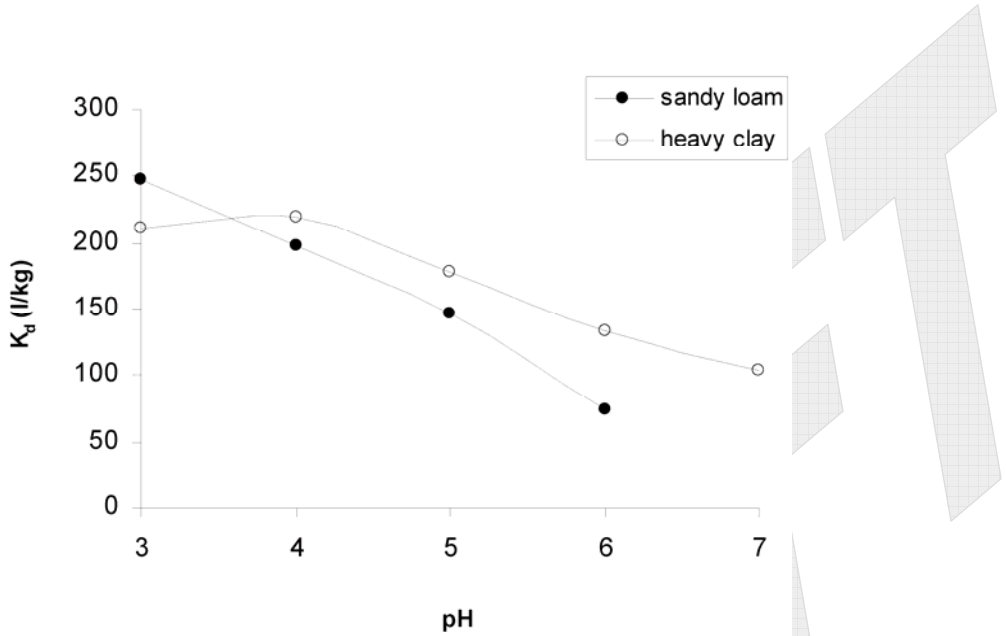

Figure 3-4 Dependence of Kd of antimony (in soils amended with 1000 mg Sb/kg as Sb_2O_3) on pH for 2 soils. Solution concentrations of antimony were measured in water extracts (S:L =1/3 kg/l) (Pauwels, 1985) (Reproduced from Degryse and Smolders (2004).

(Gerritse et al., 1982) studied the sorption of trace elements, at concentration levels closely corresponding to those to be expected for field conditions, from water, salt solutions, and sludge solutions to sandy soils and sandy loam soils. Combining two soils with nine liquid phases (including solution phases of sewage sludges) resulted in a total of eighteen soil/soil-solution phase systems. Soil adsorption was measured using radioactive tracers, obtained as chlorides or nitrates in the valency states most likely to occur in aqueous solutions (pH 4-8) under atmospheric pressure and O_2 level of the element was used. The results indicate that antimony is relatively mobile in these soils, with an adsorption constant of approximately 2-16 in the sandy soil and 20 in the sandy loam soil. Due to a combination of complexation by dissolved organic compounds and increased ionic strength of the soil solutions, the sludge application in general appeared to increase the mobility of the elements in soil. However, this did not appear to be the case for antimony.

King (1988) studied the capacity of antimony (antimony potassium tartrate) to adsorb to soil, using thirteen soils and subsoils from the southeast of the United States. Antimony adsorbed strongly to most soils (median sorption was 93 %). In all regression models most of the variation in Sb sorption was explained by the sand content (negatively correlated), while variables such as pH, organic matter, clay, presence of hydrous oxides of iron and manganese were of less importance in these soils. Once sand content was included in the regression

analysis, clay content did not explain much of the additional variation, because clay and sand content were strongly negatively correlated (r=0.84, as derived from the values given in Table 1 of King, 1988).

As previously described for sediments, several studies have correlated the adsorption of antimony in soil and sediment with the presence of iron-, manganese-, and aluminium oxides. It was also described that the hydrous oxides effectively oxidised trivalent antimony to pentavalent.

Due to the anionic character of the ($Sb(OH)_6^-$), antimony is expected to have a low affinity for organic carbon. However, results by (Tighe et al., 2005) indicate that the sorption of Sb(V) by humic acid in acid soils with high proportions of organic matter may be more important than previously suspected, although the strong Sb(V) scavenging potential of $Fe(OH)_3$ probably results in diminished role of organic matter binding in soils with high amounts of non-crystalline hydroxides. They studied the sorption of Sb(V), with concentration ranging from 0.0 to 11.3 mg Sb(V)/l, by two organic rich soils with high levels of oxalate extractable Fe over the pH range of 2.5-7 using a soil:solution ratio of 1:20. The sorption behaviour of Sb(V) was also examined in two phases mimicking those dominant in the experimental soils, namely a solid humic acid and an amorphous $Fe(OH)_3$, across the same pH range. The pH values for the points of zero charge (PZC) for soil 1, soil 2, the humic acid and the $Fe(OH)_3$ were determined to be 3.2, <2.0, 1.7 and 7.1, respectively. Sorption of antimony by the soils and the humic acid fitted a Freundlich type isotherm, with the equation parameters reflecting changes in bonding affinity corresponding to pH changes. The soils sorbed >75% of the added antimony in all trials, and 80-100% at pH values less than approximately 6.5. The $Fe(OH)_3$ retained >95% of the added Sb in all experiments. The humic acid sorbed up to 60% of the added antimony at the most acidic pH values, but the sorption decreased linearly with increasing pH to zero sorption at higher pH values, where dissolution of humic acid was visually evident (approx. pH 6-6.5). Possible mechanisms for the observed decrease in binding affinity with increasing pH were suggested. One was the decrease in positive surface charge of humic acids with increasing pH (i.e. move away from the PZC), and another was the dissolution of humic acids (discolouration of the final filtered solution was observed, even at pH values below 6, although typically not below 4-4.5), which in itself physically result in less available sorption sites regardless of other pH dependent exchange mechanisms operating.

There are also a few studies indicating at least some kind of affinity for NOM in natural waters (see section on natural organic material in aquatic transformation above).

The cationic exchange, which generally dominates the adsorption to clay for cationic metals, is not expected to be important for the anionic antimony.

D`Souza and Mistry (D´Souza and Mistry, 1980) studied the movement of ^{125}Sb following deposition onto established pastures in tropical conditions. Cuttings of napier grass were transplanted in polyethene troughs filled with medium black soil (pellustert) filled to a depth of 18 cm. The principal characteristics of the soil were pH, 8.0; cation exchange capacity, 40.5 meq; consisting of 3.7% coarse sand, 49.8% fine sand, 24.5% silt, and 22.0% clay. The troughs were initially enclosed in a polyethene tent and 100 μCi of nuclide in 100 ml distilled water was sprayed uniformly on two replicate troughs. The tent was removed after 24 hr, by which time, the spray had dried on the leaves and the troughs were then left open in the rain, and were when needed also irrigated. Weekly samplings of grasses were carried out to study

the washout of the nuclide on account of rainfall. The radioactivity in the herbage at the third cuttings (after 3 months) represent, in the main, the radionuclide absorbed from the soil as opposed to what originally was directly deposited on the plants. The changes with time in the radionuclide content of the edible herbage harvested at different times over a prolonged period extending up to 33 months post-deposition indicate that ^{125}Sb concentrations in the edible herbage approached background levels from the fifth cutting onwards, i.e. after a period of 14 months post-deposition. At the end of 33 months the soil from the troughs was sampled at various depths with the help of a soil augur to study the extent of downward movement of the radionuclide. These soil sections also contained a considerable amount of the grass root material. The vertical distribution of ^{125}Sb in soil up to 10 cm depth in through 33 months after a single foliar deposition indicated considerable movement of the fission product since the distribution of antimony at the different depths approximately were 70% in 0-2.5 cm, 17% in 2.5-5 cm, 8% in 5-7.5 cm, and 5% in 7.5-10 cm. The action of plant roots in transporting the radionuclide down in the soil column could be an additional factor influencing downward movement in the present experiments on pasture stands.

Varying the concentrations of antimony appears to only have a small influence on the sorption in soil. In the study by Vangheluwe and colleagues (2001), increased concentrations of antimony did not seem to result in decreased sorption, as has been described as a possible scenario for metals (Bockting et al., 1992), but instead the opposite was indicated with Kd values increasing with increasing concentration of antimony (Table3-42).

Table3-42 Kd values (in l/kg) for soils amended with Sb_2O_3 after 30 weeks of ageing. The solution concentrations of antimony were measured in water extracts (S:L = 1/5 kg/l) (Vangheluwe et al., 2001).

mg Sb/kg	sandy soil	loamy soil
0	>52	>45
10	31	51
20	42	53
50	38	50

3.1.3.1.4 Partition coefficients

Transport of metals between aqueous phase and soil/sediment/suspended matter should be described on the basis of measured soil/water, sediment/water and suspended matter/water equilibrium partition coefficients (Kp), instead of using common mathematical relationships based on, for example, octanol-water partition coefficients, as is usually done for organic chemical (TGD, 1996).

Kp_{soil}

U.S. EPA (1999) performed a literature survey in order to obtain partition coefficients that had been observed in field scenarios. These described the partitioning of metals between soil and soil-water, between suspended particulate matter (SPM) and surface water, between sediment and sediment-porewater, and between DOC and the dissolved organic phase in natural waters. The results for antimony are presented in Table3-43. The partitioning

coefficient for soil-water is a median value (n= 6) of log 2.4 (range 0.1-2.7). The fact that no further information was provided on the identified studies, the lowest log Kp being 0.1 (which is very low) combined with the lowest relative confidence level assigned the data results in the decision not to use the reported values from this literature survey in this RAR when deriving the partitioning coefficient for soil.

Table3-43. Partitioning coefficients (log Kp in l/kg) from literature survey U.S. EPA (1999) unless otherwise stated.

	Soil / Water (log Kp in l/kg)	Suspended matter / Water[1] (log Kp in l/kg)	Sediment / Water (log Kp in//kg)	DOC / Water (log Kp in l/kg)
Mean ± std. dev		4.8 ± 0.5		
Median	2.4		4.0	
Range	0.1 – 2.7	3.9 – 4.9	2.5 – 4.8	2.7 – 4.3[2]
n	6		3	
Relative confidence level (1=highest, 4=lowest)	4	4	Unknown	Unknown

[1]Mean, min, max from soil Kp regression equation
[2]From compilations performed in the literature survey performed by U.S. EPA

From a recent transformation experiment in soil using Sb_2O_3 and $SbCl_3$ by (Vangheluwe et al., 2001), the distribution coefficient Kd between soil and pore water was calculated to be 30 l/kg (log Kp_{soil} = 1.48) for Sb applied as $SbCl_3$ in sandy soil and 75 l/kg (log Kp_{soil} = 1.88) in a loamy soil. The transformation experiment was performed during 24 weeks, where soils were incubated at 20 ºC, with a final humidity of 22g/100g dry substance (both soils). The incubation was initiated with 2 drying/rewetting cycles. Since, according to (Bockting et al., 1992), the Kp for the adsorption of metals is not constant, but depends upon the equilibrium concentration of the aqueous phase (the Kp´s are usually inversely related to the concentration of the metal), these Kd may be to low, since the antimony concentration used (50 mg Sb/kg dw) only is found in heavily polluted soils. However, this "loading dependence" is element-specific, and seems to be quite small for antimony (see section on sorption above). Calculation of the Kp_{soil}, using a control soil in another experiment by (Vangheluwe et al., 2001), results in a partitioning coefficient in sandy soils of 81 l/kg (log Kp_{soil} = 1.91) and 150 l/kg (log Kp_{soil} = 2.18) for loamy soils. In the control the concentration was 0.57 mg Sb/kg in sandy soil and 0.45 mg Sb/kg in loamy soil. The corresponding pore water concentrations were 7 µg Sb/l and 3 µg Sb/l, respectively. The lowest applied Sb dose in this experiment (10 mg Sb/kg; Sb_2O_3), (which is higher than normally can be expected in the environment) resulted in the Kp_{soil} values of 26 l/kg (log Kp_{soil} = 1.41) and 21 l/kg (log Kp_{soil} = 1.32) for sandy and loamy soil, respectively).
Pauwels (1985) studied the distribution of Sb in soil and the uptake by plants for soils from Belgium and Algeria. The soils were treated with 32 or 1,000 mg Sb/kg (Sb_2O_3). The Kp in sandy soil (pH = 5.7) was 60 (log Kp_{soil} = 1.78) and 128 (log Kp_{soil} = 2.34) for the low and high concentrations, respectively, while in a soil with heavy clay (pH = 7.8) the Kp was 94 (log Kp_{soil} = 1.97) and 161 (log Kp_{soil} = 2.21) for the low and high concentrations, respectively.

A literature review on partition coefficients (Kp´s) with respect to adsorption experiments, with soils and sediments, was performed by (Bockting et al., 1992). The resulting partitioning

coefficients for antimony in soil and sediment were 1.93 l/kg (log value) and 3.41 l/kg (log value), respectively.

The value for soil, i.e. 1.93 l/kg (log value) was a geometric mean value of 10 measurements (from a study by King (1988), who performed batch experiments with 13 soil series samples (pH: 4.2-6.5, clay 2-63%) taken from the south east of USA. The equilibrium concentration of antimony was <1 to 100 mg/l, which ranges from background levels too much above background levels.
(Bockting et al., 1992) criticized the study of King (1988) because of the high loadings (1400 mg Sb/kg soil). One of their arguments was that for the elements that were also tested by (Buchter et al., 1989), the latter study found much higher Kd values than were found by King (1988). However, this may not be a true difference in Kd values, since differences in soil pH values between the two studies were not taken into account. If for instance Kd values of Zn are compared between both studies at the same loading (280 mg Zn/kg), it appears that both studies resulted in similar results (Figure 3-5).

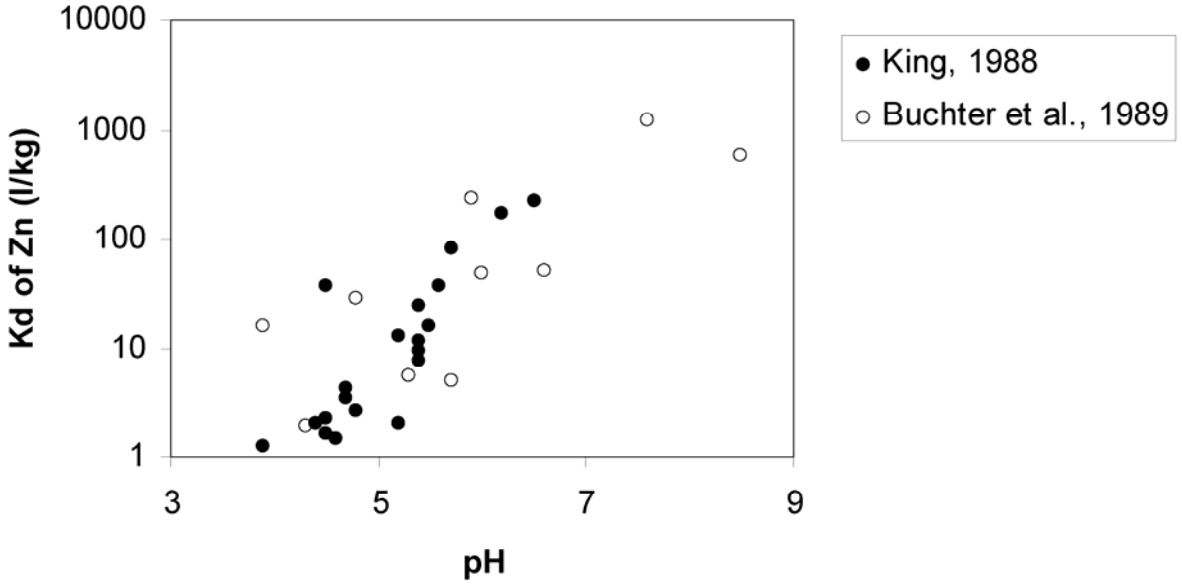

Figure 3-5 Dependence of Kd of Zn on pH measured in soil suspensions at a total zinc concentration of 280 mg Zn/kg (= actual zinc concentration in study of King (1988), and derived from Freundlich isotherm for the study of Buchter et al. (1989). (Reproduced from Degryse & Smolders (2004).

Based on the above, the study by King (1988) is considered reliable and the results will be used in this RAR. As regards the importance of pH, no pH dependence of Kd values of Sb was found in the study of King (1988), but Kd values were correlated with oxide content and with clay content. Since the pH effect on Sb sorption is quite small, this pH effect is probably hidden by the effect of oxide content, when comparing different soils.

Based on the soil classification system by USDA (USDA, 1967) the soil types sand (n = 2), sandy loam (n = 2), sandy loam-loam (n = 1), loam (n = 1), sandy clay loam-clay loam (n = 1), clay (n = 1) and silt loam (n = 1) was identified. In addition one humus rich soil sample with no data on sand, silt or clay is also available. Two additional humus rich soil samples were excluded since it was not possible to calculate partitioning coefficients from the data available. In order not to give an unproportionally large influence to the different soil types in this study (the majority consisting of only one or two values), as compared to the data from

other studies, it is pragmatically decided to combine the different loam soils, resulting in one sand soil (n = 2; Lakeland-A, Norfolk-Ap), one loam soil group (n = 7; ; Cecil-Ap, Davidson-Ap, Mecklenburg-Ap, Wahee-A, Iredell-Ap, White Store-A, Portsmouth-Ap) one clay soil group (n = 1; Hayesville-A) and one humus rich soil (n=1; Wasada-Ap). The corresponding (median; for the sand and loam soils) log values are 1.44, 2.35, 1.98, and 2.08, respectively.

(Nakamura et al., 2006) studied the mobility of antimony in Japanese agricultural soils by radiotracer experiments using ^{124}Sb tracer. The Kd values for Sb (0.1 mg Sb/kg added as $SbCl_3$, 7 days equilibration) measured in 110 Japanese soils ranged from 1 to 2065 L/kg, with a geometric mean of 62 L/kg (Log Kpsoil = 1.79) excluding one extremely high value, 2065 L/kg. Experimental measurements of Kd showed a decrease with both increasing pH and increasing phosphate concentration. The latter suggested that one aspect of the antimony sorption in Japanese soils was influenced by specific adsorption of anions such as phosphate. However, other aspects could not be explained by this specific adsorption mechanism, because only 20-40% of soil-sorbed antimony could be extracted by phosphate solution.

In a recent report by EURAS (2004) data on partitioning coefficients for antimony in the environment are evaluated. By the use of cumulative distribution functions log Kp-values were derived for soil, suspended matter and sediments (see Table3-44 below).

Table3-44 Partitioning coefficients (log Kp in l/kg) from literature survey Verdonck et al (2004).

	log Kp_{soil}	**log $Kp_{suspended\ matter}$**	**log $Kp_{sediment}$**
Median	1.79	3.57	3.30
Average	1.95	3.86	3.52
10P-90P	1.25 – 2.32	2.69-4.27	2.59-3.89
Range	1.18-2.4	1.66-4.5	2.48-4.20

The values used to derive the recommended values differ from those used in this RAR in the following respect:
Soil: The value resulting from the U.S. EPA literature survey (1999) of 2.4 is included; only two values (1.18, 2.18) from the study by Pauwels (1985) were included.
Suspended matter: The value resulting from the Orion Creek (USA) was included.
Sediment: Only three values (3.64, 2.48, and 2.48) from the Yodo river tributaries were included, and the value resulting from the U.S. EPA literature survey (1999) of 4.0 was included.

Five sources for log Kp_{soil} values for antimony has been identified:

i) The literature survey performed by the U.S. EPA (1999). The median value for log Kp_{soil} from this survey was 2.4 (range: min value 0.1, max value 2.7; n = 6). However, this result is disregarded based on the reasons discussed above.

ii) The study by (Vangheluwe et al., 2001), in which the log Kp_{soil} for sandy and loamy soils. The corresponding partitioning coefficients for the two control soils were 1.91 and 2.18, for the 10 mg Sb/kg exposure group it was 1.41 and 1.32. In another experiment using 50 mg Sb/kg the partitioning coefficients 1.48 and 1.88 were obtained for sandy and loamy soil, respectively.

iii) The study by Pauwels (1985), which resulted in the log Kp_{soil} 1.78, 2.34, 1.97, 2.21, with the two former coming from a sandy soil, and the two latter from a soil with heavy clay.
iv) The study by King (1988), which resulted in the values 1.44, 2.35, 1.98, and 2.08 for the sand, loam, clay and humus rich soils, respectively.
v) The study by Nakamura et al. (2006), which resulted in the log Kp_{soil} 1.79.

However, in order to try to give equal weight to results from different studies, performed in different types of soils, at different concentrations, it is decided to only use one value for each soil type from each study. Since the "loading dependence" for antimony in soil does not seem to be so important (see above) it is decided to take the median value of the partitioning coefficients for the different concentrations used for each soil type. This means that the data points (log Kp_{soil} values) resulting from the study by (Vangheluwe et al., 2001), that will be used in this RAR will be 1.48 for sandy soil (median value of 1.91, 1.32, and 1.48), and 1.88 for loamy soil (median value of 2.18, 1.41, and 1.88). The resulting data points from the study by Pauwels (1985) would be 2.06 for sandy soils (median value of 1.78 and 2.34) and 2.09 for the soil with heavy clay (median value of 1.97 and 2.21), the values from King (1988) are 1.44, 2.35, 1.98, and 2.08, and the value of 1.79 from (Nakamaru et al., 2006). Thus, the resulting values (log Kp_{soil}) are 1.48, 1.88, 2.06, 2.09, 1.44, 2.35, 1.98, 2.08 and 1.79.

Based on these nine data points, it is decided to use the median value (log Kp_{soil} = 1.98).

$Kp_{suspended\ matter}$

Partition coefficients for the distribution of metals between water and suspended matter are used to calculate the dissolved concentrations from total concentrations in surface water.

In a recent review by (Filella et al., 2002b), partition coefficients for the distribution of antimony between particulate matter and water (Kp_{susp}) were derived

Table3-45 Solid-water partitioning coefficients (log Kp_{susp}) in particulate material (from the review article by Filella et al. ,(2002)).

Log Kp_{susp}(l/kg)	Remark	Reference
4.5	Thames River, GB	Habib and Minski, 1982
3.4 3.5	World river average Ocean average	Martin and Whitfield, 1983
3.3	Solo River, ID	van der Sloot et al., 1989
1.66	Orion Creek water (near Van Stone Pb-Zn mine), US	Routh and Ikramuddin, 1996

Habib and Minski (1982) reported mean concentrations of antimony in soluble and suspended particulate fractions, 0.27 µg Sb/l and 8.4 mg Sb/kg, respectively, for three stations on the River Thames. The sampling covered a 10-month period, with a sampling frequency of every second week. Each sampling was performed and completed within five hours. The resulting log Kp-value is 4.49 l/kg.

(van der Sloot et al., 1989) determined metal concentrations of particulate matter in the River Solo (Indonesia). Sampling of the particulate matter involved continuous flow centrifugation of 50 L water. Particulate matter was obtained in two size/density fractions. The metal content of the particulate matter was determined using neutron activation analysis (NAA). The metal concentrations of the water samples (0.45 µm filtered) were determined with NAA after a preconcentration procedure. For the calculation of the Kp`s presented in Table3-46 above, the average metal content of the two size fractions was used. The resulting log Kp-value is 3.3 l/kg.

The study by Routh and Ikramuddin (1996) reported a log $Kp_{suspended\ matter}$ of 1.66 for the Orion Creek water (US). However, the basis for deriving this partitioning coefficient was an average of 23 water and 18 sediment concentration from the Onion Creek at a number of sampling locations near a lead-zinc mine and near two tailing ponds. Since this ratio involves the use of sediment concentration the reported partitioning coefficient is not considered relevant for deriving the partitioning coefficient between water and suspended matter. However, the partitioning coefficient is not considered useful when deriving a partitioning coefficient between water and sediment either, as it is not possible to correlate a specific water concentration with the corresponding sediment concentration. In addition, the concentrations vary between the different locations and average sediment concentrations. To conclude, it is therefore decided not to use this value in this RAR.

Veselý and colleagues (Veselý et al., 2001) derived partitioning coefficients for 41 elements, among them antimony, from water samples collected from 54 Czech rivers at 119 localities over the whole state territory in summers 1997 and 1998 under stable hydrological conditions. The analyzed river waters had a mean pH of 7.74 (range 6.9 to 8.8), ionic strength 7.8 mmol/l, specific conductance 538 µS/cm at 25 ºC, alkalinity of 1.9 mmol, and moderate mean contents of suspended particulate matter (SPM) 9.9 mg/l (range 1.0-124 mg/l). The 10^{th}, 50^{th} and 90^{th} percentile of the log $Kp_{suspended\ matter}$ were 3.54, 4.14 and 4.64, respectively. The values of Kp were calculated by dividing the total concentration of the element in SPM by its concentration in filtered water. There was no statistically significant dependence between log Kp values for antimony and loading (element loading is the sum of element concentration in the "dissolved" fraction and in the SPM).

Neal et al. (Neal et al., 2000) reported measurements of dissolved and total concentrations for the river Thames 34 km downstream of the market town of Oxford in Oxfordshire (UK) (based on weekly spot sampling). The reported average total and dissolved concentrations were 0.39 and 0.35 µg Sb/l, respectively. Using the average reported suspended matter concentration of 12.6 mg/l results in an antimony concentration on suspended matter of 3.17 mg/kg, corresponding to a log Kp-value of 3.96 l/kg. However, the reported mean value of 0.35 µg Sb/l for dissolved Sb is without outliers. When including the outliers, the mean value was 1.17 µg Sb/l, i.e. greater than the reported total concentration of 0.39 µg Sb/l. If the data that are considered as outliers for the dissolved concentration would have been left out for the calculation of the mean total concentration, this value would have been lower than 0.39 µg Sb/l. For the 'low flow' (no outliers in this category), for instance, the average total and dissolved concentration was 0.48 and 0.49 µg Sb/l. It is therefore decided not to use this value.

The literature review by (Bockting et al., 1992), described in the section on Kp_{soil} above, also included partitioning coefficient for suspended matter-water (see Table3-46).

Table3-46 Antimony log Kp´s for particulate matter from various aquatic systems. The Kp´s were calculated from the total metal content of particulate matter and corresponding water samples (modified from Bockting et al. (1992).

Log Kp (l/kg)	Location	Reference
3.52 3.64	Ysselmeer, Dutch lake. pH≈8.3	Van der Sloot et al. (1985)
4.07 4.00	Hudson River, North American River Hudson River + marine water	Li et al. (1984)
3.17 3.00 3.72	Solo River, Indonesian Rivers Wono Kromo River Porong River	Hoede et al. (1987)

(van der Sloot et al., 1985) measured several oxy-anionic metals, among them antimony, in water and particulate matter samples from the Dutch Wadden Sea, the North Sea, the estuaries of the Rhine and Scheldt and the lake Yssel. Kp's derived from data for lake Yssel (the only freshwater samples) are presented in Table3-46 above. Sampling of particulate matter involved filtration through 0.45 µm filters. The metal content was determined after leaching for 18 h with 0.1 N HCl.

(Li et al., 1984) performed batch experiments with particulate matter from the Hudson River (USA). Unfiltered Hudson River water samples (freshwater, C_L = 18 mg/L) were spiked with 13 radiotracers, including antimony. The spikes did not greatly change the natural concentration of various trace elements, as most radiotracers were carrier free and the amounts of spikes added were small. The samples were shaken for 20 days at 20°C. The aqueous radioactivity was determined in filtered (< 0.4 µm) sub samples taken at predetermined intervals using a germanium-lithium detector. Kp's for (unfiltered) Hudson river water and a 4 to 1 mixture with filtered seawater are presented in Table3-46 above (equilibration period 20 days).

(Hoede et al., 1987) determined metal concentrations of particulate matter in Indonesian river water. Sampling of the particulate matter involved continuous flow centrifugation of 50 L water. Particulate matter was obtained in two size/density fractions. The metal content of the particulate matter was determined using neutron activation analysis (NAA). The metal concentrations of the water samples (0.45 µm filtered) were determined with NAA after a preconcentration procedure. For the calculation of the Kp's presented in Table3-46 above, the average metal content of the two size fractions was used.

The literature survey performed by U.S. EPA (1999), described above under the section on Kp_{soil}, also resulted in a partitioning coefficient for antimony between suspended particulate matter (SPM) and surface water, and between sediment and sediment-porewater. The partitioning coefficient between suspended particulate matter and surface water was 4.8 ± 0.5, which was the result of a soil regression equation, and the already derived distribution coefficient between soil and water (Kp_{soil}). The literature survey included several different metals besides Sb (e.g. Ag, As, Ni, Cd, Co, Cu, Ni,…), and a general trend found was that the mean values of Kd in soil for metals was in the neighbourhood of two orders of magnitude less than the mean Kd value for suspended matter. This trend was characterized mathematically by developing a linear regression equation that was exploited to estimate mean Kd values for metals for which the literature provided an estimate of mean Kd in soil,

but not in suspended matter (or the opposite). The regression equations were developed from cases where the literature survey data provided reasonable estimates of the mean Kd for at least two of the three media. The equation used to estimate $Kp_{suspended}$ matter was based on 5 values, and had an r^2 of 0.37. The reliability of this estimated partitioning coefficient was not considered to by particularly high by the authors, since they used the lowest number for confidence level for this value (4 was used; 1 highest, 4 lowest). In addition, Sb appears as an anion as compared to the other ions, besides As, that appears as cations. It was therefore decided not to use this value in this RAR.

The following ten sources for $Kp_{suspended\ matter}$ values for antimony have been identified:

i) The study by Habib and Minski (Habib and Minski, 1982) in which the log $Kp_{suspended\ matter}$ for the river Thames in the UK was 4.5.
ii) The article by Martin and Whitfield (1983) including a log $Kp_{suspended\ matter}$ of 3.4 representing a world river average
iii) The study by (van der Sloot et al., 1989) including a log $Kp_{suspended\ matter}$ of 3. 3 for the river Solo (Indonesia)
iv) The study by Routh and Ikramuddin (1996) in which a partitioning coefficient of log 1.66 for the Orion Creek in the US was reported. However, this result is disregarded based on the reasons discussed above.
v) The study by (Veselý et al., 2001) in which the 50th percentile of log $Kp_{suspended\ matter}$ for Czech rivers was 4.14.
vi) The study by Neal et al. (2000) in which the $Kp_{suspended\ matter}$ for the river Thames in the UK was 3.96. However, this result is disregarded based on the reasons discussed above.
vii) The study by van der Sloot et al. (1985) including two log $Kp_{suspended\ matter}$ of 3.52 and 3.64 for the Dutch lake Ysselmeer
viii) The study by Li et al. (1984) in which the log $Kp_{suspended\ matter}$ for the Hudson river (North America) was 4.07.
ix) The study by Hoede et al. (1987) in which the log $Kp_{suspended\ matter}$ for the Indonesian rivers Solo, Wono Kromo, Porong, were 3.17, 3.00, and 3.72, respectively.
x) The literature survey performed by the U.S. EPA (1999). The partitioning coefficient between suspended particulate matter and surface water was 4.8 ± 0.5, which was the result of a soil regression equation, and the already derived distribution coefficient between soil and water (Kp_{soil}).. However, this result is disregarded based on the reasons discussed above.

However, in order to try to give equal weight to results from different studies performed in different waters, it is decided to only use one value for each water system. This means that the data points (log $Kp_{suspended\ matter}$ values) resulting from the River Thames will be 4.5, for the world wide average 3.4, for the River Solo 3.2 (median value of 3. 3 and 3.17), for the Czech rivers 4.14, for the Lake Ysselmeer 3.58 (median value of 3.52 and 3.64), for the Hudson River 4.07, for the River Kromo 3.00, and for the River Porong 3.72. Thus, the resulting values (log $Kp_{suspended\ matter}$) are 4.5, 3.4, 3.2, 4.14, 3.58, 4.07, 3.00, and 3.72.

Based on these eight data points, it is decided to use the median value (log $Kp_{suspended\ matter}$ = 3.65), and to perform a sensitivity analysis (see section 3.1.4.3) on the 10P (log $Kp_{suspended\ matter}$ = 3.1) and the 90P (log $Kp_{suspended\ matter}$ = 4.2). The result of the sensitivity analysis will be discussed in sections 3.1.4.4 and 3.1.4.8.

$Kp_{sediment}$

Partition coefficients for the partitioning of metals between water and suspended matter are used to calculate the concentration in sediment from the dissolved concentration in water on a local scale. When calculating the concentration in sediments on a regional scale, according to the TGD, the partitioning of metals between water and sediment should be used.

Brannon and Patrick (1985) determined Sb concentrations in interstitial water phase of 10 North American sediments. Sediments were kept under anoxic conditions. Ten freshwater sediment samples were mixed with cellulose (1% weight: weight basis) to enhance reduction of the sediment. Sb(III) was added (75 mg/kg, sediment dry weight) in the form of an antimony potassium tartrate solution. The containers were sealed and incubated for 45 days under a nitrogen atmosphere at 20˚C. After this period interstitial water from the sediments was collected by centrifugation of the sample under nitrogen atmosphere. All aqueous concentrations for unamended sediments were below detection limit. Log $Kp_{sediment}$ (l/kg) calculated from the total Sb-content of 10 Sb-amended freshwater sediments (native Sb-content and added Sb) and the corresponding interstitial concentrations ranged from 2.82 to 3.64. The average log $Kp_{sediment}$ was 3.32.

Kawamoto and Morisawa (2003) studied the distribution and speciation of antimony in river water, sediment and biota in tributaries of Yodo River, Japan. Samples were taken both upstream and downstream municipal sewage treatment plants (STP), accepting wastewater from dye works, which are the main industries in the area. The antimony concentrations in water did not correlate well with the antimony concentration in sediments, as the concentration of antimony in water upstream of the STP in the Katsura River in water was lower than downstream, 0.5 and 3.4 mg Sb/l, respectively, but the opposite was true for the concentration in sediments, 2.4 and 1.2 mg Sb/kg dry weight, respectively. Concentrations measured in water and sediments downstream another STP in Umi River were 3.4 mg Sb/l and 1.0 mg Sb/kg dry weight, which are very similar levels to those measured downstream of the STP located at Katsura River, mentioned above. Speciations of antimony revealed that the proportion of dissolved Sb(III) and Sb(V) changed depending on whether or not the sampling site was upstream or downstream a STP, and how far downstream of the STP the sampling was performed. The proportion of Sb(III) and Sb(V) upstream and far downstream of STP was 20/80 and 30/70, respectively, while it close downstream of an STP was 1/99. This difference, which was suggested to be the result of the processes in the STP, thus resulted in a non-typical water as regards the speciation of antimony. Partitioning coefficients calculated from measured concentrations in water and sediments upstream and downstream of STP in different rivers are presented in Table3-47 below.

Table3-47 Partitioning coefficients for sediment-water in Japanese rivers from Kawamoto & Morisawa (2003).

Location	log $Kp_{sediment}$ (l/kg)	Remark
Katsura River	3.68	Upstream of STP
Katsura River	2.51	Close downstream of STP
Katsura River	2.55	5 km downstream of STP
Kamo River	3.95	Upstream of STP
Uji River	2.47	Downstream of STP

Mori and co-workers (Mori et al., 1999) measured the concentration of antimony in water and sediments in Corsican Rivers upstream and downstream of an abandoned realgar mine The Bravona River spring is located on central Corsica and reaches after 38 km the Tyrrhenian Sea, on the eastern coast of Corsica. One of its affluents, the Presa River, crosses an abandoned realgar mine. Ten sampling sites were chosen: six are located on the axial course of the Bravona River, three on the Presa River (1 upstream the mine and two downstream) and one on a secondary affluent (the Alzillelo river). The Presa Rivers connects to the Bravora River between station B2 and B3. The resulting log $Kp_{sediment}$ are presented below.

Table3-48 Partitioning coefficients for sediment-water in Corsican rivers from Mori et al. (1999).

Location	log $Kp_{sediment}$ (l/kg)	Remark
Bravona River	4.20	Reference site, station B1
Bravona River	3.3	Reference site, station B2
Bravona River	2.99	Slightly polluted, station B3
Bravona River	3.05	Slightly polluted, station B4
Bravona River	3.30	Slightly polluted, station B5
Bravona River	2.69	Slightly polluted, station B6
Presa River (tributary to Bravona River)	3.51	Reference site, station P1
Presa River (tributary to Bravona River)	3.65	More polluted, station P2
Presa River (tributary to Bravona River)	2.89	More polluted, station P3
Alzillelo River (tributary to Bravona River)	3.12	Reference site, station A1

U.S. EPA (1999) performed a literature survey in order to obtain partition coefficients that had been observed in field scenarios. The outcome of this review was a median log $Kp_{sediment}$ value of 4.0 for the partitioning coefficient between sediment-water, which was calculated from three studies (2.5 –4.8; min-max). No further information was provided on the identified studies, and the relative confidence level given in the report was "unknown". It is therefore decided not to use this value in the RAR.

The literature review by Bockting et al. (1992), described in the section on Kp_{soil} above, also resulted in a partitioning coefficient for antimony for sediment-water. The value brought forward in the review was calculated using a geometrical mean of the partitioning coefficient for suspended matter-water (for values see Table3-46) that was divided with a factor of 1.5. It was assumed that Kp´s for sediments are roughly 1.5 times lower than for particulate matter. The relatively strong adsorption of metals by particulate matter is probably a result of the relatively high organic matter and clay content (size fraction smaller than 2 μm) (Bockting et al. (1992). However, this adsorption to the primarily negatively charged organic matter and clay particles is probably more relevant for cations, than for anions such as $Sb(OH)_6^-$ (the predominant antimony species).

Smock (1983) measured the concentration of seven metals, including antimony, in water, sediment and insects (gut contents and body) from a series of ten stations along the Haw and New Hope river systems in North Carolina, USA. Only sites relatively uncontaminated by metal inputs were used. Samples were collected on four dates during 6 month. Sampling was conducted only after at least 8 days without rain, a period which was assumed to allow metal

content of the sediments, water and biota sufficient time to reflect ambient conditions. In order to quantify any differences between stations, organism, water and sediment samples were collected simultaneously. Metal analyses were performed on 0.45 µm filtered samples. Analysis of metals associated with sediments was on the < 2 mm size fraction. Concentrations of antimony were determined using neutron activation analysis. A considerable temporal and spatial stability in organism metal concentration was reported. Differences in filterable and sediment-bound metal concentrations between stations and sampling dates were small and not of a sufficient magnitude to exert a significant effect on organism metal concentrations. Mean measured concentrations for all sampling sites and dates, 1 SE given in parentheses, was 0.03 (± 0.00) µg Sb/L in water and 6.06 (± 0.61) mg Sb/kg in sediment. However, it was not specifically stated whether the concentration in sediment was based on wet weight or dry weight, which will influence the size of the calculated Kd as TGD requires that sediment contain 80% water and 20% solid and a density of 1.3 kg/l. If dry weight is assumed, Kd = 6060 µg/kg/0.03 µg/l = 202000 l/kg → log Kd = 5.31. If instead the concentration is expressed on basis of wet weight, Kd = 15756 µg/kg/0.03 µg/l = 525200 l/kg → log Kd = 5.72. In summary, there is a 0.41 difference in log scale depending whether or not the presented concentration in sediment is expressed in wet or dry weight. Since it is unclear which Kd-value that should be used it is decided to not use either of them.

The following six sources for log $Kp_{sediment}$ values for antimony have been identified:

i) The study by Brannon and Patrick (1985) resulting in an averaged $Kp_{sediment}$ of 3.32 from 10 North American sediments..

ii) The study by Kawamoto and Morisawa (Kawamoto and Morisawa, 2003) in which the $Kp_{sediment}$ for the Japanese rivers (upstream and downstream of STP connected to Sb-emitting industries) were given: upstream STP – 3.68, 3.95; downstream STP – 2.51, 2.55, 2.47. Since the discharge of antimony, via the STP, changed the proportion of Sb(III) and Sb(V) upstream and downstream the STP, it is decided only to use the values obtained upstream the STPs.

iii) The study by Mori and co-workers (Mori et al., 1999) in which a number of $Kp_{sediment}$ for Corsican rivers (from reference sites to strongly polluted) were given: reference sites – 4.20, 3.3, 3.51, 3.12; polluted – 2.99, 3.03, 3.30, 2.69; heavily polluted – 3.65, 2.89.

iv) The literature survey performed by the U.S. EPA (1999). The median value for log $Kp_{sediment}$ from this survey was 4.0 (range: min value 2.5, max value 4.8; n=3). However, this result is disregarded based on the reasons discussed above.

v) The literature review by Bockting et al. (1992), included a partitioning coefficient for antimony for sediment-water, 1.93 l/kg (log-value). The value brought forward was calculated using a geometrical mean of the partitioning coefficient for suspended matter-water that was divided with a factor of 1.5. However, the basis for such a calculation is probably more relevant for cations, than for anions such as $Sb(OH)_6^-$ (the predominant antimony species), why the calculated value not will be used in this RAR.

vi) The study by Smock (1983) from which a log $Kp_{sediment}$ of 5.31 or 5.72 for the Haw and New Hope river systems in North Carolina, USA, can be calculated. Based on the uncertainty of which of these two values to use it is decided not to use either of them in this RAR.

However, in order to try to give equal weight to results from different studies performed in different waters, it is decided to only use one value for each water system. This means that the

data points (log $Kp_{sediment}$ values) resulting from the North American sediments will be 3.32, for the River Katsura will be 3.68, for the River Kamo will be 3.95, for the River Bravona 3.18 (median value of 4.2, 3.3, 2.99, 3.05, 3.3, 2.69), for the River Presa 3.51 (median value of 3.51, 3.64, 2.89), and for the River Alzillelo 3.12. Thus, the resulting values (log $Kp_{sediment}$) are 3.32, 3.68, 3.95, 3.18, 3.51, and 3.12

Based on these six data points, it is decided to use the median value (log $Kp_{sediment}$ = 3.4).

3.1.3.1.5 Biomethylation

Biomethylation of metalloids in the environment is an important transformation in the environment, which can have large influence on toxicity and biogeochemical cycling of metalloids. However, the importance of biomethylation as regards the metalloid antimony is not yet known.

Already in year 1933 Challenger and colleagues (Challenger et al., 1933) found that trimethylarsine was produced by fungal cultures (*Scopulariopsis brevicaulis*). Soon work in his laboratory demonstrated that methylation of selenium (Challenger and North, 1934), and tellurium (Bird and Challenger, 1939) also was possible. Since then biomethylation of tin (Chau et al., 1981; Guard et al., 1981; Hallas et al., 1982; Gilmour et al., 1985), mercury (Yamada and Tonomura, 1972; Hamdy and Noyes, 1975; Pan-Hou and Imura, 1982), and lead (Wong et al., 1975; Schmidt and Huber, 1976) has been detected and confirmed (Challenger, 1978). Considering the chemical similarity between antimony and the metals above, which surrounds antimony in the periodic table, and all of which have shown to be biomethylated, suggests the existence of a biomethylation pathway for antimony, but for long no experimental evidence was present. Thayer (Thayer, 1995) suggested that Challenger's mechanism for reduction and methylation of arsenic and selenium also would apply to antimony.

Oxidation of Sb^{3+} to Sb^{5+} by bacterium was reported in the mid 70´s (Lyalikova, 1974; Huey et al., 1975). Parris and Brinckman (1976) examined the oxidation of trimethylarsine and trimethylstibine by atmospheric oxygen and calculated gas-phase rate constants of 10^{-6} and 10^{3} M^{-1} s^{-1}, respectively. This indicates that the probably largest problem in detection of trimethylstibine is its fast oxidation in the gas-phase. Complete oxidation of biogenic trimethyl antimony in culture headspace to less volatile species, could account for the apparent absence of this compound in experiments that involved batch transfer of gases.

The first article of organoantimony compounds observed in nature was presented by (Andreae et al., 1981), in which methylated antimony species were found in the Gulf of Mexico, some rivers draining into the Gulf of Mexico and two German rivers. Of the two organoantimony compounds found, methylstibonic acid (MSA) and dimethylstibinic acid (DMSA), only the former was found in rivers, while both MSA and DMSA were found in marine and estuarine waters. However, due to the analytical technique used in these earlier studies, the results regarding the speciation of different methylated species cannot be trusted upon.

The methodology for analysis of antimony (organic and inorganic) in several of the earlier studies (e.g. Andreae and Froelich, 1984; Hirner et al., 1994) has been based on hydride generation of a sample or extract to produce the antimony as a volatile species (reductive derivatisation). Hydride generation of organic antimony compounds tends to produce several products from a single analyte (dismutation) (e.g. a trimethyl antimony compound may produce mono- and dimethyl antimony species during hydride generation). Nevertheless, this imperfect analytical approach still demonstrates that a carbon to antimony linkage was present in the original sample (Jenkins et al., 1998).

Brannon and Patrick (1985) noted the evolution of unidentified volatile antimony compounds during anaerobic incubation following potassium antimonyl tartrate amendment.

The studies by (Andreae et al., 1981; Bertine and Lee 1983; and Andreae and Froelich 1984), which were all questioned by (Dodd et al., 1992; and Thayer 1995), used graphite furnace atomic absorption spectrometry with hydride generation, which successfully had been used for arsenic. Due to the problematic extrapolation of analytical techniques for arsenic to similar methods for the speciation of antimony, (Dodd et al., 1992) suggested that the organoantimony results based on this procedure should be cautiously interpreted. Their criticism (Dodd et al., 1992; Dodd et al., 1996) specifically involved the problems of molecular rearrangements during the hydride generation step in batch-type generators that had not been extensively preconditioned. (Dodd et al., 1996) alleviated such problems by flushing the reaction coil (in the hydride generation phase) with acetic acid, $NaBH_4$ and distilled water prior to analysis. They reported the presence of methylstibine in extracts of a freshwater plant, namely pondweed (*Potamogeton pectinatus*), collected from a pond influenced by gold-mine drainage.

(Hirner et al., 1994) detected volatile species of antimony in gases released from domestic waste deposits, using ICP-MS. The concentration of antimony was determined (in eight gas samples from two deposits) to be 40-2400 ng Sb/m^3. Rough estimates were presented in which global emissions of several tonnes of antimony from solid wastes were estimated[9]. The same research group (Feldmann and Hirner, 1995) analyzed landfill gases and gases from sewage sludge fermentations at thermophilic and mesophilic conditions, using on-line coupling of GC with ICP-MS. One major peak was detected in both systems, which was identified as $Sb(CH_3)_3$ using comparison of retention time of standards. The concentrations measured (seven samples) in sewage gas ranged from 0.62-14.7 μg Sb m^{-3}, while those measured in landfill gas ranged from 24-72 μg Sb m^{-3}. As regards the large variation in sewage gas, there exists a tendency, that concentrations in the thermophilic fermentation are one order of magnitude higher than those measured in the mesophilic process.

Gürleyük and co-workers (1997) provided the first evidence that Sb(III) and Sb(V) are converted by micro-organisms in laboratory experiments. The organoantimony compound produced was trimethylstibine [$(CH_3)_3Sb$], which was detected in the anaerobic headspace.

(Jenkins et al., 1998) reported for the first time the formation of trimethylantimony by a characterized microorganism, *Scopulariopsis brevicaulis*, grown aerobically in the presence of inorganic antimony. No other volatile product containing antimony was detected in the culture headspace gases. The biovolatilisation occurred more readily from the inorganic

[9] The estimate is based on the measured concentration of antimony (40-2400 ng/m^3) combined with an estimated global methane emission of (30-70) x 10^6 tons per year from solid wastes and a methane content of 40% in landfill gases.

Sb(III) substrates (potassium antimony tartrate and Sb_2O_3), as compared to the Sb(V) substrates (Sb_2O_5).

(Craig et al., 1999) suggested that biomethylation to trimethylantimony occurred in aerobic environments, with subsequent abiotic oxidation to MSA and DMSA (or more complex oligomers containing the methyl antimony moiety). This could account for the presence of these involatile methylated species in fresh- and seawater, as detected by (Andreae et al., 1981; Bertine and Lee1983; and Andreae and Froelich 1984).

(Andrewes et al., 2000) found that dimethylantimony species is a true intermediate on the pathway to trimethylstibine, rather than arising from trimethylstibine oxidation. It was demonstrated that the dimethylantimony species was not an analytical artefact, formed during hydride generation process. Antimony was found to behave in the same way as arsenic with respect to the methyl source (S-adenosyl methionine) for biomethylation, and that the mechanisms of antimony and arsenic biomethylation in cultures of the model organism *Scopulariopsis brevicaulis* are probably the same. This may also be the case for other organisms.

In conclusion, similarities between antimony, and tin, lead, arsenic, selenium and tellurium which literally surround antimony in the Periodic Table, and all of which have been shown to be subject to biomethylation, suggest biomethylation pathways for antimony. However, despite these strong indications that also antimony should be able to biomethylate, biomethylation of antimony has only relatively recently been proven (in both aerobic and anaerobic environments).

Methylated antimony has been found in natural waters, where it usually account for 10% or less of the total dissolved species, and in biota (reviewed in Filella et al., 2002a).

Studies by Feldmann & Hirner (1995) detected methylated antimony species in landfill gas and in gases from sewage sludge fermentation at thermophilic and mesophilic conditions. The volatile organometallics were analyzed using ICP-MS. One major peak was detected in both systems, which was identified as $Sb(CH_3)_3$ by comparison to the retention time of the standards. Similar concentrations of $Sb(CH_3)_3$ were found in the two types of gases. Stibine (SbH_3) was detected only in landfill gas in a very low concentration.

All of this indicates that biomethylation is a way of mobilization of antimony in the environment. However, it is not possible at present to take biomethylation into account in the risk assessment due to the scarcity of (relevant and reliable) data.

3.1.3.1.6 Summary of environmental transformations

Distribution

Antimony released to the environment will eventually end up in either of the two compartments soil or sediment, depending on the release, form of antimony, meteorological conditions, etc. It may also end up in organisms.

Sorption

The most important factor as regards the mobility of antimony in soil and sediments appears to be the presence of hydrous oxides of iron, manganese, and aluminium, to which antimony may adsorb. In addition, these hydrous oxides seem to effectively oxidise dissolved trivalent antimonite ($Sb(OH)_3$) to the pentavalent antimonate ($Sb(OH)_6^-$). The effect of pH as such is probably less important as compared to the effect of the hydrous oxides. The largest effect of pH on sorption seems to be around 3-4, with decreasing sorption at higher pH-values. The decrease in bioavailable antimony due to adsorption on the hydrous oxides is not a permanent decrease, as it is dependent on both pH and oxic condition (which may change). The influence of the concentration of added antimony on sorption is small. Initial differences on sorption depending on type of antimony compound used diminish with time. Interactions between the antimony species (anionic $Sb(OH)_6^-$ or the neutral $Sb(OH)_3$) and the predominately negatively charged natural organic matter may occur, but any firm conclusion on its importance is presently hard to make.

Precipitation

Available data about antimony in soils suggests that antimony concentrations in soil solution do not depend on the slow dissolution of Sb_2O_3, but are mostly controlled by sorption reactions and precipitation of $Ca[Sb(OH)_6]_2$ at high concentrations (in the range of mg/l). The solubility of antimony compounds depends on the soil conditions (Eh/pH) and time given to dissolve. During environmental conditions antimony exists in natural waters almost exclusively in the dissolved phase in the two valence states +3 and +5. Both Sb(III) and Sb(V) ions hydrolyse easily, and Sb(III) is present as the neutral species $Sb(OH)_3$, and Sb(V) as the anion, $Sb(OH)_6^-$. Precipitation in water, as been described by (Brooke et al., 1986) using high (nominally 25-50 mg Sb/l) concentrations of $SbCl_3$, is not considered relevant in natural waters during environmentally relevant aerobic conditions since environmental levels are substantially lower (in the range of μg Sb/l).

In the presence of sulphur in anoxic systems, depending on pH, antimony will form soluble or insoluble stibnite, SbS_2^{2-} and $Sb_2S_3(s)$ (Filella et al., 2002b).

Volatilisation

Antimony will volatilise during combustion processes (smelting operations, combustion of coal, and refuse incineration) and will subsequently condense on suspended particulate matter that is predominantly less than 1 μm in size. Such fine particles are less effectively trapped by pollution control devices than are larger particles. In the atmosphere, they tend to settle out slowly; they are also removed by dry and wet deposition, which for antimony is confirmed by the fact that atmospheric fluxes are being dominated by wet deposition (Cutter, 1993; Arimoto et al., 1995).

When released into the atmosphere as an aerosol, antimony is believed to be oxidised to diantimony trioxide by reaction with atmospheric oxidants. Diantimony trioxide particles do not undergo changes in chemical composition, particle size, or morphology after emission; however a surface coating of sulphate may form (Ainsworth, 1988).

Several studies using bacteria in both aerobic and anaerobic cultures have shown that methylation of antimony may result in the formation of volatile antimony compounds (e.g. Gurleyuk et al., 1997; Jenkins et al., 1998). Volatile antimony compounds have been detected in landfill and sewage gases and it has been suggested (Hirner et al., 1994) that the global emissions of methylated antimony compounds from landfills could be in the size of several tonnes per year.

3.1.3.1.7 Distribution in wastewater treatment plants

According to Sternbeck et al. (2002a) the average removal of Sb from wastewater, as STP sludge, in four Swedish STPs was about 50% (range 34-60%). Methods used during treatment included biological-chemical (all), extra removal of nitrogen (two), and additional filters (one).

3.1.3.2 Accumulation and metabolism

Antimony has been considered to have a low to moderate bioaccumulation potential in both marine and freshwater species (US, 1979; Mulder et al., 1986). Judging by the results of (Bonotto et al., 1983; Mann and Fyfe 1988, and Mann et al., 1988) bioconcentration of antimony in aquatic algae is minor, if present. There are however no laboratory studies aiming at measuring the bioaccumulation of antimony available to verify this. However, tentative bioconcentration factors can be estimated using results from monitoring studies where the concentration of antimony has been measured in different aquatic organisms.

(Chapman et al., 1968) calculated bioconcentration factors for a number of elements, including antimony, for fish, invertebrates and plants in both marine and freshwater environments based on literature data. The concentration factors were based on the concentration of the elements expressed in ppm (in water) and ppm of wet weight (in aquatic organisms). When elemental concentrations was reported otherwise (i.e., as units of dry or ash weight of an aquatic organism), the value were converted to wet weight with the assumption that wet weight values are ten percent of values reported as dry weight and one percent of values reported as ash weight. The concentration of the element in seawater was selected from the literature to be representative of the continental shelf or estuarine waters, since the majority of seafood is harvested from these inshore waters. The representative concentration of antimony was chosen to be 0.5 µg/l, based on the following references (Goldberg, 1965; Bowen, 1966; Robertson, 1967; and Schutz and Turekian, 1965).

Chapman calculated a bioconcentration factor for antimony of 16000 (8/0.0005) in oyster, using the concentration 8 mg Sb/kg ww obtained from (Brooks and Rumsby, 1965). In the study by Brooks and Rumsby, six samples each of the three species of bivalves, *Ostrea sinuate*, *Pecten novae-zelandiae*, and *Mytilus edulis aoteanus* were sampled 18 km north of Nelson, New Zealand, at a depth of 22 m in the Tasman Bay. The content of trace elements, among them antimony, were determined in the whole animals excluding shells, in the shells, and in the individual dissected organs and also in sediment. Antimony was below the

detection limit of 30 mg Sb/kg dw in all samples, except from the muscle of the oyster *Ostrea sinuate*, where a concentration of 80 mg/kg dw was measured.

For marine fish, Chapman and co-workers derived a bioconcentration factor of 40 using a concentration of 0.02 mg Sb/kg ww derived from literature (Vinogradov, 1953). In the report by Vinogradov the chemical composition of marine organisms are collected for a number of species and elements. A value of 0.2 mg Sb/kg dw in this book is given for both the spiny dogfish (*Squalus acanthias*) (eviscerated) and Goldsinny (*Ctenolabrus rupestris*) (whole), both values originating from the article by Noddack and Noddack (1939a).

Maher (1986) measured antimony in marine organisms and waters from South Eastern Australia using HG-AAS. Water was filtered through a 0.45 µm filter. Samples of macroalgae were washed with distilled water to remove salts, freeze dried and ground to pass a 200 µm sieve. Animals were separated into component tissues and composites prepared by combining the tissues of five freeze dried specimens of each sample. The concentration measured in the marine water (140ºE, 51.5ºS) was 0.17 ± 0.02 µg Sb/l. The concentrations measured in biota are presented together with BCFs in Table3-49, below. Tentative bioconcentration factors have been calculated using the concentrations in biota and in water measured by Maher (1986), and a conversion factor of 10 between concentrations in dry weight and wet weight. The resulting bioconcentration factors for algae, mollusc tissues, crustacean tissues, and fish muscle are 55 - 114, 18 - 35, 10, 68, and 5-6, respectively.

Table3-49 Concentrations of antimony in different marine species and corresponding tentative BCFs outside of South Eastern Australia (Maher, 1986).

Sample	Concentration (mg/kg dry weight)	Tentative BCF
Algae		
Ecklonia radiata	0.094 ± 0.005	55
Sargassum lacerifolium	0.121 ± 0.006	71
Ulva sp.	0.193 ± 0.008	114
Mollusc tissues (*Mytilus edulis planulatus*)		
Mantle	0.031 ± 0.003	18
Visceral mass	0.047 ± 0.004	28
Abductors	0.060 ± 0.005	35
Crustacean tissues (*Helograpsus sp.*)		
Muscle	0.018 ± 0.002	11
Visceral mass	0.116 ± 0.006	68
Pisces (muscle)		
Arripus georgianus	< 0.009	<5
Semir hamphus australis	0.010 ± 0.001	6
Stillaginodes punctatus	< 0.009	<5

In a large survey by (Hall et al., 1978) trace element levels, including antimony, were determined in tissues of finfish, mollusca, and crustacean taken from 198 sites around the coastal United States, including Alaska and Hawaii. Muscle was analysed from 159 species of finfish, liver from 82, whole fish from 17, mollusca from 18, and crustacean from 16 species.

The mean levels of antimony in most finfish muscles and livers fell in the range of 0.5-0.9 mg/kg. Most species of whole finfish had antimony levels between 1.0 and 3.0 mg Sb/kg. Most shellfish species displayed mean antimony levels between 0.8 and 1.0 mg Sb/kg. The report does not clearly state on what basis (dry weight, wet weight etc.) the reported concentrations are given. The rapporteur assumes that they are given on a wet weight basis. Tentative BCF values for whole finfish and shellfish would be in the range of 5000-15000, and 4000-5000 respectively using a concentration of antimony in marine waters of 0.2 µg Sb/l (Filella et al., 2002a).

Ute and Bligh (1971) measured the concentration of a number of metals, including antimony, in freshwater fish from a lake free of major industrial development (Moose Lake) and from lakes in a highly industrialised area (Lower Great Lakes Basin). All samples were composite samples consisting of at least 2.5 kg or three fish. In the majority of samples the number of fish used were larger than three. Samples were prepared as follows: headless dressed fish (at least three) were ground and thoroughly mixed and stored at -40 ºC until analysis. Antimony was analysed using Neutron Activation. The concentration of antimony in the water of the lakes was not measured in the study. Tentative BCFs have been calculated by the rapporteur using an estimated antimony concentration in surface water of 0.3 µg Sb/l. The resulting BCF values are presented together with the actual concentration in fish in Table3-50 below.

Table3-50 Concentrations of antimony in fish from lakes in the Great lakes area and corresponding tentative BCFs.

Location	Species	Concentration (mg Sb/kg ww)	Tentative BCF
Moose Lake	*Coregonus clupeaformis*	0.0022	7
	Esox lucius	0.0032	11
Lake Ontario	*Coregonus clupeaformis*	0.0031	10
Lake St. Pierre	*Esox lucius*	0.0037	12
Lake Erie	*Esox lucius*	0.0043	14
	Osmerus mordax	0.0035	12
	Perca flavescens	0.0031	10

Couillard et al. (2008) evaluated the relationships between concentrations accumulated by specimens of the amphipod *Hyalella azteca* and concentrations in water for 27 metals (including antimony) in a field deployment defined in time in two metal-contaminated rivers in northwestern Québec, Canada. The amphipods were placed along with natural food items in small acrylic cages and left in six riverine sites for 17 days. Based on the findings by (Borgmann et al., 2007), which showed that the dissolved phase was the dominant route of metal accumulation for 24 (including antimony) of the 27 metals, Couillard and co-workers (2008) concluded that biouptake of metals in nature by *Hyalella* mainly was attributed to bioconcentration and therefore adequately represented a field BCF.
The use of *Hyalella* for deriving BCF is supported by the findings of (Borgman et al., 2004) who developed a mechanistically based saturation model for bioaccumulation of metals in *Hyalella azteca* in the laboratory. Although this results in a BCF that decreases with increasing concentrations of metal in the water, a background-corrected BCF at low water concentration can be calculated from the slope of the bioaccumulation curve at metal concentrations in water approaching 0 (Norwood et al., 2007). The determination of BCFs in the field study by (Couillard et al., 2008) could not be determined using the full bioaccumulation curve because none of the transplant sites had enough metal levels to reach the plateau region of bioaccumulation corresponding to maximum uptake. In addition, metal

bioaccumulation from food had to be considered. Presently, no criteria exist regarding the measurement of BCFs and BAFs in the field (Arnot and Gobas, 2006). Therefore, the approach taken by Couillard and co-workers (2008) included key characteristics of methodologies for deriving BCF-values (OECD, 1993; Borgmann et al. , 2004), namely:
- the requirement of having at least three, low exposure (i.e. substantially below acute toxicity), treatment levels per metal for the test species (Couillard et al., 2008) used three deployment per river);
- for a given metal the requirement of obtaining an absorption isotherm with a slope of approximately 1; this isotherm is defined as the log-log relationship between the chemical concentration in the test-organism and that in the water (OECD, 1993). This condition is equivalent to reaching steady-state between organism and water "compartments" for the metal studied.
However, for all sites, except one, either the measured concentration of antimony in the *Hyalella* or in the water, or both, were below the detection limit. Thus, one of the two defined "key characteristics" above could not be fulfilled for antimony. The BCF-value calculated for the only remaining site where the mean measured concentration of both the Hyalella and the water were above the detection limit, is 0.056. Although not fully reliable the study indicates that the bioaccumulation of antimony in Hyalella is low.

Summary conclusion
No fully reliable bioaccumulation studies are available and measured data from different aquatic organisms have been used to calculate tentative BCF-values. For marine fish the BCFs vary between 40 and 15000 whereas for freshwater fish the BCF values are lower the highest being 14. For invertebrates tentative BCFs in the range of 4000-5000 have been calculated. As opposed to these values a study with caged specimens of *Hyallella azteca* indicate a BCF-value of approximately 0.06. As there is a considerable uncertainty in these BCF-values the risk characterization for secondary poisoning will be performed both using both a BCF of 40 and a BCF of 15000.

3.1.3.2.1 Terrestrial organisms

There are no laboratory studies on bioaccumulation of antimony in earthworm available. However, concentrations of antimony in soil and invertebrates at locations close to an antimony smelter in UK have been measured by Ainsworth *et* al. (Ainsworth et al., 1990b). The highest antimony concentrations were found in earthworms (*oligochaeta*). The dry weight concentrations (including gut contents) were 398 ± 94 mg/kg, 213 ± 65 mg/kg and 109 ± 28 mg/kg 100 m, 250 m and 450 m downwind of the smelter. Based on measurements in the Figure 3 in Ainsworth *et* al. (Ainsworth et al., 1990b) result in the following approximations for the 0-5, 5-10, 10-15 and >15 cm depths for the 100 m, 250 m and 450 m sites; 367, 253, 220 and 171 mg Sb/kg dw, 204, 155, 114 and 94 mg Sb/kg dw, and 180, 180, 163 and 135 mg Sb/kg dw, respectively. Ainsworth gives earthworm body burden:soil ratios, which were 1.03, 1.05 and 0.6 for the 100 m, 250 m and 450 m sites, respectively. Based on these data a BCF of 1 for earthworms is taken forward to the risk characterisation.

3.1.3.3 Analysis of antimony

Several recent reviews have dealt with the subject of antimony analysis and speciation (Krachler et al., 2001; Nash et al., 2000; Smichowski et al., 1998).

Previous studies on antimony have almost all focused on total antimony concentration, with only a few studies being available on individual antimony species.

Determination of total antimony concentration is far easier than the speciation of the various antimony species; inorganic (Sb(III), Sb(V), and organic antimony compounds. Since most analytical techniques for antimony determination use liquid samples, analysis of environmental water samples requires minimal sample preparation, while determinations in solid materials require sample preparation in the form of matrix digestions techniques that sufficiently dissolve the solid sample matrix and release the analyte into solution (Nash et al., 2000).

Total concentrations of antimony are now readily achievable as a result of technological advances with analyte detection system and the development of ICP-AES and in particular ICP-MS techniques (Nash et al., 2000). Determination of total antimony from solid sample have also benefited from these detection systems, even though liquid sample introduction often is required, and pre-analysis matrix digestion can be problematic.

Problems specific for solid matrices when determining total antimony is a poor recovery, which may result from loss of antimony due to volatilisation, and incomplete matrix decomposition (Nash et al., 2000). The use of microwave digestion techniques has proven successful to reduce volatilisation, which taken together with a careful choice of digest reagents (in order to ensure adequate matrix dissolution) will improve the recovery. Since the same analytical techniques, as for the aqueous samples described below, often are used, interfering matrix ions may cause problems, which may be particularly pronounced for solid matrices. Techniques to overcome this problem for aqueous samples, i.e. solvent extraction and selective adsorption techniques are described below.

Problems encountered at the determination of total antimony in naturally derived water samples are due to the low analyte concentration (often in the μg/l range), and analytical interferences due to the concentration of problematic matrix ions (Nash et al., 2000). Preconcentration or separation of the analyte prior to analysis can alleviate the problems with low concentration; however, improvements in analyte detection depend upon the original sample volume used and the sample concentration factor (Vien and Fry, 1988). Solvent extraction and selective adsorption may remove many problematic matrix ions, thereby removing problems with analytical interferences due to presence of hydride-forming elements (besides Sb, such as As, Bi, and Sn) and precipitation of elements (such as Cu, Pb, Cr, Fe, Co, and Ni) (Nash et al., 2000).

Owing to different solution chemistry, simultaneous separation and analysis of both inorganic and organic Sb compounds still represents a great challenge for analytical chemists (Krachler, Emons and Zheng, 2001).

However, numerous difficulties are associated with Sb speciation, many of them basically being caused by the “special” chemistry of Sb (Krachler et al., 2001). Problems encountered are:

1. Low extraction efficiency in most matrices.

2. The lack of suitable Sb standards, due to the "troublesome" chemistry of Sb, which hampers the extension of separation procedures, and the identification of unknown peaks in chromatograms (Krachler et al., 2001). As regards methylated Sb compounds, only trimethylated Sb compound are available, i.e. trimethylantimony oxide ($TMSbCl_2$), trimethylantimony dihydroxide ($TMSb(OH)_2$) and trimethylantimony oxide (TMSbO), were the former is the only commercially available and the two latter only are available within the scientific community. Attempts to synthesise soluble mono- and dimethylated Sb compounds to be used for speciation studies have failed so far because; they tend to polymerise upon dissolution, cannot even be synthesised as a monomer, or are not stable under ambient air (Krachler et al., 2001). In addition, mono- and dimethylated Sb compounds can sometimes only be dissolved under drastic conditions, which is not desirable for speciation in liquid extracts at all. To illustrate this unfavourable behaviour of antimony, the synthesis of monomethylated antimony may serve as an example: alkaline hydrolysis of $MeSbCl_2$ with NaOH yields $(MeSbO)_3$. This compound, characterised by MS, is air-sensitive and not soluble in water, benzene or dimethyl sulfoxide, although soluble under alkaline conditions. Using the standards $TMSbCl_2$ or TMSbO in hydride generating (HG) system have resulted in severe molecular rearrangements, originating from the demethylation of the trimethylantimony standards during the analysis. This demethylation leads to the formation of Me_2SbH, $MeSbH_2$, SbH_3, and Me_3Sb. This problem has been extensively discussed in the literature, but has not yet been unequivocally solved (Dodd et al., 1996).

3. Lack of stability of the investigated Sb compounds throughout the entire analytical process (sample collection, and processing such as drying, extraction, separation, and detection) (Krachler et al., 2001). A study by Krachler and Emmons (2001) revealed that Sb(III) is easily oxidised within some hours to Sb(V). This finding raises the question whether or not the concentrations of Sb(V), found to be the major Sb species in extracts of various matrices, may have been overestimated owing to oxidation of Sb(III) to Sb(V) during sample manipulation. The quick oxidation of Sb(III) to Sb(V), mentioned above was observed using laboratory water. All the studies having reported Sb(V) (as the difference between Sb_{total} and Sb(III) under anoxic conditions used environmentally sampled brackish-marine water, which may behave somewhat differently in this respect as compared to laboratory water. In fact (Andreae et al., 1981) reported of oxidation of Sb(III) reference standards solutions at low concentrations, and that the rate of oxidation appeared to depend on the quality of the water used to make the dilutions. They suspected that small amounts of oxidant, e.g. active chlorine, present in the water might be responsible for the oxidation. In fact, a somewhat unorthodox, but highly efficient, solution to this problem was to make the 10 and 0.1 µg/l intermediate standards for Sb(III) using water from a local river, which typically contained less than 5 ng/l of Sb(III). These solutions were very stable even at the 0.1 µg/l level. In the studies were Sb(V) where detected during anoxic conditions, the water samples analysed by Bertine and Lee (1983) were initially treated with citrate buffer (which is known to act stabilising on Sb(III), the samples analysed by (Andreae et al., 1984) was rapidly frozen before analysis, and the samples by Cutter (1991, 1992) were refrigerated before being quickly analysed on-board ship.

To conclude, the reason for the relatively limited knowledge on antimony, its different species and behaviour, is to some extent explained by the "difficult chemistry" of antimony and the relatively recent advances in analytical techniques.

3.1.4 Calculated and measured concentrations

The concentrations of antimony in the environment can be estimated via calculations or via available measured data.

In section 3.1.4.1 below, different background concentrations (natural, baseline, ambient and realistic worst case ambient) are described. In section 3.1.4.2 calculated $PEC_{regional}$ for the different compartments are compared with the realistic worst case concentrations for the corresponding compartments.

3.1.4.1 Definition of terms expressing different environmental concentrations

In order to interpret data on environmental concentrations of antimony, four definitions are presented below in order to explain the difference between natural background, baseline background, ambient and realistic worst case ambient concentrations.

3.1.4.1.1 Natural background concentration

The natural concentration of an element in the environment reflects the situation before any human activity disturbed the natural equilibrium. As a result of historical and current anthropogenic input from diffuse sources the direct measurement of natural background concentrations is not possible in the European environment.

3.1.4.1.2 Baseline background concentration

The baseline background concentration is the concentration of an element corresponding to very low anthropogenic pressure (i.e. close to the natural background concentration but not identical). A good example of baseline background concentrations in Europe is the high quality data in the Forum of European Geological Survey´s Directors (FOREGS; http://www.gsf.fi/foregs/geochem/)) database, which include measurements performed in 26 European countries using standardised methods for sampling and analysis.

3.1.4.1.3 Ambient concentration

The ambient concentration is the sum of the natural background concentration of an element and diffuse anthropogenic input in the past or present (i.e. influence of point sources not included).

3.1.4.1.4 Realistic worst case ambient concentration

A realistic worst case (RWC) ambient concentration (representing total measured) to be used in the regional risk characterisation will be calculated as follows (which is in accordance with the methods used in the risk assessments for other metals):

RWC-ambient $PEC_{country}$: median value of all 90th percentiles that have been derived for different sites, rivers/catchments or regions within this country.

In case the amount of data for a country is to limited for deriving site/river/catchment-specific 90P-values, the 90th percentile of all its data is considered as the RWC-ambient PEC. For countries with only one ambient data point, that single value will be used as an initial estimate for the RWC-ambient $PEC_{country}$. Measured values coming from polluted sites or mining areas are not included for the derivation of a regional RWC-ambient PEC, since those values are influenced by point sources and therefore not representative for ambient (i.e., diffuse) contamination on a regional scale.

RWC ambient country-specific data are only available for a limited number of EU-countries, as compared to the FOREGS database which includes measurements from the majority of the EU countries + Norway. However, since the FOREGS-database is made up of measurements corresponding to low anthropogenic pressure, the use of country-specific measurements from the FOREGS database for those countries lacking RWC ambient data may result in an underestimation of the environmental concentrations intended to represent $PEC_{regional}$.

In order to determine whether or not country-specific FOREGS 90P are suitable to be used for those countries lacking RWC-ambient $PEC_{country}$ data, a comparison between available RWC-ambient $PEC_{country}$ and the corresponding country-specific FOREGS 90P value were performed below for freshwater, freshwater sediment and soil (data are presented in more detail in sections 3.1.4.4.2 and 3.1.4.5.2).

Freshwater

As can be seen from Table3-51 below, the ratios between the RWC-ambient $PEC_{country}$ concentrations and their corresponding country-specific FOREGS 90P values, range from about two to eighteen. This means that using the FOREGS 90P for these countries would have resulted in lower concentrations in all cases.

Table3-51 Concentration ratios of RWC-ambient PECcountry and country-specific FOREGS 90P for freshwater

Compartment	Concentration ratios RWC-ambient PECcountry / country specific FOREGS 90P
Freshwater	1.5, 1.9, 2.1, 3.2, 5.3, 6.7, 8.9, 18

An RWC-ambient concentration for freshwater only using country-specific RWC ambient data results in a median concentration of 0.72 µg Sb/l (total measured), while including country-specific FOREGS 90P data for those countries lacking country-specific RWC ambient data results in a median concentration of 0.26 µg Sb/l (only using country-specific FOREGS 90P-data results in a median concentration of 0.19 µg Sb/l).

The combination of the fact that

i) the country-specific FOREGS 90P data was a factor of two to eighteen lower as compared to the corresponding country-specific RWC ambient data

ii) the inclusion of country-specific FOREGS 90P data (for those countries lacking RWC ambient data) lowered the RWC-ambient concentration (from 0.72 µg Sb/l to 0.26 µg Sb/l)

result in the conclusion that the RWC-ambient concentration for freshwater will be based on country-specific RWC ambient data only.

Freshwater sediment

As can be seen from Table3-52 below, the ratios between the RWC-ambient $PEC_{country}$ concentrations and their corresponding country-specific FOREGS 90P values, range from just about one to eight. This means that using the FOREGS 90P for these countries would have resulted in lower concentrations in all cases except one (ratio = 0.98), i.e. in 80% of the cases.

Table3-52 Concentration ratios of RWC-ambient PECcountry and country-specific FOREGS 90P for freshwater sediment

Compartment	Concentration ratios RWC-ambient PECcountry / country specific FOREGS 90P
Freshwater sediment	1.0, 1.2, 2.7, 7.0, 8.4

An RWC-ambient concentration for freshwater sediment only using country-specific RWC ambient data results in a median concentration of 3.0 mg Sb/kg dw, while including country-specific FOREGS 90P data for those countries lacking country-specific RWC ambient data results in a median concentration of 1.72 mg Sb/kg dw (only using country-specific FOREGS 90P-data results in a median concentration of 1.38 mg Sb/kg dw).

The combination of the fact that

iii) the country-specific FOREGS 90P data was a factor of just above one to eight lower as compared to the corresponding country-specific RWC ambient data

iv) the inclusion of country-specific FOREGS 90P data (for those countries lacking RWC ambient data) lowered the RWC-ambient concentration (from 3 mg Sb/kg dw to 1.7 mg Sb/kg dw)

result in the conclusion that the RWC-ambient concentration for freshwater sediment will be based on country-specific RWC ambient data only.

Soil

As can be seen from Table3-53 below, the ratios between the RWC-ambient $PEC_{country}$ concentrations and their corresponding country-specific FOREGS 90P values, range from about point four to three. This means that using the FOREGS 90P for these countries would have resulted in both higher and lower concentrations. The reason for this may be due to that the number of data for the individual countries is very low. Two of the three countries with only one data point have a ratio below one.

Table3-53 Concentration ratios of RWC-ambient PECcountry and country-specific FOREGS 90P for freshwater sediment

Compartment	Concentration ratios RWC-ambient PECcountry / country specific FOREGS 90P

Soil	0.42, 0.50, 1.2, 1.5, 1.6, 2.6, 2.8

An RWC-ambient concentration for soil only using country-specific RWC ambient data results in a median concentration of 1.73 mg Sb/kg dw, while including country-specific FOREGS 90P data for those countries lacking country-specific RWC ambient data results in a median concentration of 1.59 mg Sb/kg dw (only using country-specific FOREGS 90P-data results in a median concentration of 1.03 mg Sb/kg dw).

Since i) the inclusion of the country-specific FOREGS 90P values did not increase the RWC-ambient concentration and ii) the RWC-ambient concentration for freshwater and freshwater sediment is calculated only using country-specific RWC-values, the same approach will be used for soil as well.

3.1.4.2 Comparison between calculated $PEC_{regional}$ and the RWC-ambient concentration

Based on the available monitoring, ambient background concentrations in the different compartments have been chosen. The reasoning behind the choice of values is given in sections 3.1.4.4- 3.1.4.8.2. The ambient concentrations, consists of the natural background (present only due to natural causes) and the emission of metal from diffuse sources of human origin (present due to anthropogenic activities, historical and present). The values are presented in Table3-54 below.

The calculated $PEC_{regional}$ are steady state concentrations based only on the estimated total emissions of antimony. However, as the time for reaching steady state is not taken into consideration and the natural background concentrations are not included, the modelled $PEC_{regional}$ do not represent the current situation. Therefore, the measured RWC-ambient concentrations will be used instead of the calculated $PEC_{regional}$ as background concentrations in the calculations of local PECs.

Table3-54 Ambient background concentrations of antimony in the environment.

Compartment	Concentration	Section in the RAR
Freshwater	0.72 μg Sb/l	3.1.4.4.2
Sediment	0.65 mg Sb/kg ww	3.1.4.4.2
Soil	1.5 mg Sb/kg ww	3.1.4.5.2
Air	2.6 ng Sb/m^3	3.1.4.6.2
Marine water	0.20 μg Sb/l	3.1.4.8.2
Marine sediment	0.65 mg Sb/kg ww	3.1.4.8.2

In Table3-55 the measured ambient antimony concentrations are compared to the regional PECs calculated by EUSES. The results of three different EUSES calculations are given:

i. The basic calculation including estimated emissions from all scenarios and lifecycle stages except for the emissions from hunting, shooting and fishing.
ii. Emissions from hunting, shooting and fishing included in the estimation of emissions from "unintentional" sources of antimony.
iii. Emissions from "unintentional" sources of antimony excluded

Table3-55 $PEC_{regional}$ compared to ambient background concentrations

EUSES modelling	$PEC_{regional}$					
	Surf. water (µg/l)	Sediment (mg/kg ww)	Marine water (µg/l)	Marine sed. (mg/kg ww)	Air (ng/m^3)	Natural soil (µg/kg ww)
Standard scenario*	0.36	0.65	0.038	0.069	0.16	4.74
Hunting, shooting and fishing included**	1.26	2.27	0.14	0.25	0.21	6.22
Unintentional sources excluded***	0.35	0.63	0.036	0.066	0.085	2.54
Measured PEC_{reg}	**0.72**	**0.65**	**0.20**	**0.65********	**2.6**	**1.5×10^3**

*Median values for partitioning coefficients used, emissions from 100% incineration

**As the basic modelling except the estimation of emissions from "unintentional" sources of antimony includes also worst case emissions of antimony from bullets and sinkers

***As the basic modelling except the emissions of antimony from unintentional sources excluded.

**** No monitoring data available. Based on freshwater sediment monitoring data

The PECs estimated by EUSES are in the same order of magnitude as the PECs based on monitoring data for the aquatic compartment. For the air compartment, the PECs estimated by EUSES are approximately one order of magnitude lower than the PECregional based on monitoring data. For the marine compartment the modelled PECs are lower than the measured regional PEC. For natural soil the correlation is poor, with the modelled PECs being a factor 200-600 lower than the measured.

The regional PEC based on measured data has been chosen because of the uncertainties in release estimations, especially for other sources than diantimony trioxide use, and because EUSES does not take the natural background into account.

3.1.4.3 Influence of bioavailability

The bioavailability of Sb is most probably different in the different compartments. However, no bioavailability corrections have been performed as any reliable information on how to proceed presently exists.

3.1.4.4 Aquatic compartment – (incl. sediment)

Antimony will enter the aquatic compartment, either as via atmospheric input, or via transport from soil via water. Depending on which form it is introduced into the water body, the time to reach the sediment may vary. Once in the sediment it may bind to hydrous oxides of Fe, Mn, or Al, or to bind to sulphur, depending on the redox situation.

3.1.4.4.1 Calculation of predicted environmental concentrations (PEC)

The ambient background concentrations used as $PEC_{regional}$ are 0.72 µg Sb/l for freshwater and 0.65 mg Sb/kg ww for freshwater sediment (see section 3.1.4.4.2).

The TGD suggests that the local concentration in the sediment is predicted based on the local concentration in water as:

$$PEC_{local\ sediment} = Kp_{suspended\ matter} * PEC_{local\ water}$$

In analogy with the risk assessment for Cadmium, an alternative approach has been used in this risk assessment. This approach use a measured regional PEC for the sediment to which a local fraction (proportional to the local concentration in water) is added. This option has been preferred because of the preference for measured rather than predicted sediment concentrations, and because the contribution of the local discharge to the sediment concentrations is taken into account via suspended matter Kp in line with the TGD. Local $PEC_{sediment}$ is then calculated according to the equation below:

$$PEC_{local}(ww) = PEC_{regional}(ww) + Kp_{SPM} * C_{localwater} \frac{Fsolid_{susp} * RHOsolid}{RHOsusp}$$

where

PECregional = 0.65 mg Sb/kg ww, which is the chosen RWC-ambient concentration for sediment based on measured data

Concentration in surface water during emission period (dissolved); $C_{local\ water}$ = From EUSES [mg/kg]

Fraction solid in suspended matter; $Fsolid_{susp}$ = 0.1 [m_{solid}^3 x m_{susp}^{-3}]; default value from TGD

Bulk density of solid phase; RHOsolid = 2500 [kg_{solid} x m_{solid}^{-3}]; default value from TGD

Bulk density of (wet) suspended matter; RHOsusp = 1150 [kg x m^{-3}]; default value from TGD

Partition coefficient suspended matter; $Kp_{SPM\ median} = 10^{3.65}$ [l/kg]; $Kp_{SPM\ 10P} = 10^{3.1}$ [l/kg]; $Kp_{SPM\ 90P} = 10^{4.2}$[l/kg] ; see 3.1.3.1.4

$\frac{Fsolid_{susp} * RHOsolid}{RHOsusp}$ *converts the local fraction added from dry weight to wet weight*

As a sensitivity analysis, the PEC for local sediment has been calculated for three values of Kp suspended matter (see section 3.1.3.1.4), the median, the 10P and the 90P of available Kp_{SPM} data. As the selection of value of Kp_{SPM} influences the local concentration in water, a new calculation of the local concentration in water has been made for each Kp_{SPM}, and the values pertinent for the relevant Kp_{SPM} has been used in the calculation of PEC for local sediment.

The PEC for water, sediment and STP are summarised in Table3-56 below.

Table3-56. Summary of PECs for water, sediment and STP.

Scenario	**$PEC_{local,\ water}$ (µg Sb/l) - dissolved**	**$PEC_{local,\ sediment}$ (mg Sb/kg ww)**			**PEC_{STP} (µg Sb/l) (effluent concentration)**
		Kp_{spm} 10P	**Kp_{spm} median**	*Kp_{spm} 90P*	
Production site, P1	39.5	*11.76*	38.33	*115.73*	not relevant*

Scenario	$PEC_{local, water}$ (µg Sb/l) - dissolved	$PEC_{local, sediment}$ (mg Sb/kg ww)			PEC_{STP} (µg Sb/l) (effluent concentration)
Production site, P5	0.88	*0.70*	0.80	*1.12*	1.7
Use as flame-retardant in plastics and rubber - formulation	4.7	*1.79*	4.51	*12.47*	42.5
Use as flame-retardant in plastics and rubber – formulation, assuming release to air	0.72	*0.65*	0.65	*0.65*	not relevant*
Use as flame-retardant in plastics and rubber – industrial use	2.1	*1.05*	2.01	*4.78*	14.9
Use as flame-retardant in textiles – formulation, generic site	28.5	*4.65*	27.65	*42.0*	148
Use as flame-retardant in textiles – formulation, site AMI-19U	4.7	*1.79*	4.52	*12.50*	90
Use as flame-retardant in textiles – formulation/processing, site FT-2	2.2	*1.07*	2.06	*4.96*	15.5
Use as flame-retardant in textiles – formulation, site 96	3.1	*1.32*	2.92	*7.61*	25
Use as flame-retardant in textiles – industrial use (application to textiles), generic site	69.4	*20.35*	67.36	*204.62*	733
Use as flame-retardant in textiles – industrial use (application to textiles), site RC74U	2.4	*1.13*	2.26	*5.58*	17.7
Use as catalyst (PET industry) -polymerisation	4.0	*1.59*	3.84	*10.40*	35
Use as catalyst (PET industry) – industrial use (shaping)	0.75	*0.67*	0.68	*0.83*	not relevant*
Use in paint as flame-retardant - formulation	1.32	*0.82*	1.23	*2.43*	3.2
Use in paint , pigment production - formulation	1.47	*0.87*	1.38	*2.89*	4.0
Use in paint – formulation, site AMI3G	3.2	*1.36*	3.05	*7.99*	100
Use in glass – formulation	0.72	*0.65*	0.65	*0.65*	not relevant*
Use in glass – industrial use ProcGlass 2	3.07	*1.32*	2.93	*7.64*	not relevant*
Use in glass – industrial use ProcGlass 3	0.90	*0.70*	0.83	*1.19*	not relevant*
Regional (based on monitoring data)	0.72	*0.65*	0.65	*0.65*	-

* Either no emissions to waste water, not connected to STP or emission figure representing emissions after sewage treatment

The PECs calculated with the median value of Kp suspended matter has been chosen for the risk characterisation. The median value has been chosen rather than the 90-percentile to avoid using worst-case values for several independent parameters influencing the same PEC, as this could potentially give a too extreme worst case. In this case the median of the 90-percentile of monitored environmental concentrations of antimony in sediment has been selected as a regional background PEC, and therefore the median value of $Kp_{suspended\ matter}$ (which also has a major impact on the local PECs for sediments, see resulting PECs in Table3-56 above) has been used for the risk characterisation.

3.1.4.4.2 Measured levels

Levels in surface water

Baseline background concentrations of antimony in stream waters measured in the FOREGS-project results in values that range about three orders of magnitude from < 0.002 to 1.21 µg Sb/l (excluding two outliers up to 2.91 µg Sb/l), with a median value of 0.07 µg Sb/l (see Table3-57 and Figure 3-6).

Table3-57 Concentrations of Sb in European stream water according to FOREGS http://www.gsf.fi/publ/foregsatlas/article.php?id=15).

Unit	Number of samples	Min	Median	Mean ± SD	90P	Max
µg Sb/l	807	<0.002	0.07	0.109 ± 0.177	0.21	2.91

DRAFT

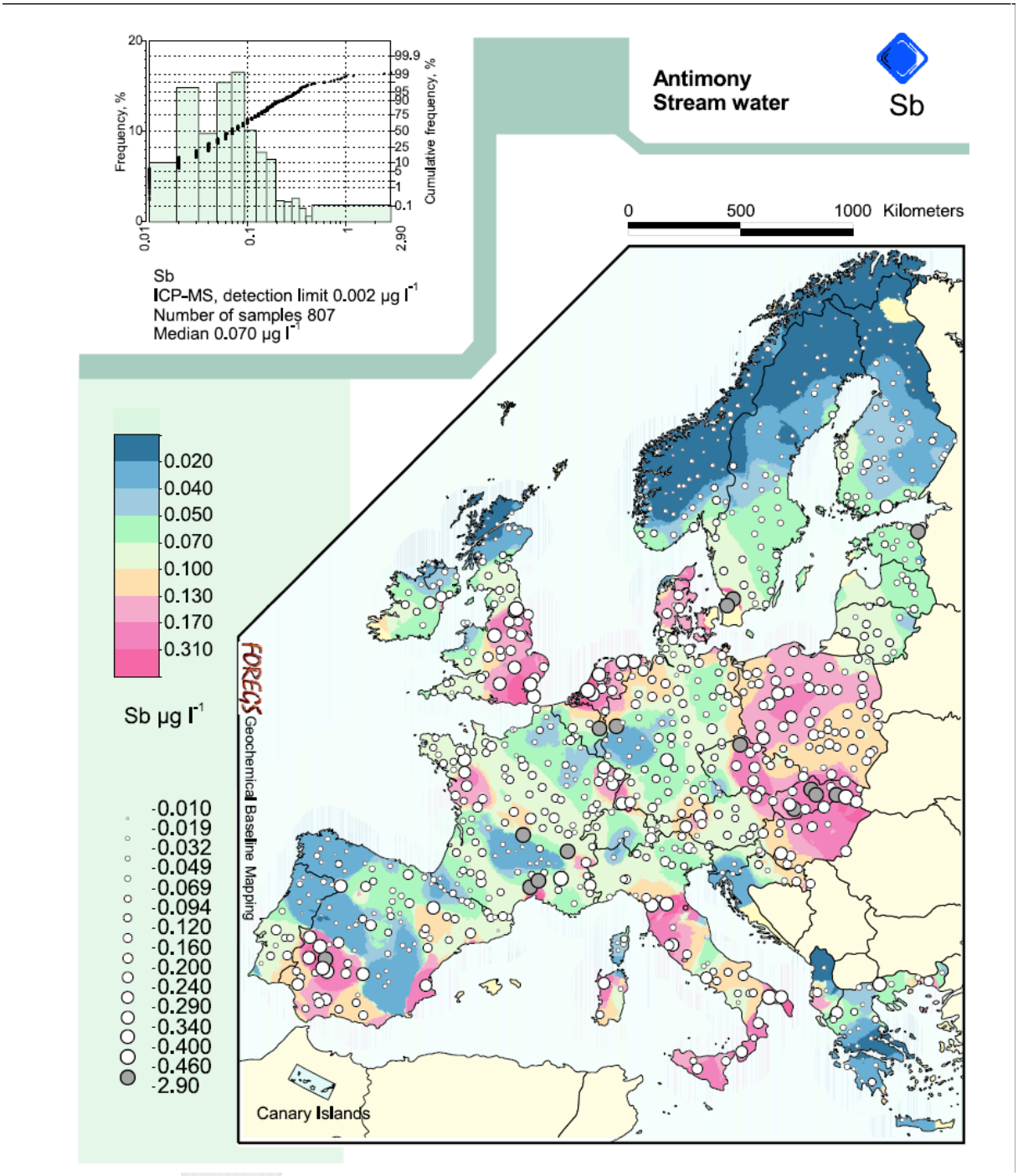

Figure 3-6 FOREG map showing measured concentrations of Sb in European stream water (http://www.gsf.fi/publ/foregsatlas/maps/Water/w_icpms_sb_edit.pdf).

Measured antimony concentrations in rivers, lakes, ground water and precipitation from other sources besides the FOREGs database are presented in Table3-59, Table3-60, Table3-61 and Table3-62.

In the lower end of the concentration range are the concentration measured in Swedish lakes (Naturvårdsverket, 1999) with a median value of 0.035 µg Sb/l, and in the French lake Pavin where a concentration of 0.032 µg Sb/l has been measured. In the upper end of this concentration range is the Alzillelo River on Corsica, France, with a concentration of about

3.0 µg Sb/l. Higher concentrations, up to 400 µg Sb/l, have been measured in association to industrial activities, especially mining.

The concentrations measured in snow and rainwater varies widely depending on the location of sampling, from 0.06 µg Sb/l (median value of precipitation during one year) measured in the west of Sweden, to 36 µg Sb/l, measured in snow in the city of Warsaw, Poland. Stuyfzand (1992) proposed that antimony contaminated rainwater probably was the most important antimony source for shallow groundwater.

Calculation of RWC-ambient PEC

A RWC-ambient PEC of 0.63 µg Sb/l is calculated using ambient data from eight countries (see Table3-58). The country specific FOREGS 90P values range from similar (Sweden) to about 18 times smaller (France) as compared to measured ambient concentrations.

Table3-58 Calculation of RWC-ambient PEC for antimony in freshwater. Country specific 90P FOREGS data are included for comparative purposes only.

Country	RWC-ambient PEC (90P) µg Sb/l	Data used	FOREGS -90P µg Sb/l
Austria			0.25
Belgium			0.66
Bulgaria			-
Cyprus			-
Czech Republic			0.26
Denmark			0.18
Estonia			0.10
Finland			0.07
France	3.3	1.5, 0.34, 0.49, 0.33, 3, 1, 4, 0.032	0.18
Germany	1.52	0.05, 0.06, 0.10, 1.0, 5.2, 2.3, 0.23, 0.08, 0.23, 0.31, 0.53, 0.31, 0.00037, 0.10, 0.17, 0.12, 0.11	0.17
Greece			0.15
Hungary			0.24
Ireland			0.20
Italy			0.27
Latvia			0.08
Lithuania			0.10
Luxembourg			-
Malta			-
Netherlands	0.83	0.85, 0.8, 0.6, 0.37, 0.48	0.39
Poland	0.60	0.60	0.19
Portugal			0.14
Romania			-
Slovakia			0.75

Slovenia			-
Spain	0.53	0.54, 0.39	0.28
Sweden	0.13	0.05, 0.07, 0.08, 0.018 (= 0.0365/2)*, 0.14, 0.13, 0.063	0.09
United Kingdom	2.0	0.42, 0.13, 0.27, 0.35, 1.9, 2, 2.1, 0.08, 0.16, 0.17, 0.15	0.30
Norway	0.32	0.35, 0.032	0.06
Median	0.72		0.19** 0.26***

*Half detection limit

**Median of FOREGS country-specific 90P values

***Median value of RWC values when available and country-specific FOREGS 90P-values for those countries with no RWC-value

In conclusion, the RWC-ambient background concentration of antimony in freshwater (dissolved) used in this RAR is 0.72 µg Sb/l.

Table3-59 Measured antimony concentrations in European rivers.

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
France Rhône River (Avignon)	1.5 (16% standard dev. – pooled estimate from dupl. determin.)		1966, June	Precipitation (sulphide) + NAA	Kharkar et al., 1968
Rhône River (Arles) Isère (Domène)	0.34 0.33		05-09-81 02-09-81	HG-AAS	Andreae and Froelich, 1984
Rhône River	0.49			NAA	Guieu et al., 1993
CORSICA Alzillelo River Bravona River	3.0 1.0–41		1990, winter, spring, summer	HG-AAS Unpolluted Upstream-downstream confluence with Presa River Upstream-	Mori et al., 1999

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Presa River	4.0-385			Downstream Realgar mine	
Germany					
Iller kanal	0.050 (±10%)			NAA	Schramel et al., 1973
Donau (Ulm)	0.060 (±10%)				
Donau (Böfingerhalde)	0.10 (±10%)				
Lech (Augsburg)	1.0 (±10%)				
Lech kanal (Gersthofen)	5.2 (±10%)				
Lech kanal (Langweid)	2.3 (±10%)				
Rhine River (Oppenheim)	0.0004	Sb(III)		HG-AAS	Andreae et al., 1981
	0.23	Sb(V)			
	0.0001.2	MSA			
	n.d.	DMSA			
Main River (Frankfurt)	0.0003	Sb(III)			
	0.31	Sb(V)			
	0.00018	MSA			
	n.d.	DMSA			
Rhine (Bregenz)	0.080		12-05-82	HG-AAS	Andreae and Froelich, 1984
Rhine (Oppenheim)	0.23		02-04-82		
Main (Frankfurt)	0.53		06-12-82		
Danube (Neuberg)	0.31		10-05-82		
Elbe (below Hamburg)	<0.00037		09-06-81		
Netherlands					
Rhine	0.85		1975, February	HG-NAA	Van der Sloot et al., 1985
Rhine River (Lobith)	0.80			HG-AAS	Haring et al., 1982
Rhine River (Gorkum)	0.60				
Norway					
11 rivers	0.35 (weighted mean)		1971, May-October	NAA	Salbu et al., 1979

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Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
	<0.16-2.1 (min-max; individual obs.)				
Poland Vistula River	0.60 ± 0.040			HMDE-DPASV Surface water (non-filtered)	Postupolski and Golimowski, 1991
Portugal Tejo River (Santarem) Tejo River (Vilafranca)	0.10 0.23		30-04-82 06-12-82	HG-AAS	Andreae, 1983
Spain Guadalquivir (Coria)	0.54		18-12-82	HG-AAS	Andreae, 1983
Manzanares River	n.d. 0.39	Sb(III) Sb(V)		HPLC-HG-ICP-AAS	Smichowski et al., 1995
Sweden Skellefte River Ume River Ångerman River	0.050 0.070 0.080	Unfiltered	9-11-64 23-1-64 17-5-64	NAA	Landström and Wenner, 1965
Creek in Gideå, north	<0.0365		1990, September	ICP-MS Measurements were performed on ^{125}Sb	Carbol et al., 1995
United Kingdom Gowny River Cowny River	0.42 0.13			Precipitation (zirconium oxide) + colorimetry (ion pair $SbCl_6$ with crystal violet)	Abu-Hilal and Riley, 1981
Thames River	0.27 (±57%) 0.086-0.86 (min-max)		23-10-78 – 17-08-79	NAA 3 stations	Habib and Minski, 1982
Thames River (downstream of Oxford)	0.39 0.35	Total Dissolved	11-04-97 – 01-11-98	ICP-MS Weekly spot sampling	Neal et al., 2000
Glenshanna River (Scotland)	5.3 n.d.	Total Sb(III)		HG-AAS Disussed mine entrance	Mohammad et al., 1990

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
	5.3	Sb(V)			
	60	Total		Soil heap drainage	
	2.2	Sb(III)			
	58	Sb(V)			
	14	Total		River (2 m downstream soil heap)	
	n.d.	Sb(III)			
	14	Sb(V)			
			1996, February, April, July, and October	ICP-OES and ICP-MS	Jarvie et al., 2000
Trent River (Cromwell Lock)	1.9			Non-tidal	
Trent River (Torksey)	2.0			Freshwater tidal	
Trent River (Gainsborough	2.1			Freshwater tidal	

Table3-60. Measured antimony concentrations in European lakes.

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
France					
Lake Pavin			1990, August	HG-AAS	Takayanagi and Cossa, 1997
				Small meromictic lake	
	0.032	Total		Surface	
	0.016	Sb(III)			
	0.016	Sb(V)			
	0.043	Total		Oxic (deep)	
	n.d.	Sb(III)			
	0.043	Sb(V)			
	<0.0073	Total		Anoxic	
Germany					
Lake Constance	0.10-0.17		1986-1987	HG-AAS	Stabel et al., 1991
Lake Constance	0.12	Constant with depth (mean 9,	03-04-90		Sinemus et al., 1992

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
		samples 0-140m)		FIA-HG-GF-AAS	
	0.11 0.12 0.11 0.11	surface 20 m 60 m 140 m	14-11-90		
Bitterfeld Pfälzer Bergland	31 2.0 28 96 9.0 85	Total Sb(III) Sb(V) Total Sb(III) Sb(V)		IC-ICP-AES Polluted by industrial waste Polluted by mining processes	Ulrich, 1998
Netherlands Lake Yssel	0.37-0.48		1978, April	HG-NAA 3 stations	Van der Sloot et al., 1985
Norway 41 lakes	0.032 (mean)		1977	NAA	Allen and Steinnes, 1979
Sweden South North	 0.14 0.13		 1980, March-April 1980, March-April	In situ dialysis + NAA 20 small acidic lakes, 2.5 m depth 18 small lakes, samples 2 m below the ice	Borg, 1983
242 lakes	0.035 (median) 0.010-0.063 (10-90 perc.)				Naturvårdsverket, 1999b
United Kingdom Lake Celyn	0.080			Precipitation (zirconium oxide) + colorimetry (ion pair $SbCl_6$ with crystal violet)	Abu-Hilal and Riley, 198
Loch Ewe (NW Scotland)	0.16 0.003	Total Sb(III)		HG-AAS Spring diatom bloom, no temporal variation	Apte et al., 1986

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Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Loch Ewe (NW Scotland)	0.17 0.002 0.15 0.003	Total Sb(III) Total Sb(III)	26-03-83 09-04-83	HG-AAS	Apte and Howard, 1986

Table3-61. Measured antimony concentrations in European ground waters.

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Sweden				NAA	Landström and Wenner, 1965
Near Skellefteå	0.010		11-8-64	2 locations	
	0.020		7-9-64		
Brån	0.030		23-1-64	2 locations	
	0.20		23-1-64		
Sollefteå	0.040		17-5-64	3 locations	
	0.060		17-5-64		
	0.50		17-5-64		
Poland					
Warsaw	0.13			HMDE-DPASV Surface water (non-filtered)	Postupolski and Golimowski, 1991

Table3-62. Measured antimony concentrations in European snow and rain water.

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Netherlands	0.89			Rain	Stuyfzand, 1991
Poland				HMDE-DPASV	Postupolski and Golimowski, 1991
Warsaw (City)	36	Sb(III)		Snow	
Warsaw (suburb)	3.0	Sb(III)			
	5.1	Total			

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Szczyrk (S Poland)	0.030	Total			
Sweden Gårdsjön	0.066±0.026 (mean±stdev) 0.060 (median) 0.044-0.140 (min-max)		990104-000107	ICP-EAS/ICP-SMS Precipitation during one year, monthly sampling	Eriksson, 2001
Arup (South of Sweden) Gårdsjön (West) Bredkälen (North) Mjölsta (East)	0.12, 0.095, 0.11 0.13, 0.21, 0.14 0.041, 0.036, 0.067 <d.l., 0.14, 0.098 0.011, 0.011, <d.l. <d.l., 0.032, 0.034 0.17, 0.17, 0.12		010701-010801 011201-020101 010710-010802 011127-020111 010703-010731 011204-020104 011127-020114	ICP-MS detection limit 10 ng Sb/l	Sternbeck et al., 2002a

Levels in sediment

Baseline background concentrations of antimony in stream water sediments measured in the FOREGS-project results in values that range over three orders of magnitude from < 0.02 to 34.1 mg Sb/kg dw, with a median value of 0.615 mg Sb/kg dw (see Table3-63 and Figure 3-7).

Table3-63 Concentrations of Sb in European stream sediment according to FOREGS (http://www.gsf.fi/publ/foregsatlas/article.php?id=15).

Unit	Number of samples	Min	Median	Mean ± SD	90P	Max
mg Sb/kg dw	848	<0.02	0.615	1.07 ± 1.88	2.10	34.1

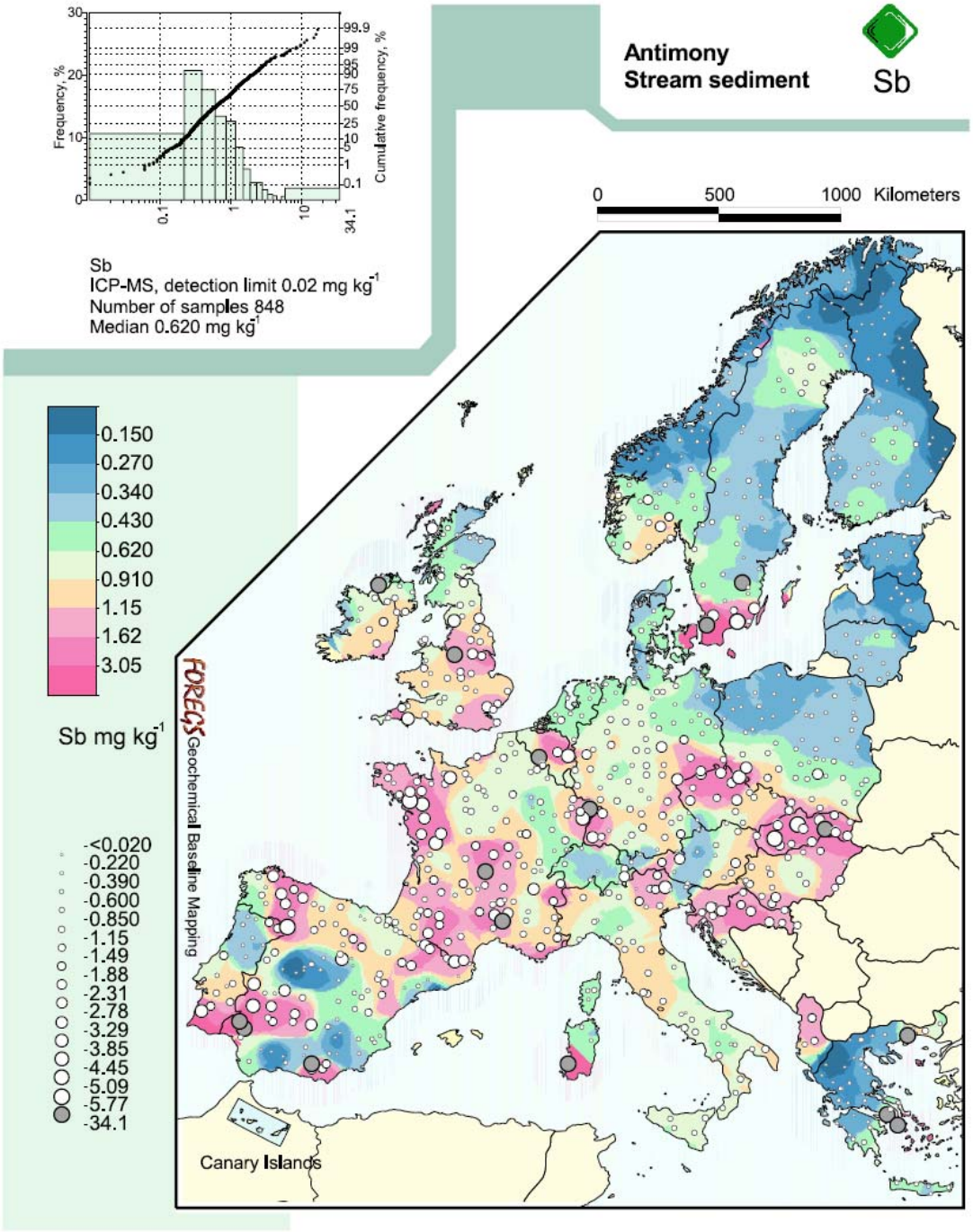

Figure 3-7 FOREG map showing measured concentrations of Sb (mg/kg dw) in European stream sediment (http://www.gsf.fi/publ/foregsatlas/maps/StreamSed/s_icpms_sb_edit.pdf).

Measured antimony concentrations in sediments from other sources besides the FOREGs database are presented in Table3-65 below.

In the lower end of concentrations measured are concentrations measured in Norwegian freshwater lakes 0.07-3 mg Sb/kg dw (10P-90P) (Rognerud and Fjeld, 2001), and in the upper end contaminated sediments of the River Rhine 7.95-96.1 mg Sb/kg dw (min-max) (Dissanayake et al., 1983).

Calculation of RWC-ambient PEC

A RWC-ambient PEC of 3 mg Sb/kg dw (0.65 mg/kg ww) in freshwater sediments is calculated using ambient data from five countries (see Table3-64). The country-specific FOREGS 90P values range from similar in size (Italy and UK) to eight times smaller (Austria) as compared to measured ambient concentrations.

Table3-64 Calculation of RWC-ambient PEC for antimony in freshwater sediments. Country specific 90P FOREGS data are included for comparative purposes only.

Country	RWC-ambient PEC (90P) mg Sb/kg dw	Data used	FOREGS -90P mg Sb/ dw
Austria	15.1	9.62, 10.01, 12.21, 12.01, 13.37, 16.83	1.80
Belgium			7.28
Bulgaria			-
Cyprus			-
Czech Republic			2.88
Denmark			0.64
Estonia			0.45
Finland			0.52
France			2.78
Germany	9.7	0.9, 0.4, 11.9	1.39
Greece			0.73
Hungary			1.34
Ireland			1.72
Italy	1.6	0.2, 0.14, 0.74, 2.03, 1.02	1.35
Latvia			0.39
Lithuania			0.54
Luxembourg			-
Malta			-
Netherlands			0.87
Poland			0.90
Portugal			2.21
Romania			
Slovakia			5.55
Slovenia			2.46
Spain			2.86
Sweden			1.38
United Kingdom	2.3	0.92, 0.71, 0.7, 2.9	2.36
Norway	3	3	1.12

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Median	3		1.38* 1.72**

***Median of FOREGS country-specific 90P values**

****Median value of RWC values when available and country-specific FOREGS 90P-values for those countries with no RWC-value**

In conclusion, the RWC-ambient background concentration of antimony in freshwater sediments used in this RAR is 3 mg Sb/kg dw (0.65 mg Sb/kg ww).

Table3-65. Measured antimony concentrations in European sediments.

Location	Concentration (Sb mg/kg dw)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Austria Danube River and Danube Canal (Vienna)	9.62, 10.01, 12.21, 12.01, 13.37, 16.83 (medians from the stations) 8.22-22.22 (range)		1973, August – 1974, March	NAA 6 stations, no significant variation over time	Rehwoldt et al., 1975
Germany Markt RedwitzFaulschlamm (Großlappen) Bodenschlamm (Inn) Isarwerkkanal (München) Bodenschlamm (Zulauf-Speicher)	0.9 (±10%) 46.5 (±10%) 36.5 (±10%) 4.1 (±10%) 1.5 (±10%) 6.0 (±10%)			NAA Upstream large chemical industry producing Hg Downstream ditto Samples were selected in such a way that contamination could be expected	Schramel et al., 1973
Rhine (Ginsheimer Altrhein)	7.95-96.1 (min-max)			NAA 5 stations, 0-15 cm Polluted	Dissanayake et al., 1983
Rhine	3 38 1	Sb(III) Sb(V) TMSbO		ICP-OES "surface sediments", polluted	Hirner et al., 1990
River in West Germany	0.2-9.8 (min-max) 0.1-1.2 0.1-0.9	$SbMeH_2$ $SbMe_2H$ $SbMe_3$		LTGC-ICP-MS	Krupp et al., 1996
Italy Arno River Venice Lagoon	0.2 0.14-0.74			HG-AFS	Barghigiani et al., 1995

Location	Concentration (Sb mg/kg dw)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Lagoon Sacca di Goro Po River delta	2.03 1.02		1992, April – 1992, June	NAA Heavy industrialised coast line High water renewal Slow water renewal	Bondavalli et al., 1996
Norway Norwegian freshwater lakes	0.29 (median) 0.07-3 (10P-90P) 0.04 (median) 0.009-0.14 (10P-90P)		1996-1997	ICP-MS 210 lakes, no significant local source of pollution Upper 0.5 cm 30-50 cm depth	Rognerud and Fjeld, 2001
United Kingdom Southampton Water Town Quay Woolston Hamble Calshot Mersey River	 0.92 2.3 0.88 0.71 0.67 0.7 2.9			NAA Surface 10 cm 15 cm Surface 10 cm Surface Surface	Leatherland and Burton, 1974

WWTP levels

Measured antimony concentrations in STP-sludge are presented in Table3-66 below.

The levels of antimony in 48 municipal wastewater treatment plants have been reported by Eriksson (2001). The levels found were in the range 0.6 mg Sb/kg_{dw} to 18 mg Sb/kg_{dw}, with the median level being 1.3 mg Sb/kg_{dw}, and the 90P 3.4 mg Sb/kg dw. The samples were collected spring-early summer 2000. The sample with the highest concentration of antimony was collected from wastewater treatment plants (Borås) that had possible contribution from the textile industry.

Sternbeck et al. (2002a) studied 4 municipal wastewater treatment plants that differed substantially as regards type of loading. The emission of antimony calculated as person equivalents (pe) was rather constant for the three WWTP with the lowest concentrations,

ranging from 0.039-0.078 g Sb/pe and year (equivalent to 0.57-3.1 mg Sb/kg dw) while it was substantially increased, 1.2 g Sb/pe and year (16 mg Sb/kg dw) for the WWTP connected to the textile industries.

Table3-66 Measured antimony concentrations in European wastewater treatment plants.

Location	Concentration (mg Sb/kg dw)	Period	Remark	Reference
Sweden	0.6-18 (mean ± S.D.: 2.4 ± 3.0) (median: 1.3) (90-perc.:3.4	2000, spring-summer	ICP-QMS Values from 48 WWTPs spread across the country.	Eriksson, 2001
Sweden	16 3.1 1.7 0.57	2001	ICP-MS Values from 4 WWTP with different loadings Household, textile industries, graphical Household, surface water, various industries industry, surface water Household, laundry Only household	Sternbeck et al., 2002a

Local levels

Not much data on measured local concentrations of antimony in the aquatic compartment near sites using diantimony trioxide is available. The data available is presented below.

Production

The concentration of antimony and other metals in the sediment of the water stream into which releases from production site P-1 will enter, has been investigated. (EURAS, 2008) The sediment was sampled over a distance of 70 m. Throughout the 70 m long trajectory; 70 sediment samples have been taken by means of a piston sampler. The top and bottom 3 cm of each 'piston-like' sample were investigated. The sediment was on average 10-20 cm thick, dependent on the location. The 70 samples of the top layer (top 3 cm) were mixed to one 'mixture sample'. The samples of the bottom layer (bottom 3 cm) of the sediment were also mixed to one sample. Two samples were analysed. The first sample contains the top 3 cm of the sediment. The second sample contains the bottom 3 cm of the sediment. Both samples are analysed for eight elements (As, Cr, Cu, Cd, Hg, Pb, Ni, Zn) and additionally on antimony. The content of antimony was 130 mg Sb/ kg dw in the top 3 cm sample and 120 mg Sb/ kg dw in the bottom sample. Conversion to wet weight using the reported percentage of dry matter (measured to 76.4% in the top sample and 78.1% in the bottom sample) gives 99 mg Sb/ kg ww and 94 mg Sb/ kg ww respectively.

Earlier investigations showed significantly higher levels of antimony in the sediment, 2 400 mg Sb/kg dw in samples from July 1998, and 323 mg Sb/kg dw in samples from January 2003. The sampling in these investigations was somewhat different, but it is clear that the levels of antimony in the sediment have been significantly reduced.

The company has reduced emissions through installation of a filter and other measures. In recent years, the company made some efforts in reducing the waste water volume, through reuse and recycling of water. Treated waste water is only emitted to surface water in case of heavy rainfall.

For the receiving aquatic compartment only P-4 provides measured data from 1998 (EURAS bvba, 2003). The concentration of Sb in water and sediment were measured in one location 100 m upstream and in one location 100 m downstream from the plant. The Sb concentrations (dissolved) reported in water were 5 µg Sb/l and 10 µg Sb/l, respectively. In sediment the concentrations 100m upstream and 100 m downstream were 22 mg Sb/kg and 206 mg Sb/kg, respectively. It is unclear whether the concentrations given are based on dw or ww. Assuming dry weight concentrations the corresponding ww concentration would be approx 45 mg/kg ww. The site has been used for mining exploitation, metallurgical treatment and as a slag dump for 100 years why the concentrations measured downstream do not only stem from emissions from the present activities.

Textile industries

Antimony has several known use areas within the textile industry. It has been noticed that certain WWTPs in Borås (a city situated in a region in Sweden with a large textile industry) has high concentrations of antimony in sludge. Measurements performed by Eriksson (2001) showed that the concentration of antimony in sludge from Gässlösa WWTP (located in Borås) was the highest of the 48 different WWTP tested in Sweden. The reason for this is believed that process water from textile industry is connected to this WWTP. Two of these industries measure Sb in their process water, and the concentrations in these waters normally fall below 100 µg/l.

Measurements were also performed on process water from a laundry, and the WWTP (Rimbo) to which it is connected.

The concentrations of antimony in the inflow and outflow of Rimbo WWTP cannot directly be compared since the residence time within the WWTP is about a week. As for the comparison between the inflow and outflow from Gässlösa WWTP, where the concentration of antimony was higher in the outflow as compared to the inflow, the authors had no explanation (several alternative explanation was presented).

Table3-67 Measured local concentrations in water and sludge at a WWTP (Gässlösa) which serves textile industries, the River Viskan to which Gässlösa WWTP releases its outflow process water, at a laundry and to the WWTP (Rimbo) to which the laundry is connected. (Sternbeck, Palm and Kaj, 2002).

Site	Concentration	Sample	Remark	Date
WWTP (Gässlösa)	16.0 mg Sb/kg dw	Sludge		Oct.1-12, 2001
	3.9 µg Sb/l	Water	Inflow, weekly sample, very muddy	Oct. 1-7, 2001
	2.9 µg Sb/l	Water	Inflow, weekly sample, filtered	Oct. 1-7, 2001
	5.7 µg Sb/l	Water	Outflow, unfiltered	Oct. 1-7, 2001

Site	Concentration	Sample	Remark	Date
	5.04 μg Sb/l	Water	Outflow, filtered	Oct. 1-7, 2001
	1.79 μg Sb/l	Water	Inflow, weekly sample, very muddy	Oct. 8-14, 2001
	6.09 μg Sb/l	Water	Outflow	Oct. 8-14, 2001
River Viskan	0.11 μg Sb/l	Water	Upstream 1 km of Gässlösa WWTP outflow, surface water	Oct. 11, 2001
	0.79 μg Sb/l	Water	Downstream 0.5 km of Gässlösa WWTP outflow, surface water	Oct. 11, 2001
WWTP (Rimbo)	1.74 mg Sb/kg dw	Sludge		Oct. 22-28, 2001
	0.16 μg Sb/l	Water	Inflow, weekly sample, very muddy	Oct. 22-28, 2001
	0.26 μg Sb/l	Water	Outflow	Oct. 22-28, 2001
	0.1 μg Sb/l	Water	Inflow, weekly sample, very muddy	Nov. 12-18, 2001
	0.22 μg Sb/l	Water	Outflow	Nov. 12-18, 2001
Laundry (Rimbo)	2.01 μg Sb/l	Water	Processing water	Oct. 3, 2001
	2.37 μg Sb/l	Water	Processing water	Oct. 3, 2001

Glass production (processing)

A glasswork, ProcGlass-2, located in Sweden uses about 50% of the produced glass raw material in the country. Emissions to air and water are measured on a regularly basis. In addition, measurements on antimony in water moss downstream of the glasswork are also regularly measured. Another site, ProcGlass-3 uses less than 10 t antimony trioxide per year. The levels of antimony and other elements in a small stream near the site are measured and reported. The water flow in the stream is low.

Water samples for ProcGlass-2 were taken on the same day (Sep. 5, 2001) at five different locations. Samples of sediments were taken at four different locations. Results are shown in Table3-68.

Table3-68 Measured local concentrations in water and sediment at and nearby producers of artistic glass in Sweden.

Compartment	Concentration	Location/year	Reference
ProcGlass-2			(Sternbeck, Palm and Kaj, 2002).
Water	3.01 μg Sb/l	Outflow of day water and cooling water within the factory area	
	4.63 μg Sb/l	Outside of the factory area	
	0.10 μg Sb/l	Pond, close upstream	
	3.08 μg Sb/l	Downstream (Riveberg)	
	3.55 μg Sb/l	Downstream (Flygsfors)	

Compartment	Concentration	Location/year	Reference
Sediment	68.6 mg/kg dw	Downstream in creek, 0-2.5 cm	
	65.8 mg/kg dw	Downstream in creek, 7.5-10 cm	
	1.79 mg/kg dw	Pond, close upstream, 0-2.5 cm	
	3.79 mg/kg dw	Pond, close upstream, 7.5-10 cm	
	17.5 mg/kg dw	Pond, downstream, 0-2.5 cm (Flygsfors)	
	4.11 mg/kg dw	Pond, downstream, 6-11 cm (Flygsfors)	
	12.9 mg/kg dw	Pond, downstream, 0-2.5 cm (a bit upstream Flygsfors)	
ProcGlass-3			
Water, small stream near site	13 µg Sb/l	yearly average 1995	Davidsson, 2007
	10 µg Sb/l	yearly average 1996	
	9.8 µg Sb/l	yearly average 1997	
	7.8 µg Sb/l	yearly average 1998	
	6.2 µg Sb/l	yearly average 1999	
	5.4 µg Sb/l	yearly average 2000	
	3.3 µg Sb/l	yearly average 2001	
	5.1 µg Sb/l	yearly average 2002	
	0.6 µg Sb/l	yearly average 2003	
	4 µg Sb/l	yearly average 2004	
	3.8 µg Sb/l	yearly average 2005	
	4.2 µg Sb/l	yearly average 2006 (based on measurements up to October 2006)	

Comparison between predicted and measured levels

Except for production of antimony, textile industry and glass production there are no measured data relevant for comparison with the predicted local PECs. The predicted local concentrations of antimony in surface water range from background concentrations (0.72 µg Sb/l) up to 69 µg Sb/l.

The highest predicted local concentrations of antimony in freshwater sediment fall in the range of 27 to 70 mg Sb/kg ww. This is higher than the highest measured sediment concentrations not identified as being close to a point source, the highest being approx. 21 mg Sb/kg ww measured in a polluted area of the river Rhine; but is similar to levels found near point sources (glass producer and antimony trioxide producer).

Production

Water

The predicted local PEC for production (site P1, which is the site modelled in EUSES) is 39.5 µg Sb/l (see Table3-56). No measured data are available from the recipient. Measured data from site P4 give a surface water concentration of 10 µg Sb/l 100 m downstream the site. It shall be noted that site P4 reports zero emissions to water. However, there has been mining exploitation, metallurgical treatment and slag dump on site for 100 years. There are also increased levels upstream from plant.

Sediment

The estimated local PEC in sediment for production site P1 is 38 mg Sb/ kg ww (see Table3-56), which is lower than the measured value of 94 to 99 mg Sb/ kg ww, but in the same order of magnitude. The measured value also includes historical releases which are reported to have been significantly higher than the more recent releases.

Conclusion

Overall it is deemed appropriate to use the PEC derived from EUSES-modelling for the local risk characterisation for sediment near production site P-1.

Because of the historical pollution at site P4, it is not possible to compare measured to predicted environmental concentrations

Textile industries

Water

There are no data available on direct emissions from textile industry to the aquatic compartment. However, the influence of textile industry on this compartment appears obvious when examining data from a municipal WWTP in Borås, Sweden, to which textile industries are connected. Concentrations of antimony measured in River Viskan waters upstream and downstream of the WWTP were 0.11 µg Sb/l and 0.79 µg Sb/l, respectively. The concentration of 0.79 µg Sb/l measured 500 m downstream the WWTP is considered relevant for comparison.

This value is almost 2 orders of magnitude lower than the predicted local PEC, which is 67 µg/l. However, the measured values does not exclusively represent emissions from textile industry but is “diluted” with household wastewater.

STP

The concentration measured in the outflow from the WWTP was 6.09 µg Sb/l. This value is 2 orders of magnitude lower than the predicted local PEC, which is 733 µg Sb/l.

Conclusion

Based on the present data it is not possible to judge if the predicted local PEC for use in textile industry is reasonable or not.

Glass production (processing)

The measured data is not from the site reporting releases.

Water

Concentrations of antimony measured in waters upstream and downstream of a glass producer were 0.1 µg Sb/l and 3.08 – 3.55 µg Sb/l, respectively. These values are similar to those estimated by EUSES, which are 0.90 and 3.07 µg Sb/l, respectively.

Sediment

Measurements of antimony in sediments in ponds upstream and downstream also indicates emission to the aquatic compartment since concentrations measured are substantially lower upstream, 1.79 mg Sb/kg dw (0-2.5cm) and 3.79 mg Sb/kg dw (7.5-10 cm), as compared to downstream, 17.5 mg Sb/kg dw (0-2.5cm) and 4.11 mg Sb/kg dw (7.5-10 cm). The predicted local concentrations in sediment near glass-producing sites are 2.9 and 0.8 mg Sb/kg ww corresponding to 13.3 and 3.7 mg Sb/kg dw respectively (using TGD default conversion factor). This is within the range of reported measured values.

Conclusion

Based on a limited data set, the PEC for sediment and surface water seem to be relevant for risk characterisation for glass-producing sites.

3.1.4.5 Terrestrial compartment

Antimony, being a natural element, will naturally occur in soils as a result weathering of parent rock material. The concentrations of antimony in soils are highest in soils from sedimentary rocks such as argillaceous sediments and shale (Fergusson, 1990). The average concentration of antimony in the earth's crust has been suggested to be approximately 0.2 - 0.3 mg/kg (Lisk, 1972; Bowen, 1979; Wedepohl, 1995). The average concentration of antimony in soils is about 0.5 mg/kg (Reiman and Caritat, 1998) to 1 mg/kg (Bowen, 1979), but wide ranges have been reported (see Table3-73 below).

3.1.4.5.1 Calculation of predicted environmental concentrations (PEC)

The ambient concentrations used as $PEC_{regional}$ is 1.7 mg Sb/kg dw corresponding (with EUSES default conversion factors) to 1.5 mg Sb/kg ww. (see section 3.1.4.5.2).

The PEC for soil are summarised in Table3-69 below.

Table3-69. Summary of PECs for soil.

Scenario (generic unless otherwise stated)	PEC agric soil, 30d average (mg Sb/kg ww)	PEC agric soil, 180d average (mg Sb/kg ww)	PEC grassland, 180d average (mg Sb/kg ww)
Production site, P1	1.51	1.51	1.53
Production site, P5	1.56	1.56	1.52
Use as flame-retardant in plastics and rubber - formulation	3.04	3.03	2.1

Scenario (generic unless otherwise stated)	PEC agric soil, 30d average (mg Sb/kg ww)	PEC agric soil, 180d average (mg Sb/kg ww)	PEC grassland, 180d average (mg Sb/kg ww)
Use as flame-retardant in plastics and rubber – formulation assuming release to air	1.5	1.5	1.5
Use as flame-retardant in plastics and rubber – industrial use	2.04	2.04	1.71
Use as flame-retardant in textiles – formulation, generic site	1.5	1.5	1.5
Use as flame-retardant in textiles – formulation, site AMI-19U	4.75	4.75	2.76
Use as flame-retardant in textiles – formulation/processing, site FT-2	2.06	2.06	1.72
Use as flame-retardant in textiles – formulation, site 96	2.4	2.4	1.85
Use as flame-retardant in textiles – industrial use (application to textiles) generic site	28	28	12
Use as flame-retardant in textiles – industrial use (application to textiles) site RC74U	2.14	2.14	1.75
Use as catalyst (PET industry) – polymerisation	2.77	2.76	1.99
Use as catalyst (PET industry) – industrial use (shaping)	1.5	1.5	1.5
Use in paint as flame-retardant - formulation	1.62	1.62	1.54
Use in paint, pigment production - formulation	1.65	1.65	1.56
Use in paint – formulation, site AMI3G	5.12	5.11	2.9
Use in glass – formulation	1.5	1.5	1.5
Use in glass – industrial use Proc Glass 2	1.5	1.5	1.5
Use in glass – industrial use Proc Glass 3	1.5	1.5	1.5
Regional (based on monitoring data)			1.5

3.1.4.5.2 Measured levels

Baseline background concentrations of antimony in soil (topsoil and subsoil) measured in the FOREGS-project results in values in topsoil that range over three orders of magnitude from < 0.02 to 31.1 µg Sb/kg dw, with a median value of 0.60 mg Sb/kg dw (see Table3-70 and Figure 3-8). The median ratio topsoil/subsoil is 1.15. Concentrations detected in subsoil are presented in Table3-71.

Table3-70 Concentrations of Sb in European topsoil according to FOREGS http://www.gsf.fi/publ/foregsatlas/article.php?id=15).

Unit	Number of samples	Min	Median	Mean ± SD	90P	Max
mg Sb/kg dw	840	0.02	0.60	1.04 ± 2.04	1.91	31.1

Table3-71 Concentrations of Sb in European subsoil according to FOREGS http://www.gsf.fi/publ/foregsatlas/article.php?id=15).

Unit	Number of samples	Min	Median	Mean ± SD	90P	Max
mg Sb/kg dw	783	<0.02	0.47	0.836 ± 1.73	1.53	30.3

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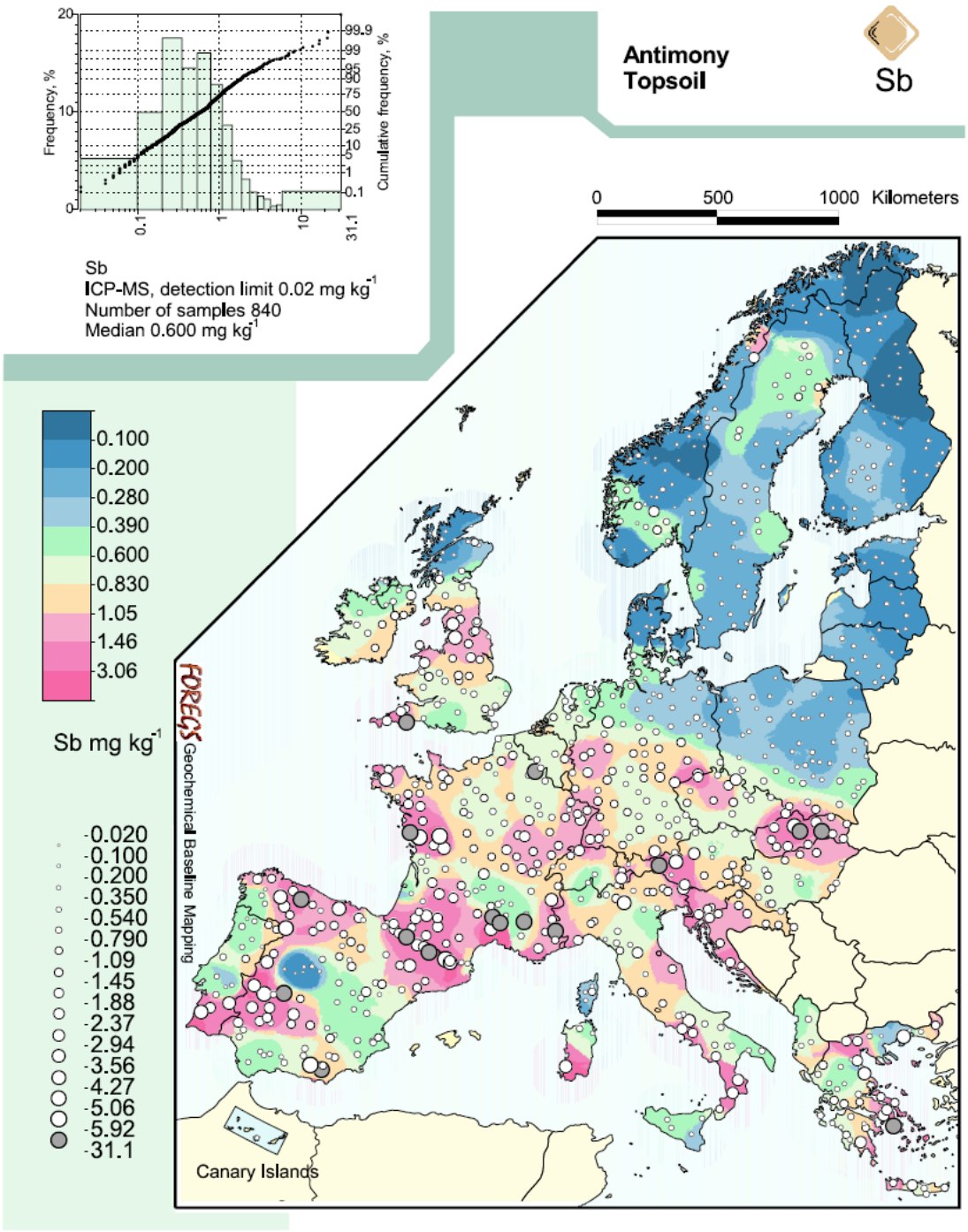

Figure 3-8 FOREG map showing measured concentrations of Sb (mg/kg dw) in European top soil (http://www.gsf.fi/publ/foregsatlas/maps/Topsoil/t_icpms_sb_edit.pdf).

Measured antimony concentrations in European soils from other sources besides the FOREGS database are presented in Table3-73.

In the lower end of the concentration range is concentrations measured in Sweden and Norway, with median values of about 0.3 mg Sb/kg, and in the upper end concentrations measured in rural regions of Northumberland, UK, with a concentration of about 7 mg Sb/kg. Higher concentrations, up to about 300 mg Sb/kg and above, have been measured in association to copper ore and industrial activities, especially smelters.

Calculation of RWC-ambient PEC

A RWC-ambient PEC of 1.7 mg Sb/kg dw (1.5 mg/kg ww) in soil is calculated using ambient data from 7 countries (see Table3-72). The country-specific FOREGS 90P values range from a factor two larger (Spain and Sweden) to a factor of almost three lower (Germany and UK) as compared to measured ambient concentrations.

Table3-72 Calculation of RWC-ambient PEC for antimony in soil. Country specific 90P FOREGS data are included for comparative purposes only.

Country	RWC-ambient PEC (90P) mg Sb/kg dw	Data used	FOREGS -90P mg Sb/kg dw
Austria			3.1
Belgium	1.6	0.1, 0.5, 0.1, 0.5, 0.1, 0.5, 0.1, 3, 0.1, 0.5, 0.1, 0.5	0.99
Bulgaria			-
Cyprus			-
Czech Republic			1.91
Denmark			0.17
Estonia			0.42
Finland			0.33
France			2.61
Germany	3.7	1, 4	1.27
Greece			2.71
Hungary			0.98
Ireland			1.03
Italy			2.57
Latvia			0.25
Lithuania			0.30
Luxembourg			-
Malta			-
Netherlands	1.73	1.73	0.95
Poland			0.70
Portugal			3.52
Romania			-
Slovakia			12.2
Slovenia			1.69
Spain	1.8	1.8	3.87
Sweden	0.25	0.25	0.58
United Kingdom	5.13	0.7, 1, 0.64, 6.9	1.76
Norway	1.26	2.2, 1.26, 0.83, 0.22, 1.09, 0.33, 0.25, 0.41, 0.31, 0.23, 0.17	0.58

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Median	1.73		1.03* 1.59**

*Median of FOREGS country-specific 90P values

**Median value of RWC values when available and country-specific FOREGS 90P-values for those countries with no RWC-value

In conclusion, the RWC-ambient background concentration of antimony in soil used in this RAR is 1.7 mg Sb/kg dw (1.5 mg Sb/kg ww).

Table3-73. Measured antimony concentrations in European soils.

Location	Concentration (Sb mg/kg dw)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Austria					Fritsche, 2003
Liesingtal	1.4, 1.1, 1.1, 1.1, 1.0		2002	FIFU-AAS 31 samples, different soil types and locations near motorways all over Austria	
Knittelfeld	4.8, 2.5, 1.2, 1.1, 0.9				
Rankweil	2.9, 2.6, 1.4, 1.1, 0.8				
Brennersee	3.4, 5.1, 2.3, 1.9, 1.7				
Tangente (SW of)	3.4, 3.3, 3.8, 2.9, 4.0				
Wien (Stadtpark)	3.7, 4.0				
Graz (Stadtpark)	1.9, 1.9				
Lungau	0.5, 0.4				
Belgium				GF-AAS 130 soil samples from all over Belgium in areas as far away as possible from point sources. Normal levels in top soils (0-10 cm)	De Temmerman et al., 1984
				Quaternarian parent material	
	<0.2-0.5			Sand	
	<0.2-0.5			Sandy-loam	
	<0.2-0.5			Loam and clay (polder)	
	0.1-3.0			Alluvium (clay)	

Location	Concentration (Sb mg/kg dw)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
	<0.2-0.5 <0.2-0.5 1.0			Palaeozoic and Mesozoic parent material Sand (woods) Stony loam-heavy clay Upper limit of "normal" concentration in unpolluted soil	
Germany	222-333			GF-AAS + HG-AAS Soil samples associated with gray copper ore	Gebel et al., 1997
	1.0 4.0			AAS Agricultural (unpolluted) soil Luvisol Chernozem	Hammel et al., 1998
Bavaria	232 268			HPLC-ICP-MS Polluted soil, 2 samples	Lintschinger et al., 1997
Bitterfeld	5.8 2.8 2.4 1.3			PAA Industrial area Loam from flood plain forest around Bitterfeld-Greppin (Accumulation in Ah horizon, decrease with depth but at 1 m depth (Gor-horizon acts as a barrier) PAA 10 cm 50 cm 100 cm 150 cm NAA	Schulze et al., 1997

Location	Concentration (Sb mg/kg dw)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
	9.0 4.0 3.6 2.0			10 cm 50 cm 100 cm 150 cm	
Bitterfeld	>100**			IC-ICP-MS Strongly polluted	eUlrich, 1998
Bitterfeld	2487***			FIA-ICP-MS Strongly polluted	Ulrich, 2000
Nordpfälzer Bergland	13 ± 2 82 ± 6 153 ± 14 18 ± 1 1317 ± 508			AAS Soil samples from a historical (1442-1932) mining area Fallow meadow Agriculture used meadow Vineyard Vineyard Former mine dump	Hammel et al., 1998
Netherlands Soils, Holland Marsh	1.85, 2.11 1.73 0.66 0.58 0.93 0.75 0.7		1970, June	PAA Market-garden soils Surface 0-7.5 cm 7.5-15.0 cm 15.0-22.5 cm 22.5-30 cm 30.0-37.5 cm 37.5-45 cm	Chattopadhyay and Jervis, 1974
Lekkerkerk	0.24-3.2 0.32-3.2 0.4-3.0			Highly polluted soils Comparison of three techniques Colorimetry HG-AAS GF-AAS	Haring et al., 1982
Norway	2.2		1977, Summer	NAA Humus layer, Mean values S. Norway (0-60 km	Allen and Steinnes, 1979; Steinnes, 1980

Location	Concentration (Sb mg/kg dw)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
				from the coast)	
	1.26			S. Norway (60-120 km from the coast)	
	0.83			E. Norway, central part	
	0.22			E. Norway, northern part	
	1.09			W. Norway, coast	
	0.33			Møre/Trøndelag, coast	
	0.25			Møre/Trøndelag, inland	
	0.41			Nordland, coast	
	0.31			Nordland, inland	
	0.23			Troms/Finnmark, coast	
	0.17			Troms/Finnmark, inland	
				Surface soil (decrease along a northly gradient)	Steinnes et al., 1997
South	2.4				
North	0.22				
Spain				ICP-MS	Cabrera et al., 1999
Guadiamar valley	0.71-3.31 (mean: 1.80)			Soils unaffected by acidic mining sludges	
	0.89-323 (mean: 13.7)			Soils affected by a toxic flood from acidic mining sludges	
Sweden				ICP-EAS/ICP-QMS	Eriksson, 2001
	0.25±0.08 (mean±stdev) 0.25 (median) 0.07-0.41 (min-max)			Topsoil, n=25 from different regions in Sweden covering different types of soil and parent material	
	0.21±0.10 (mean±stdev) 0.23 (median)			Subsoil, n=25 from different regions in Sweden covering different types of soil and parent material	

Location	Concentration (Sb mg/kg dw)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
	0.02-0.38 (min-max)				
United Kingdom South East England	 0.7			AAS Rural soil (acid sandy soil, pH=4.9)	Cornfield, 1977
Scotland	~1			Arable surface soils	Mitchell and Burridge, 1979
Scottish soils	 0.29-1.3 (mean = 0.64)			SSMS 10 samples, different soil types and locations	Ure et al., 1979
Northumberland	 6.9 7.6 170-360 90-200 120-180			Rural soils 0-5 cm 10-15 cm Proximity to Sb smelter (concentrations decrease with depth) 100 m from smelter 250 m from smelter 450 m from smelter	Ainsworth et al., 1990a
Derbyshire	 23.4 21.8 17.1 1.43 1.11 1.2 103 41.3		1990, May	ICP-AES *Old mining area* Winster village and farm A Topsoil (0-15 cm) Subsoil (15-20 cm) Subsoil (30-45 cm) Control site (farm B) Topsoil (0-15 cm) Subsoil (15-20 cm) Subsoil (30-45 cm) Old smelter site (stone Edge) Topsoil (0-15 cm) Subsoil (15-20 cm)	Li and Thornton, 1993

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Location	Concentration (Sb mg/kg dw)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
	17.3			Subsoil (30-45 cm)	
				Smelter surrounding area	
	1.1			Topsoil (0-15 cm)	
	1.03			Subsoil (15-20 cm)	
	0.63			Subsoil (30-45 cm)	
				Old zinc mining area	
Shipman			1990, August	Shipman mining area	
	37.9			Topsoil (0-15 cm)	
	36.1			Subsoil (15-20 cm)	
	28.9			Subsoil (30-45 cm)	
				Old Red sandstone area	
	1.01			Topsoil (0-15 cm)	
	0.68			Subsoil (15-20 cm)	
	???			Subsoil (30-45 cm)	
				Old mining area	
				Old mining site	
	2.21			Topsoil (0-15 cm)	
Cornwall	2.32		1991, December	Subsoil (15-20 cm)	
	1.95			Subsoil (30-45 cm)	
				Old mining site, Wheal sister Farm	
	0.65			Topsoil (0-15 cm)	
	0.53			Subsoil (15-20 cm)	
	0.51			Subsoil (30-45 cm)	
				Old As calciner and Sn smelter sites	
				New Mill Farm	
	5.49			Topsoil (0-15 cm)	
	6.33			Subsoil (15-20 cm)	
	3.91			Subsoil (30-45 cm)	

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Location	Concentration (Sb mg/kg dw)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Thetford Forest	0.0 8.7			ICP-MS 0-10 cm soil horizon Control soil (pH 3.3) Contaminated soil (pH 3.9)	Hartley et al., 1999

*Mobile content (NH4NO3 extraction)
**Sb(V) (90%), Sb(III) (10%), TMSbO (only in small concentrations)
***Concentration in soil extracts, units g/l. Sb(V), Sb(III), and TMSbO values determined.

Local levels

Not much data on measured local concentrations of antimony in the terrestrial compartment near sites using diantimony trioxide is available except for measurements close to a PVC cable producing company in Sweden. The data available is presented below.

Cable-producer

About 30-40 t Sb_2O_3 /y are used for flame-retardant properties. The production plant has no emissions to the aquatic compartment, but emissions to the atmosphere is possible through the ventilation in the building in which the Sb_2O_3 is used. Soil samples were taken in Nov. 2001 at four different locations at increasing distance from the building. Measured data are presented in Table3-74 below.

Table3-74 Measured local concentrations in soil at a PVC cable producing plant (Sternbeck, Palm and Kaj, 2002).

Location (distance from building)	Concentration in soil (mg Sb/kg dw)
500 m SW	0.80
250 m ESE	2.44
800 m NE	0.063
2000 m ESE	0.18

The highest soil concentration (2.44 mg Sb/kg dw) was found closest to the building were the Sb_2O_3 is used.

Comparison between predicted and measured levels

The predicted local concentrations of antimony in soil range from background concentrations (1.7 mg/kg ww) to 7 mg/kg ww. Concentrations of up to one order of magnitude higher than this has been measured in mining areas and in the proximity of smelters. No measured data representative for comparison to predicted local concentrations are available except for measured data from a PVC-cable producer in Sweden.

Cable-producer

The predicted local PEC for industrial use of diantimony trioxide as flame-retardant plastics and rubber is 2.04 mg Sb/kg ww and 1.71 mg Sb/kg ww in agricultural soil and natural soil, respectively. The measured concentration in soil in the dominant direction of the wind 250 m from a cable-producer was 2.1 mg Sb/kg ww and 0.16 mg Sb/kg ww at a distance of 2000 m from the factory.

Conclusion

It is hard to draw firm conclusions regarding the reliability of the predicted local PEC based on the very limited measured data available. However, in this case the concordance between predicted and measured values was good.

3.1.4.6 Atmosphere

Most of the antimony that is released to the atmosphere from anthropogenic sources results from metal smelting and refinement, combustion of coal, refuse and sludge incineration, and road traffic.

3.1.4.6.1 Calculation of predicted environmental concentrations (PEC)

The RWC ambient background concentration used as $PEC_{regional}$ is 2.6 ng Sb/m^3 (see section 3.1.4.6.2).

The PEC for air are summarised in Table3-75 below.

Table3-75 Summary of PECs for air.

Scenario	$PEC_{local(air)}$ (ng Sb/m^3)
Production site, P1	986
Production site, P5	2.6
Use as flame-retardant in plastics and rubber - formulation	2.6
Use as flame-retardant in plastics and rubber – formulation, assuming release to air	41
Use as flame-retardant in plastics and rubber – industrial use	2.6
Use as flame-retardant in textiles – formulation, generic site	5.0
Use as flame-retardant in textiles – formulation, site AMI-19U	3.2
Use as flame-retardant in textiles – formulation/processing, site FT-2	2.6
Use as flame-retardant in textiles – formulation, site 96	2.6
Use as flame-retardant in textiles – industrial use (application to textiles), generic site	3.0
Use as flame-retardant in textiles – industrial use (application to textiles), site RC74U	2.7
Use as catalyst (PET industry) - polymerisation	2.6
Use as catalyst (PET industry) – industrial use (shaping)	2.6
Use in paint as flame-retardant - formulation	5.4

Scenario	$PEC_{local(air)}$ (ng Sb/m^3)
Use in paint, pigment production - formulation	6.1
Use in paint – formulation, site AMI3G	23
Use in glass – formulation	17
Use in glass – industrial use Proc Glass 2	23
Use in glass – industrial use Proc Glass 3	3.4
Regional (based on monitoring data)	2.6

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3.1.4.6.2 Measured levels

Measured antimony concentrations in European air are presented in Table3-76.

The measured concentrations in European air normally range from background values of about 0.1 ng Sb/m^3 in remote areas of Norway, to several tenths of ng Sb/m^3 and above in areas with heavy traffic or in regions with metal smelting and manufacturing industries. The concentrations (from several thousand to several tens of thousands of ng Sb/m^3) of methylated antimony measured in sewage gas and landfill gas (see section 3.1.2.7.2 above) indicate that these kinds of sources may, at least at a local scale, be important.

Since the atmospheric half-life of antimony particles has been estimated to be shorter than one week, it is not expected that historical anthropogenic activities should contribute to the atmospheric background concentration.

Calculation of RWC-ambient PEC

The data available to estimate a RWC-ambient PEC for air consists of:

- three measurements performed in Norway at a remote location on airmasses corresponding to Atlantic, Central European Air and air from Ireland and UK which range from 0.07-0.53 ng Sb/m^3
- six measurements from minor Austrian villages/cities (1500-12500 inhabitants) with mean values ranging from 1.0 – 4.4 ng Sb/m3.
- one measurement from an urban site in the Helsinki Metropolitan area in Finland (2 km NE of the centre of Helsinki, 14 m from a road with an average traffic load of 14 000 vehicles/day), and one measurement from a rural site situated in a forested area 25 km NW of central Helsinki, 1 km from the closest major road. The total concentrations (fine + coarse particles) for the two sites were 1.55 and 0.40 ng Sb/m^3, respectively.
- two measurements from Munich, Germany. One from a site with heavy traffic intensity (approx. 120 000 vehicles/day) and one from a residential district NE of Munich with modest traffic influence. The measured concentrations were 13.4 and 3.2 ng Sb/m^3, respectively.
- two measurements performed at two traffic tunnels (subdivided in inlet and outlet concentrations) with heavy traffic in Gothenborg, Sweden. Arithmetic mean values of the measured concentrations from the two tunnels were 51.5 and 70.5 ng Sb/m^3.
- one measurement from the centre of Stockholm, Sweden, with high traffic intensity and one from the countryside, next to the motorway E4. The measured concentrations were 20 and 1.8 ng Sb/m^3, respectively.

- two measurements performed in the inland part of England in industrial areas with ferrous and non-ferrous smelting and manufacturing. The arithmetic mean of the two measurements is 39.5 ng Sb/m^3.

Using data points that are representative for ambient (i.e. diffuse) contamination on a regional scale from the available dataset in order to calculate a RWC-ambient PEC for air, similarly as previously been done for freshwater, freshwater sediment and soil using the median of 90P country-specific data, is not straight forward. This since it is difficult to select values that are not influenced by point sources (e.g. traffic). Only using values measured at rural or remote sites, without influence of e.g. traffic or industrial activities would on the other hand not result in a relevant RWC ambient PEC for air either.

Only using country specific values not influenced by point sources, but considered relevant for an industrialised region (i.e. excluding the measurements performed at a remote location in Norway) would result in 2.4 ng Sb/m^3 (median of 0.40 and 4.35 (90P of 1, 1.4, 1.9, 4.3, 1.2, 4.4) and 0.40). This approach assumes that that the measured Austrian values are not influenced by traffic, which appears most uncertain considering the range of the individual measurements. Performing an urban scenario including measurements from cities with varying traffic intensities (Munich in Germany, Helsinki in Finland and Stockholm in Sweden) results in 12 ng Sb/m^3 (median of 12 (90P of 3.2 and 13), 1.44 (90P of 0.40 and 1.55), and 20).

It is therefore pragmatically decided to also calculate an arithmetic mean, a geometric mean and a median value using all available data, and compare all of these values with the available data, and thereafter choosing the value considered most relevant.

	Concentration (ng Sb/m^3)
Median of country-specific 90P (AU, FIN)	2.4
Median of country-specific 90P urban scen. (DE, FIN, SE)	12
Arithmetic mean value of all available data	12
Geometric mean value of all available data	2.6
Median value of all available data	1.9

Based on the available measured data it is decided to use the geometric mean value of 2.6 ng Sb/m^3. This value is supported by the median of country-specific 90P, which is based on a very limited dataset; it is close to the median of all available data and is in the range of concentrations measured in urban areas with low to modest traffic intensity. The selected value is higher than concentrations measured in rural and remote areas, but on the other hand lower than the concentrations measured in areas with high traffic intensity and/or in areas with smelting and manufacturing industries.

In conclusion, the RWC-ambient background concentration of antimony in air used in this RAR is 2.6 ng Sb/m^3.

Table3-76. Measured antimony concentrations in European air.

Location	Concentration (Sb ng/m³)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Austria				FIFU-AAS	Umweltbundesamt, 2004
Illmitz	<0.082 – 2.0 (mean = 1.0)		2004, Jan – Feb	10 samples PM10	
Kittsee	<0.082 – 3.5 (mean = 1.4)			10 samples PM10	
Wörgl	1.2 – 2.4 (mean = 1.9)		2003, Oct - Nov	5 samples PM10	
Vomp	2.5 – 6.1 (mean = 4.3)			6 samples PM10	
Brixlegg	0.19 – 3.2 (mean = 1.2)			4 samples PM10	
Inntal	1.2 – 6.7 (mean = 4.4)			5 samples PM10	
Finland			1996, April – 1997, June	ICP-MS	Pakkanen et al., 2001
Helsinki (central)	0.77 ± 1.4 0.78 ± 0.52			$PM_{2.3}$ $PM_{2.3-15}$	
NW Helsinki (rural)	0.36 ± 0.4 0.036 ± 0.033			$PM_{2.3}$ $PM_{2.3-15}$	
Germany			1994, May-October	ICP-MS PM10-fraction	Dietl et al., 1997
Munich (central)	13.4 (range 9.0-16.7)			High traffic density	
NE Munich (residential district)	3.2 (range 1.69-5.0)			Very light traffic density	
Municipal putrification plant	618-14720 (min-max) 23900-71600	Methylated		GC-ICP-MS Sewage gas	Feldmann and Hirner, 1995

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Location	Concentration (Sb ng/m³)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Domestic waste deposit	(min-max)	Methylated		Landfill gas	
Norway Nordmoen	 0.07 0.54 0.13		1988, April-May Atlantic air UK & Ireland air Central Europe air	ICP-MS	Hillamo et al., 1988
Oslo	6 ± 0.5[a]		1994, summer	ICP-MS Street dust, unit mg/kg	de Miguel et al., 1997
Sweden Gothenburg (Tingstad) Gothenburg (Lundby) 	 28 ± 10 (range: 17-40) 75 ± 35 (range: 56-131) 34 ± 12 (range: 20-49) 107 ± 14 (range: 91-127)		 1999, November 2000, April 	ICP-MS Road tunnels with heavy traffic Tunnel inlet Tunnel outlet Tunnel inlet Tunnel outlet	Sternbeck et al., 2002b
Stockholm (central) S Stockholm	20 ± 12 (range: 5-42) 1.8 ± 1.3 (range: 0.4-3.9)		2003	ICP-MS PM10-fraction Urban air Country side (next to motorway E4)	Sternbeck, 2003
United Kingdom Inland part of England	50 29		1980, Aug.- Oct 1980, Nov-Jan	NAA Industrial area, with ferrous and non-ferrous metal smelting and manufacturing	Pattenden et al., 1982

[a]Mean ± S.E.

Local levels

Some data on measured local concentrations of antimony in the atmospheric compartment near sites using diantimony trioxide is available. The data is presented below.

Production

Limited information is available on environmental monitoring data in the vicinity of the Sb_2O_3 producing plants (see Table3-77). Aerial deposition data are available for plant P-1 and P-3. Plant P-1 provided measured air concentrations in the vicinity of the plant (sampling points Environment Agency). No recent information on air concentrations was reported by plant P-4 and P-5.

Site P-1 reports measured deposition rates in the vicinity of the site at different distances /downwind directions (250 m, 1000 m, NE, NW) and for different years. There is influence from neighbouring sources, an industrial lead recycling site handling materials containing antimony. From 2000 to 2005 there is a general decrease in aerial deposition rates over time for all sampling stations. This is a similar trend as observed for stack air emissions for P-1. Site P-3 reported on-site aerial deposition from three sampling points.

The TGD takes aerial deposition in the surroundings of the plant (area with radius of 1000 m) into account for the calculation of a PEC_{soil}. Measured data can be of use for comparison of calculated data (on the basis of emissions) with site specific industry measurements.

Table3-77 Measured local concentrations in air and in aerial deposition from Sb_2O_3 producing plants in the EU (EURAS, 2006).

Plant Nº	**AIR**				
	Average conc. in air		**Aerial deposition**		**Year**
	Location	**(µg Sb/m³)**	**Location**	**Unit**	
P-1			Wet deposition	Average (mg Sb/m².month)	
P-1	-	-	250 NW	18.9 (9 dp)	1998
P-1	-	-	250 NE	26.0 (9 dp)	1998
P-1	EA, Station BEE02	0.2	250 NW	22.7 (9 dp)	1999
P-1	-	-	250 NE	28.6 (9 dp)	1999
P-1	-	-	Border NW	28.71	2000
P-1	-	-	100 m NW	18.8	2000
P-1	-	-	250 m NW	11.8	2000
P-1	-	-	1000 m NW	7.79	2000
P-1	-	-	Border NE	112	2000
P-1	-	-	100 m NE	79.7	2000
P-1	-	-	250 NE	32.5	2000
P-1	-	-	1000 m NE	24.3 (9 dp)	2000
P-1			Wet deposition	Average	

Plant Nº	AIR				
	Average conc. in air		Aerial deposition		Year
	Location	(µg Sb/m³)	Location	Unit	
				mg Sb/(m² d)	
P-1	-	-		0.78 mg Sb/(m² d)	2001
P-1	-	-		0.68 mg Sb/(m² d)	2002
P-1			250 m NE**	1.08 mg Sb/(m² d)	2000
P-1			250 m NW**	0.39 mg Sb/(m² d)	2000
P-1			1000 m NE**	0.81 mg Sb/(m² d)	2000
P-1			1000 m NW**	0.26 mg Sb/(m² d)	2000
P-1			250 m NE**	1.41 mg Sb/(m² d)	2001
P-1			250 m NW**	0.55 mg Sb/(m² d)	2001
P-1			1000 m NE**	0.62 mg Sb/(m² d)	2001
P-1			1000 m NW**	0.30 mg Sb/(m² d)	2001
P-1			250 m NE**	0.55 mg Sb/(m² d)	2002
P-1			250 m NW**	0.30 mg Sb/(m² d)	2002
P-1			1000 m NE**	0.26 mg Sb/(m² d)	2002
P-1			1000 m NW**	0.18 mg Sb/(m² d)	2002
P-1			250 m NE**	0.75 mg Sb/(m² d)	2003
P-1			250 m NW**	0.69 mg Sb/(m² d)	2003
P-1			1000 m NE**	0.50 mg Sb/(m² d)	2003
P-1			1000 m NW**	0.57 mg Sb/(m² d)	2003
P-1			250 m NE**	0.32 mg Sb/(m² d)	2004
P-1			250 m NW**	0.13 mg Sb/(m² d)	2004
P-1			1000 m NE**	0.27 mg Sb/(m² d)	2004
P-1			1000 m NW**	0.08 mg Sb/(m² d)	2004
P-1			250 m NE**	0.26 mg Sb/(m² d)	2005
P-1			250 m NW**	0.13 mg Sb/(m² d)	2005
P-1			1000 m NE**	0.14 mg Sb/(m² d)	2005
P-1			1000 m NW**	0.06 mg Sb/(m² d)	2005
P-1	EA, station BEEE01; 1200 m E of site	0.045 µg Sb/m³			2004
P-1	EA, station BEEE02; 500 m N of site	0.268 µg Sb/m³			2004
P-1	local Sb background in air	0.001 µg Sb/m³ (detection limit)			
P-3				Average (mg/m².d)	
P-3	-	-	Border factory (3 sampling points)	1.3-16 (3 dp)	1999

Plant Nº	AIR				
	Average conc. in air		Aerial deposition		Year
	Location	(μg Sb/m³)	Location	Unit	
P-3			3 on-site sampling points, 3 Owen gauges	11.1 mg Sb/(m² d)	2001
P-3			3 on-site sampling points, 3 Owen gauges	4.2 mg Sb/(m² d)	2002
P-3			3 on-site sampling points, 3 Owen gauges	2.9 mg Sb/(m² d)	2003
P-3			3 on-site sampling points, 3 Owen gauges	6.2 mg Sb/(m² d)	2004
P-4	-	-	-	-	1998
P-5	-	<3.5	-	-	2000

-No data available.
[a]Mining exploitation, metallurgical treatment and slag dump on the site for 100 years.
** Prevailing wind direction:SW ; Sampling performed by Environment Agency, Nilu recipient, wet deposition; Influence from neighbouring sources: Industrial Pb recycling site (contains Sb) in surroundings

Cable-producer

(Sternbeck et al., 2002) performed a study, which included screening of some potential Sb-emitters including a production facility for PVC-cables. About 30-40 t Sb_2O_3 /y are used for flame-retardant properties. The production plant has no emissions to the aquatic compartment, but emissions to the atmosphere is possible through the ventilation in the building in which the Sb_2O_3 is used. Air samples were taken at two separate occasions (in Nov. 2001) 50 and 200 m from the building, both at production and when no production was performed (windy at the time of sampling). Large amounts of copper and aluminium are also used at the facility, why also magnesium and manganese were used as markers of minerogent antimony in this area. The results are shown in Table3-78 below.

Table3-78 Measured local concentrations in air at a PVC cable producing plant, samples taken in Nov. 2001 at two separate occasions (Sternbeck, Palm and Kaj, 2002).

Location (distance from building)	Concentration in air (ng Sb/m³)
50 m	2.24 ng/m³
50 m (windy)	1.47 ng/m³
200 m	0.45 ng/m³
200 m (windy)	0.66 ng/m³

The difference in air concentration between the occasions when Sb_2O_3 was used in production or not was rather low, which may be explained by resuspended material from the ground also being measured at the windy occasion when Sb_2O_3 not was used in production.

Glass production (processing)

(Sternbeck et al., 2002) performed a study, which included screening of some potential Sb-emitters, including a producer of art glass. This glasswork uses about 50% of the produced glass raw material in Sweden. A rough approximation that it therefore also uses about 50% of the Sb_2O_3 imported by the glass manufacturing industry and used when producing glass

results in about 35-40 t/y. Emissions to air and water are measured on a regularly basis. In addition, measurements on antimony in water moss downstream of the glasswork are also regularly measured.

Air samples were taken at two separate (in Sept. 2001) occasions 200 and 1000 m from the two chimneys. The dominating wind direction is North-East. The results are shown below in Table3-79.

Table3-79 Measured local concentrations in air at and nearby a producer of artistic glass samples taken at two separate occasions in Sept. 2001 (Sternbeck, Palm and Kaj, 2002).

Concentration in air (ng Sb/m^3)	Location
0.31	200 m NE
6.16	200 m NE
28.9	1000 m NW
10.2	1000 NW

The glass manufacturing plant reports decreasing aerial deposition rates with increasing distance from the plant (0.0023 mg/m^2 d measured 80 m from plant to 0.0004 mg/m^2 d 900 m from plant) (year 2001 data) (see Table3-80).

Table3-80 Measured local concentrations of aerial deposition at a glass producing plants in the EU. (EURAS, 2003).

Plant Nº	Aerial deposition		Year
	Location	(mg/m^2 d)	
ProcGlass-1	P1 (80m) P2 (80m) P3 (200m) P4 (200m) P5 (900m)	0.0023 0.0015 0.0008 0.0004 0.0004	2001

Comparison between predicted and measured levels

The predicted local concentrations in air range from about 1 to 23 ng Sb/m^3 with the exception of near one production site for antimony trioxide with a local PEC of near 1µg Sb/m^3. Measured antimony concentrations range from about 0.1 ng/m^3 in unpolluted areas to 100 ng/m^3 in urban areas with heavy traffic. Predicted local concentrations of antimony in air are compared to measured concentrations for the local scenarios for which measured data are available (see below). This indicates that traffic or other sources are in most cases more relevant for concentrations in air in urban areas than sites using antimony trioxide.

Production

The predicted local PEC in air for site P1 (which is the site modelled by EUSES) is 1.0 µg/m^3. A measured concentration of 0.3 µg Sb/m^3 in air close to site P1 has been reported. The representativity of this measurement cannot be assessed. For site P5 a measured value of <3.5 µg Sb/m^3 has been reported. However, no information on the sampling point is given.

Basing the comparison on aerial deposition gives another picture. Measured deposition in the vicinity of site P-1 vary, but are lower than the predicted deposition of 1.49 µg Sb/m^2 day, but in the same order of magnitude.

Conclusion

The predicted concentration in air for site P1 seem to be overestimated compared to measured data. Comparison of predicted aerial deposition rates to measured rates indicates that the predicted aerial deposition rate is slightly overestimated. No firm conclusions can be drawn since the measured data set is limited and the representativity is difficult to assess. Therefore, it is decided to base the risk characterization on predicted concentrations.

Cable producer

The predicted local PEC in air for industrial use of diantimony trioxide as flame-retardant plastics and rubber is 2.6 ngSb/ m^3. Measured concentrations from a PVC cable producing plant is 1.5 – 2.2 ng Sb/m^3 50 m from the building

Conclusion

It is hard to draw firm conclusions regarding the reliability of the predicted local PEC based on the very limited measured data available. However, in this case the concordance between predicted and measured values was good.

Glass formulation

For glass formulation no measured concentrations in air are available however, measurements of aerial deposition rates have been performed. Measured deposition rates in the vicinity of site MN62-S (which is the site modelled in EUSES) range from 1.5µg Sb/m^2*day to 2.3 µg Sb/m^2*day 80 m from the facility. The predicted deposition rate for this site was 0.02 µg Sb/m^2*day.

Conclusion

Comparison of predicted aerial deposition rates to measured rates indicates that the predicted emissions to air are underestimated. It is however hard to draw firm conclusions regarding the reliability of the predictions based on the very limited measured data available. It is therefore decided to base the risk characterisation on predicted values.

Glass production (processing)

The predicted local PECs in air for glass production is 23 and 3 ng Sb/m^3. Measured concentrations in the dominating wind direction 200 m north east of the chimneys was 0.3 and 6.2 ng Sb/m^3 at two different sampling occasions. However, higher concentrations 10.2 ng Sb/m^3 and 28.9 ng Sb/m^3 were measured 1000 m north west from the chimneys.

Conclusion

It is hard to draw firm conclusions regarding the reliability of the predicted local PEC based on the very limited measured data available. However, in this case the concordance between predicted and measured values was good.

3.1.4.7 Non-compartment specific exposure relevant to the food chain

3.1.4.7.1 Calculation of predicted environmental concentrations (PEC)

The PECs calculated for fish and earthworms for the assessment of secondary poisoning are summarised in Table3-81 below.

A BCF-value of 40 and 15000 have been used for fish (see section 3.1.3.2 above), and a value of 1 has been used for earthworm.

Table3-81 Calculated concentrations in freshwater fish and in earthworms.

Scenario	Concentration in fish (mg Sb/kg ww)		Concentration in earthworm
	BCF = 40	BCF = 15000	(mg Sb/kg ww)
Production site, P1	0.18	68.2	0.17
Production site, P5	0.031	11.5	0.17
Use as flame-retardant in plastics and rubber - formulation	0.094	35.4	0.25
Use as flame-retardant in plastics and rubber – formulation assuming release to air	0.029	10.8	0.17
Use as flame-retardant in plastics and rubber – industrial use	0.052	19.4	0.20
Use as flame-retardant in textiles – formulation, generic site	0.46	173	0.47
Use as flame-retardant in textiles – formulation, site AMI-19U	0.083	31	0.35
Use as flame-retardant in textiles – formulation/processing, site FT-2	0.050	18	0.20
Use as flame-retardant in textiles – formulation, site 96	0.057	21	0.22
Use as flame-retardant in textiles – industrial use (application to textiles), generic site	1.2	434	1.7
Use as flame-retardant in textiles – industrial use (application to textiles), site RC74U	0.045	16.9	0.20
Use as catalyst (PET industry) -polymerisation	0.083	31	0.24
Use as catalyst (PET industry) – industrial use (shaping)	0.029	11.2	0.17
Use in paint as flame-retardant- formulation	0.039	14.5	0.17
Use in paint, pigment production - formulation	0.041	15.5	0.18
Use in paint – formulation, siteAMI3G	0.076	28.5	0.37
Use in glass - formulation	0.029	10.8	0.17
Use in glass – industrial use Proc Glass 2	0.045	16.8	0.19
Use in glass – industrial use Proc Glass 3	0.030	11.4	0.17

3.1.4.7.2 Measured levels

Information on measured antimony concentrations is presented in Table3-82 below.

Levels in aquatic biota

Information on concentrations of antimony in aquatic biota in general is scarce.

(Sternbeck et al., 2002a) performed measurements on livers of pikes and perch upstream and downstream a glass manufacturer in Sweden. The results showed increased concentrations downstream the glass manufacturer. These concentrations correlated with water concentrations measured in the respective waters.

Krachler *et al*. (1999) reports a concentration of about 400 µg Sb/kg dw in livers of freshwater bream from an urban-industrialized region of Germany.

Ute and Bligh (1971) measured the concentration of a number of metals, including antimony, in freshwater fish from a lake free of major industrial development (Moose Lake) and from lakes in a highly industrialized area (Lower Great Lakes Basin) in Canada. All samples were composite samples consisting of at least 2.5 kg or three fish. In the majority of samples the number of fish used was larger than three. Samples were prepared as follows: headless dressed fish (at least three) were ground and thoroughly mixed and stored at -40 ºC until analysis. Antimony was analysed using Neutron Activation.

Table3-82 Concentrations of Sb in aquatic organisms.

Species	Location	Concentration (µg Sb/kg)	Period	Remark	Reference
	Sweden			ICP-MS Liver	Sternbeck et al., 2002a
Perch	Lake Orranäs	0.85 µg Sb/kg ww (= 4 µg Sb/kg dw)		Upstream glass manufacturer	
	Lake Smedsfors	6.69 µg Sb/kg ww (= 30.8 µg Sb/kg dw)		Downstream glass manufacturer	
Pike	Lake Orranäs	1.26 µg Sb/kg ww (= 5.9 µg Sb/kg dw)		Upstream glass manufacturer	
	Lake Smedsfors	4.26 µg Sb/kg ww (= 19.9 µg Sb/kg dw)		Downstream glass manufacturer	
Bream	German freshwater	3.8 ± 0.7		HG-AAS Liver n = 4 Urban-industrialised region	Krachler et al., 1999
	Canada		1971		(1971)

Species	Location	Concentration (µg Sb/kg)	Period	Remark	Reference
Coregonus clupeaformis	Moose Lake Lake Ontario	2.2 (ww) 3.1 (ww)			
Esox lucius	Moose Lake Lake St. Pierre Lake Erie	3.2 (ww) 3.7 (ww) 4.3 (ww)			
Osmerus mordax	Lake Erie	3.5 (ww)			
Perca flavescens	Lake Erie	3.1 (ww)			

Levels in terrestrial biota

Information on measured antimony concentrations are presented in Table3-83- Table3-86 below.

Invertebrates

Concentrations of antimony in soil and invertebrates at locations close to an antimony smelter in UK have been measured by Ainsworth *et* al. (Ainsworth et al., 1990b). The highest antimony concentrations were found in earthworms (*oligochaeta*). The dry weight concentrations (including gut contents) were 398 ± 94 mg Sb/kg, 213 ± 65 mg Sb/kg and 109 ± 28 mg Sb/kg 100 m, 250 m and 450 m downwind of the smelter.

Mammals

Krachler et al. (1999), performed measurements on livers from deer inhabiting an urban-industrialised region and from deer inhabiting an agrarian region of Germany (see Table3-83 below).

(Ainsworth et al., 1990b) performed measurements on shrews, voles, and rabbits close to Sb-smelter and in a control site.

Table3-83. Concentrations of Sb in mammals.

Species	Location	Concentration (µg Sb/kg dw)	Period	Remark	Reference
	Germany		1997	HG-AAS	Krachler et al., 1999
Deer (1-y old roe deer)		8.3 ± 1.2		Urban-industrialised region Liver n = 5	
Deer (1-y old roe deer)		5.9 ± 0.5		Agrarian region Liver n = 3	
Deer (2-3-month old				Liver n = 3	

Species	Location	Concentration (µg Sb/kg dw)	Period	Remark	Reference
fawns)		6.4 ± 0.3			
Microtus agrestis (short-tailed field vole)	United Kingdom	0.30 ± 0.18 0.31± 0.14 0.18± 0.09		100 m from Sb smelter Liver Lung Kidney n = 21	Ainsworth et al., 1990b
Oryctolagus cuniculus (rabbit)		0.68 ± 0.38 0.31± 0.067 0.28± 0.11 0.15± 0.089		250 m from Sb smelter Liver Lung Kidney Femur n = 5	
Sorex araneus (common shrew)		0.49 ± 0.47 0.11± 0.073 0.33± 0.17		450 m from Sb smelter Liver Lung Kidney n = 6	
		0.04 ± 0.024 0.15± 0.10 0.15± 0.12		Control site Liver Lung Kidney n = 6	

Eriksson (2001) performed measurements on the concentration of antimony in manure from Swedish cows and pigs (see Table3-84 below).

Table3-84 Measured antimony concentrations in European manure.

Location	Concentration (Sb mg/kg) in dw	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Sweden	0.079 (mean) 0.067-0.091 (min-max) 0.31		2000, summer	ICP-EAS/ICP-QMS Floating, milk cows, n=4 Floating, pigs, n=4	Eriksson, 2001

	(mean) 0.28-0.35 (min-max) 0.31 (mean) 0.28-0.35 (min-max)			Solid, pigs, n=4	

Plants

Measurements of antimony in plants are presented in Table3-85 below.

Krachler et al. (1999), analyzed leaves and shoots of poplar, spruce, beech, and elder in various locations in Germany. According to the authors, washing (briefly rinsed with high-purity water) removed approximately 20% of the antimony found in elder leaves collected beside a well frequented motorway, whereas washed leaves from the residential area removed about 30% of the antimony, as compared to non-washed leaves. A strong association of the concentration of antimony in elder leaves with traffic was established. Concentrations of antimony in poplar leaves was higher in the densely populated urban-industrialized area of the city of Leipzig, as compared to the concentrations measured in leaves from another urban-industrialized region with a distinctly lower population and traffic density (Saartal). One reason for the distinct difference in measured concentration between beech and spruce collected in the agrarian region may be the different geometry and surface properties of their leaves and needles, respectively.

Ainsworth and Cooke measured concentrations of antimony in various species of grass close to a Sb-smelter and at a control site (Ainsworth et al., 1990a). The further from the smelter, the lower the measured concentration.

Table3-85. Concentrations of antimony in plants.

Species	Location	Concentration (µg Sb/kg dw)	Period	Remark	Reference
Spruce (needles)	Austria Leoben	14-113	1995-2001	ICP-MS Industrialised region	Weiss et al., in Press
Spruce (needles)	Austria Linz	21-568 21-711	1996 (1 needleye ar) 1996 (2 needleye ar)	ICP-MS Industrialised / urban region	Offenthaler et al., 2004

Species	Location	Concentration (µg Sb/kg dw)	Period	Remark	Reference
			(n=117)		
Maple		32-158 (n=15)			
Poplar		21-384 (n=104)			
	Austria			FIFU-AAS	Zechmeister et al., 2005
Moss		90-2560 (n=62)	2003	Locations near motorways all over Austria	
	Germany			HG-AAS	Krachler et al., 1999
Poplar (leaves)		147 ± 3	1997	Urban-industrialised region n = 6	
Spruce (shoots)		51 ± 5		n = 3 Agrarian region n = 4	
Beech (leaves)		39 ± 2		n = 6	
Spruce (shoots)		64 ± 1			
Elder (virgin leaves)		589 ± 30	1998, autumn	Directly beside motorway A n = 4	
		209 ± 13		50 m from motorway A n = 3	
		591 ± 26		Directly beside motorway B, height fraction 0.8 - 2 m n = 4	
		468 ± 19		Directly beside motorway B, height fraction > 2 m n = 5	
				Residential area 1 n = 4	
		198 ± 9		Residential area 2 n = 4	
		153 ± 5		Leipzig n = 3	
Poplar (virgin leaves)		150 ± 3	1998, autumn	Saartal n = 4	

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Species	Location	Concentration (µg Sb/kg dw)	Period	Remark	Reference
		131 ± 4			
	United Kingdom			HG-AAS Bulk samples from at least 10 different areas	Ainsworth et al., 1990a
				Close to Sb-smelter	
Agropyron repens		167 ± 100 14 ± 7	1985, April – September (6 occasions)	Distance 100 m Distance 250 m	
Dactylis glomerata		92 ± 37 16 ± 10		Distance 100 m Distance 250 m	
Festuca rubra		159 ± 68 20 ± 5		Distance 100 m Distance 250 m	
Dactylis glomerata (live)		0.2 0.3 0.1 ± 0.02 (live) 0.32 ± 0.07 (dead)	1985, March 1985, June 1986, April, n=6	Control site (rural grassland), unwashed	
Other broad-leaved grasses (live)		0.1 0.2 0.2	1985, March 1985, June 1985, August		

Birds

The only measurements in birds, to our knowledge, comes from a study by Krachler et al. (1999), in which 3 pigeon eggs from an urban-industrialized region of Germany were analyzed (see Table3-86 below).

Table3-86 Concentration of antimony in birds.

Species	Location	Concentration (µg Sb/kg dw)	Period	Remark	Reference
Pigeon (eggs)	Germany	2.1 ± 0.2	1997	HG-AAS Urban-industrialised region n = 3	Krachler et al., 1999

3.1.4.7.3 Comparison between predicted and measured levels

Aquatic organisms

Measured concentrations in freshwater fish are in the range of a few µg Sb/ kg ww.

Predicted $PEC_{sec.\ poisoning}$ range from 29 µg Sb/kg ww for sites with no local emission to water to 1 200 µg Sb/kg ww in local scenario with the highest predicted emission to water using a BCF value of 40. Using a BCF value of 15 000 results in PEC values ranging from 10 800 µg Sb/ kg ww to 434 000 µg Sb/kg ww.

Except for glass production there are no measured data relevant for comparison with the predicted local PECs.

Glass production

The predicted local $PECs_{sec.\ poisoning}$ using a BCF of 40 for glass production is 45 and 30 µg Sb/kg ww. Measured concentrations 4 km downstream a glass production site in perch and pike liver were 6.7 and 4.3 µg Sb/kg ww, respectively. The concentrations in whole fish are expected to be similar (see Table3-82).

Conclusion

Using a BCF of 40 gives a fairly good prediction of $PEC_{sec.\ poisoning}$ even though it seems to over predict the concentrations in fish at least for glass production. Using a BCF of 15 000 gives totally unrealistic predictions of concentrations in fish. The BCF of 40 will therefore be used in the risk characterization.

Terrestrial organisms

The predicted concentrations in earthworm range from 0.17 mg Sb/kg ww to 1.7 mg Sb/kg ww. Measured concentrations of antimony in soil and invertebrates at locations close to an antimony smelter in UK have been measured. The highest antimony concentrations were found in earthworms (*oligochaeta*). The dry weight concentrations (including gut contents) were 398 ± 94 mg Sb/kg, 213 ± 65 mg Sb/kg and 109 ± 28 mg Sb/kg 100 m, 250 m and 450 m downwind of the smelter. Recalculation to wet weight concentrations using an approximate conversion factor of 0.1 results in 40, 21, and 11 mg Sb/kg ww 100 m, 250 m and 450 m downwind of the smelter, respectively.

Conclusion

The measured data indicate that for heavily contaminated soils the terrestrial $PEC_{sec.\ poisoning}$ may be underestimated. The measured data from the antimony smelter in the UK are however not directly comparable to the predictions for site P1 performed in this RAR.

3.1.4.8 Exposure assessment for the marine environment

3.1.4.8.1 Calculation of predicted environmental concentrations (PEC)

Predicted local environmental concentrations in the marine environment have been calculated with EUSES 2.0.3.

The ambient background concentrations used as $PEC_{regional}$ are 0.2 µg Sb/l for marine water and 0.65 mg Sb/kg wwt (3 mg/kg dw) for marine sediment (see section 3.1.4.8.2).

The TGD suggests that the local concentration in the sediment is predicted based on the local concentration in water as:

$$PEC_{local\ sediment} = Kp_{suspended\ matter} * PEC_{local\ water}$$

In analogy with the risk assessment for Cadmium, an alternative approach has been used in this risk assessment. This approach use a measured regional PEC for the sediment to which a local fraction (proportional to the local concentration in water) is added. This option has been preferred because of the preference for measured rather than predicted sediment concentrations, and because the contribution of the local discharge to the sediment concentrations is taken into account via suspended matter Kp in line with the TGD. Local $PEC_{sediment}$ is then calculated according to the equation below:

$$PEC_{local}(ww) = PEC_{regional}(ww) + kp_{SPM} * C_{localwater} \frac{Fsolid_{susp} * RHOsolid}{RHOsusp}$$

where

PECregional =0.65 mg Sb/kg ww, which is the chosen background for sediment based on measured data

Concentration in surface water during emission period (dissolved); $C_{local\ water}$ = From EUSES [mg/kg]

Fraction solid in suspended matter; $Fsolid_{susp}$ = 0.1 [$m_{solid}^3 x\ m_{susp}^{-3}$]; default value from TGD

Bulk density of solid phase; RHOsolid = 2500 [$kg_{solid} x\ m_{solid}^{-3}$]; default value from TGD

Bulk density of (wet) suspended matter; RHOsusp = 1150 [kg x m^{-3}]; default value from TGD

Partition coefficient suspended matter; $Kp_{SPM\ median} = 10^{3.65}$ [l/kg]; $Kp_{SPM\ 10P} = 10^{3.1}$ [l/kg]; $Kp_{SPM\ 90P} = 10^{4.2}$[l/kg] ; see 3.1.3.1.4

$\frac{Fsolid_{susp} * RHOsolid}{RHOsusp}$ *converts the local fraction added from dry weight to wet weight*

As a sensitivity analysis, the PEC for local sediment has been calculated for three values of Kp suspended matter (see section 3.1.3.1.4), the median, the 10P and the 90P of available Kp_{spm} data. As the selection of value of Kp_{spm} influences the local concentration in water, a new calculation of the local concentration in water has been made for each Kp_{spm}, and the

values pertinent for the relevant Kp_{spm} has been used in the calculation of PEC for local sediment.

The PECs calculated with the median value of Kp suspended matter has been chosen for the risk characterisation. The median value has been chosen rather than the 90-percentile to avoid using worst-case values for several independent parameters influencing the same PEC, as this could potentially give a too extreme worst case. In this case the median of the 90-percentile of monitored environmental concentrations of antimony in sediment has been selected as a regional background PEC, and therefore the median value of $Kp_{suspended\ matter}$ (which also has a major impact on the local PECs for sediments, see resulting PECs in Table3-87 below) has been used for the risk characterisation.

For secondary poisoning PECs for top predators do not need be calculated, since both BMF_1 and BMF_2 are 1 (BCF = 40 or 15000), which means that the possible risks to top predators are covered already by the risk assessment for predators. The for $PEC_{top\ predator}$ have been included for reference however.

The PEC for water, sediment and secondary poisoning are summarised in Table3-87 below.

Table3-87 Estimated PECs for antimony for the local marine risk assessment.

Scenario	$PEC_{local, seawater}$ (µg/l)	$PEC_{local, sed}$ (mg/kg ww)			**$PEC_{oral\ predator}$ (mg/kg)**		**$PEC_{top\ predator}$ (mg/kg)**	
		Kp_{spm} 10P	**Kp_{spm} median**	*Kp_{spm} 90P*	**BCF = 40**	**BCF = 15000**	**BCF = 40**	**BCF = 15000**
Production site, P1	-	-	-	-	-	-	-	-
Production site, P5*	0.23	*0.66*	0.68	*0.74*	0.0084	3.14	0.0081	3.03
Plastic and rubber – formulation	1.00	*0.88*	1.42	*3.02*	0.021	7.91	0.010	3.98
Plastic and rubber – formulation assuming release to air	0.20	*0.65*	0.65	*0.65*	0.0080	3.0	0.0080	3.0
Plastic and rubber – industrial use	0.48	*0.73*	0.92	*1.48*	0.013	4.72	0.0089	3.34
Textiles – formulation generic site	2.98	*1.45*	3.35	*8.92*	0.0514	19.2	0.017	6.25
Textiles – formulation, site AMI19U	-	-	-	-	-	-	-	-
Textiles – formulation/processing, site FT-2	-	-	-	-	-	-	-	-
Textiles – formulation, site 96	-	-	-	-	-	-	-	-
Textiles – industrial use, generic site	13.9	*4.59*	13.95	*41.31*	0.23	87.7	0.053	19.9
Textiles – industrial use, site RC74U	-	-	-	-	-	-	-	-
PET - formulation	0.9	*0.84*	1.29	*2.61*	0.019	7.04	0.010	3.81

Scenario	$PEC_{local, seawater}$ (µg/l)	$PEC_{local, sed}$ (mg/kg ww)			**$PEC_{oral\ predator}$ (mg/kg)**		**$PEC_{top\ predator}$ (mg/kg)**	
		Kp_{spm} 10P	**Kp_{spm} median**	*Kp_{spm} 90P*	**BCF = 40**	**BCF = 15000**	**BCF = 40**	**BCF = 15000**
PET – industrial use	0.20	*0.65*	0.66	*0.67*	0.0080	3.0	0.0080	3.0
Paint FR – formulation	0.26	*0.67*	0.71	*0.83*	0.0090	3.37	0.0082	3.07
Paint pigment - formulation	0.28	*0.67*	0.72	*0.87*	0.0092	3.47	0.0083	3.09
Paint – formulation site AMI3G	-	-	-	-	-	-	-	-
Glass – formulation	0.20	*0.65*	0.65	*0.65*	0.0080	3.0	0.0080	3.0
Glass – industrial use ProcGlass 2**	0.44	*0.72*	0.67	*1.35*	0.0096	3.6	0.0083	3.1
Glass – industrial use ProcGlass 3	-	-	-	-	-	-	-	-
Regional (based on monitoring data***)	0.20	*0.65*	0.65	*0.65*				

- No emissions to seawater

* Not known if the the recipient is marine- or fresh water

** Not located by the sea, included as a theoretical realistic worst case

*** The regional value for marine sediment is based on monitored data for freshwater sediment

3.1.4.8.2 Measured levels

Levels in marine water

Measured antimony concentrations in marine water are presented in Table3-88.

The concentrations measured in marine waters, adjacent to the EU, are normally within the range of 0.20-0.40 µg Sb/l. (Filella et al., 2002a) concluded that the concentration of antimony in oceans is about 0.2 µg Sb/l, and that the decrease in antimony concentrations as well as in data scatter with time reflect the parallel improvement of the analytical techniques available. The relatively lower concentrations measured in the brackish waters of the Baltic Sea have been attributed to a higher particle flux and longer water residence time, compared to the oceans (Andreae and Froelich, 1984).

It is reasonably to think that at least some of the Sb in marine waters is due to anthropogenic activity, even though the importance of it is substantially lower as compared to the freshwater discussed above. For the present risk assessment an (high) estimate of 0.2 µg Sb/l was chosen as a ambient background of antimony in marine waters (dissolved).

In conclusion, the ambient background concentration of antimony in marine water (dissolved) used in this RAR is 0.20 µg Sb/l.

Table3-88. Measured antimony concentrations in marine waters adjacent to Europe.

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Adriatic Sea (north)	0.19-0.53 (mean 0.31)			Precipitation (MnO2) + NAA 5 locations	Strohal et al., 1975
Atlantic Ocean (north west)	0.24			Precipitation (sulphide) + NAA	Schutz and Turekian, 1965
Atlantic Ocean, Portuguese coast	0.16		26-04-82	HG-AAS Surface water	Andreae, 1983
Atlantic Ocean (50-70°N)	0.0037-0.023 0.12-0.15 (mean 0.14)	Sb(III) Sb(V)		Preconcentration of hydrides on active carbon + NAA 0-4000 m	Middelburg et al., 1988
Atlantic Ocean, French coast off Nantes Mouth Loir	 0.21 0.22 0.003	 Total Total Sb(III)		HG-AAS	Takayanagi and Michel, 1996
Atlantic Ocean (54-68°N)	 0.23 0.16 n.d. 0.16 n.d. n.d.	 Total Sb(III) MSA Total Sb(III) MSA	1993	HG-GC-PID Surface water Deep water	Cutter and Cutter, 1998
Baltic Sea Station BY5 Station BY11 Station BY15 Station BY23 	 0.038-0.11 0.0007-0.0097 <0.0007 0.039-0.083 0.018-0.073 0.0002-0.019 0.020-0.068 0.0016-0.0050 <0.0007 0.043-0.095 <0.0006-0.0085	 Total MSA DMSA Total MSA DMSA Total Sb(III) Sb(V) MSA DMSA Total MSA	 11-06-81 12-06-81 13-06-81 15-06-81 	HG-AAS 10-95 m, 10 samples 10-210 m, 15 samples 10-235 m, 19 samples 10-65 m, 6 samples 	Andreae and Froelich, 1984

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Station BY26	<0.0012-0.0049 0.0006-0.0011 0.063-0.083 <0.0006-0.011 <0.0012 0.046-0.067	DMSA Total Sb(III) Sb(V) MSA DMSA	14-06-81,	10-90 m, 8 samples	
Dutch Wadden Sea	0.18-0.35		April 1978	HG-NAA 12 stations	Van der Sloot et al., 1985
Irish Sea	0.13-0.40 (median 0.22)			Precipitation (MnO_2) + solvent extraction (CCl_4) + colorimetry (rhodamine B)	Portmann and Riley, 1966
Irish Sea	0.26			Precipitation (zirconium oxide) + colorimetry (ion pair $SbCl_6$ with crystal violet) Surface water	Abu-Hilal and Riley, 1981
Gullmar fjord, Sweden	<0.50			Spectrophotometry Shallow water	Noddack and Noddack, 1939b
Mediterranean (east) Tyro Basin Bannock Basin	 0.20-0.24 0.0073-0.044 0.18-0.21 0.34-0.85 0.11-0.64 0.20-0.32 0.20-0.24 0.023-0.037 0.16-0.21 0.67-0.77 0.39-0.65 0.12-0.28	 Total Sb(III) Sb(V) Total Sb(III) Sb(V) Total Sb(III) Sb(V) Total Sb(III) Sb(V)	 1987, 21/5-5/6 1987, 21/5-5/6	HG-NAA 10-3347 m (oxic) 3579-3563 m (anoxic) 2-3300 m (oxic) 3320-3470 m (anoxic)	Van der Weijden et al., 1990
Mediterranean, Spanish coast	 0.14 n.d.	 Total Sb(III)		FIA-HG-AAS Industrial area	De la Calle Guntinas et al., 1991
Mediterranean ,				HPLC-HG-ICP-AAS	Smichowski et al.,

Location	Concentration (Sb µg/l)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Spanish coast Santander Denia	 n.d. 0.25 n.d. 0.29	 Sb(III) Sb(V) Sb(III) Sb(V)			1995
Mediterranean (west)	 0.16 0.22 0.16	 Total Sb(III) Total		HPLC-HG-ICP-AAS	Takayanagi et al., 1996
Mediterranean, Open-ocean	0.16 0.0021	Total Sb(III)		HG-AAS	Takayanagi and Michel, 1996
North Sea	0.30			HG-AAS	Haring et al., 1982
North Sea, Belgian coast	0.30-0.82			HMDE-DPASV 3 m, 5 samples	Gillain et al., 1979)
North Sea, Belgian coast	0.12 0.16			RRDE-ASV HMDE-DPASV 3 m, 5 samples	Brihaye et al., 1983
North Sea, Belgian coast	0.050-0.38 (median 0.12) <0.005-0.039 0.015-0.25	Total Sb(III) Sb(V)	1983, July	RDE-ASV, HMDE-DPASV 19 stations	Gillain and Brihaye, 1985
North Sea, Southern Bight and British Channel	0.20-0.37		1981, October	HG-NAA 10 stations	Van der Sloot et al., 1985
Ocean average	0.33			Precipitation (sulphide) + NAA	Schutz and Turekian, 1965

Levels in marine sediment

Measured antimony concentrations in marine sediment are presented in Table3-89.

Cato (1997) presented measurements from upper (0-2 cm), and deeper (40-55 cm) marine sediments from the Skagerrak and Kattegat Seas, which form a transition area between the brackish Baltic Sea in the east and the North Sea in the west (which has an almost oceanic salinity). However, no conclusion can be drawn from the fact that the reported concentration increased with increasing depth since some elements (not described which) were reported to display large variations between the two sea areas and within different regimes of each of the seas.

The information on antimony concentrations in marine sediments is scarce, why the concentration chosen for freshwater sediments will be used for the marine sediments as well.

In conclusion, the ambient background concentration of antimony in marine and freshwater sediments used in this RAR is 3 mg Sb/kg dw corresponding (with EUSES default conversion factors) to **0.65 mg Sb/kg ww.**

Table3-89. Measured antimony concentrations in marine sediments.

Location	Concentration (Sb mg/kg$_{dw}$)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
Greece Upper Saronikos Gulf	0.16-18 (min-max)		1973	NAA Athens sewage outfall area 20 samples Lower concentrations further away from outfall area ~20 cm maximum penetration	Papakostidis et al., 1975
Upper Saronikos Gulf Elefsis Bay Keratsini Bay Piraeus Harbour	0.4 2 15 65		1975, beginning of	NAA 0-5 cm Background level Industry discharge Athens outfall Fertilizer factory	Grimanis et al., 1977
Sweden S Baltic Sea and the Stockholm Archipelago	1.22 (median) 0.16-8 (min-max)				Cato, In prep.
Skagerrak and Kattegat	0.68 (mean) 0.66 (median) 0.2-2.14 (range) n=149 0.88 (mean) 0.83 (median) 0.4-2 (range) n=70		1987-1992	ICP-MS Marine sediments 0-2 cm 40-55 cm	Cato, 1997
Baltic Sea	0.8 (median) 0.16-4.7 (min-max)			Taken at 0.55 m depth, brackish water	Naturvårdsverket, 1999a

Location	Concentration ($Sb\ mg/kg_{dw}$)	Speciation (Total, unless otherwise specified)	Period	Remark	Reference
North Adriatic Sea	20			NAA, Under strong influence of the North Italian industrial region, open sea sample taken at 35 m depth	Strohal et al., 1975

Levels in marine biota

Information on measured antimony concentrations in marine biota is presented in Table3-90.

(Sternbeck et al., 2002a) performed a screening study on livers of Herring caught on the East coast of Sweden in the brackish Baltic Sea, and on the West coast of Sweden in the marine Skagerrak Sea (see Table3-82 below). The liver samples originated from samples taken 1990 and 1999. No geographical trends could be seen during 1990, while the concentrations during 1999 were higher in the samples from the marine Skagerrak Sea. No trend in time could be concluded from the limited material.

Concentrations in the North Sea in fish muscle have been reported to be about 8 µg Sb/kg ww (Vadset, pers. communication in Sternbeck et al., 2002a).

Maher (1986) measured antimony in marine organisms and waters from South Eastern Australia using HG-AAS. Water was filtered through a 0.45 µm filter. Samples of macroalgae were washed with distilled water to remove salts, freeze dried and ground to pass a 200 µm sieve. Animals were separated into component tissues and composites prepared by combining the tissues of five freeze dried specimens of each sample.

In a large survey by (Hall et al., 1978) trace element levels, including antimony, were determined in tissues of finfish, Mollusca, and crustacean taken from 198 sites around the coastal United States, including Alaska and Hawaii. Muscle was analysed from 159 species of finfish, liver from 82, whole fish from 17, Mollusca from 18, and crustacean from 16 species. The mean levels of antimony in most finfish muscles and livers fell in the range of 0.5-0.9 mg/kg. Most species of whole finfish had antimony levels between 1.0 and 3.0 mg/kg. Most shellfish species displayed mean antimony levels between 0.8 and 1.0 mg/kg. The report does not clearly state on what basis (dry weight, wet weight etc.) the reported concentrations are given. The rapporteur assumes that they are given on a wet weight basis. However, the high concentrations reported in this study can be questioned when comparing them with all other measurements of antimony in fish including freshwater fish which are orders of magnitudes lower.

Table3-90 Concentrations of antimony in marine organisms.

Species	Location	Concentration (μg Sb/kg)	Period	Remark	Reference
Herring	Baltic Sea			ICP-MS Liver	Sternbeck et al., 2002a
	Bergsjöfjärden	0.3 μg Sb/kg ww	1990		
	Landsort	0.25 μg Sb/kg ww			
	Utlängan	0.28 μg Sb/kg ww			
	Bergsjöfjärden	0.1 μg Sb/kg ww	1999		
	Landsort	<0.11 μg Sb/kg ww*			
	Utlängan	0.11μg Sb/kg ww			
	Skagerrak Sea				
	Väderöarna	0.36	1990		
		0.57	1999		
Fish (no further information given)	North Sea	8 μg Sb/kg (ww)		Muscle	Vadset, pers. communication in Sternbeck et al., 2002a
	marine waters South Eastern Australia		1986		Maher, 1986
<u>Algae</u>					
Ecklonia radiata		94 ± 5 (dw)			
Sargassum lacerifolium		121 ± 6 (dw)			
Ulva sp.		193 ± 8 (dw)			
<u>Mollusc tissues</u> (*Mytilus edulis planulatus*)					
Mantle		31 ± 3 (dw)			
Visceral mass		47 ± 4 (dw)			
Abductors		60 ± 5 (dw)			
<u>Crustean tissues</u> (*Helograpsus sp.*)					
Muscle		18 ± 2 (dw)			
Visceral mass		116 ± 6 (dw)			
<u>Pisces (muscle)</u>					
Arripus georgianus		< 9 (dw)			
Semir hamphus australis		10 ± 1 (dw)			
Stillaginodes punctatus		< 9 (dw)			
	198 sites around the coastal United States		1971		Hall et al., 1978
<u>finfish</u>		500 - 900			
muscle & liver					
whole fish		1000 - 3000			
Mollusca, and crustacean		800 and 1000			

* Below detection limit

3.1.4.9 Comparison between predicted and measured levels

Water and sediment

The predicted local concentrations of antimony in marine water range from background concentrations, 0.2 µg/l to 14 µg/l. Measured concentrations range from about 0.1 µg Sb/l to 0.8 µg Sb/l (see Table3-88). However, there are no measured data comparable to the estimated local PECs.

The predicted local concentrations of antimony in marine sediment range from 0.65 mg Sb/kg ww to 14 mg Sb/kg ww. The information on antimony concentrations in marine sediments is scarce. Measured concentrations range from 0.04 µg Sb/kg ww to 14 mg Sb/kg ww. The highest concentrations were measured outside a fertilizer factory.

The predicted local PECs are used in the risk characterisation in the absence of representative measured data.

Marine organisms

Measured concentrations in marine fish range from 0.1 µg Sb/kg ww to 3000 µg Sb/ kg ww.

Predicted $PEC_{oral\ predator}$ range from 8 µg Sb/kg ww for sites with no local emission to water to 230 µg Sb/kg ww in local scenario with the highest predicted emission to water using a BCF value of 40. Using a BCF value of 15000 results in PEC values ranging from 3 000 µg Sb/ kg ww to 87 700 µg Sb/kg ww.

Predicted $PEC_{oral\ top\ predator}$ range from 8µg Sb/kg ww for sites with no local emission to water to 53 µg Sb/kg ww in the local scenario with the highest predicted emission to water using a BCF value of 40. Using a BCF value of 15000 results in PEC values ranging from 3 000 µg Sb/ kg ww to 19 900 µg Sb/kg ww.

Conclusion

Using a BCF value of 15000 gives unrealistic predictions of concentrations in prey for predators and top predators. The BCF of 40 will therefore be used in the risk characterization.

3.2 EFFECTS ASSESSMENT: HAZARD IDENTIFICATION AND DOSE (CONCENTRATION) - RESPONSE (EFFECT ASSESSMENT)

3.2.1 Antimony toxicity

As described in the section on environmental fate, antimony prevails as Sb(III) and Sb(V) in the environment. Both Sb(III) and Sb(V) ions hydrolyse easily, and Sb(III) is present as the neutral species $Sb(OH)_3$, and Sb(V) as the anion, $Sb(OH)_6^-$. According to thermodynamic calculations, antimony should almost exclusively be present as Sb(V) in oxic systems, and as Sb(III) in anoxic systems. Even though the dominant species in oxic waters is Sb(V), Sb(III) has been detected in concentrations much above what is predicted, and the reverse is true for Sb(V) in anoxic systems.

The toxicity of antimony is expected to be exerted through its ions.

3.2.1.1 Differences in compounds used in the toxicity tests

When using an Sb-compound, other then Sb_2O_3, as a source of Sb in toxicity testing, an increase in counter ions and protons will occur which may influence the interpretation of an observed response.

How much protons that are produced, and if this will affect the pH or not depends on the conditions in the test system, the type of Sb-compound used, and the amount of the Sb-compound added. Below is this process illustrated using the easily soluble $SbCl_3$. The last step, not included in the equations, is the oxidation of the trivalent $Sb(OH)_3$ to the pentavalent $Sb(OH)_6^-$, which is the dominating specie during oxic conditions.

$$SbCl_3 + 3H_2O \rightarrow Sb(H_2O)_3Cl_3 \rightarrow Sb(H_2O)^{3+} + 3Cl^- \rightarrow$$
$$Sb(H_2O)_2OH^{2+} + H^+ \rightarrow Sb(H_2O)(OH)_2^+ + H^+ \rightarrow Sb(OH)_3 + H^+$$

The more sparingly dissolvable Sb_2O_3 can also be used as a source of Sb, but due to the O-Sb bonds in the crystal structure that needs to be broken, less Sb per time unit will be dissolved. In order to remove an Sb atom from the crystal structure, a bond first needs to be established between the oxygen in a water molecule and an Sb in the crystal structure. Secondly the bond between an oxygen atom in the crystal structure and the Sb needs to be broken. Since each Sb is bound to three oxygen atoms in the crystal structure, three water molecules are needed to remove one Sb atom. In the process of removing an Sb atom, each of the remaining oxygen atoms in the crystal structure to which the Sb atom originally were bound, will take one proton from the water molecule responsible for breaking that particular crystal structure O-Sb bond. The process of removing Sb, from the crystal structure will thus not result in any net change in the amount of protons. The last step, not included in the equations, is the oxidation of the trivalent $Sb(OH)_3$ to the pentavalent $Sb(OH)_6^-$, which is the dominating specie during oxic conditions.

$$Sb_2O_3 + 3H_2O \rightarrow Sb_2O_3(H_2O)_3 \rightarrow Sb(OH)_3$$

An important difference between different antimony compounds, as regards the toxicity observed in toxicity tests, will be the time needed to dissolve and liberate ions able of exerting toxicity. Antimony chlorides, being more soluble, will dissolve faster than oxides, not saying that the latter will not dissolve, just that the liberation of ions will be slower for oxides. Dissolution of antimony compounds will be faster in the aquatic compartment, as compared to the terrestrial compartment. (Vangheluwe et al., 2001) performed transformation experiments in two different soils (one sandy soil and one loamy soil) with the antimony compounds $SbCl_3$ and Sb_2O_3, both at the concentration 50 mg Sb/kg dw. After two weeks, only 10 to 15 % of the Sb_2O_3 was transformed to a soluble form, and after 12 weeks almost half was transformed, which shows that the oxide dissolves and results in soluble forms of Sb. It is thus likely that the difference in solubility between the salt and the oxide will disappear in the long run. As previously described (see section 3.1.3.1.3), the sorption reactions seem to be more important than the dissolution processes as regards the fate of antimony in soil.

Since most toxicity tests of antimony are performed under oxic conditions using trivalent antimony compounds such as $SbCl_3$ and Sb_2O_3, the toxicity exerted will, depending on the duration of the test, reflect a mixture of toxicity from trivalent and pentavalent antimony ions. In the relatively rare cases in this RAR where a compound with pentavalent antimony has been used (oxic conditions), the toxicity observed will most probably only reflect the toxicity of the pentavalent $Sb(OH)^-_6$.

3.2.1.1.1 Aquatic toxicity tests

It has been reported that the use of high concentrations of $SbCl_3$, dissolved in water, have resulted in precipitation of antimony (Brooke et al., 1986). What is likely being observed is an initial formation of chloroantimonate(III) species, which in aqueous solutions are weak, and hydrolyses to oxychloride (SbOCl), which has low solubility in water, and is further hydrolysed to Sb_2O_3 (Filella et al., 2002b; Filella and May, 2003). The percentage of Sb lost from solution in the study by (Brooke et al., 1986), due to precipitation, during a 96-h test ranged from a low 6% for the lowest exposure concentration (nominally 25 mg Sb/l) to 76% for the highest concentration (nominally 50 mg Sb/l). The apparent trend was to loose a greater percentage of antimony from solution with increasing nominal concentration. The authors improved the methods by introducing mixing and filtering to remove the precipitate from the solution. The maximum concentration maintained in a solution without organisms for 96 h was 35.0 mg Sb/l, using a nominal concentration of 250 mg Sb/l. Nominal bioassay concentrations at 50 mg Sb/l and 100 mg Sb/l using Sb_2O_3 resulted in the measured soluble concentrations of 3.4 and 5.0 mg Sb/l, respectively. Solutions of Sb_2O_3 in lab water at the nominal concentrations of 28, 58, and 110 mg Sb/l had the measured dissolved concentrations during the 96 h experiment period of 1.9, 2.6, and 3.3 mg Sb/l, respectively.

As obvious from the study above, nominal concentrations may be very misleading as regards the actual amount of dissolved antimony. At equal nominal concentration, easily soluble compounds like $SbCl_3$ results in more dissolved antimony during the exposure periods used in toxicity experiments than less soluble compounds like Sb_2O_3. However, even the easily soluble compound $SbCl_3$ may at high nominal concentrations result in reduced amount of soluble antimony.

Filella and May (2003) conducted a critical review of all available thermodynamic data on antimony and developed a computer speciation model of antimony in multi-component

solutions, representative for different environmental conditions. Based on the limited data set and the subsequent speciation calculations it was shown that antimony is exclusively present as the pentavalent $Sb(OH)^-_6$ in oxic freshwater systems and as the trivalent $Sb(OH)_3$ in anoxic conditions, at all pH values of environmental relevance for aquatic systems. The formation of chloride-antimony species does not appear to be of importance under environmentally relevant conditions as no Sb(III)-chloride was observed under seawater conditions, and the concentration of possible Sb(V)-chloride could not be calculated due to scarcity on data. The very few studies available on Sb(V)-chloride binding had been performed under extremely acidic conditions to prevent hydrolysis and could thus not be used, as it was difficult to establish the strength of such interactions under dilute conditions relative to other antimony species and accordingly no thermodynamic relationship of this kind has been published.

There is, based on the available information on antimony known by the rapporteur, nothing that indicates that the difference observed in toxicity in aquatic systems between different inorganic antimony compounds of the same valence, such as for instance $SbCl_3$ vs. Sb_2O_3, would be due to different antimony species exerting different degrees/kind of toxicity. Instead, an observed difference in toxicity at equal nominal doses of antimony is most probably just a reflection of differences in solubility, which means that a more soluble antimony compound will result in more dissolved antimony able of exerting toxicity. It may however at higher concentrations of Sb-compounds also reflect an increased presence of counter ions and/or protons.

3.2.1.1.2 Terrestrial toxicity tests

The knowledge on antimony toxicity and the factors modifying it in soil is limited (as is the knowledge on antimony toxicity in the terrestrial compartment as such).

During the slow transformation of Sb_2O_3 in soil into soluble Sb, the bioavailable Sb concentration (i.e. Sb in soil solution) will gradually increase and there will, as a consequence of that, not be a constant “toxic pressure” during the exposure period. As long as full equilibrium not is reached, the total Sb toxicity will be underestimated.

However, the use of soluble Sb salts for toxicity testing introduces other changes in soil chemistry, besides increasing the concentration of Sb, such as increased concentrations of counter ions and protons. These changes in soil chemistry also have the potential to affect the observed toxic response.

At TC NES I ´07 it was concluded that the preferred exposure regime for terrestrial toxicity studies is sufficiently aged soil spiked with Sb_2O_3. This would ensure i) a constant Sb pore water concentration (i.e. constant toxic pressure) during the entire test period and ii) avoids toxic effects due to increased concentrations of counter ions (e.g. chloride, sulphate) and/or protons.

3.2.1.2 Differences in valence

The initial proportion of the two valences initially may differ in a toxicity test, due to the antimony compound used, from what would have been observed in the environment under same conditions (e.g. oxic system, Eh/pH, etc.). However, their relative proportion will with

time change to the proportions observed, i.e. predominantly Sb(V) in oxic systems and predominantly Sb(III) in anoxic systems.

In toxicity tests performed under oxic conditions, most trivalent antimony will thus sooner or later oxidize to pentavalent antimony, and the opposite will occur in tests performed under anoxic conditions, i.e. most pentavalent antimony will be reduced to trivalent antimony. The extent of this transformation from trivalent antimony to pentavalent antimony will depend on the physico-chemical water conditions, including the presence of organic material (which may delay the oxidation process), but also on the experimental protocol used (static, flow-through, etc.).

Rates of transformation from trivalent to pentavalent antimony and the reverse in natural waters are not particularly well studied (see section 3.1.3.1.2). Based on the toxicity data available for antimony an emphasis will here be put on the oxic system and the oxidation of Sb(III) to Sb(V). No systematic studies on Sb(III) oxidation kinetics under natural water conditions occur. Published observations seem to indicate that, at natural antimony levels, Sb(III) is gently oxidised to Sb(V): By adding known amount of both species (2 µg Sb/l) to lake and waste water, it was observed that Sb(III) appeared unstable since none could be detected after standing for 6 h (Sun et al., 1982). Krachler and Emons (2001) found that a freshly prepared aqueous standard solution containing 0.07 µg Sb(III)/l was easily oxidized within some hours to Sb(V). Methods of preserving trivalent antimony in oxic condition can be found in the literature, e.g. addition of organic substances such as tartaric acid (Sun et al., 1982), lactic, citric, ascorbic acids (de la Calle Guntinas et al., 1992) to natural or synthetic solutions which has been shown to have a stabilising effect on Sb(III). None of the toxicity studies included has taken such steps to preserve the trivalent form of antimony, nor have any measurements on species differentiation been performed in order to assure that the observed or non-observed effects would be correlated to a specific state of valence.

Toxicity results obtained from studies using pentavalent antimony compounds performed under oxic conditions probably reflects the toxicity of pentavalent antimony only, since the possible presence and contribution of toxicity of trivalent antimony during the test appears to be negligible.

There are, at least to our knowledge, at present no available toxicity studies which also include redox speciation measurements. Since trivalent antimony will oxidize to pentavalent antimony (in an oxic environment), any conclusions on differences in toxicity between tri- and pentavalent antimony, based on studies without information on measured redox speciation, will therefore become speculative.

In the literature, trivalent antimony has been described as being more toxic than pentavalent antimony (Sloof et al., 1992). However, since the number of studies including comparative toxicity data on both trivalent and pentavalent antimony (with similar solubility) in general is rare, and for reliable studies in particular, is very rare, any firm conclusion on differences in toxicity between the two valences is difficult to make. The studies by (Appelgate et al., 1957), using $SbCl_3$ and $SbCl_5$ when testing the toxicity on the freshwater species *Lepopomis machrochirus, Oncorhynchus mykiss,* and *Petromyzon marinus* (using nominal concentrations), and by MacPhee and Ruelle (1969), using $SbCl_3$ and $SbCl_5$ when testing the toxicity on the freshwater species *Oncorhynchus kisutch, Oncorhynchus tshavrytscha*, and *Ptychocheilus oregonensis* (using nominal concentrations), indicate similar toxicity in freshwater. The study by He and Yang (1999), who studied the effect on antimony on rice during germination and growth using the trivalent antimony potassium tartrate ($2KSbOC_4H_4O_6 \cdot H_2O$) or the pentavalent antimonite ($KSb(OH)_6$), indicate that trivalent

antimony are somewhat more toxic then pentavalent antimony, which is also in accordance with the results by Arzamastev (1964), who studied the acute toxicity of $SbCl_3$ and $SbCl_5$ on rat and Guinea pig using oral exposure. The resulting LD_{50} values for the trivalent antimony was 360 and 305 mg Sb/kg bw for rat and Guinea Pig, respectively, and for the pentavalent antimony 455 and 365 mg Sb/kg bw for rat and Guinea pig, respectively.

The study by Takayanagi (2001), using $SbCl_3$, $SbCl_5$, and $K[Sb(OH)_6]$ when testing the toxicity on the marine fish *Pargus major*, indicate that the pentavalent antimony, especially when originating from $SbCl_5$, may be more toxic. The results also indicate that Sb(III) and Sb(V) when originating from $SbCl_3$ and $K[Sb(OH)_6]$, respectively, are of similar toxicity (difference less than a factor of two), agree with the results from literature of a "similar degree" of toxicity. The toxicity value for the trivalent antimony compound $SbCl_3$ is in addition well in line with result using the same compound on freshwater species, with LC_{50} values of 12.4 mg Sb/l and 14.4 mg Sb/l for marine and freshwater, respectively. However the difference in toxicity between the $SbCl_3$ and $K[Sb(OH)_6]$ on one hand, and $SbCl_5$, on the other, deviates on two issues from what can be expected. Firstly, the large difference (13 times) in toxicity between a trivalent and a pentavalent antimony compound of similar solubility, and secondly the large difference (more than a factor of seven) between Sb(V) originating from two pentavalent compounds which both dissolve and had their concentrations measured. These two differences have not found any support in the literature. It is therefore decided not to use the results from the $SbCl_5$ by Takayanagi (2001) in this RAR.

(Flynn et al., 2003) studied the difference in toxicity between tri- and pentavalent antimony using the lux-marked bacterial biosensors *Escherichia coli* HB101 and *Pseudomonas fluorescens*. The bacterial biosensors were during 15 minutes exposed to a range of trivalent (added as KSbO-tartrate, i.e. $C_4H_4O_7SbK0.5H_2O$) or pentavalent ($KSb(OH)_6$) antimony concentrations in deionised water at pH 5.5. Deionised water at pH 5.5 was used as control. The pentavalent antimony did not yield bioluminescence up to nominal concentrations of 100 mg Sb/l, while the trivalent antimony nominal concentration yielding 50% inhibition (EC_{50}) ranged between 0.7 (*E. coli* HB101) and 5.7 mg Sb/l (*P. Fluorescens*). Despite the fact that only nominal concentrations were used and information on statistics is lacking, this information is still valuable, as it can be expected that most of the antimony originating from KSbO-tartrate remained trivalent during the short exposure period (15 minutes), which indicate that trivalent antimony is more toxic than pentavalent, at least to bacteria.

To conclude, since the results of toxicity studies using a trivalent antimony compound probably to a lesser or larger extent is the result of a mixture of trivalent and pentavalent antimony ions, and there are no conclusive evidence supporting a significant difference in toxicity between the two valences, it is decided not to differentiate between relevant and reliable toxicity results originating from tri- or pentavalent antimony studies.

3.2.1.3 Influence of abiotic factors on the toxcity

Based on the content in the environmental fate (see section 3.1.3) bioavailability is, at least to some degree, expected to be reduced in sediment and soil, mostly due to the presence of aluminium, iron and manganese hydrous oxides. Reduced toxicity of antimony, due to possible sorption, may therefore exist in soils and sediments with higher contents of these hydrous oxides, as compared to soils and sediments with lower contents. Since the sorption has been shown to decrease with increasing pH, the antimony toxicity may also increase with increasing pH. There exists however no studies where the possible influence of presence of

hydrous oxides, pH, or other abiotic factors for that matter, on the toxicity of antimony have been studied. Therefore there is presently not enough reliable and relevant ecotoxicity data available in this RAR to perform a useful evaluation of the issue.

To conclude, it is at present not possible to include abiotic factors in order to adjust derived PNEC or PEC for bioavailability of Sb.

3.2.2 Sources of ecotoxicological data

The ecotoxicological data in this report are derived from original papers on the subject, published in international journals and from research projects. The toxicity data presented in the tables below either represent the total measured concentration of antimony or the nominal concentration of antimony.

3.2.3 Selection of ecotoxicological data

The toxicity tests are evaluated with regard to their adequacy, which addresses the reliability and relevance of test data, and completeness according to what is outlined in the TGD:

- Reliability: covering the inherent quality of a test relating to test methodology and the way that the performance and results of a test are described.
- Relevance: covering the extent to which a test is appropriate for a particular risk assessment.

The assessment of data adequacy involves a review of individual data elements with respect to how a study is conducted and how the results are interpreted in order to accept (or reject) a study in accordance with the purpose of the assessment. Only data considered reliable and relevant is considered valid for use in the risk assessment. A study considered reliable and relevant has the notation “R”, while studies considered not reliable and/or not relevant has the notation “NR” in the effect tables below. Note that studies considered reliable not necessarily are considered relevant for the purpose of this risk assessment.

3.2.3.1 Reliability

Standardised tests, e.g. according to OECD guidelines, are used as reference when test methodology and test conditions, performance and data/treatment/reporting are evaluated. Non-standardised test results may also be reliable, but require a more thorough check on compliance with reliability criteria(s) before being considered reliable.

A detailed description of the methods used in the study, measurements and observations performed should be provided. Test conditions should be suitable for the test organism. Minimum requirements (incl. maximum acceptable control mortality) for endpoints such as mortality, growth, and reproduction need to be fulfilled. Information on dose-response and statistics should also be presented.

3.2.3.2 Relevance

The toxicity data study relevant ecotoxicological parameters such as survival, growth and/or reproduction. Only an effect resulting from exposure of antimony is considered relevant for the effect assessment, i.e. a study will be rejected in case there exist an indication that impurities or other substances influence the observed response.

3.2.4 Aquatic compartment (incl. sediment)

As a consequence of the above, toxicity studies in the aquatic compartment

1. only presenting nominal antimony concentrations will not be considered reliable,
2. will not be rejected only based on which antimony compound that was used in the test, or wheter or not a tri- or a pentavalent compound was used, as long as the results are considered reliable and relevant.

3.2.4.1 Toxicity test results

The number of valid studies is limited. The results from only three studies, i.e. Kimbal (1978), TAI Inc. (1990), and LISEC (2001), provide all valid EC_{50}s and NOECs for fish, invertebrate, and algae.

Toxicity results for marine species are scarce. Only the results from Takayanagi (2001) are considered reliable (with the exception of the results using $SbCl_5$ which are not considered reliable).

None of the NOECs used to derive the PNECs in the aquatic compartment are considered to be confounded by the additions of counter ions (i.e. chloride) and/or protons resulting from the use of $SbCl_3$.

3.2.4.1.1 Fish

A total of 18 studies with 15 different species were found. However, most of these studies could not be used since they were not considered valid due to reasons of reliability (e.g. test concentrations above solubility of test compound, nominal concentrations only, and/or lack of information on test protocol).

The lowest valid toxicity data for fish are summarized in Table3-91.

Table3-91 presenting lowest valid LC_{50}, LOEC, and NOEC for fish.

Organism	Water	Exposure period	Endpoint	LC_{50}/LOEC/NOEC	Value (mg Sb/l)	Reference
Pimephales promelas	Fresh	4 d	Mortality	LC_{50}	14.4	Brooke et al., 1986
Pimephales promelas	Fresh	28 d	Growth (length)	LOEC	2.31	Kimball, 1978

Pimephales promelas	Fresh	28 d	Growth (length)	NOEC	1.13	Kimball, 1978
Pargus major	Marine	4 d	Mortality	LC_{50}	6.9	Takayanagi, 2001
----------	Marine	----------	----------	LOEC	---------	----------
----------	Marine	----------	----------	NOEC	---------	----------

Acute toxicity

Fresh water

Three valid studies of acute toxicity to freshwater fish considered valid are available, (i.e. Kimball 1978; Brooke et al., 1986; and TAI 1990).

In the study by Kimbal (1978), 8 week-old juvenile *Pimephales promelas* were exposed to trivalent antimony ($SbCl_3$) in a flow-through system, performed in duplicates with six concentrations (range: 1.0 - 27.6 mg Sb/l) and a control, where each group consisted of 10 fishes. The tests were performed with hard well water (more details on the test are provided in Table3-92 below). The resulting EC_{50}s for the 4 and 8 d exposures were 21.9 and 20.2 mg Sb/l, respectively.

In the study by (Brooke et al., 1986) juvenile *Salmo gairdneri* were exposed in a static test design to trivalent antimony ($SbCl_3$) for 4 days, and the LC_{50} were determined. The tests were performed in duplicates with two concentrations (11.4 and 25.7 mg Sb/l) and a control, where each group consisted of 10 fishes. The mortality in the highest of the two dose groups was 45 %, why no LC_{50} could be determined, and the study only resulted in larger-than values. This study also contained results of static tests performed on *Pimephales promelas*, which resulted in the LC_{50} 20.8 mg Sb/l, 17.4 mg Sb/l, and 14.4 mg Sb/l for the time periods 1 d, 2 d, and 4 d, respectively. These results are considered reliable even though no dose-response relationship was included since the methodology used is well described, the antimony concentrations were measured, the water characteristics remained within the tolerance limits of the test species, the estimated LC_{50}-values was within the range of the test concentrations used, and in addition the effect values presented were very similar to those reported by Kimbal (1978) using the same fish species. More details on the test are provided in Table3-92 below.

In the study by TAI Inc. (1990) juvenile *Ictalurus punctatus* were exposed to trivalent antimony ($SbCl_3$) in a static test design for 4 days, in moderately-hard reconstituted culture water. The tests were performed in duplicates with the five measured concentrations (nominal concentrations within brackets) 8.5 (15.6), 15.4 (31.25), 19.8 (62.6), 24.6 (125), and 21.22 (250) mg Sb/l, and a control with 0 mg Sb/l (measured concentration), where each group consisted of 10 fishes. The resulting LC_{50} was 24.6 ± 2.6 mg Sb/l.

Thus, the lowest valid value for acute toxicity to freshwater fish is 14.4 mg Sb/l for *Pimephales promelas*, which originates from the study by (Brooke et al., 1986).

The reason why the study by Curtis and Ward (1981) is considered unreliable, even though it in Table3-92 is indicated that the reported concentration was measured, is that the reported >-effect concentration not is considered to represent a dissolved antimony concentration. This conclusion is based on the fact that (i) the reported ">-than concentration by far exceededs the water solubility of the diantimony trioxide, (ii) the concentrations used are not presented, (iii)

the test substances in the study were added either directly or in the form of a stock solution in deionized water and the solutions were briefly stirred with a glass rod before a water sample was removed for analysis (i.e. no initial pretreatment of the Sb_2O_3 to ensure that it was properly dissolved before it was added to the test solution), (iv) initially water samples were not filtered before analysis, filtering through 0.45 -µm filters before analysis was performed at a later stage (unclear when and for which chemicals), and the fact that (v) it is specifically mentioned that the results of the analysis primarily were used for information about the physicochemical behaviors of the toxicants rather than for computing LC_{50}s (which cast some doubts whether or not the nominal or measured concentrations were used).

The reasons why the results by (Doe et al., 1987) on *Oncorhyncus mykiss* is considered unreliable, even though it in Table3-92 is indicated that the reported concentration was measured, are that there is no information presented on (i) the number of concentrations and which concentrations that were used, (ii) dose-response curves (no raw data available), (iii) number of replicates (if any?), and (iv) what statistics that was used to calculate the presented LC_{50}-values.

Marine water

There are three studies available on acute toxicity to marine fishes, as can be seen from Table3-92 below. However, only the study by Takayanagi (2001) is considered reliable.

In the study by Takayanagi (2001) 3 months-old *Pargus major* were exposed to trivalent ($SbCl_3$) or pentavalent antimony ($SbCl_5$ or $K[Sb(OH)_6]$) under static conditions with a control and an unknown number of concentrations (range: $SbCl_3$, 7.8-25.7 mg Sb/l; $SbCl_5$, 0.40-1.06 mg Sb/l; $K[Sb(OH)_6]$, 2.8-10.3 mg Sb/l), where each group consisted of eight fishes. The tests were performed using natural seawater, with a salinity of 33.7 ppt, passed through sand and activated-charcoal filters. Each aquarium was aerated. The pH was determined every day and the reported ranges were 4.9-7.8, 7.8-8.1, and 7.8-8.1, for $SbCl_3$, $SbCl_5$, and $K[Sb(OH)_6]$, respectively. For the $SbCl_3$ test, a decrease in the pH of the test solution was observed. Therefore, a low pH seawater prepared with HCl for use as control for the $SbCl_3$ test in order to assess pH effects. All test fish survived in the HCl-adjusted seawater, and therefore pH was considered a negligible factor, and the mortality found in the $SbCl_3$ dilution waters was considered to have been caused by the $SbCl_3$. The concentrations of antimony were measured at the beginning and end of the experiments using the hydride-generation atomic absorption method. The resulting EC_{50} for the 1 d, 2 d, 3 d, and 4 d exposure were 15.5, 15.5, 15.2, 12.4, and 0.93, 0.93, 0.93, 0.93, and 6.9, 6.9, 6.9, 6.9, for $SbCl_3$, $SbCl_5$, and $K[Sb(OH)_6]$, respectively (all concentrations in mg Sb/l). However, the results from using the pentavalent $SbCl_5$ appears questionable (see 3.2.1.2 above) and will therefore not be used.

The study by Dorfman (1977) results in larger-than values in a study where the presented effect concentration exceeds the solubility of the antimony compound used, the concentrations are nominal, and the study is therefore not considered reliable. The other study, by (Heitmuller et al., 1981), which indicate that the LC_{50} (1, 2, 3, 4 d) of Sheepshead Minnow (*Cyprindon variegatus*) is larger than 6.2 but smaller than 8.3 mg Sb/l, is not considered valid either. Vital information such as, antimony compound used, test concentrations, pH, dissolved oxygen concentration at the end of the test, is missing. In addition to this, only nominal concentrations are given.

The lowest valid acute toxicity to marine water fish is 6.9 mg Sb/l for *Pargus major*, which originates from the study by Takayanagi (2001).

Table3-92 presenting acute toxicity tests for freshwater and marine fish.

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
Fresh water										
Ictalurus punctatus juveniles, length about 1 inch (= 25 mm)	$SbCl_3$	Sb(III)	Moderately-hard reconstituted culture water; Alk=120-171 pH=6.8-8.1 H=153-180 DO=6.2-8.9 mg/l C=600-2300	Static T=21-25	M	4	Mortality	24.6	TAI Environmental Sciences, 1990	R
Pimephales promelas 8 week-old juvenile (12-16 mm length);	$SbCl_3$	Sb(III)	Hard well water; Alk = 232 DO = 6.88 pH = 8.16	Continuous flow T = 25	M	4 8	Mortality	21.9 20.2	Kimball, 1978	R
Pimephales promelas 30-days old, length 19 ± 1 mm, weight 0.099 ± 0.025 g	$SbCl_3$	Sb(III)	Municipal water for the City of Superior, WI. The water was declorinated by charcoal filtration and sodium sulfite adition.; Alk=38.2 pH=7.10 H=48.5 DO=76.3	Static T=21.4	M	1 2 4	Mortality	20.8 17.4 14.4	Brooke et al., 1986	R
Salmo gairdneri juveniles, length 23 ± 1 mm, weight 0.123 ± 0.013 g	$SbCl_3$	Sb(III)	Municipal water for the City of Superior, WI. The water was declorinated by charcoal filtration and	Static T=15.5	M	4	Mortality	>25.7 (45%)	Brooke et al., 1986	R

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
			sodium sulfite adition.; Alk=36.9 pH=7.12 H=52.2 DO=70.9							
Brachydanio rerio weight = 0.28±0.06 length = 2.87±0.3	Sb_2O_3	Sb(III)	ISO/DIS 7346/1 DO = 8.1-9.5 pH = 7.5-7.8 H = 241.8	Static T = 22.3-22.7	N	4	Mortality	>834 (0%)	Janssen Biotech, 1990b	NR
Leuciscus idus age/length/weight?	$K[Sb(OH)_6]$	Sb(V)	?	Static T = ?	N	2	Mortality	774 (50%)	Juhnke and Ludemann, 1978	NR
Lepomis macrochirus larvae (≤10 cm)	$SbCl_3$ $SbCl_5$	Sb(III) Sb(V)	Lake Huron water; pH = 7.5-8.2	Static T = 12.8	N	1	Mortality	>2.67 (0%) >2.04 (0%)	Appelgate et al., 1957	NR
Lepomis macrochirus juvenile, weight 0.32-1.2 g	Sb_2O_3	Sb(III)	Well-water; ALK = 28-34 C = 93-190 DO = 7.0-8.8 H = 32-48 pH = 6.7-7.8	Semi-static T = 22 ± 1	N	1 4	Mortality	>443(% effect ?) >443(% effect ?)	Buccafusco et al., 1981	NR
Oncorhynchus mykiss larvae (≤10 cm)	$SbCl_3$ $SbCl_5$	Sb(III) Sb(V)	Lake Huron water; pH = 7.5-8.2	Static T = 12.8	N	1	Mortality	>2.67 (0%) >2.04 (0%)	Appelgate et al., 1957	NR
Oncorhynchus mykiss fingerling, mean weight 1.2 g	$2KSbOC_4H_4O_6 \cdot H_2O$	Sb(III)	H = 25 pH = 6.7-7..2	Semi-static T = 15	M	4	Mortality	37 (50%)	Doe et al., 1987	NR
Oncorhynchus kisutch length= 5-10 cm	$SbCl_3$	Sb(III)	Rochat Creek water;	Static	N	1	Loss-of-equilibrium	>5.34 (0%)	MacPhee and Ruelle, 1969	NR

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Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
	$SbCl_5$	Sb(V)	Alk = 7 H = 0-17 pH = 7.2	T = 10.6		1	Mortality	>4.07 (0%)		
Oncorhynchus tshavrytscha length= 5-10 cm	$SbCl_3$ $SbCl_5$	Sb(III) Sb(V)	Rochat Creek water; Alk = 7 H = 0-17 pH = 7.2	Static T = 10.6	N	1 1	Loss-of-equilibrium Mortality	>5.34 (0%) >4.07 (0%)	MacPhee and Ruelle, 1969	NR
Oreochromis mossambicus; larvae age: 3 d and 30 d	$SbCl_3$	Sb(III)	Freshwater	Static; Larvae (age: 3 d) T = 26-29 Static; Larvae (age: 30 d) T= 26-29	N	2 4 2 2	Mortality Mortality Body length Yolk length Body Na Body K Body Ca Body Mg Body length Body Na Body K Body Ca Body Mg	19 19 1.95 (NOEC) / 9.5 (LOEC) >19 (NOEC) / >19 (LOEC) >19 (NOEC) / >19 (LOEC) 1.95 (NOEC) / 9.5 (LOEC) <0.4 (NOEC) / 0.4 (LOEC) 1.95 (NOEC) / 9.5 (LOEC) >24 (NOEC) / >24 (LOEC) >24 (NOEC) / >24 (LOEC) 19 (NOEC) / 24 (LOEC) >24 (NOEC) / >24 (LOEC) 19 (NOEC) / 24 (LOEC)	Lin and Hwang, 1998	NR
Petromyzon marinus larvae: 7.5-12.5 cm	$SbCl_3$ $SbCl_5$	Sb(III) Sb(V)	Lake Huron water; pH = 7.5-8.2	Static T = 12.8	N	1	Mortality	>2.67 (0%) >2.04 (0%)	Appelgate et al., 1957	NR
Pimephales promelas age/length/weight ?	$2KSbOC_4H_4O_6 \cdot H_2O$	Sb(III)	Soft water; Alk = 18 H = 20 pH = 7.4 Hard water;	Exploratory test only T = ? Exploratory test	N	4	Mortality	>20 (%effect ?)	Tarzwell and Henderson, 1960	NR

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Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
	$SbCl_3$ Sb_2O_3	Sb(III) Sb(III)	Alk = 360 H = 400 pH = 8.2 Soft water Hard water Soft water Hard water	only T = ? Exploratory test only T = ? Exploratory test only T = ?				>12 (%effect ?) 9 17 >80 (%effect ?) >80 (%effect ?)		
Pimephales promelas age/length/weight ?	Sb_2O_3	Sb(III)	Reconstituted soft water; ALK = 30-35 C^2 = 120-160 H = 40-48 pH = 7.2-7.9	Static T = 22 ± 1	M	4	Mortality	>696 (0%)	Curtis and Ward, 1981	NR
Ptychocheilus oregonensis length= 5-10 cm	$SbCl_3$ $SbCl_5$	Sb(III) Sb(V)	Rochat Creek water; Alk = 7 H = 0-17 pH = 7.2	Static T = 10.6	N	1 1	Loss-of-equilibrium Mortality	>5.34 (0%) >4.07 (0%)	MacPhee and Ruelle, 1969	NR
Marine water										
Pargus major Juvenile 30 days posthatch weight (g) 2.74 ±0.16 2.94 ±0.15	$SbCl_3$	Sb(III)	Natural seawater (filtered through sand and activated-charcoal filters) S = 33.7	Static T= 20 ±1 Each aquarium was aerated pH range 4.9-7.8 7.8-8.1	M	1 2 3 4	Mortality	15.5 15.5 15.2 12.4	Takayanagi, 2001	R

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Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
2.88 ±0.18	$SbCl_5$	Sb(V)		7.8-8.1		1 2 3 4		0.93 0.93 0.93 0.93		NR
	$K[Sb(OH)_6]$	Sb(V)				1 2 3 4		6.9 6.9 6.9 6.9		R
Cyprindon variegatus juvenile 14-28 days posthatch length = 8-15 mm	Sb (unknown Sb-compound)	Sb(?)	Natural seawater (filtered, 5 μm filter)	Static T = ?	N	1 2 3 4	Mortality	>6.2<8.3 >6.2<8.3 >6.2<8.3 >6.2<8.3 6.2 (0%)	Heitmuller et al., 1981	NR
Fundulus heteroclitus average weight = 2.7 range 1.1-6	Sb_2O_3	Sb(III)	Seawater Low salinity (S=6) High salinity (S=21)	Static T = ?	N	1 2 3 4 1 2 3 4	Mortality	>1000 (% effect ?) >1000 >1000 >1000 >1000 >1000 >1000 >1000	Dorfman, 1977	NR

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Alk = alkalinity (as mg $CaCO_3$/l); C^1 = specific conductance (µmhos/cm); C^2 = specific conductance (µS/cm); DO = dissolved oxygen (mg O_2/l); H = hardness (as mg $CaCO_3$/L); S = salinity (‰); T = temperature (°C)

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Long-term toxicity

Fresh water

There are only two studies considered valid, i.e. Kimball (1978) and LeBlanc and Dean (1984). However, only the former, i.e. Kimball (1978) can be used to derive a PNEC since the latter study only contains larger-than values (see Table3-93 below).

LeBlanc and Dean (1984) exposed eggs from *Pimephales promelas* in a flow-through system, using two replicates, with five concentrations (range: 0.6 - 7.5 µg Sb/l) and two controls (one regular and one vehicle control (HCl), where each replicate consisted of 55 eggs. The tests were performed in well water with a hardness of 28-40 mg $CaCO_3$/l (see Table3-93 below for more details). Since there were no statistically different effect (endpoints: mortality, larvae lengths, and weights) as compared to the control, not even the highest concentration used (7.5 µg Sb/l), the resulting unbounded NOEC-value was > 7.5 µg Sb/l.

Chronic test performed by Kimball (1978), used embryo-larva of *Pimephales promelas* in a flow-through system, with four replicates and six concentrations (range: 0.52 – 19.11 mg Sb/l) and a control, where each group consisted of 20 eggs. The tests were performed with hard well water (more details on the test are provided in Table3-93 below). The lowest resulting NOEC-value from this study, using reduction of length as an indicator of toxicity, is 1.13 mg Sb/l. This value, 1.13 mg Sb/l, is also the lowest valid NOEC for fresh water fish in long-term toxicity tests.

There are several studies by Birge and co-workers (Birge, 1978; Birge et al., 1980), which report NOEC-values below that concentration, i.e. 1.13 mg Sb/l. However, even though these studies are well performed, with measured test concentrations etc., neither of these studies are considered valid in this RAR. The reason for this is that the test concentrations used are never reported, which makes it difficult to decide whether or not the calculated NOEC values are included in the tested concentration range, or not.

The reasons why the result by (Doe et al., 1987) on rainbow trout is not considered reliable are presented in the section on acute toxicity above.

Exposure via the food

Tamulinas (1979) performed a study in which *Icatalurus punctatus* was exposed to Sb_2O_3 via the food for 90 days. Four concentrations (range: 2000 – 8000 mg Sb/kg) and a control, where each group consisted of 10 catfish, were used. All of the fish survived the 90-day exposure period. There were no specific signs of intoxication other than anorexia, which was observed in the two highest exposure groups (6000 and 8000 mg Sb/kg). All of the fish, at the beginning of the experiment, readily consumed the food, however, as the experiment progressed, all of the fish reduced their food intake and periodically would not eat the diet. The fish at the 8000 mg Sb/kg exposure group exhibited the highest incidence of anorexia. The most consistent changes seen throughout the oral exposure experiments were lowered packed cell volume, increased spleen weight, and increased serum glutamic oxaloacetic transaminase, which is indicative of damage to the erythrocytes. Total serum protein decreased significantly in the 8000 mg Sb/kg group. The serum enzyme profile of the catfish exposed to diantimony trioxide showed indications of muscle or liver damage. However, the

underlying reason for the effects observed, whether it was avoidance of the food containing high concentrations of Sb_2O_3 resulting in starvation or if it was a toxicological response from exposure to Sb_2O_3 is difficult to decide. However, given the very high concentrations tested the results will not be used further in the RAR

Marine water

There is no valid long-term toxicity data on marine species. The only result available is from a study on *Cyprinodon variegatus*, which resulted in a larger-than value of 0.0075 mg Sb/l. For more information see Table3-93 below. However, since there is a lack of important information regarding the experimental set-up etc., this result cannot be properly evaluated and are therefore not considered valid.

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Table3-93 presenting long-term toxicity tests for freshwater and marine fish.

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/l; % effect)	LOEC (mg Sb/l; % effect)	E(L)C_{50} (mg Sb/l; % effect)	Reference	Reliable & Relevant
Fresh water												
Pimephales promelas embryo-larva (16 - 40 h old)	$SbCl_3$	Sb(III)	Hard well water; Alk = 234 DO = 6.87 pH = 7.97	Continuous flow T = 25.1	M	28	Posthatch survival Growth (length) Growth (weight)	4.50 1.13 2.31	9.31 2.31 4.50		Kimball, 1978	R
Pimephales promelas embryo-larva (<48 h old)	Sb_2O_3	Sb(III)	Well water; C = 140-170 DO =8.7±0.4 pH = 6.2-7.3 H = 28-40	Continuous flow T = 25±1	M	30	Posthatch survival Growth (length) Growth (weight)	> 0.0075 > 0.0075 > 0.0075			LeBlanc and Dean, 1984	R
Carassius auratus embryo-larva	$SbCl_3$	Sb(III)	DO=near saturation H=195±5.4 pH=7.4±0.1	Semi-static T=22.0±1.0	M	7	Mortality	0.111 (1%) (0.0001-0.663)*		11.3 (3.99-55.0)*	Birge, 1978	NR
Icatalurus punctatus length 20-25 cm	Sb_2O_3	Sb(III)	Municipal water free of chlorine	Semi-static T = ?	N	30	Mortality, gross pathological changes, blood parameters (packed cell volume (serum glutamic oxaloacetic transaminase, glutamic pyruvic transaminase)	>100			Tamulinas, 1979	NR
	$2KSbOC_4H_4$	Sb(III)				30		>4				

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/l; % effect)	LOEC (mg Sb/l; % effect)	E(L)C_{50} (mg Sb/l; % effect)	Reference	Reliable & Relevant
fingerlings: length 7.5-12.5 cm	$O_6 \cdot H_2O$ Sb_2O_3	Sb(III)				120	in addition to the parametrers mentioned above, also a) serum electrolytes (Na+, K+) b) spleen weight	>4	4 (only one dose tested) 4 (only one dose tested)			
Oncorhynchus mykiss fingerling, mean weight 1.2 g	$2KSbOC_4H_4O_6 \cdot H_2O$	Sb(III)	H=25 pH=6.7-7.2	Semi-static	M	30	Mortality			16	Doe et al., 1987	NR
Oreochromis mossambicus 3-day-old larvae	$SbCl_3$	Sb(III)	Freshwater	Semi-static; Larvae (age: 3d) T = 26-29	N	16	Body length Body Na (µg/larva) Body K (µg/larva) Body Ca (µg/larva) Body Mg (µg/larva)	>1.95 >1.95 >1.95 >1.95 >1.95	9.5 (100% mortality		Lin and Hwang, 1998	NR
				Semi-static; Larvae (age: 30d) T = 26-29		16	Body length Body Na (µg/larva) Body K (µg/larva) Body Ca (µg/larva) Body Mg (µg/larva)	<9.5 <9.5 <9.5 <9.5 <9.5				
Salmo gairdneri embryo-larva	$SbCl_3$	Sb(III)	DO=near saturation H=104±2.0 pH=7.4±0.1	Semi-static T=13.0±0.5	M	28	Mortality	0.0286 (1%) (0.0046-0.0722)*		0.58 (0.34-0.92)*	Birge, 1978	NR
Salmo gairdneri embryo-larva	$SbCl_3$	Sb(III)	DO=9.3-10.1 H=92-110 pH=6.9-7.8	Semi-static T=12-13	M	28	Mortality	0.0489 (1%) (0.0248-0.0792)*	0.157 (10%) (0.101-0.216)*	0.66 (0.53-0.79)*	Birge et al., 1980	NR

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/l; % effect)	LOEC (mg Sb/l; % effect)	$E(L)C_{50}$ (mg Sb/l; % effect)	Reference	Reliable & Relevant
30-day-old larvae							Body length Body Na (μg/larva) Body K (μg/larva) Body Ca (μg/larva) Body Mg (μg/larva)	<9.5 <9.5 <9.5 <9.5 <9.5	9.5 9.5 9.5 9.5 9.5			
Marine water												
Cyprinodon variegatus	Sb_2O_3	Sb(III)	Salt water	?	?	3	Mortality			>0.0075	Anonymous, 1978	NR

*95% confidence interval; Alk = alkalinity (as mg $CaCO_3$/l); C^1 = specific conductance (umhos/cm); C^2 = specific conductance (μS/cm); DO = dissolved oxygen (mg O_2/l); H = hardness (mg $CaCO_3$/L); S = salinity (‰); T = temperature (˚ C)

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3.2.4.1.2 Aquatic invertebrates

A total of 13 studies with 8 different species were found. However, most of these studies cannot be used since they are not considered valid due to reasons of reliability (e.g. test concentrations above solubility of test compound, nominal concentrations only, and/or lack of information on test protocol).

Table3-94 below summarize the lowest valid toxicity data for aquatic invertebrates. The lowest LC_{50}, resulting from a study by TAI (1990) using *Chlorohydra viridissimus*, is 1.77 Sb mg/l. This is very similar in size to the NOEC at 1.74 mg Sb/l, resulting from a study by Heijerick and Vangheluwe (2003b) using *Daphnia magna*. The endpoint used to derive the NOEC was reproduction.

Table3-94 presenting lowest valid LC_{50}, LOEC, and NOEC for aquatic invertebrates.

Organism	Water	Exposure period	Endpoint	LC_{50}/LOEC/NOEC	Value (mg Sb/l)	Reference
Chlorohydra viridissimus	Fresh	4 d	Mortality	LC_{50}	1.77	TAI Environmental Sciences, 1990
Daphnia magna	Fresh	21 d	Reproduction	LOEC	3.13	Heijerick and Vangheluwe, 2003b
Daphnia magna	Fresh	21 d	Reproduction	NOEC	1.74	Heijerick and Vangheluwe, 2003b
----------	Marine	----------	----------	LC_{50}	---------	----------
----------	Marine	----------	----------	LOEC	---------	----------
----------	Marine	----------	----------	NOEC	---------	----------

Acute toxicity

Fresh water

There are only two studies resulting in reliable results, as regards acute toxicity to aquatic invertebrates, i.e. Kimbal (1978) and TAI (1990). However, there is also a study by (Brooke et al., 1986) that, even though not consider reliable in the context of this RAR, provides strong indications that the coelenterate hydra is the most sensitive invertebrate tested so far.

In the study by Kimbal (1978), <1 d old daphnids were exposed in a static test design to trivalent antimony ($SbCl_3$) for 2 or 4 days. This study was performed with six concentrations (range: 1.65 – 44.15 mg Sb/l) and a control, using four replicates with ten neonates for each concentration. Mortality was used as endpoint (for more information on the study, see Table3-95 below). In the 2-day exposure regime, the daphnids were exposed to antimony ($SbCl_3$) either with or without feeding. The resulting LC_{50} were 12.1 and 18.8 mg Sb/l, respectively. At the 4-day exposure regime, the antimony exposure was only performed together with feeding, and the resulting LC_{50} was 12.1 mg Sb/l (see Table3-95 below).

In the study by TAI (1990) two hydra species (*Hydra oligogactis*, *Chlorohydra viridissima*), one snail (*Physa heterostropha*), one midge (*Chironomus tetans*) and the amphipod *Hyalella azteca* were tested in a static test design to trivalent antimony ($SbCl_3$) for 4 days. All tests were performed with five concentrations and a control, using duplicate replicates with 5 individuals for each test chamber. The measured concentrations used (nominal concentration within brackets) for the *Hyalella azteca* were 8.5 (15.6), 15.4 (31.25), 19.8 (62.6), 24.6 (125), and 21.22 (250) mg Sb/l, and a control with 0 mg Sb/l (measured concentration) and for *Physa heterostropha* and *Chironomus tetans* 3.93 (15.6), 7.29 (31.25), 17.13 (62.6), 22.55 (125), and 24.14 (250) mg Sb/l, and a control with 0 mg Sb/l (measured concentration). The measured concentrations used (nominal concentration within brackets) for the two hydra species *Hydra oligogactis* and *Chlorohydra viridissima* were 1.18 (0.5), 1.52 (1.0), 2.56 (2.5), 4.48 (5.0), and 5.42 (6.25), and a control with 0 mg Sb/l (measured concentration). Using mortality as endpoints the resulting LC_{50}s were 1.95, 1.77, 14.2, 4.1, and 21.6 mg Sb/l for the *Hydra oligogactis*, *Chlorohydra viridissima*, *Physa heterostropha*, *Chironomus tetans*, and *Hyalella azteca*, respetively. For more information on the study, see Table3-95 below. In the study by TAI (1990) described above, the hydras turned out to be the most sensitive invertebrate.

This observation is further supported by the study by (Brooke et al., 1986), in which also the *hydra* turned out to be the most sensitive of several tested invertebrates; (*Hydra sp.*), amphipods (*Gammarus pseudolimnaeus*), annelids (*Lumbriculus variegates*), and caddisflies (*Pycnopsche sp.*). Adult hydroids (*Hydra sp*.) were exposed in a static test design to trivalent antimony ($SbCl_3$) for 4 days, and an EC_{50} (tentacles clubbed and/or shortened body column and tentacles) was determined. These tests were performed in quadruplicate with five concentrations (range: 0.3 - 3.3 mg Sb/l) and a control, with each replicate consisting of 10 hydroids (for more information on the study, see Table3-95 below). The resulting EC_{50} (1 d) was 2 mg Sb/l (1.8-2.2 = 95% conf. int.), EC_{50} (2 d) was 1.0 mg Sb/l (conf. limits not reliable, according to author), and EC_{50} (4 d) which was 0.5 mg Sb/l (0.5-0.6 = 95% conf. int.). The tests on amphipods (*Gammarus pseudolimnaeus*), annelids (*Lumbriculus variegatus*), and caddisflies (*Pycnopsche sp.*) were performed in duplicate with two concentrations (11.4 ± 3.9 and 25.7 ± 2.2 mg Sb/l) and a control, with each replicate consisting of ten individuals. The EC_{50} for these three species all resulted in larger-than values (> 25.7 mg Sb/l). However, the result of the hydra is not considered reliable but only indicative, since the endpoint used (tentacles clubbed and/or shortened body column and tentacles) is subjective, and no information is provided whether or not a dose-response relationship exists.

The hydra results from the study by (Brooke et al., 1986) are only slightly lower then those reported by TAI (1990), i.e. 0.5 mg Sb/l (*Hydra sp*.) as compared with 1.77 mg Sb/l (*Chlorohydra viridissimus*) and 1.95 mg Sb/l (*Hydra oligactis*). This slight difference may be due to several reasons such as inter laboratory differences, different sensitivities between the different hydra species tested, or the fact that the criteria used for defining the endpoint mortality probably differ in sensitivity. The criteria for hydra mortality in TAI study was the beginning of the break down of the tissue integrity and an associated bacterial growth enveloping the animals, while it in the study by (Brooke et al., 1986) was clubbed tentacles and/or shortened body column and tentacles.

Thus, most sensitive of the aquatic invertebrates is the hydra, and the lowest valid LC_{50} for acute toxicity is 1.77 mg Sb/l.

The reasons why the study by (Doe et al., 1987) on acute toxicity of *Daphnia magna* is considered unreliable, even though it is indicated that the reported concentrations were measured, is that there is no information presented on (i) the number of concentrations and

which concentrations that were used, (ii) dose-response curves (no raw data available), (iii) number of replicates (if any?), and (iv) what statistics that have been used to calculate the presented LC_{50}-values.

Borgman et al. (2005) reported an acute LC50 value of 0.687 mg Sb/l for *Hyalella azteca*, which is not considered reliable. Because of the objective of the study (large screening of metal toxicity for categorization of substances on the Canadian Domestic Substances List) several modifications have been made to the standard experimental design. The main reason why the abovementioned result from this study cannot be considered reliable is because antimony was clearly not the only toxicity-inducing factor in the test medium. This study used metal standards for toxicity testing. For antimony, a metal standard containing 20% HCl was used. The acid in the metal standards was neutralized by adding a solution of 1M $NaHCO_3$ and 1M KOH in a 19:1 ratio. Next to a control treatment (containing normal test medium), an acid control was used (containing acid and neutralizing solution additions equal to the amount added in the tests with acidified metal standards). For metal standards supplied in 20% HCl, survival in the acid controls for the 1000 µg/L treatment dropped to 32%, indicating that the organisms were adversely affected by the blank test medium. Therefore, toxicity in the metal treatments was most likely overestimated in this study. The LC_{50}s derived for antimony in the tests using the metal standards is not considered reliable. Toxicity tests were also conducted using sodium antimonate ($NaSbO_3$). However, no toxicity was observed at the highest exposure concentrations (= 1 mg Sb/l nominal = 0.197 mg Sb/l measured) used.

Marine water

There are no valid acute toxicity data on marine species. The only result available is from a study on *Mysidopsis bahia* which resulted in a larger-than value of 4.15 mg Sb/l (for more information see Table3-95 below). However, there is a lack of important information regarding the experimental set-up etc., therefore the study is not considered reliable, and cannot be used in the risk assessment.

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Table3-95 presenting acute toxicity tests for aquatic invertebrates in fresh- and marinewaters.

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
Fresh water										
Chironomus tentans Second instar larvae	$SbCl_3$	Sb(III)	Moderately-hard reconstituted culture water; Alk=51-85 pH=7.3-7.8 H=85-103 DO=5.4-8.7 mg/l C=290-1400	Static T=21-25	M	4	Mortality	4.1	TAI Environmental Sciences, 1990	R
Chlorohydra viridissima adult	$SbCl_3$	Sb(III)	Moderately-hard reconstituted culture water; Alk=85-103 pH=7.5-7.8 H=85-103 DO=5.1-8.4 mg/l C=300-490	Static T=21-25	M	4	Mortality	1.77	TAI Environmental Sciences, 1990	R
Daphnia magna <1d old	$SbCl_3$	Sb(III)	Hard well water; pH=8.16; DO=7.87	Static T=20	M	2 4	Mortality	12.1 (fed) 18.8 (not fed) 12.1 (fed)	Kimball, 1978	R
Gammarus pseudolimnaeus adult, length= 7.1 mm	$SbCl_3$	Sb(III)	Municipal water for the City of Superior, WI. The water was declorinated by charcoal	Static T=15.5	M	4	Mortality	>25.7 (25%)	Brooke et al., 1986	R

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Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
			filtration and sodium sulfite adition.; Alk=36.9 pH=7.12 H=52.2 DO=70.9							
Hyalella azteca juvenile	$SbCl_3$	Sb(III)	Moderately-hard reconstituted culture water; Alk=85-171 pH=7.1-8.2 H=136-188 DO=6.6-8.4 mg/l C=600-2300	Static T=21-25	M	4	Mortality	21.6	TAI Environmental Sciences, 1990	R
Hydra oligactis adult	$SbCl_3$	Sb(III)	Moderately-hard reconstituted culture water; Alk=85-103 pH=7.5-7.8 H=85-103 DO=5.1-8.6 mg/l C=300-490	Static T=21-25	M	4	Mortality	1.95	TAI Environmental Sciences, 1990	R
Lumbriculus variegatus length=23 mm	$SbCl_3$	Sb(III)	Municipal water for the City of Superior, WI. The water was declorinated by charcoal filtration and	Static T=15.5	M	4	Mortality	>25.7 (5%)	Brooke et al., 1986	R

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Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
			sodium sulfite adition.; Alk=36.9 pH=7.12 H=52.2 DO=70.9							
Physa heterostropha small juveniles (1-2 mm)	$SbCl_3$	Sb(III)	Moderately-hard reconstituted culture water; Alk=51-85 pH=7.1-7.8 H=85-103 DO=5.4-8.6 mg/l C=290-1400	Static T=21-25	M	4	Mortality	14.2	TAI Environmental Sciences, 1990	R
Pycnopsche sp. length=18 mm	$SbCl_3$	Sb(III)	Municipal water for the City of Superior, WI. The water was declorinated by charcoal filtration and sodium sulfite adition.; Alk=36.9 pH=7.12 H=52.2 DO=70.9	Static T=15.5	M	4	Mortality	>25.7 (5%)	Brooke et al., 1986	R
Caenorhabditis elegans	$SbCl_3$	Sb(III)	AM	Static T=20	N	4	Mortality	>20	Williams and Dusenbery, 1990	NR

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Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
young adults (3-4 d)										
Carcinus maenas	?	?	?	Static	N	3	Immobilization	>204	Amiard-Triquet et al., 1982	NR
Daphnia magna age unknown	Sb_2O_3	Sb(III)	Filtered aerated tubewell water; Alk=390-415 H=235-260 DO=5.2-6.5 pH=7.2-7.8	Static T=11.5-14.5	N	1 2	Immobilization	555.3 (453.8-726.3) 423.5 (361.5-496.0)	Khangarot and Ray, 1989	NR
Daphnia magna age<24h	Unknown salt form According to EPA (1988) the salt form used was Sb_2O_3	Sb(III)	Deionized reconstituted well water; H=173±13 pH=8.0±0.2	Static T = 22 ± 1	N	1 2	Mortality	>530 (0%) >530 (0%) According to US-EPA (1988) this results in: >443 (0%)	LeBlanc, 1980	NR
Daphnia magna age<4±4h	$SbCl_3$	Sb(III)	Lake Eire water	Static T = 25	N	2.67	Immobilization	19.7	Anderson, 1948	NR
Daphnia magna age<24h	Sb_2O_3	Sb(III)	ISO-6341, with added micronutrients; pH=7.6-7.9 H=240 DO=8.3-9.2	Static T=19.8-20.6	N	2	Immobilization	>834 (0%)	Janssen Biotech, 1990a	NR
Daphnia magna age<24h	$KSbC_4H_4O_7 \cdot 1/2H_2O$	Sb(III)	ISO-6341	Static T = ?	N	2	Immobilization	20	Knie et al., 1983	NR

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
Daphnia magna age<24h	$2KSbOC_4H_4O_6 \cdot H_2O$	Sb(III)	H = 250 ± 25 pH= 7.8 ± 0.2 H = 31 H = 45 H = 92 H = 220	Static T = 20	M	2	Mortality	6.7 5 5 5 6.7	Doe et al., 1987	NR
Hyalella azteca age<24h	Sb (HCl standard)	Sb(III)	H = 18 pH = 8.27 Alk = 14 DOC = 0.28 C^2 = 447 H = 124 pH = 8.46 Alk = 84 DOC = 1.1 C^2 = 730	Static T = 24-25	M N	7 7	Mortality	0.687 >3.15	Borgmann et al., 2005	NR
	$NaSbO_3$		H = 18 pH = 7.39 Alk = 14 DOC = 0.28 C^2 = 66 H = 124 pH = 8.21 Alk = 84 DOC = 1.1 C^2 = 345		N N	7 7		>0.197 >1		
Hydra sp. adult	$SbCl_3$	Sb(III)	Municipal water for the City of Superior, WI. The water	Static T=23.6	M	1 2 4	Body column and tentacles clubbed (i.e. shorter than controls)	2.0 (1.8-2.2)* 1.0 (**) 0.5 (0.5-0.6)*	Brookee, 1986	NR

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
			was declorinated by charcoal filtration and sodium sulfite adition.; Alk=41.5 pH=7.7 H=46.9 DO=90.7							
Tubifex tubifex age/length/weight not known	Sb_2O_3	Sb(III)	Tubewell water; pH=7.6 DO=5.8 H=245 ALK=400	Static T=30	N	1 2 4	Immobilization	108? (926-1330)* 920 (840-1181)* 678 (610-840)*	Khangarot, 1991	NR
Marine water										
Mysidopsis bahia age/length/weight not known	Sb_2O_3	Sb(III)	Salt water	?	?	4	Mortality	>4.15 (50%)	Anonymous, 1978	NR

Alk = alkalinity (mg $CaCO_3$/l); AM = artificial medium; C^1 = specific conductance (µmhos/cm); C^2 = specific conductance (µS/cm); DO = dissolved oxygen (mg O_2/l);

DOC = Dissolved Organic Material (mg/L); H = hardness (mg $CaCO_3$/L); T = temperature (ºC); * =95% confidence interval; ** = confidence limits not reliable.

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Long-term toxicity

Fresh water

There is one study available providing reliable long-term toxicity data for invertebrates, which is the one, performed by Heijerick and Vangheluwe (2003b). In this study a 21-d chronic toxicity test was performed with *Daphnia magna* using $SbCl_3$ as the test substance. This study, which is in accordance with the OECD 211 guideline, was conducted as semi-static test, with full renewal of the test medium three times/week. The test was performed with five measured concentrations (range: 0.056 – 9.96 mg Sb/l) and a control, with ten test chambers per concentration, using one juvenile (<24h-old) daphnia in each chamber. Mortality of adult organisms is noted and newborn neonates were counted and removed when the exposure medium was renewed. Two endpoints, based on survival and reproduction (net reproduction rate, R, and the intrinsic rate of natural increase, rm) were determined. The resulting 21-d NOEC for reproduction of *Daphnia magna* was 1.74 mg Sb/l, with a LOEC of 3.13 mg Sb/l. The LC_{50} (21-d) was 4.77 mg Sb/l, the $EC_{50,R}$ (21-d) was 3.82 mg Sb/l, and the $EC_{50,rm}$ (21-d) was 4.86 mg Sb/l.

The study by Kimbal (1978) only contains reliable results for a 7-d screening test on *Daphnia magna*. The 7-d screening test was conducted as a static renewal test with exposure to trivalent antimony ($SbCl_3$). The test was performed with four replicates, six concentrations (range: 1.9 – 52.2 mg Sb/l) and a control, with ten test chambers per replicate, using one 2 w old adult daphnia in each chamber. Once the daphnids began reproducing, the neonates and the molts of the adults were counted. Survival was recorded at each renewal period (three times weekly). The resulting 7-d NOEC for reproduction of *Daphnia magna* was 3.9 mg Sb/l. The LC_{50} for the 7-d screening study was 14.5 mg Sb/l. The set-up for the 28-d test was identical to the screening test, except that neonate daphnia initially were placed into the beakers, instead of two week old adults and that the concentration range used were 0.52- 7.05 mg Sb/l. However, the results from the 28-d exposure period could not be used due to the high control mortality (30%). For more information on the study see Table3-96 below.

The reasons why the study by (Doe et al., 1987) on chronic toxicity of *Daphnia magna* is considered unreliable, even though it in is indicated that the reported concentrations were measured, are that there is no information presented on (i) the size of the group(s?) (only information given is that there was 1 animal/50 ml), (ii) the number and which concentrations that were used, (ii) dose-response curves (no raw data available), (iii) number of replicates (if any?), and (iv) what statistics that have been used to calculate the presented LC_{50}-value and NOECs. The 30 day LC_{50} is 2.7 mg Sb/l and NOECs for reproduction (30d) and growth (33d) are 1.7 mg Sb/l and 0.8 mg Sb/l, repectively. However, the only information on concentrations used is a legend in a figure indicating that one control, and the two doses 1.7 mg Sb/l and a 3.7 mg Sb/l have been used.

Marine water

There is no data on long-term toxicity of antimony to aquatic invertebrates available for marine species.

Table3-96 presenting long-term toxicity tests for aquatic invertebrates in freshwater.

Organism	Compound/ Valency	Medium	Test conditions	Exposure period (d)	Endpoint	NOEC (mg Sb/l)	LOEC (mg Sb/l)	E(L)C50 (mg Sb/l)	Reference	Reliable & Relevant

Organism	Compound/ Valency	Medium	Test conditions	Exposure period (d)	Endpoint	NOEC (mg Sb/l)	LOEC (mg Sb/l)	E(L)C50 (mg Sb/l)	Reference	Reliable & Relevant
Daphnia magna 2w old	$SbCl_3$/Sb(III)	Hard well water; pH=8.5; DO=7.36; Alk=225;	Semi-static; Measured conc. T=20; Screening test	7	Reproduction (mean young/female) Mortality	3.9	7.8	14.5	Kimball, 1978	R
<1d old				28	Reproduction (mean young/female) (mean young/female survivor) Mortality	4.16 4.16	7.05 7.05	4.51		NR
Daphnia magna <1d old	$SbCl_3$/Sb(III)	ISO-test medium H=250 pH=7.3; DO>80% saturation	Semi-static; Measured conc. T=20;	21	Reproduction Mortality	1.74	3.13	4.77	Heijerick and Vangheluwe, 2003b	R
Daphnia magna <1d old	$2KSbOC_4H_4O_6 \cdot H_2O$ /Sb(III)	H = 250 ± 25 pH= 7.8 ± 0.2	Semi-static; Measured conc. T=20;	30 30 33	Reproduction Mortality Growth	1.7 0.8	? ?	2.7	Doe et al., 1987	NR

Alk = alkalinity (mg $CaCO_3$/l); DO = dissolved oxygen (mg O_2/l); H = hardness (mg $CaCO_3$/L); T = temperature (ºC)

3.2.4.1.3 Algae

A total of 8 studies with 8 different species of algae were found. However, only three of these studies were considered reliable, while the remaining five studies could not be used since they were not considered valid due to reasons of reliability (e.g. test concentrations above solubility of test compound, nominal concentrations only, and/or lack of information on test protocol).

Table3-97 summarizes the lowest valid toxicity data for algae. The lowest reliable effect concentration is the EC_3value of 2.11 mg Sb/l, using the endpoint growth rate, which resulted from tests performed by Heijerick and Vangheluwe (2004) on the unicellular green algae *Raphidocelis subcapitata* (previously *Selenastrum capricornutum*). This value is used as NOEC in this RAR.

Table3-97 presenting lowest valid LC_{50}, LOEC, and NOEC for algae.

Organism	Water	Exposure period	Endpoint	LC_{50}/LOEC/NOEC	Value (mg Sb/l)	Reference
Raphidocelis subcapitata	Fresh	3 d	Growth rate	EC_{50}	> 36.6	Heijerick and Vangheluwe, 2004
Raphidocelis	Fresh	3 d	Growth rate	NOEC		Heijerick and

Organism	Water	Exposure period	Endpoint	LC_{50}/LOEC/NOEC	Value (mg Sb/l)	Reference
subcapitata					2.11	Vangheluwe, 2004
----------	Marine	----------	----------	LC_{50}	--------	----------
----------	Marine	----------	----------	LOEC	--------	----------
----------	Marine	----------	----------	NOEC	--------	----------

Fresh water

There are only three reliable studies, in which the effect of antimony on growth was studied.

In the study by (Heijerick and Vangheluwe, 2004) exponentially growing cultures of the unicellular green algae *Raphidocelis subcapitata* were exposed to various concentrations of trivalent antimony ($SbCl_3$) for a period of 72 h. This study was performed with three replicates, seven concentrations (range: 1.22 – 36.6 mg Sb/l) and a control (<0.002 mg Sb/l), using growth as endpoint, resulting in the same NOEC value, 2.11 mg Sb/l, for both biomass growth (EbCx) and growth rate (ErCx), with the latter being the preferred parameter according to the revised ISO 8692 guideline. The resulting NOEC for algae is thus 2.11 mg Sb/l, with a corresponding LOEC of 4.00 mg Sb/l (see Table3-99 below)

In the study by (LISEC, 2001) exponentially growing cultures of the unicellular green algae *Raphidocelis subcapitata* were exposed to various concentrations of trivalent antimony (Sb_2O_3) for a period of 72 h. This study was performed with three replicates, six concentrations (range: 0.074 – 2.4 mg Sb/l) and a control, using growth as endpoint, and resulted in two NOEC values, 0.323 and 0.396 mg Sb/l. While the former was calculated for growth (biomass), the latter was calculated for growth rate, which is the preferred parameter according to the revised ISO 8692 guideline. The resulting NOEC for algae is thus 0.396 mg Sb/l, with a corresponding LOEC of 1.32 mg Sb/l (see Table3-99 below). However, this NOEC value, which was the highest concentration not significantly different from the control, will not be taken forward to the calculation of $PNEC_{aquatic}$. Instead will the highest concentration tested, with an inhibition of growth rate of 3% be used. The reason for using the highest tested concentration 2.4 mg Sb/l, instead of the NOEC or the LOEC ($EC_{2.3}$), is that it is not totally clear whether or not the true beginning of a real dose-response curve is observed, since the highest concentration tested only resulted in an inhibition of (3%) using the recommended endpoint growth rate. This study has an unusually low variation between the replicates, and a very low effect, even at the highest concentration tested (see Table3-98 below).

Table3-98 Inhibition of growth rate of *Raphidocelis subcapitata* due to exposure of Sb_2O_3 (LISEC, 2001).

Measured conc. (mg Sb/l)	0	0.074	0.156	0.323	0.396	1.32	2.4
Inhibition growth rate (%)	-	0.8	0.8	0.1	1.0	2.3	3.0
Relative standard deviation (%)	2.1	2.2	1.8	1.1	1.8	3.5	1.2

A confirmatory test would most likely result in a higher NOEC-value due to the normally occurring much larger variations. A recent review of data from 41 algal tests in Germany and USA indicates that EC_{10} on average corresponds reasonably well with the NOEC (Heitmanm and Staveley, 2003). The choice of taking the highest concentration tested forward in the calculation of $PNEC_{aquatic}$ may therefore still be considered as protective, since the inhibition at this concentration only is 3 % (for more information see Table3-99 below). The value taken forward to the calculation of $PNEC_{aquatic}$ is thus 2.4 mg Sb/l. As a result of the low effect at the highest concentration used, i.e. 3%, no EC_{50} could be determined.

In the study by (Brooke et al., 1986) *Lemna minor* were exposed in a static test design to trivalent antimony ($SbCl_3$) for 4 days. The endpoint studied was reduction in frond production. The test was performed in quadruplicate with five concentrations (range: 1.6 – 25.5 mg Sb/l) and a control, with 20 fronds in each test chamber (for more information on the study, see Table3-99 below). No EC_{50} could be obtained. However, in the highest exposure group, i.e. 25.5 mg Sb/l, a significant ($p<0.05$) reduction in frond production occurred. The reduction measured 32%. The resulting NOEC from this study was 12.5 mg Sb/l, with the corresponding LOEC 25.5 mg Sb/l.

In the study by de Jong (1965) *Chlorella vulgaris* were exposed in a static test design to trivalent antimony ($Sb_2O_32SbOCl$) for 90-120 days. The endpoint studied was growth. Results reported from the study were a NOEC of 0.032 mg Sb/l and a LOEC of 0.064 mg Sb/l. However, the results from this study are not considered reliable due to a number of reasons, (i) as that the test concentrations used were not measured nor given, (ii) it is not clear whether or not the effect concentrations given represent the salt used or the metal, (iii) no information were given on hardness or alkalinity, the temperature was only given as "room temperature",(iv) pH at the start of the experiment was reported to be "just below 7" for the basal medium used but no measurements of the pH were performed in the test solutions used (at any time point).

The study by LISEC (1994) on *Selenastrum capricornutum* is considered unreliable, despite that measured concentrations were reported to be used. The reasons are that (i) the reported concentrations were not based on the dissolved concentrations but instead on nominal concentrations which were used since the total concentration, constituting of "whole media" (=dissolved and dispersed amount of test material) were within 10% of the nominal concentrations, (ii) the measured concentrations in the filtrate differed substantially between the samples taken at the start (0 h) and the end (72 h) of the experiment (with higher concentrations at the end of the experiment). In addition, the EC_{50} based on growth rate is extrapolated since only a 16 % inhibition in growth rate was observed in the highest test concentration, which exceeds the maximum solubility of Sb_2O_3.

Marine water

There is no available useful data on antimony toxicity to marine algal species. The only study available (Anonymous, 1978), which was performed on the diatom *Skeletonema costatum,* lack important information regarding the experimental set-up etc., and can therefore not be used in the risk assessment (see Table3-99 below).

Table3-99 presenting toxicity tests for primary producers in fresh- and marine water.

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/l)	LOEC (mg Sb/l; % effect)	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
Fresh water												
Lemna minor	$SbCl_3$	Sb(III)	Municipal water for the City of Superior, WI. The water was declorinated by charcoal filtration and sodium sulfite adition.; Alk=46.5 pH=7.2 H=60.5 DO=93.8	Static T=24	M	4	Reduction in frond production	12.5	25.5 (32%)	>25.5	Brooke et al., 1986	R
Raphidocelis subcapitata	Sb_2O_3	Sb(III)	Algal medium (recommended in OECD Guideline 201).	Static T=23.8-24.2	M	3 3	growth (b) growth rate (r)	0.323 0.396	0.396 (4.4%) 1.32 (2.3%)	> 2.4 (9.2%) > 2.4 (3%)	LISEC, 2001	R
Raphidocelis subcapitata	$SbCl_3$	Sb(III)	Algal medium (recommended in OECD Guideline 201), using artificial dissolved organic	Static T = 23 ± 1	M	3 3	growth (b) growth rate (r)	2.11 2.11	4.00 (27.2%) 4.00 (7.9%)	> 36.6 > 36.6	Heijerick and Vangheluwe, 2004	R

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/l)	LOEC (mg Sb/l; % effect)	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
			matter instead of EDTA									
Chlorella vulgaris	$Sb_2O_3$2SbOCl	Sb(III)	AM	Static room temp.	N	90-120	Growth	0.032	0.064		Dooren De Jong, 1965	NR
Microcystis aeruginosa *Scenedesmus quadricauda*	$K[Sb(OH)_6]$	SbV	AM	Static T = ?	N	8 8	Growth Growth	23 >982			Bringmann and Kuhn, 1978	NR
Selenastrum capricornutum	Sb_2O_3	Sb(III)	AM	Static T=23.6 – 24.1	M Measurement on "Whole media" (= dissolved and dispersed amount of test material)	3 3	Growth (b) Growth rate (r)	8.3 8.3		28.4 55.8	LISEC, 1994	NR
Selenastrum capricornutum	Sb_2O_3	Sb(III)	?	?	?	4	Inhibition of chlorophyll a			0.61	Anonymous, 1978	NR
Natural phytoplankton populations	$Sb[K(SbO)C_4H_4O_6 \cdot ½H_2O]$	Sb(V) ?	Filtered natural seawater, enriched with N, P and Si.	Static T = 12 - 16	N	4	Growth (inhibition of chlorophyll a)			EC_{50} was not measured	Hollibaugh et al.e	NR
Marine water												
Skeletonema costatum	Sb_2O_3	Sb(III)	Salt water	?	?	4	Inhibition of chlorophyll a			>4.2	Anonymous, 1978	NR

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Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/l)	LOEC (mg Sb/l; % effect)	E(L)C50 (mg Sb/l; % effect)	Reference	Reliable & Relevant
							Reduction in cell number			>4.2		
Thalassiosira aestevalis	Sb[K(SbO)$C_4H_4O_6 \cdot ½H_2O$]	Sb(V) ?	Filtered natural seawater, enriched with N, P and Si.	Static T = 12 - 16	N	4	Growth (inhibition of chlorophyll a)			EC_{50} was not measured	Hollibaugh et al., 1980	NR

Alk = alkalinity (mg $CaCO_3$/l); AM = artificial medium; DO = dissolved oxygen (mg O_2/l); H = hardness (mg $CaCO_3$/L); T = temperature (ºC)

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3.2.4.1.4 Microorganisms

There is only one study on microorganisms available that is considered relevant and reliable. In this study (EPAS, 2005) the inhibition of $SbCl_3$ on nitrification in activated sludge was assessed according to the ISO 9509 protocol (Water quality – Method for assessing the inhibition of nitrification of activated sludge micro-organisms by chemicals and waste waters). The protocol was slightly modified as the NH_4-N concentration in the prescribed medium had an inhibitory effect on the nitrification. The test was performed by aerating a nitrifying sludge in the presence and absence of Sb. The nominal Sb concentrations used ranged from 1.1 mg Sb/l to 1200 mg Sb/l. The protocol was modified by washing the sludge with tap water instead of demineralised water and medium because the proposed buffer was not sufficient to ensure a stable pH. By modifying the protocol this way, an activity of the sludge of 4.5 mg N/(g.h) could be calculated, which lies within the proposed interval of 2 to 6.5 mg N/(g.h). Each test solution was set up in double. The flasks were incubated for 4 hr at a constant temperature and aerated in the dark. No inhibitory effects could be observed at a measured concentration of 2.55 mg Sb/l. Based on measured Sb concentrations an EC_{50} of 27 mg Sb/l was derived (C.I. 14-52 mg Sb/l).

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Table3-100 presenting toxicity tests for microorganisms.

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/l)	E(L)C50 (mg Sb/l)	Reference	Reliable & Relevant
Nitrifying bacteria	$SbCl_3$	Sb(III)	Medium according to ISO 9509	Static	M	0.17	Inhibition of nitrification	2.55	27	EPAS, 2005	R
Entosiphon sulcatum	$K[Sb(OH)_6]$	Sb(V)	Cell-medium	Static T = 25	N	3	Growth	86		Bringmann, 1978	NR
Tetrahymena pyriformis	$SbCl_3$	Sb(III)	PPYSm medium (proteose peptone yeast-extract substrate modified)	Static T = 28	N	1.5	Growth (Relative doubling time)		Microplate: 6 Flask: 16	Sauvant et al., 1995a	NR
Tetrahymena pyriformis	$SbCl_3$	Sb(III)	PPYSm medium (proteose peptone yeast-extract substrate modified)	Static T = 28	N	0.125 0.25 0.375	Growth (Relative doubling time)		60 38 20	Sauvant et al., 1995b	NR

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3.2.4.1.5 Amphibians

The study by Birge (1978) on *Gastrophryne carolinensis* is the only study of antimony toxicity to amphibians (see Table3-101 below). However, since this study contains no information of the concentrations used in the experiments, the results cannot be used in the RAR.

Table3-101 presenting toxicity data for amphibians.

Organism	Compound/ Valency	Medium	Test conditions	Exposure period (d)	Endpoint	NOEC (mg Sb/l)	E(L)C_{50} (mg Sb/l)	Reference	Reliable & Relevant
Gastrophryne carolinensis embryo larval	$SbCl_3$/ Sb(III)	Reconstituted water H = 92-110; D.O = 9.3-10.1; pH = 6.9-7.8,	Semi static measured conc. T = 22	7	Mortality	0.0038 (LC_1)	0.3	Birge, 1978	NR

Alk = alkalinity (mg $CaCO_3$/l); DO = dissolved oxygen (mg O_2/l); H = hardness (mg $CaCO_3$/L); T = temperature (ºC)

3.2.4.1.6 Calculation of Predicted No Effect Concentration (PNEC) for surface water

There are reliable and relevant short-term toxicity data available for a number of species including fish, daphnia and algae. However, the most sensitive species in the acute toxicity tests was the green hydra *Chlorohydra viridissimus*, with an LC_{50} –value of 1.77 mg Sb/l.

Relevant and reliable long–term toxicity studies are available for fish, daphnia and algae. The most sensitive organisms in the long-term toxicity tests seem to be fish. The lowest NOEC (1.13 mg Sb/l) has been derived in a study on the fish *Pimephales promelas.* According to the TGD, an assessment factor of 10 shall be applied on the lowest NOEC when NOECs from three trophic levels are available.

There is however a footnote (Notes: c) in the TGD, when deriving a $PNEC_{aquatic}$, that says that an assessment factor of 50 should be used on the lowest of three NOECs covering three trophic levels when such NOECs have not been generated from that trophic level showing the lowest L(E)C50 in the short term test. According to Appendix IV in the TGD, fish (lowest NOEC) and hydra (lowest L(E)C50) both belong to the trophic level "Secondary consumers". As a consequence of that this footnote does not apply on this dataset, and an assessment factor of 10 should be used.

Using the assessment 10 on the lowest NOEC results in the following $PNEC_{aquatic}$:

$PNEC_{aquatic}$ = 0.113 mg Sb/l

3.2.4.1.7 Calculation of Predicted No Effect Concentration (PNEC) for STP

There is one relevant and reliable study available on microorganisms that can be used to derive the $PNEC_{microorganisms}$. Based on measured Sb concentrations a NOEC of 2.55 mg Sb/l and an EC_{50} of 27 mg Sb/l was derived.

The $PNEC_{microorganisms}$ can be calculated from the EC_{50} from a test performed with nitrifying bacteria by dividing the EC_{50} by 10 or can be set equal to the NOEC from such a test (TGD, 2003). This would result in a $PNEC_{microorganisms}$ of 2.55 or 2.7 mg Sb/l. Selecting the lowest results in

$PNEC_{micro\text{-}organisms}$ = 2.55 mg Sb/l

3.2.4.2 Toxicity test results for sediment organisms

Fresh water

Three reliable and relevant chronic sediment toxicity tests with different single species are available. The test species all have different exposure routes, feeding habits and ecological niches: (1) the bottom-dwelling *Hyalella azteca* (crustacean) is a surface deposit and filter feeder, (2) *Chironomus riparius* (insect) burrows within the sediment with a combined surface and subsurface feeding behaviour, and (3) *Lumbriculus variegatus* (oligochaete) is a head-down deposit feeder that feeds well below the sediment-water interface.

In this study (Heijerick and Vangheluwe, 2003c), the amphipod *Hyalella azteca* was exposed to various concentrations of trivalent antimony ($SbCl_3$) in a sediment-water system (see Table3-102 below). The amphipods were exposed for 28 days. After this period, the amphipods were separated from the sediment and placed in sediment-free chambers for another 14 days. During this period, survival (day 28, 35, 42), growth (day 28, 42) and reproduction (number of young per female produced from day 28 to 42) were measured. This study was performed with 13 replicates, five concentrations (range: 30.8 - 249 mg Sb/kg ww; 44 - 355 mg Sb/kg dw) and a control, with each replicate consisting of 10 amphipods (for more information on the study, see table 3.2.11 below). The 13 replicates were used as follows: four replicates were used for the 28-day growth and survival endpoints and eight for the measurements of survival and reproduction on day 35 and 42 (incl. growth). The remaining replicate was used for the performance of chemical measurements. The overlying water was renewed at least three times a week (±75%). The most sensitive endpoint was growth (both weight and length) after 28 days of exposure, which resulted in a NOEC of 87 mg Sb/kg ww (124 mg Sb/kg dw).

Larvae of the midge *Chironomus riparius* were exposed to a concentration range of $SbCl_3$ in a sediment-water system (Heijerick and Vangheluwe, 2005a; Table3-102). The test procedure was based on the OECD draft proposal for a new guideline 218 “Sediment-Water Chironomid Toxicity Test Using Spiked Sediment” (Draft February 2001, adopted April 2004). Two-day old larvae were exposed to spiked sediment until the larvae transformed to the adult phase. Mortality and growth of the larvae, and emergence to midges were determined after 14 and 28 days of exposure. The study was performed with six concentrations (23 - 445 mg Sb/kg ww; 33 - 636 mg Sb/kg dw) and a control (<1.4 mg Sb/kg ww; <2.0 mg Sb/kg dw), and 11

replicates per concentration of which five replicates were used to determine survival and growth after 14 days of exposure, five to determine emergence after 28 days of exposure, and the remaining replicate to perform chemical analyses. The overlying water was renewed at least three times a week (±75% renewal). The most sensitive endpoint was growth (weight), which resulted in a NOEC of 78 mg Sb/kg ww (112 mg Sb/kg dw).

Adults of the oligochaete *Lumbriculus variegatus* were exposed to a concentration range of $SbCl_3$ in a sediment-water system (Heijerick and Vangheluwe, 2005b; Table3-102). The test procedure was based on the OECD draft "Bioaccumulation; Sediment test using benthic oligochaetes" (January 2000; 3rd revised draft) and the EPA Guideline 600/R-99/064 "Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates", Section 13, Test Method 100.3 "*Lumbriculus variegatus* bioaccumulation test for sediments". The test organisms were exposed for 28 days to the spiked sediment. At the end of the exposure period survival, reproduction and growth were monitored. The study was performed with six concentrations (23 - 445 mg Sb/kg ww; 33 – 636 mg Sb/kg dw) and a control (<1.4 mg Sb/kg ww), and 7 replicates per concentration of which six replicates were used to determine survival, reproduction and growth after 28 days of exposure and the remaining replicate to perform chemical analyses. The overlying water was renewed at least three times a week (±75% renewal). The most sensitive endpoint was growth (weight), which resulted in a NOEC of 78 mg Sb/kg ww (112 mg Sb/kg dw).

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Table3-102 presenting toxicity tests for sediment-living organisms in freshwater.

Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/kg ww.)	NOEC (mg Sb/kg dw.)	LOEC (mg Sb/kg ww.; % effect)	LOEC (mg Sb/kg dw.; % effect)	Reference	Reliable & Relevant
Chironomus riparius Age = 2 days	$SbCl_3$	Sb(III)	Artificial sediment (in % of sediment dry weight): peat 5%, quartz sand 75%, kaolinite clay 20%, pH 7.0 ± 0.5, organic carbon 2±0.5%, calcium carbonate 0.05-0.1%, water 30%. EPA medium as overlying water H=234-252 pH=7.52-7.79 DO=4.90-5.45	Semi-static T=20 ± 1	M	14 28	Survival Growth (weight) Emergence	≥ 445 <u>78</u> ≥ 445	≥ 636 112 ≥ 636	> 445 120 > 445	> 636 172 > 636	Heijerick and Vangheluwe, 2005a	R
Hyalella azteca Age = 6-7 d	$SbCl_3$	Sb(III)	Artificial sediment (in % of sediment dry weight): peat 5%, quartz sand 75%, kaolinite clay 20%,	Semi-static T=23	M	28 42	Growth (weight) Growth (length) Survival Growth (weight) Growth (length)	86.8 86.8 190.4 190.4 190.4	124 124 272 272 272	190.4 (29%) 190.4 (8%) 248.5 (21%) 248.5 (34%)	272 (29%) 272 (8%) 355 (21%) 355 (34%)	Heijerick and Vangheluwe, 2003c	R

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Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/kg ww.)	NOEC (mg Sb/kg dw.)	LOEC (mg Sb/kg ww.; % effect)	LOEC (mg Sb/kg dw.; % effect)	Reference	Reliable & Relevant
			organic carbon 2±0.5%, calcium carbonate 0.05-0.1%, water 30-50%. Borgman medium as overlying water H=148±21.6 pH=7.92-7.95 DO=5.52-5.92				Reproduction (number of young per female)	190.4	272	248.5 (17%) 248.5 (89%)	355 (17%) 355 (89%)		
Lumbriculus variegatus Age = 7 days	$SbCl_3$	Sb(III)	Artificial sediment (in % of sediment dry weight): peat 5%, quartz sand 75%, kaolinite clay 20%, pH 7.0 ± 0.5, organic carbon 2±0.5%, calcium carbonate 0.05-0.1%, water 30%. EPA medium as overlying	Semi-static T=25 ± 1	M	28	Survival / reproduction Growth (weight)	120 <u>78</u>	172 112	244 120	348 172	Heijerick and Vangheluwe, 2005b	R

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Organism	Compound	Valency	Medium	Test conditions	Nominal/ Measured	Exposure period (d)	Endpoint	NOEC (mg Sb/kg ww.)	NOEC (mg Sb/kg dw.)	LOEC (mg Sb/kg ww.; % effect)	LOEC (mg Sb/kg dw.; % effect)	Reference	Reliable & Relevant
			water H=222-267 pH=7.73-7.94 DO=5.0-5.9										

DO = dissolved oxygen (mg O_2/l); T = temperature (ºC)

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3.2.4.2.1 Calculation of Predicted No Effect Concentration (PNEC) for sediment organisms

Reliable and relevant chronic NOEC values are available for three species with different living and feeding conditions. According to the TGD (2003) the $PNEC_{sediment}$ shall be derived from the lowest NOEC divided by an assessment factor of 10. The lowest NOEC (78 mg Sb/kg ww or 112 mg Sb/kg dw) has been derived for the midge *Chironomus riparius* and the oligochaete *Lumbriculus variegatus*.

This results in the following $PNEC_{sediment}$:

$PNEC_{sediment}$ = 11.2 mg Sb/kg dw (7.8 mg Sb/kg ww).

3.2.4.3 Predicted no effect concentration (PNEC) for the marine compartment

3.2.4.3.1 Water

The only reliable and relevant toxicity study of antimony available for the marine environment is an acute toxicity study on the marine fish *Pagrus major*, which resulted in a LC_{50} of 6.9 mg Sb/l. Therefore, toxicity data from tests on freshwater organisms are used to derive a $PNEC_{marine\ water}$. Relevant and reliable long–term toxicity studies are available from three trophic levels. The lowest NOEC (1.13 mg Sb/l) has been derived in a study on the fish *Pimephales promelas.*

According to the TGD, an assessment factor of 100 should be applied to the lowest of three long-term NOECs from three freshwater or saltwater species (normally algae and/or crustaceans and/or fish) representing three trophic levels.

There is however a footnote (Notes: c) in the TGD, when deriving a $PNEC_{marine\ water}$, that says that an assessment factor of 500 should be applied to the lowest NOEC when the lowest NOEC has not been derived from the taxonomic group showing the lowest EC_{50}-value. This should apply to the available data since the hydra (lowest $L(E)C_{50}$) belongs to a different taxonomic group than the fish (lowest NOEC). But since the same freshwater data set is used for both the freshwater and marine water assessment, and the "default" difference between freshwater and marine water is a factor of 10, it would be inconsistent to use an assessment factor of 10 for freshwater and an assessment factor of 500 for marine water.

Based on the above, it is decided to use an assessment factor of 100 on the lowest of the three available freshwater NOECs, which results in the following PNECmarine water:

$PNEC_{marine\ water}$ = 11.3 µg Sb/l

3.2.4.3.2 Sediment

As no studies on marine sediment organisms are available the $PNEC_{marine\ sediment}$ is derived from studies on freshwater sediment species. Reliable and relevant chronic NOEC values are available for three freshwater sediment species with different living and feeding conditions. The freshwater NOEC is 78 mg Sb/kg ww or 112 mg Sb/kg dw. According to the TGD, the assessment factor used should be 50, which results in the following $PNEC_{marine\ sediment}$:

$PNEC_{marine\ sediment}$ = 2.24 mg Sb/kg dw (1.6 mg Sb/kg ww).

3.2.5 Terrestrial compartment

As a consequence of the outcome of TC NES I ´07 (see section 3.2.1.1.2), the preferred exposure regime for terrestrial toxicity studies is sufficiently aged soils spiked with Sb_2O_3.

3.2.5.1 Toxicity test results

It was at TC NES I ´07 decided that both the critical NOECs on plants (Oorts et al., 2005; 5.8 mg Sb/kg dw) and invertebrates (Kuperman et al., 2002) and (Phillips et al., 2002) ; both 58 mg Sb/kg dw), respectively, would be overruled by the results of new plant and invertebrate tests, if considered valid by the Rapporteur.

Two new toxicity studies, which both are considered valid and performed in the same Sb_2O_3-spiked sufficiently aged soil, generate the toxicity data for invertebrates, plants and microrganisms that are used to derive $PNEC_{soil}$ (and thereby overrule the results by (Oorts et al., 2005; Kuperman et al., 2002 and Phillips et al., 2002).

The first of these two studies were performed by (Smolders et al., 2007) on the endpoints lettuce emergence and growth (ISO 11269-2), barley root elongation (ISO 11269-1) and nitrification rate (ISO 14238).

The second study was performed by (Moser 2007) on the endpoints mortality and reproduction of *Folsomia candida* (ISO 11267).

The NOEC used are presented in Table3-103 below, and are not considered to be confounded by the addition of counter ions and/or protons.

Table3-103 presenting lowest reliable NOEC for different trophic levels of terrestrial organisms.

Organism	Exposure period	Endpoint	LC_{50}/LOEC/NOEC	Value (mg Sb/kg dw)	Reference
Hordeum vulgare	5 d	Barley root elongation	NOEC	999	Smolders et al., 2007
Folsomia candida	28 d	Reproduction	NOEC	999	Moser, 2007
Native microorganisms	7 d	Potential Nitrification Rate	NOEC	2930	Smolders et al., 2007

3.2.5.1.1 Plants

There are four studies available on Sb toxicity to plants, but as stated above, only the study by (Smolders et al., 2007),which results in a bounded NOEC of 999 mg Sb/kg dw, will be used when deriving $PNEC_{soil}$.

Study 1

(Smolders et al., 2007) studied the toxicity of Sb to plants (lettuce emergence and growth and barley root elongation) in Sb_2O_3 amended soil, which had aged for 31 w in field (starting 2006-12-19). The dose levels used were 3, 90, 322, 999, 2930 and 10119 mg Sb/kg dw (measured concentrations).

For the plant growth assay using lettuce (ISO 11269-2), the number of seedling that emerge per pot and the shoot yield of the plants were determined. The soils were fertilized and pre-incubated for six days before twenty lettuce (*Lactuca sativa*) seeds were sown per pot. Four pots per concentration were used. The plant emerged after three days and at day six, the number of seedlings was counted (see Figure 3-9) and the plants were thinned out to 5 plants per pot. Fourteen days after 50% of the control plants (day 3) had emerged, plants were harvested and the fresh biomass was weighted (see Figure 3-10)

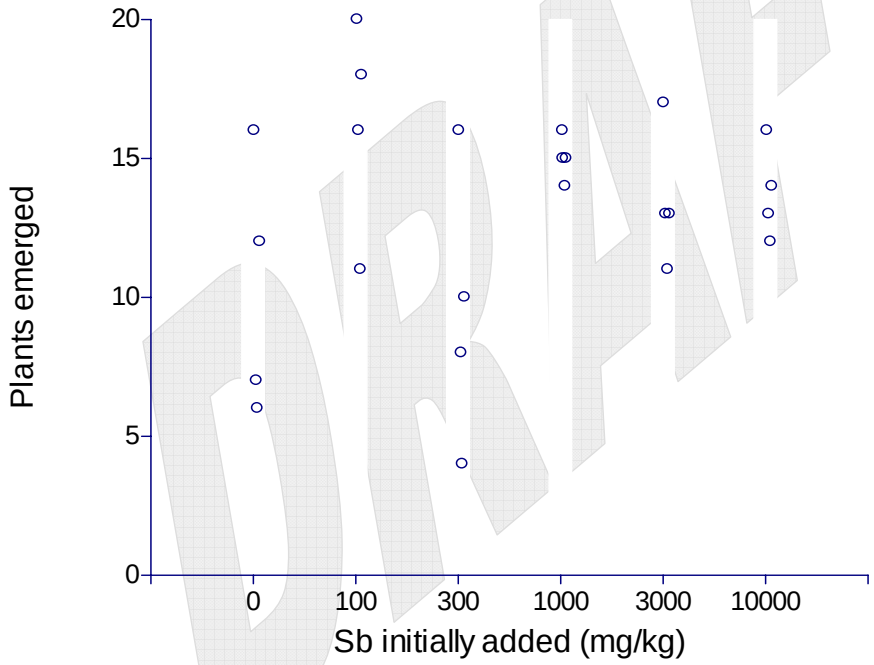

Figure 3-9 Dot plot showing the number of lettuce plants (twenty seeds per pot, four pots per dose) emerged exposed to antimony at different doses.

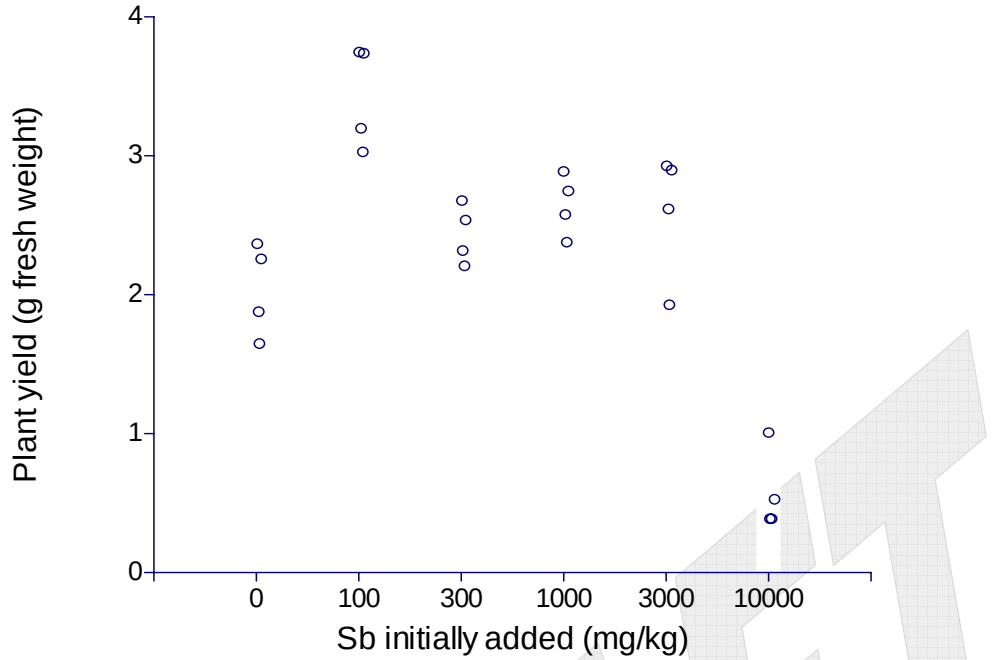

Figure 3-10 Dot plot showing the number of plant yield of lettuce plants (five seedlings per pot, four pots per dose) exposed to antimony at different doses.

According to the guideline ISO 11269-2 from 1995 (which was the guideline used in this study), the emergence shall be sufficient to provide 5 healthy seedlings per pot in the control. In the study, 6 healthy seedlings were recorded in 1 control replicate, and 7-16 seedlings in 3 replicates, resulting in an arithmetic mean number of 10 seedlings per control pot. However, according to the guideline ISO 11269-2 from 2005, the emergence should be at least 7 healthy seedlings out of 10 planted seeds which mean that the lettuce emergence is not valid according to these newer validity criteria.

The lettuce assay resulted in a bounded NOEC of 2930 mg Sb/kg dw, a LOEC of 10119 mg Sb/kg dw and an EC_{10} of 4517 mg Sb/kg dw on lettuce biomass (shoot) yield and an unbounded NOEC of 10119 mg Sb/Kg dw on lettuce seed emergence (see Figure 3-10).

To conclude, this means that the lettuce results (emergence and growth) from this study are valid and reliable according to the guideline to which is was performed, but that the results on emergence are not compliant with the criteria in the most recent ISO guideline (from 2005) and according to those therefore not considered valid. However, even if the emergence in the control group would have been 70% (and the study therefore would have been valid according to the most recent ISO guideline), an effect on emergence would still not have been possible to detect. Based on this, the results are still considered useful since they provide a very strong indication that growth is a more sensitive endpoint than emergence for lettuce.

In the survey of the barley root elongation assay (ISO 11269-1), barley (*Hordeum vulgare*) was pre-germinated for three days and four seeds were planted in each pot. Three replicate per Sb-concentration were used. After five days of growth, intact roots were washed out of the soil matrix and the length of the longest root on each plant was recorded. The mean length of the longest root of all replicate samples per soil was determined. There are no validity criteria given in the guideline. However, a mean root elongation in the controls at test end of 9.1 cm with a relative standard deviation of 5.1% indicates that the results from the root elongation test are valid.

The root elongation study resulted in a bounded NOEC of 999 mg Sb/kg dw, a LOEC of 2930 mg Sb/kg dw and an EC_{10} of 1931 mg Sb/kg dw (see Figure 3-11).

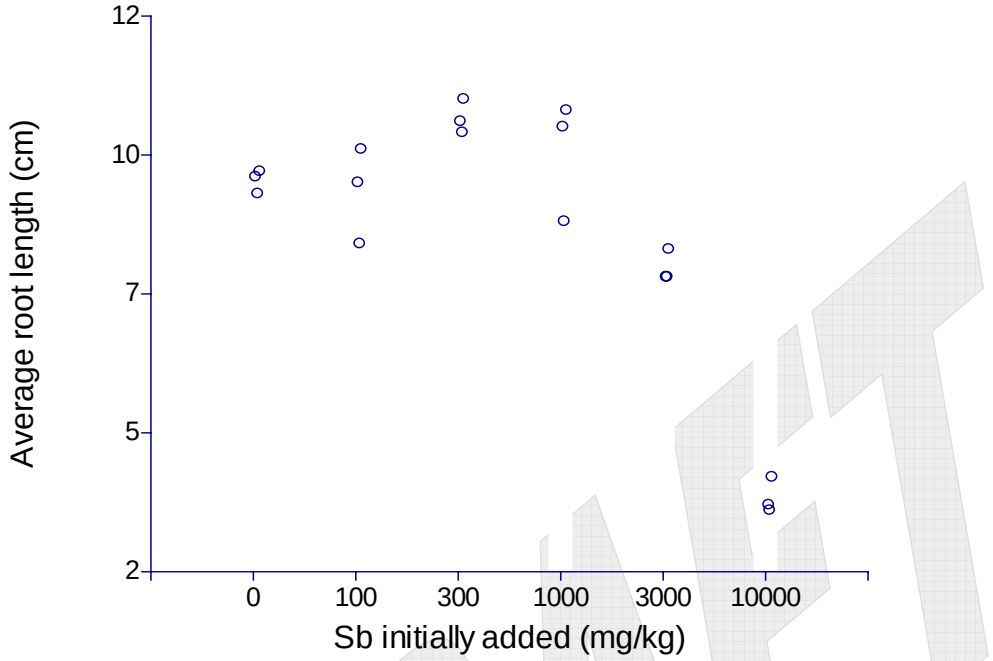

Figure 3-11 Dot plot showing the average root length of barley (four seeds per pot, three pots per dose) exposed to antimony at different doses.

The Sb pore water concentration corresponding to the bounded NOECs of 999 and 2930 mg Sb/kg dw were 9.7 and 18.7 mg Sb/l, respectively.

In order to answer the question whether or not the soil amended with Sb_2O_3 and aged for 31 weeks was fully reacted and equilibrated, the authors used modelling of the Sb_2O_3 solubilization process in soil based on all previously available data. It was found that 31 weeks of ageing in field was sufficient in order to have fully reacted and equilibrated soils at high doses, but not at low. For example, the model predicts a Sb pore water concentration of 26 mg Sb/l at 999 mg Sb/kg dw whereas the observed concentration in this study was 9.7 mg Sb/l.

As long as equilibrium is not reached the pore water concentration will gradually increase during the course of time. How much the pore water concentration will have increased, from the initiation of the study until the termination, is not possible to know since the pore water concentration was only measured at the end of the study. However, the difference in pore water concentration between the initiation and termination of the bioassay is not considered to be of importance considering that the test started at 31 w of ageing and the bioassay continued for another 5 d for barley (which resulted in the lowest NOEC). It is neither considered to be of importance for the lettuce which continued for an additional 12 d, as compared to the barley. As a consequence of that, a sufficiently constant toxicity pressure is considered to have been maintained during the bioassays.

To conclude, the NOEC of 999 mg Sb/kg dw for barley root elongation from this study will be used in order to derive $PNEC_{soil}$, but this value still has to be corrected for the fact that the equilibrium pore water concentration was not reached (see section 3.2.5.1.4).

Study 2

In the second study, (Oorts et al., 2005) studied the toxicity of Sb to plants (lettuce shoot yield) in freshly spiked soil using Sb_2O_3 (measured conc.; 0.6, 12.4, 34.5, 67, 124, 422, 897, 1804 mg Sb/kg dw.) or $SbCl_3$ (measured conc.; 0.6, 10, 43.2, 73.1, 159, 384, 836, 1741 mg Sb/kg dw.), and in five year aged Sb_2O_3 spiked soil (measured conc.; 0.4, 5.8, 28.4, 71.7 and 116 mg Sb/kg dw.). In addition, a plant growth test was also performed in field in the aged Sb_2O_3 spiked soil.

In order to study the potential effect of the counter ion (chloride) resulting from spiking with $SbCl_3$, soil was also spiked with $CaCl_2$ at equivalent chloride doses using identical preincubation and spiking procedures. Soil solution from all Sb amended soils was extracted 4 weeks after spiking by centrifugation and analysed using ICP within two days after extraction. Soil solution was also analysed with respect to pH and electrical conductivity.

The lettuce shoot yield test was based on ISO 11269-2. Twenty lettuce seeds (*Lactuca sativa* (cv. Pontiac) were sown in three replicate pots for each treatment. Following emergence, seedlings were thinned to 5 plants per pot. Plants were harvested 24 days after sowing. At harvest, shoots were cut just above the soil surface and dried at 70 ºC for at least 48 h, and the dry matter yield was recorded.

In addition to the plant test performed in lab, a field study using aged Sb_2O_3 spiked soil was also performed, using 18 d old pre-cultivated plants. Four plants were planted on each container with aged soil (control, and the originally added concentrations 10 mg Sb/kg, 50 mg Sb/kg and 250 mg Sb/kg). After an exposure period of 2 months, the plants were harvested (2005-09-15). Above ground biomass and root weight were measured for each plant. Edible parts were selected and oven dried at 50 ºC. Dry plant material was ground and used for Sb analysis. Total metal concentrations were determined by boiling nitric acid digestion and subsequent analysis with ICP-OES.

The use of freshly spiked Sb_2O_3 resulted in an unbound NOEC of 1804 mg Sb/kg dw (nominal added concentration of 2000 mg Sb/kg dw.).

The use of freshly spiked $SbCl_3$ resulted in a NOEC of 43 mg Sb/kg dw (nominal added concentration of 50 mg Sb/kg dw.).

The use of five-year-old aged Sb_2O_3 spiked soil in a lab test resulted in dose-response effects of Sb, and a NOEC of 28 mg Sb/kg dw. (nominal dose, 50 mg Sb/kg dw.). (Oorts et al., 2005) did not consider this result relevant and searched for a confounding factor, and found that the container with the highest concentration was located next to a hedge row that had been treated with an herbicide (CANYON, mixture of glyphosphate, diuron and diflufenican). The second highest concentration resulted from mixing soil from the highest dose with a lower dose, and had therefore also been potentially contaminated with the herbicide. Thus, the effects observed in the two highest doses may be confounded due to the presence of a herbicide.

Another test using the aged Sb_2O_3 spiked soil in a field study with 18 d old precultivated plants resulted in unbound NOECs of 116 mg Sb/kg dw. for both lettuce shoot yield and root yield. However, the experimental set-up between the lettuce grown in the laboratory and in

the field differed in several important aspects, why any conclusions based on the difference in toxicity are difficult to make (e.g. the field tests were initiated with 18 d old pre-cultivated plants while seeds were used in the lab tests).

Since the original test design for the aged Sb_2O_3 spiked soils tested at lab included four doses and a control, removing the two highest doses thus results in two doses and a control. The highest remaining dose is 28 mg Sb/kg dw Our evaluation of the data using a step-down approach results in a NOEC of 5.8 mg Sb/kg dw (nominal added concentration of 10 mg Sb/kg dw). There are however a number of reasons that raises concern against this value; (i) there are only two doses above the control (of which the lowest is the NOEC and the other is LOEC), (ii) the NOEC value resulting from $SbCl_3$ is 43 mg Sb/kg (for which the observed effect is attributed to the effect of the chloride), (iii) the Sb pore water concentration, which is used as a simplified measure of toxic pressure, was far below the critical pore water concentration in the freshly spiked soils (the largest pore water concentration in the aged soils was 0.7 mg Sb/l, whereas the lowest LOEC in freshly $SbCl_3$ spiked soils were 3.6 mg Sb/l), and (iv) the maximum internal Sb concentrations in the shoot for the five year aged Sb_2O_3 spiked soils (1 mg Sb/kg dw) was far below the values at which toxicity was observed in soil freshly spiked with Sb_2O_3 or $SbCl_3$ (>10 mg Sb/kg dw).

It is, as mentioned in section 3.2.5.1, decided not to use this NOEC when deriving $PNEC_{soil}$.

Study 3

This study is the dissertation thesis by Pearce (1978) named "The effect of antimony on plants". In general, the longer the plants were grown in antimony treated medium (medium coarse sand), the higher the antimony level in the plant, particularly in the roots. All concentrations presented are nominal concentrations.

A general observation was that antimony toxicity is most pronounced at the roots, which is noticed as stunting and disturbances of root growth, and the fact that effects on root growth are noticed at lower antimony concentrations than effects on shoot growth. The negative effect of antimony on nodulation in legumes was considered to be due to both bacterial inhibition and suppression of lateral root formation. The effect of antimony in forms other than diantimony trioxide produces similar symptoms as the trioxide, though often at lower concentrations.

Some plants, e.g. *Amaranthus* and *Polygonum*, grew very poorly in high levels of Sb_2O_3 (4000 mg Sb/kg) and developed no true leaves. In such cases, root development was very poor and rapidly deteriorated with treatment time. *Amaranthus* grown in the presence of antimony sulphate or antimony tartrate developed very poor roots at lower concentrations, 400 and 40 mg Sb/kg, respectively.

However, in other species, e.g. *Rumex obtusifolius* high levels of Sb_2O_3 (4000 mg Sb/kg) resulted in dwarfed plants, which appeared to have healthy, though small, shoots and leaves but very poor roots. *Zea mays* and *Hordeum vulgare* did not exhibit such large reductions in fresh and dry weights in response to antimony, but the roots of these plants were somewhat stunted and thickened at Sb_2O_3 concentrations of 40 000 mg Sb/kg. This latter effect, i.e. on roots, seemed to be an important symptom of antimony injury in most plants tested and was accompanied by both a reduction in number and blackening of the apices.

In contrast to the deleterious effects of antimony at high concentrations, enhanced growth was observed for some plant species at low concentrations.

The effect of germination of *Pisum sativum* in the presence (40 and 400 mg Sb/kg) or absence (control) of Sb_2O_3 was studied. No nodules were observed in either of the antimony concentrations used. Experiments on nodulation were also carried out on two species of *Trifolium*. In both, *Trifolium pratense* and *Trifolium repens* nodulation was inhibited, however at a higher concentration of Sb_2O_3, i.e. 4000 mg Sb/kg Sb. This suggests that inhibition of nodulation by antimony may be a general phenomenon not only restricted to peas. In addition, Pearce (1978) studied the growth in agar of the strain *Rhizobium* 1045 exposed to either diantimony trioxide or tartar emetic. No growth was observed at 40 and 400 mg Sb/kg for both antimony compounds, which indicate that antimony may exert microbial toxicity (see table 3.2.15).

However, considering that the NOEC/LOEC-values are based on visual endpoints presented in nominal concentrations (measurements were not performed on the concentrations tested), measured weights and lengths were reported but without any calculated statistics, and the medium used (medium coarse sand), the results from this study can only be used as indications of Sb toxicity.

Study 4

He and Yang (1999) studied the effect on antimony on rice during germination and growth. Antimony was applied either as antimony potassium tartrate (III) or as potassium antimonate (V). Two types of exposure regimes were performed, one germination study, and one pot study.

In the *germination study*, rice seed were grown in culture medium supplemented with antimony potassium tartrate (III) or potassium antimonate (V) to provide the Sb-concentrations: 0, 1, 5, 10, 20, 50, 100, 200, 500, and 1000 mg Sb/l. Three groups of cultures, and each culture was run in triplicate. It is unclear how many rice seedlings each culture contained.

Using the trivalent antimony compound ($2KSbOC_4H_4O_6 \cdot H_2O$) resulted in a stimulation of sprout growth in the low range concentrations (1-50 mg Sb/l), with an increase of 17% at the 50 mg dose, as compared to the control. However, in the high range concentrations (100-1000 mg Sb/l), the sprout growth decreased (with 37% at the 1000 mg dose), as compared to the control. The root growth was retarded at all concentrations (1-1000 mg Sb/l), with a decrease of 92% at the highest dose, as compared to the control.

Using the pentavalent antimony compound ($KSb(OH)_6$) resulted in a decrease of both sprout and root growth over the entire range of concentrations (1-1000 mg Sb/l), with a decrease in length of 64% and 100%, respectively, as compared to the control.

In the *pot study* antimony was added to the soil either as antimony potassium tartrate (III) or potassium antimonate (V) to provide the Sb-concentrations: 0, 1, 5, 10, 50, 150, 300, 600, and 1000 mg Sb/kg of soil. Each treatment was made in four replicates. The rice (Yuhong 1) was grown as a test crop in the experiment, and was after 2 weeks, planted in each pot.

Using trivalent antimony did not result in any damage in the transplanted rice seedlings after three days, as they started to turn green. By the tenth day, the difference in the way rice was growing was obvious. At the highest concentrations (300-1000 mg Sb/ml) the leaves became

yellow and the growth was reduced. The effect on the root system was most obvious, with few and short root systems, and more red-brown fibrils. There was a clear dose-dependent relationship, where increasing amounts of antimony applied resulted in decreased growth and productivity of the rice. The biomass (g/pot), plant height (cm), relative ratio of rice tillering (%), effective ear of rice (ears/pot), weight of 1000 grains (g), and yields of rice (g/pot) at the highest dose were only 32%, 61%, 68%, 26%, 94%, and 8% to that observed for the control, respectively.

Using the pentavalent antimony mainly resulted in a drop in rice yields. The yield of rice at highest antimony concentration was 80%, as compared to the control.

The authors wrote that if a 10% drop in rice yield is regarded as a critical value of damage of antimony, based on the results observed in the pot experiments, the critical concentrations of Sb(III) and Sb(V) in soils are 150 and 300 mg Sb/kg, respectively. The drop for Sb(III) at 50 mg Sb/kg dw. was 10%, and 21% at 150 mg Sb/kg dw., and these values can therefore be regarded as a tentative NOEC and LOEC resulting from this study.

The results shows that the phytotoxicity of Sb(III) is higher than that of Sb(V). However, no NOEC/LOEC/ECx were calculated (using statistical methods), nor is it possible from the data in the article to calculate any. It is therefore not possible to derive any NOEC from this study, that can be used to derive a $PNEC_{soil}$, and the results can only be used as indications of no-effect levels.

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Table3-104 presenting LC_{50}, LOEC, and NOEC for plants.

Organism	Compound	Valency	Medium	pH	OC (%)	CEC*	WHC** (%)	Clay (%)	Sand (%)	Silt (%)	Growing period (d)	Endpoint	NOEC (mg Sb/kg; % effect)	LOEC (mg Sb/kg; % effect)	E(L)C50 (mg Sb/kg; % effect)	Reference	Reliable & Relevant
Hordeum vulgare	Sb_2O_3	Sb(III)	Loam	6.5-6.7	0.9	9.8	53	14	10	76	5	Root elongation	999	2930	1931 (10%) 6937 (50%)	Smolders et al., 2007	R
Lactuca sativa		Sb(III)										Seed emergence	>10119				R / NR***
												Weight (shoots; fresh weight)	2930	10119	4517 (10%) 7514 (50%)		
Amaranthus caudatus	Sb_2O_3	Sb(III)	Medium coarse sand								4 weeks	Length (shoots)	400 (+8%) (very healthy plants, sligtly larger than controls)	800 (24%) (smaller than controls, less vigorous)	2000 (59%) (upturned cotyledons, very straggly roots)	Pearce, 1978	NR
												Weight (shoots; fresh weight)	400 (+21%)		<2000 (92%)		
												Length (roots)	400 (+11%)	800 (45%)			
												Weight (roots; fresh weight)	800 (+50%)		>2000 (35%)		
														800 (12.5%)	2000		
														2000 (50%)			
Chenopodium album												Weight (shoots; dry weight)	400 (+37%) (plants larger than	4000 (75%) (stunted	<4000 (75%)		

Organism	Compound	Valency	Medium	pH	OC (%)	CEC*	WHC** (%)	Clay (%)	Sand (%)	Silt (%)	Growing period (d)	Endpoint	NOEC (mg Sb/kg; % effect)	LOEC (mg Sb/kg; % effect)	E(L)C50 (mg Sb/kg; % effect)	Reference	Reliable & Relevant
Hordeum vulgare												Weight (roots; dry weight) Length (shoots) Weight (shoots; dry weight) Weight (roots; dry weight)	controls) 400 (+64%) 40000 (5%) 40000 (+16%) 4 (+29%) (as control)	plants with yellow mottling of leaves) 4000 (80%) >40000 >40000 40 (45%) (as control)	<4000 (80%) 40000 (stunted roots, purple pigment to leaf edges)		
Lactusa sativa	Sb_2O_3 (freshly spiked) $SbCl_3$ (freshly spiked) Sb_2O_3 (5 year aged soil)	Sb(III)	Loam	7.0 5.94 -7.04 7.42 – 7.58	0.9 1.5-1.8	9.84 11.2 – 11.7	pF=2.0	14	10	76	24 d	Weight (shoots)	>1804 43 28 5.8	73 72		Oorts et al., 2005	NR

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Organism	Compound	Valency	Medium	pH	OC (%)	CEC*	WHC** (%)	Clay (%)	Sand (%)	Silt (%)	Growing period (d)	Endpoint	NOEC (mg Sb/kg; % effect)	LOEC (mg Sb/kg; % effect)	E(L)C50 (mg Sb/kg; % effect)	Reference	Reliable & Relevant
	Sb_2O_3 (5 year aged soil)										65 d	Weight (shoots) Weight (roots)	>116 >116				
Oryza sativa	$2KSbOC_4H_4O_6 \cdot H_2O$	Sb(III)	Culture medium								3	Growth (sprout) Growth (root)			<1000 mg/l (37%) >1000 mg/l (92%)	He and Yang, 1999	NR
	$KSb(OH)_6$	Sb(V)									3	Growth (sprout) Growth (root)			>1000 mg/l (64%) >1000 mg/l (100%)		
	$2KSbOC_4H_4O_6 \cdot H_2O$	Sb(III)	Loam	6.8	2.52	11,95					10	Visual appearance		300			
											98	Biomass (g/pot)	50 (10%)	150 (21%)			
												Growth (height)	600 (8%)	1000 (39%)			
												Yields of rice (g/pot)	50 (9%)	150 (15%)			
	$KSb(OH)_6$	Sb(V)									10	Visual appearance		300			
											98	Biomass (g/pot)	150 (10%)	300 (11%)			
												Growth (height)	>1000 (1%)	1000 (39%)			
												Yields of rice (g/pot)	150 (4%)	300 (12%)			
Pisum sativum	$2KSbOC_4H_4O_6 \cdot H_2O$											Length (shoots & roots	>4000			Pearce, 1978	NR

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Organism	Compound	Valency	Medium	pH	OC (%)	CEC*	WHC** (%)	Clay (%)	Sand (%)	Silt (%)	Growing period (d)	Endpoint	NOEC (mg Sb/kg; % effect)	LOEC (mg Sb/kg; % effect)	E(L)C50 (mg Sb/kg; % effect)	Reference	Reliable & Relevant
Polygonum tinctorium												Weight (shoots; fresh weight)	>4000				
												Weight (roots; fresh weight)	400 (0%) (leaf "burns", stringy roots with several pink nodules)	4000 (15%) (decayed roots especially, some nodules)			
												Length (shoots)	-	-	4000 (chlorotic, stunted, pigmented roots)		
												Weight (shoots; dry weight)	-	-	<4000 (91%)		
												Weight (roots; dry weight)	-	-	4000		
Rumex obtusifolius												Weight (shoots; fresh weight)	400 (+1%) (reduced lateral roots – thick, upturned ends. Red pigmentation on leaf margins)	4000 (92%) (very small plants, pigmented leaves, roots very poor)	<4000 (92%)		

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Organism	Compound	Valency	Medium	pH	OC (%)	CEC*	WHC** (%)	Clay (%)	Sand (%)	Silt (%)	Growing period (d)	Endpoint	NOEC (mg Sb/kg; % effect)	LOEC (mg Sb/kg; % effect)	E(L)C50 (mg Sb/kg; % effect)	Reference	Reliable & Relevant
Zea mays												Weight (roots; fresh weight)	400 (+26%)	4000 (99%)	<4000 (99%)		
												Length (shoots)	>40000				
												Weight (shoots; dry weight)	400 (+17%) (chlorotic shoots, "peggy roots)	4000 (20%) (spindly, chlorotic shoots, „peggy" roots)	40000 (senescent shoots, very stunted roots)		
												Weight (roots; dry weight)	4000 (+20%)	40000 (35%)			

*Effective CEC, i.e. CEC measured at pH of the soil.: Unit cmolc/kg

**Water Holding Capacity

***Valid and reliable according to the ISO guideline to which is was performed (from 1995) but the results on emergence are not compliant with the criteria in the most recent ISO guideline (from 2005). However, even if the emergence in the control group would have been 70% (as compared to the actual emergence of 60%) and the study therefore would have been valid according to the most recent guideline (from 2005), an effect on emergence would still not have been possible to detect. Based on this, the results are still considered usefull since they provide a very strong indication that growth is a more sensitive endpoint than emergence for *Lactusa sativa*.

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3.2.5.1.2 Invertebrates

There are seven studies available on antimony toxicity to invertebrates, but as stated in section 3.2.5.1 above, only the study by Moser (2007), which results in a bounded NOEC of 999 mg Sb/kg dw, will be used when deriving $PNEC_{soil}$.

Acute toxicity

There are no acute toxicity tests on invertebrates available.

Long-term toxicity

Study 1

Moser (2007) studied the toxicity of Sb to the springtail *Folsomia candida* (reproduction and mortality) in approx. 32 w aged Sb_2O_3 amended soil. The soil used in this study, including the different test concentrations used, is identical to the soil that was used when testing the toxicity to plants and microorganisms by (Smolders et al., 2007). This study was carried out according to ISO 11267 (1999). Ten (10-12 d old) synchronised springtails were put onto 30 g moist soil in a glass vessel during an exposure period of four weeks. Five replicates were used for each test concentration and the measured test concentrations used were 90, 322, 999, 2930, 10119 mg Sb/ kg soil dw. Additionally, the springtails were also tested in uncontaminated field soil control and an untreated artificial soil as control, which was based on OECD 207. The positive control used on springtail reproduction was the herbicide Betosip (a.i. 159 g/L phenmedipham). Granulated dry yeast served as food for the springtails and was added onto the soil surface in the beginning of the test and after 14 days. At the end of the exposure period, the number of juveniles in each test vessel was counted after floating. The mortality of the springtails was also recorded. At day 28, the pH-value and the moisture of the artificial coil for each concentration in additional vessels without springtails were determined. It was shown that the pH or the moisture did not diverge from the guideline recommendations.

For both mortality and reproduction, the highest effect observed was below 50% in any of the treatments. Consequently, LC_{50} and EC_{50} values could not be calculated but were estimated as being >10119 mg Sb/kg soil dw for each one of the endpoints. The $NOEC_{reproduction}$ was determined as 999 mg Sb/kg dw. $LOEC_{reproduction}$ was derived as 2930 mg Sb/kg dw. All values refer to measured concentrations.

The results from this study are considered reliable. However, since the soil used in this study is identical to the soil used by (Smolders et al., 2007) (see plants and micro organisms), the same concerns, as presented for the study by (Smolders et al., 2007), of having a NOEC from a soil that was not fully in equilibrium are also valid for the results from this study. As a consequence of that, the same correction of the NOEC of 999 mg Sb/kg dw needs to be performed when calculating $PNEC_{soil}$ (see section 3.2.5.1.4).

Study 2

Simini and co-workers (2002) exposed the earthworm *Eisenia fetida*, measuring adult survival and cocoon production according to ISO 11268-2 (International Organization for Standardization, 1998a), to various concentrations of $Sb_2(SO_4)_3$ in a natural sandy loam. The

original ISO method was designed for use with artificial soil (USEPA Standard Artificial Soil), however, research of the authors showed that the test could also successfully be conducted using natural soils (Kuperman et al., 2004). The method was modified for use with natural soils having physical and chemical characteristics that support a relatively high level of metal bioavailability. After performing a range-finding test, a definitive test was performed with an exposure period of three weeks using seven concentration groups, besides the control. The nominal added concentrations, besides the control, used for reproduction were 60, 86, 104, 124, 149, 179, and 215 mg Sb/kg, Four replicates were used per concentration, with five earthworms per replicate. Toxicity tests using the salt carrier control $CaSO_4 \cdot 2H_2O$ were performed in order to evaluate the effect of sulphate. The positive control was 4-nitrophenol. All soil treatment concentrations and controls were subjected to simulated aging/weathering procedures, which included alternating wetting/air-drying cycles for three weeks prior to commencement of definite tests. Hydration and moisture equilibration followed the weathering and aging process before exposing organisms in definitive studies.

Sulphate control treatments showed no statistically significant effect on reproduction as compared with negative controls. Juvenile or cocoon production in positive controls (4-nitrophenol) were within the baseline established for the laboratory cultures of *E. fetida*. The decrease in pH in the highest Sb treatment was below 1.0 pH unit, as compared with untreated soil (i.e. negative control). In the sulphate control, soil pH decreased by less than 1.0 pH unit in 7000 and 35000 mg SO_4^- treatment, as compared with negative controls.

Only about 58% of nominal Sb concentrations were recovered. The nominal concentrations were used when determining the ecotoxicological parameters of Sb with the results of definitive tests of Sb toxicity resulting in NOEC/LOEC values for adult survival of 617/697 mg Sb/kg and 60/86 for juvenile production. The authors recommended that the nominal values should be adjusted to 58% of the nominal values, as that was the average recovery. This may not necessarily be an exposure problem, but could instead be an analytical problem. However, since it presently not is known which of the two alternatives (or both?) that is true, it is decided to use the conversion factor of 0.58 which for *Eisenia fetida* results in the NOECs 35 mg Sb/kg and 358 mg Sb/kg for juvenile production and adult survival, respectively. Since the Sb soil concentration never was measured during the course of the experiment it is not possible to know if a steady-state was reached after the three weeks of aging/weathering, but it is not considered likely. As a consequence, the toxicity pressure has probable not been constant during the exposure period.

The results from this study can only be used as supportive evidence. According to the ISO protocol on which the study was based, the earthworm reproduction test is considered valid only if the coefficient of variation for the mean number of cocoons from the control species is ≤30% at the end of the test. The coefficient of variation for the reproductive performance of the control in the study by (Simini et al., 2002) could be calculated from the raw data in annex and was 45%.

To conclude, the results from this study are considered unreliable.

Study 3

(Oorts et al.; 2005) studied the toxicity of Sb to the springtail *Folsomia candida* (reproduction and mortality) in freshly spiked soil (Sb_2O_3 or $SbCl_3$), and in five year aged Sb_2O_3 amended soil. The study was carried out according to ISO 11267 (1999). Ten (10-12 d old) synchronised springtails were exposed to 30 g moist soil in a glass vessel during an exposure

period of four weeks. Granulated dry yeast was added onto the soil surface every week and served as food for the springtails. At the end of the exposure period, the number of juveniles in each test vessel was counted after floating. The mortality of the springtails was also recorded.

A large variation in the toxicity data in several of the exposed groups is noticeable and reduces the possibility of detect deviations from the control. The authors, who used ANOVA and the *post hoc* test Duncan, reported an unbound NOEC of 2000 mg Sb/kg dw nominal concentration (measured concentration of 1864 mg Sb/kg dw), for number of juvenile and adult mortality. The use of the step-down approach with the Jonckheere-Terpstra test results in a NOEC of 1000 mg Sb/kg dw nominal concentration (= measured concentration of 897 mg Sb/kg dw) and a LOEC of 2000 mg Sb/kg dw nominal concentration (measured concentration of 1864 mg Sb/kg dw).

An unbounded NOEC of 1804 mg Sb/kg dw, (measured concentration) results from a freshly Sb_2O_3 spiked soil. However, using statistical methods considered more appropriate for dose-response studies (step-down approach), as compared to the method used by (Oorts et al., 2005), which was ANOVA and Duncan´s method, results in a bound NOEC of 897 mg Sb/kg dw. However, due to the increasing porewater concentration, which indicates that the toxicity could still increase during the exposure period, a derived NOEC (not being unbound) using freshly spiked Sb_2O_3 soils may be too high. Having the same soil solution concentration in an exposure regime with constant toxic pressure (i.e. aged soil) may have resulted in lower NOEC-values.

The data from freshly $SbCl_3$ spiked soils results in a NOEC < 20 mg Sb/kg dw (measured concentration of 10 mg Sb/kg dw). This "smaller-than-value" was considered unreliable by (Oorts et al., 2005) since they considered that the weight of evidence showed no toxic effect in freshly added Sb_2O_3 to doses up to 500 mg Sb/kg dw nominal concentration (measured concentration of 422 mg Sb/kg dw). The difference in response between the control and the first five $SbCl_3$ doses, up to 500 mg Sb/kg dw, nominal concentration was considered to be within the non-significant variation in response observed for the Sb_2O_3 amended soils.

It is however considered difficult to draw any firm conclusions on absence of toxicity for the freshly spiked Sb_2O_3 soils up to 500 mg Sb/kg dw nominal concentration, given the variation in response, within and between the different dose groups, especially for the number of juveniles. The NOEC < 20 mg Sb/kg nominal dose resulting from the freshly spiked $SbCl_3$, which was considered unreliable by (Oorts et al., 2005), has a measured Sb soil solution concentration of 0.62 mg/l. This is almost identical to the Sb soil solution concentration of 0.63 mg Sb/l, which was measured in the freshly spiked Sb_2O_3 nominal concentration 200 mg Sb/kg dw. group, a dose group with three out of four values being lower than the lowest of the control values. In addition, the toxic pressure between two concentrations likely differed (see the discussion in plant study by (Oorts et al., 2005) above). The Sb soil solution concentration measured in the freshly spiked Sb_2O_3 500 mg nominal dose group, was 9 mg Sb/l, in which again three out of four values being lower than the lowest of the control values.

The effect observed in the aged Sb_2O_3 spiked soil study at the two highest doses may be explained by the presence of a herbicide, which was a mixture including substances having shown to have toxicity towards earthworms. The Sb soil solution concentration in the highest of the remaining doses, i.e. 24 mg Sb/kg dw., is 0.745 mg Sb/l. The four data points at that concentration indicate a response, but the difference is not significant.

However, as explained above, values from this study will not be used in the RAR.

Study 4

In the fourth study (LISEC, 2002c), the microdrile oligochaete *Enchytraeus crypticus* was exposed to a range of nominal concentrations (1, 10, 100, and 1000 mg Sb_2O_3/kg dw), in addition to the control, for 14 days using artificial soil (see Table3-105 below). The endpoints studied were mortality and reproduction. This study was performed with one replicate per concentration. No effect on survival or reproduction was observed. The results from this study are not considered reliable since soil with freshly spiked Sb_2O_3 was used. In addition, only nominal total concentrations are reported.

Study 5

In the fifth study (Heijerick and Vangheluwe, 2003a), the microdrile oligochaete *Enchytraeus albidus* was exposed to various concentrations of trivalent antimony ($SbCl_3$) in an artificial soil and the endpoints mortality and reproduction were studied. The standard soil used is described and recommended by OECD (1984), (for more information on the study, see Table3-105 below). This study was performed with 5 replicates per concentration, five concentrations (measured conc. range: 59.1 – 6980 mg Sb/kg dw) and a control (measured conc. 0.37 mg Sb/kg dw), with each replicate consisting of 10 enchytraeids with fully developed clitellum. Adult enchytraeids were exposed for 21 days (which is the period in which cocoons are produced). After this period, the adult enchytraeids were removed from the soil and the survival was examined. The cocoons were incubated for another 21 days (i.e. a total of 42 days) after which reproduction was measured (number of young enchytraeids). The resulting NOEC and LOEC were 760 mg Sb/mg kg dw. and 2012 mg Sb/mg kg dw., respectively, for both endpoints. One of the three performance criteria specified in the OECD test guideline 220 (Enchytraeid reproduction test) that should be met in the control in order for the test to be valid is not fulfilled. This since the coefficient of variation around the mean number of juveniles that should not be higher than 50% at the end of the reproduction phase was 81%. The results from this study are therefore considered not reliable.

Study 6

In the sixth study, Kuperman and co-workers (2002) exposed the enchytraeid *Enchytraeus crypticus*, measuring adult survival and juvenile production according to ISO 16387 (International Organization for Standardization, 2003) to various concentrations of $Sb_2(SO_4)_3$ in a natural sandy loam. The original ISO methods were designed for use with artificial soil (USEPA Standard Artificial Soil). These methods were modified for use with natural soils having physical and chemical characteristics that support a relatively high level of metal bioavailability. The modifications included different soil hydration levels that have lower water holding capacity compared with the artificial soil and shorter duration test for *E. crypticus* (28 vs. 42 d) because of shorter generation time of this species compared with *Encytraeus albidus*, for which the ISO 16387 test conditions were optimized.

In the enchytraeid *Enchytraeus crypticus* reproduction test ten enchytraeid adults with eggs in the clitellum were placed on top of prepared soil in each container and exposed to $Sb_2(SO_4)_3$

for four weeks. The nominal concentrations of Sb used were 0, 100, 140, 196, 274, 384, 538, 753, and 1054 mg Sb/kg. Four replicates were used per concentration. After two weeks, soil in each test container was carefully searched and adult worms were removed and counted. The remaining test substrate, including any cocoons laid during the first two weeks of the test, was incubated for additional two weeks. After four weeks from the start of the test, soil in the test containers was fixed with ethanol and Rosebengal biological stain was added. Staining continued for a minimum of 24 hours. The content of each test container was wet-sieved and transferred to a counting tray and worms were counted. Measurement endpoints included number of surviving adults after 14 days and number of juveniles produced after 28 days.

The authors were not able to adequately measure the Sb concentrations in the soils. Despite change of methods and improved efficiency of Sb extraction in average only 58% of the added antimony could be measured. For this reason, nominal concentrations were used to determine the ecotoxicological parameters for Sb. Using methods developed by the authors, the amended soils were weathered and aged for three weeks before definitive testing. Hydration and moisture equilibration followed the weathering and aging process before exposing organisms in definitive studies. The decrease in pH in the highest Sb treatment was 1.2 pH unit, as compared with untreated soil (i.e. negative control). In the sulphate control, soil pH decreased by less than 1.0 pH unit in either 7000 or 35000 mg SO_4^- treatment, as compared with negative control. Sulfate control treatments showed no statistically significant ($p>0.05$) effect on adult survival or reproduction measurement endpoints as compared to the negative controls. Juvenile or cocoon production in positive controls (4-nitrophenol for the *enchytraeid* test and carbamate in the collembolan test) were within the baseline established for the laboratory cultures of *E. crypticus*. Results of definitive tests of Sb toxicity resulted in NOEC/LOEC values for adult survival of 384/538 mg Sb/kg. For juvenile production the resulting NOEC/LOEC were 100/140 mg Sb/kg. The authors recommended that the nominal values should be adjusted to 58% of the nominal values, as that was the average recovery. This may not necessarily be an exposure problem, but could instead be an analytical problem. However, since it is presently not known which of the two alternatives (or both?) that is true, it is decided to use the conversion factor of 0.58 which for *Enchytraeus crypticus* results in NOECs of 58 mg Sb/kg and 223 mg Sb/kg for juvenile production and adult survival, respectively. Since the Sb soil concentration never was measured during the course of the experiment it is not possible to know if a steady-state was reached after the three weeks of aging/weathering, but it is not considered likely. As a consequence, the toxicity pressure has probable not been constant during the exposure period.

The results indicate that Sb excert toxicity on *Enchytraeus crypticus* with the resulting NOEC for the reproduction of *Enchytraeus crypticus* is 58 mg Sb/kg dw. There are however some concerns against using this value since i) the result was obtained using $Sb_2(SO_4)_3$ which per definition means that counter ions (i.e. sulphate) and protons also was added to soil which may have influenced the results, ii) it is not known how the toxic pressure may have varied over the exposure period, and iii) the authors had difficulties measuring the Sb concentration in soil.

It is, as already stated in section 3.2.5.1, decided not to use this NOEC when deriving $PNEC_{soil}$.

Study 7

In the seventh study, Phillips and co-workers (2002) exposed the collembolan *Folsomia candida,* measuring adult survival and juvenile production according to ISO 11267 (International Organization for Standardization, 1998b), to various concentrations of $Sb_2(SO_4)_3$ in a natural sandy loam.

In the *Folsomia candida* reproduction test 10-12 day-old juveniles were exposed to $Sb_2(SO_4)_3$ for four weeks. The nominal concentrations of Sb used were 0, 100, 140, 196, 274, 384, 538, 753, and 1054 mg Sb/kg. Four replicates were used per concentration. After the exposure period, purified water was added to each test chamber to bring the level up to half its volume. After gentle mixing with a spatula, the chamber was examined under a dissecting microscope for the presence of juveniles and adults. The juveniles that floated to the surface were counted and removed. This procedure was repeated until no other springtails floated to the surface. The chamber was given a final mixing and examined once more to ensure all individuals were counted. Adult survival and juvenile production were used as endpoints.

The authors were not able to adequately measure the Sb concentrations in the soils. Despite change of methods and improved efficiency of Sb extraction in average only 58% of the added antimony could be measured. For this reason, nominal concentrations were used to determine the ecotoxicological parameters for Sb. Using methods developed by the authors, the amended soils were weathered and aged for three weeks before definitive testing. Hydration and moisture equilibration followed the weathering and aging process before exposing organisms in definitive studies. The decrease in pH in the highest Sb treatment (nominal concentration = 1054 mg Sb/kg dw.) was 1.2 pH unit, as compared with untreated soil (i.e. negative control). In the sulphate control, soil pH decreased by less than 1.0 pH unit in either 7000 or 35000 mg SO_4^- treatment, as compared with negative control. Sulfate control treatments showed no statistically significant ($p>0.05$) effect on adult survival or reproduction measurement endpoints as compared to the negative controls. Juvenile or cocoon production in positive controls (4-nitrophenol for the *enchytraeid* test and carbamate in the collembolan test) were within the baseline established for the laboratory cultures of *F. candida*. Results of definitive tests of Sb toxicity resulted in NOEC/LOEC values of 100/126 mg Sb/kg for adult survival. Juvenile production was significantly ($p=0.045$) decreased in the lowest positive Sb treatment compared with the negative control, producing an unbound LOEC of 100 mg Sb/kg (based on Fisher´s least-significant-difference pairwise comparison test). Bounded NOEC and LOEC values of 100 and 126 mg/kg were determined using the more conservative Bonferroni mean comparison test. The authors recommended that the nominal values should be adjusted to 58% of the nominal values, as that was the average recovery. This may not necessarily be an exposure problem, but could instead be an analytical problem. However, since it is presently not known which of the two alternatives (or both?) that is true, it is decided to use the conversion factor of 0.58 which for *Folsomia candida* results in the NOECs 58 mg Sb/kg for both juvenile production and adult survival. The results indicate that Sb excert toxicity on *Folsomia candida* and that a NOEC for reproduction may be around 58 mg Sb/kg, the authors suggested that it could be even lower. The *post hoc* test used by the authors, Fisher's least-significant-difference pairwise comparison test (Fisher's LSD) does not correct for multiple comparisons which means that it is easier to find statistical significance with the Fisher's LSD test (it has more power) than with other multiple comparisons tests that correct for multiple comparisons, but that also means it is too easy to be mislead by false positives (you'll get bogus 'significant' results in more than 5% of the experiments). In order to correct for this the authors used the Bonferroni correction on the p-values resulting from *Folsomia candida* reproduction test, which resulted in a NOEC of 58 mg Sb/kg. The Bonferroni correction is generally overly conservative, especially for large *k*. However, also the use up the step-down

approach results in a NOEC of 58 mg Sb/kg. There are however some concerns against using this value since i) the result was obtained using $Sb_2(SO_4)_3$ which per definition means that counter ions (i.e. sulphate) and protons also was added to soil which may have influenced the results, ii) it is not known how the toxic pressure may have varied over the exposure period, and iii) the authors had difficulties measuring the Sb concentration in soil.

It is, as already stated in section 3.2.5.1, decided not to use this NOEC when deriving $PNEC_{soil}$.

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Table3-105 Toxicity tests performed on terrestrial invertebrates

Organism	Compound	Valency	Medium	pH	OC (%)	CEC*	WHC** (%)	Clay (%)	Sand (%)	Silt (%)	Equil. time (d)	Duration (d)	Endpoint	NOEC (mg Sb/kg)	LOEC (mg Sb/kg; % effect)	E(L)C50 (mg Sb/g; % effect)	Reference	Reliable & Relevant
Folsomia candida	Sb_2O_3 (31 w aged soil)	Sb(III)	Loam	6.0 – 6.6	0.9	9.8	53	14	10	76		28	Mortality Reproduction	999	2930	>10119 (LC_{50}) >10119 (EC_{50})	Moser, 2007	R
Eisenia fetida	$Sb_2(SO_4)_3$ (3-w aged soil)	Sb(III)	Sandy loam	4.39-5.29	1.2	4.27	100	11	71	18		21	Adult mortality Juvenile production	358[a] 35[a]	404[a] 50[a]		Simini et al., 2002	NR
Encytraeus albidus	$SbCl_3$	Sb(III)	Artificial soil	6.0	10		50	20	70		7	21 42	Mortality Mortality Reproduction	760 760 760	2012 (100%) 2012 (100%) 2012 (100%)	< 2012 (LC_{50}) < 2012 (LC_{50}) < 2012 (EC_{50})	Heijerick and Vanghel uwe, 2003a	NR
Encytraeus crypticus	Sb_2O_3	Sb(III)	Artificial soil	6.71-7.7	10		40-60	20	69		7	14	Mortality Reproduction	>1000 >1000		>1000 >1000	LISEC, 2002c	NR
Encytraeus crypticus	$Sb_2(SO_4)_3$	Sb(III)	Sandy loam	4.08-5.29	1.2	4.27	100	11	71	18	21	28	Mortality Reproduction	223[a] 58[a]	312[a] 81[a]		Kuperman et al., 2002	NR
Folsomia candida	$Sb2O_3$ (freshly spiked)	Sb(III)	Loam	7.0	0.9	9.8	pF=2.0	14	10	76	7	28	Mortality Reproduction	897[b] (>1804[b]) 897[b]	1804[b] 1804[b] 1741[b]		Oorts et al., 2005	NR

	$SbCl_3$ (freshly spiked) Sb_2O_3 (5 year aged soil)			5.94-7.04 7.42-7.58	 1.5 – 1.8	 11.2 – 11.7							Mortality Reproduction Mortality Reproduction	(>1804 [b]) (836 [b] (<10 [b] (28 [b] (28 [b]	 (72 [b] (72 [b]			
Folsomia candida	$Sb_2(SO_4)_3$	Sb(III)	Sandy loam	4.57-5.29	1.2	4.27	100	11	71	18	21	28	Mortality Reproduction	58 [a] 58 [a]	73 [a] 73 [a]		Phillips et al., 2002	NR

* Effective CEC, i.e. CEC measured at pH of the soil. Unit cmol c/kg
** Water Holding Capacity

[a]58% of nominal values

[b]Measured concentration. Values within parenthesis represent the values derived by Oorts et al. (2005)

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3.2.5.1.3 Microorganisms

There are four studies available on antimony toxicity to the microflora, but as stated in section 3.2.5.1 above, only the study by (Smolders et al., 2007) , which results in a bounded NOEC of 2930 mg Sb/kg dw, will be used when deriving $PNEC_{soil}$.

Study 1

(Smolders et al., 2007) studied the toxicity of Sb to native microorganisms in 31 weeks aged Sb_2O_3 amended soil (measured conc. = 3, 90, 322, 999, 2930 and 10119 mg Sb/kg dw.) using a nitrification assay based on ISO 14238-1 (1997). The test measures the Substrate Induced Nitrification rate (SIN), which the nitrification rate induced after substrate (NH_4^+) supply measured during 28 days incubation. Soils were preincubated for 3 days at a moisture content of 23% taken into account the volume of $(NH_4)_2SO_4$ (80 mg N/ml) that was added later. Afterwards, $(NH_4)_2SO_4$ was added to 50 g soil to a final concentration of 100 mg NH_4-N/kg fresh soil by thoroughly mixing. Soils were incubated in an incubation room at 20 °C. After 0, 7 (additional sampling compared to the ISO-protocol) and 28 days, the NO_3^- and NH_4^+ concentrations were measured colorimetrically in a shaken and centrifuged soil extract (5 g soil in 12.5 ml KCl 1 M, 2 h end-over-end shaking, 15 min centrifugation at 3500g) and the SIN was calculated as the daily production of NO_3^-/kg soil, based on the measurements between day 0 and day 28. The nitrate production rate was also calculated in the initial period (0-7 days) during which there was unlimited substrate present; this endpoint is termed the Potential Nitrification Rate (PNR, or also nitrification potential in the literature). This means that two endpoints were defined: the SIN and the PNR. The ISO guideline only recommends the latter, however the PNR endpoint has been reported as being more sensitive than the SIN endpoint since the former is measured in the initial period after NH_4^+ addition, i.e. as long as the substrate is still abundantly present (Smolders et al., 2001). The test was performed in duplicate per soil. No validity criteria are given in the guideline ISO 14238-1 (1997), however the nitrification rate in the control soil was considered to be within expected limits and the results are therefore considered reliable.

The SIN-endpoint resulted in an unbounded NOEC of >10119 mg Sb/kg dw, while the PNR-endpoint resulted in a bound NOEC of 2930 mg Sb/kg dw and a LOEC of 10119 mg Sb/kg dw. The pore water concentrations corresponding to the PNR, NOEC and LOEC were 18.7 and 54.4 mg Sb/l, respectively. It was not possible to derive EC_{10} or EC_{50} for the PNR-endpoint.

As already mentioned, when presenting the results for plants by the same authors (i.e. Smolders et al., 2007), the calculation of $PNEC_{soil}$ needs to correct for the fact that the results were obtained from not fully equilibrated soils (see section 3.2.5.1.4).

Study 2

(Oorts et al., 2005) studied the toxicity of antimony to native microorganisms in freshly spiked soil (Sb_2O_3 or $SbCl_3$; control (measured conc., 0.6 mg Sb/kg dw), nominally added concentrations 20, 50, 100, 200, 500, 1000 and 2000 mg Sb/kg dw.), and in five year aged Sb_2O_3 amended soil (measured conc. = 0.4, 5.8, 28.4, 71.7 and 116.3 mg Sb/kg dw). Seven days after metal spiking, triplicate 100 g subsamples of each Sb concentration of each soil were amended with 100 mg NH_4-N/kg fresh soil. The increase in soil NO_3-N in a period of 28

days after substrate addition was measured by analysing the soil nitrate colorimetrically in a centrifuged soil extract (1 M KCl, 10 g subsample, L/S = 2.5, 2 h end-cover-end shaking, n=3). Two different endpoints were measured; the substrate induced nitrification (SIN) and the potential nitrification rate (PNR). SIN was calculated as the percentage of added NH_4-substrate transformed into NO_3-N during 28 days (OECD 216). PNR (mg NO_3-N/kg fresh soil/day) was calculated as the slope of the regression of soil nitrate concentration against time. The PNR endpoint has been reported as being more sensitive than the SIN endpoint since the former is measured in the initial period after NH_4^+ addition, i.e. as long as the substrate is still abundantly present (Smolders et al., 2001). For the $SbCl_3$ amended samples, PNR was calculated for a seven day incubation period; while it was calculated for a ten day period for all the other soils. The incubation time selected was identical for each sample within a dose-response study (i.e. for each Sb dose).

There was no toxicity observed in either of the Sb_2O_3 spiked soils, i.e. the freshly spiked soil and the 5 year aged soils, for any of the two endpoints Potential Nitrification Rate (PNR) and Substrate Induced Nitrification (SIN). However, the reason for the lack of response in the aged Sb_2O_3 soil might be that the microflora has adapted to the different Sb-concentrations during the five years of aging, and that no response therefore is to be expected.

This is in contrast to the freshly spiked $SbCl_3$ soils where $NOEC_{PNR}$ is 73 mg Sb/kg dw. (nominally added concentration, 100 mg Sb/kg dw.) and $NOEC_{SIN}$ is 384 mg Sb/kg dw. (nominally added concentration, 500 mg Sb/kg dw.). The potential effect of the counter ion, i.e. Cl^-, was tested separately. Increased concentrations of chloride decreased the PNR but not the SIN (even a slight stimulation was observed with increasing concentrations of chloride). The measured PNR and SIN in the separate chloride toxicity experiment was about 70-80 % of the PNR and SIN in the control of the freshly spiked $SbCl_3$ soils. In addition, the dose-response pattern with decreasing PNR with increasing concentrations of chloride (measured as electrical conductivity) differed. It is therefore not clear cut to determine how much of the decrease in PNR that resulted from increased concentrations of chloride. According to the authors (Oorts et al., 2005), about 50% of the decrease was explained by increased concentrations of chloride. The rest of the decrease in PNR and all of the decrease of SIN was by (Oorts et al., 2005) proposed to be due to reduced pH, since the nitrification process has been reported to be sensitive to changes in pH.

The response observed for SIN is easier to interpret since there was no observed effect of the counter ion. The observed response may therefore be assumed to either result from increased concentrations of Sb and/or decreased pH. The NOEC for SIN in the freshly spiked $SbCl_3$ soil the measured Sb soil solution concentration measured was 13.1 mg Sb/l. This can be compared with the Sb soil solution concentration measured in the highest freshly Sb_2O_3 spiked soil, which was 13.9 mg Sb/l. Despite probable differences in toxic pressure in these two exposure regimes, these two results both appears to indicate that the toxicity to microorganisms, measured via SIN, is low. The reason for large response in the freshly spiked $SbCl_3$ soils, despite the relatively small increase in the measured Sb soil solution concentration, from 13.1 mg Sb/l (pH = 6.68) with 96% substrate used, via 14.4 mg Sb/l (pH = 6.58) with 73% substrate used, to 15.85 mg Sb/l (pH = 5.94) with 15% substrate used is not clear. Especially the decrease from 96% substrate used at pH = 6.58 to 73% substrate used at pH = 6.58 appears large if the reduction of pH is a major factor. This since the decrease in pH from the control with 100% of the substrate used at pH = 7.04 to pH = 6.68 only reduced the substrate used to 96%.

There was no response in SIN in the aged Sb_2O_3 spiked soils up to soil solution concentrations of 4.2 mg Sb/l, which may depend on adapted microflora, but is on the other hand in agreement with the NOECs in the freshly spiked soils, since no toxicity was observed at that Sb concentration level.

Study 3

In this study by LISEC (2002), the effect of Sb_2O_3 on the nitrogen transformation activity of soil organisms was studied. The soil used was a sandy loam (for more information, see Table1-106 below). The test substance was applied using quartz sand as carrier. The test was performed with three replicates, (with an additional fourth replicate for Sb measurement), and five concentrations (range: 2.6-823 mg Sb/mg dw.) and a control (concentration below detection limit 1.8 mg Sb/kg dw.). The test was terminated after 28 days. On day 28, the quantities of nitrate were determined. There were no significant difference in the nitrogen transformation activity between the highest concentration used (823 mg Sb/kg dw.) and the control, resulting in a LOEC > 823 mg Sb/kg dw. The value of this unbound NOEC is unclear due to the uncertainty associated with a lack of response using freshly spiked Sb_2O_3 soils. It is not possible to know how the Sb soil solution concentration has changed (most probable increased) during the exposure period. It can therefore not be excluded that the absence of observed effect, resulting in an unbound NOEC, may be the consequence of to low Sb soil solution concentration to long during the exposure period. To conclude, the results from this study are not considered valid.

Study 4

Pearce (1978) studied the growth in agar of the strain *Rhizobium* 1045 exposed to either diantimony trioxide or tartar emetic. No growth was observed at 40 and 400 mg Sb/kg for both antimony compounds, which indicate that antimony may exert microbial toxicity. However, the results will not be used in the development of a $PNEC_{soil}$ since they are only considered to be of indicative value to a risk assessment process.

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Table3-106 presenting toxicity tests for microorganisms

Organism	Compound	Valency	Medium	pH	OC (%)	CEC*	WHC** (%)	Clay (%)	Sand (%)	Silt (%)	Duration (d)	Endpoint	NOEC (mg Sb/kg)	LOEC (mg Sb/kg; % effect)	E(L)C50 (mg Sb/g; % effect)	Reference	Reliable & Relevant
Native soil microflora	Sb2O3 (31 w aged soil)	Sb(III)	Loam	6.5 - 6.7	0.9	9.8	53	14	10	76	28 7	Substrate Induced Nitrification (SIN) Potential Nitrification Rate (PNR)	> 10119 2930	10119		Smolders et al., 2007	R
Native soil microflora	Sb2O3 (freshly spiked) SbCl3 (freshly spiked) Sb2O3 (5 year aged soil)	Sb(III) Sb(III) Sb(III)	Loam	7.0 5.94-7.04 7.42 – 7.58	0.9 1.5 – 1.8	9.8 11.2 – 11.7	pF=2.0	14	10	76	28 28 28 28	Substrate Induced Nitrification (SIN) Potential Nitrification Rate (PNR) SIN PNR SIN PNR	>1804 >1804 384 73.1 >116 >116	 836 158.6		Oorts et al.e	NR R
Native soil microflora	Sb_2O_3	Sb(III)	Sandy loam	6.3	1.2		36		55.4		28	Nitrogen transformation activity	>823	>823	>823 (0%)	LISEC, 2002a	NR
Rhizobium 1045	Sb2O3 2KSbOC4H4O6·H2O	Sb(III)	Agar									Growth			<40 (100%) <40 (100%)	Pearce, 1978	NR

*Effective CEC, i.e. CEC measured at pH of the soil. Unit cmolc/kg.

**Water Holding Capacity

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3.2.5.1.4 Calculation of Predicted No Effect Concentration (PNEC) for the terrestrial compartment

Valid bounded chronic NOECs are available for plants (999 mg Sb/kg dw), invertebrates (999 mg Sb/kg dw), and microorganisms (2390 mg Sb/kg dw).

The NOEC of 999 mg Sb/kg dw for plants results from the barley root elongation assay. There are no indications from the other available studies on antimony-phytotoxicity that other endpoints are more sensitive compared to root growth.

There are, based on results from the study by (Smolders et al., 2007) on lettuce, strong indications that emergence not is a sensitive endpoint (this since (i) even if the emergence in the control group would have been 70%, no effect would have been possible to detect, and (ii) there is no dose-response relationship) and that growth not is a sensitive endpoint, (this since observable effects on growth requires high concentrations of antimony, at least for lettuce). It is from studies by Pearce (1978) and He and Yang (1999) brought forward that the Sb-toxicity on plants is most pronounced at the roots, which is what was tested on barley by (Smolders et al., 2007). It is therefore not considered probable that tests on barley on emergence and/or growth would have resulted in lower NOEC. Indirect evidence suggests that the root elongation assay of barley is more sensitive than longer term shoot growth assay for the same plant. Results from Rooney and colleagues (2006) from 18 soils showed a mean ratio of 2.1 (median 1.8) of ED10 of tomato shoot growth to ED10 of barley root elongation for Cu and 3.7 (median 2.7) for Co. (A ratio >1 means thas that the long term assay is less sensitive as compared to the short term root elongation assay.) Furthermore, toxicity tests with Co showed that barley is even less sensitive than tomato when both were tested according to the same shoot growth test protocol (McGrath et al., personal communication in International Antimony Oxide Industry Association, 2007). It should be noted that it is not known if the relative toxicity of Sb amongst different species is the same as for the metals mentioned. However, taken together this suggests that the root elongation assay is a more sensitive assay as compared to a long term growth assay for barley.

According to the TGD, the resulting $PNEC_{soil}$ will be the lowest of these three NOECs divided by an assessment factor of 10. Since the lowest NOEC is 999 mg Sb/kg dw the resulting $PNEC_{soil}$ thus becomes 99.9 mg Sb/kg dw (999 mg Sb/kg dw/10).

These NOECs result from studies using soil spiked with Sb_2O_3 and aged for 31 weeks before testing. During the ageing period, a substantial part, of the Sb_2O_3 was transformed into soluble Sb. It is expected that further transformation of Sb_2O_3 after this ageing period would be very slow and as a result a sufficiently constant toxic pressure is considered to have been obtained during these bioassays. Using Sb_2O_3 amended soils avoids confounding effects of counter-ions or lowered pH, and therefore observed toxic effects can be attributed to the increasing Sb dose only.

However, since not all Sb_2O_3 had dissolved during the aging period used, a standard TGD procedure dividing the lowest NOEC (999 mg Sb/kg dw; *Hordum vulgare*-root elongation and *Folsomia candida*-reproduction) with an assessment factor of 10, resulting in a $PNEC_{soil}$ of 99 mg Sb/kg dw, would underestimate the toxicity. This since the equilibrium pore water concentration was not reached at this test concentration during the study.

Instead $PNEC_{soil}$ is based on the porewater concentration measured at NOEC (9.7 mg Sb/l), divided with an assessment factor of 10 (cfr. Standard TGD-procedures) which is multiplied with the equilibrium solid:liquid distribution coefficient (K_d) for Sb in this soil. The K_d value for the soil used in the present studies by (Smolders et al., 2007 and Moser 2007) is 38 l/kg, which is the value

observed for the Sb_2O_3 amended soil aged for five years and for the soluble $SbCl_3$ added to soil (Oorts et al., 2005).

The resulting $PNEC_{soil}$ after having performed this calculation is 37 mg Sb/kg dw (= 9.7 mg Sb/l/ 10) x 38 l/kg).

(Vijver et al., 2001) found that dermal uptake of metals is the most important uptake route for metals in earthworms and that soluble metal pools are the best descriptors of metal accumulation in earthworms. These results indicate that the proposed correction is valid also for soil invertebrates.

To conclude:

$PNEC_{soil}$ = 37 mg Sb/kg dw.

3.2.6 Atmosphere

No data is available on atmospheric toxicity of antimony.

3.2.7 Secondary poisoning

3.2.7.1 Mammalian toxicity

The mammalian toxicity data for diantimony trioxide are described in detail section 4. A summary of the available relevant data is given below.

- Repeated dose toxicity.

Two repeated dose oral studies indicate that diantimony trioxide may be toxic to the liver. This is based on an increase in liver weight in one study, supported by significantly elevated ASAT and ALP levels and slight disorder and cloudy swelling in hepatic cord observed in another study. However, no histological changes in the liver were observed to support an adverse effect on liver, and the absence of any other evidence of antimony intoxication suggests that these findings are incidental to treatment. A NOAEL corresponding to 1686 mg/kg/d (males) and 1879 mg/kg/d (females), corresponding to the highest doses tested, can be derived from this study.

- Fertility.

  For fertility there are two studies in animals, one on female rats and one on male rats and mice. The inhalation study on female rats indicated that antimony might have an adverse effect on fertility after repeated exposure to 250 mg/m^3. The gavage study on male rats and mice showed no testicular toxicity after repeated doses up to 1200 mg/ kg bw.

- Development

For developmental toxicity, there are two animal studies available. The first study indicates that diantimony trioxide may cause embryo mortality and other embryo toxicity (hemorrhage into the fetal cerebral membrane and liver and enlargement of the kidney cavity and the cerebral ventricles)

after inhalation exposure of female rats, 24h/ day, throughout gestation, at 0.082 mg diantimony trioxide /m^3 or higher. The second study, which was also an inhalation study, 6 hr/day, throughout gestation, showed no developmental toxicity at 2.6, 4.4 or 6.3 mg diantimony trioxide /m^3. The different results in these studies are not easily explained, however, it should be noted that in the first study there was a continuous exposure of diantimony trioxide throughout gestation while in the second, animals were exposed for 6h/ day. It might be possible that the recovery period between the exposures in the second study was beneficial for the animal's well being.

3.2.7.2 Calculation of $PNEC_{oral}$

The reproduction and developmental toxicity studies available for diantimonytrioxide are, all except one, inhalation exposure studies. Even though the inhalation exposure studies reveal effects they are not considered relevant to use in the assessment of secondary poisoning. In the study performed with oral exposure of male rats and mice no testicular toxicity was seen after repeated doses up to 1200 mg/ kg bw.

It can be discussed if the effects on liver seen in the two repeated dose oral studies are relevant on a population level. However, it is decided to use these data, i.e. a NOAEL of 1686 mg/kg bw/day for female rats from a 90 d repeated dose study in the derivation of a $PNEC_{oral}$ for secondary poisoning.

Using the conversion factor 20 (rats >6 weeks) for the NOAEL to NOEC conversion and an assessment factor of 90 as suggested in the TGD results in a PNECsec posisoning of 374.8 mg Sb/kg food. This value is also used for the assessment of secondary poisoning in the marine environment.

3.3 RISK CHARACTERISATION [10]

The risk assessment has considered all the main stages during which antimony may be released into the environment due to production, use and waste disposal of diantimony trioxide. The regional background concentrations used in the modelling are based on measured RWC ambient levels of antimony in Europe (see 3.1.4).

3.3.1 Aquatic compartment (incl. sediment)

3.3.1.1 Water

$PNEC_{aquatic}$ = 0.113 mg Sb/l, was derived by dividing the lowest NOEC (1.13 mg Sb/l) from a study on the fish *Pimephales promelas* with an assessment factor of 10, since NOECs from three trophic levels are available.

[10] Conclusion (i) There is a need for further information and/or testing.
Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.
Conclusion (iii) There is a need for limiting the risks; risk reduction measures which are already being applied shall be taken into account.

The risk characterisation ratios for surface water are shown in Table3-107 below. Predicted PEC:s have only been included for the realistic worst cases, not for all sites giving emission information. The reason is that all realistic worst case predicted PEC:s are below the PNEC. Reported emission data for all sites are presented in the release section.

Table3-107. PEC/PNEC ratios for the aquatic compartment.

	PEC_{water} (µg Sb/l)	RCR (PEC/PNEC)
Production, site P1	39.5	0.35
Production, site P5	0.88	0.008
Formulation as flame-retardant in plastics and rubber	4.7	0.06
Formulation as flame-retardant in plastics and rubber, assuming releases to air	0.72	0.04
Processing as flame-retardant in plastics and rubber	2.12	0.02
Formulation as flame-retardant in textiles, generic site	28.5 (14.6)*	0.25 (0.13)*
Formulation as flame-retardant in textiles, site AMI19U	4.71	0.04
Formulation/application of back coating as flame-retardant in textiles, site FT-2	2.17	0.02
Formulation as flame-retardant in textiles, site 96	3.06	0.03
Application of textile back-coating, generic site	69.4	0.62
Application of textile back-coating, site RC74U	2.38	0.02
Production of PET polymer – i.e. formulation	4.0	0.041
Production of PET articles – i.e. industrial use	0.75	0.007
Use in paint flame-retardant – formulation	1.32	0.011
Use in paint, pigment production	1.47	0.012
Use in paint, site AMI 3G - formulation,	3.19	0.03
Formulation in glass	0.72	0.006
Glass-manufacturing Proc Glass 2	3.1	0.03
Glass-manufacturing Proc Glass 3	0.90	0.008
Regional	0.72	0.006

*The PEC_{wate} is calculated assuming that the waste water is not treated in a municipal STP.The figures within brackets represent the PEC and RCR if STP treatment is assumed.

All risk characterization ratios are below 1, which indicates that the risk to surface water is low both from regional and local sources.

Conclusions to the risk assessment for surface water:

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those that are already being applied.

This conclusion applies to all scenarios.

3.3.1.2 Sediment

$PNEC_{sediment}$ = 7.8 mg Sb/kg ww was derived by dividing the lowest NOEC (midge *Chironomus riparius* or oligochaete *Lumbriculus variegates*) with an assessment factor of 10, since NOECs from three species with different living and feeding conditions are available.

The risk characterisation ratios for sediment are shown in Table3-108. Predicted PEC:s have only been included for the realistic worst cases and if these give PEC > PNEC also for other sites in the same use pattern, but not for all sites giving emission information. Reported data for all sites are presented in the release section.

It is the PEC derived using the median of the available partition coefficients of suspended matter that has been used for the risk characterisation.

Table3-108. PEC/PNEC ratios for sediment

	$PEC_{local, sediment}$ (mg Sb/kg wet weight)	RCR (PEC/PNEC)
Production, site P1	38.33	*4.9*
Production, site P5	0.80	*0.10*
Formulation as flame-retardant in plastics and rubber	4.51	*0.58*
Formulation as flame-retardant in plastics and rubber, assuming releases to air	0.65	*0.083*
Processing as flame-retardant in plastics and rubber	2.01	*0.26*
Formulation as flame-retardant in textiles, generic site	27.65	*3.54*
Formulation as flame-retardant in textiles, site AMI19U	4.52	*0.58*
Formulation/application of back-coating as flame-retardant in textiles, site FT-2	2.06	*0.26*
Formulation as flame-retardant in textiles, site 96	2.92	*0.38*
Application of textile back-coating, generic site	67.36	*8.64*
Application of textile back-coating, site RC74U	2.26	*0.29*
Production of PET polymer – i.e. formulation	3.84	*0.49*
Production of PET articles – i.e. industrial use	*0.68*	*0.087*
Use in paint flame-retardant – formulation	1.23	*0.16*
Use in paint, pigment production	1.38	*0.18*
Use in paint, site AMI 3G - formulation,	3.05	*0.39*
Formulation in glass	*0.65*	*0.083*
Glass-manufacturing ProcGlass 2	*2.93*	*0.38*
Glass-manufacturing ProcGlass 3	*0.83*	*0.11*
Regional (PEC based on monitoring data)	*0.65*	*0.083*

The PEC/PNEC ratio is well below 1 for the regional risk characterization, but above 1 in some local scenarios. The risk characterisation is conservative in the way that the bioavailability is

assumed to be 100%. There is however, no information available that makes it possible to take bioavailability into account.

Four sites producing diantimony trioxide have reported information on releases making it possible to make a completely site-specific prediction of PEC. Of these one (site P1) had a PEC/PNEC ratio >1 which is also supported by measured concentrations of antimony in sediment near the site (see section Comparison between predicted and measured levels) whereas the other three had PEC/PNEC ratios <1.

For the use areas formulation and industrial processing of flame-retardant textile back-coatings, the PEC/PNEC ratios are above 1 only for the generic (formulation and processing) sites. The PEC calculation for the generic textile formulation site is performed assuming no municipal STP treatment. The reason for this is that the emission factor used represents emissions after on-site treatment and it is not assumed to be a realistic worst case to assume both on site and off site sewage treatment. However, even if it was assumed that the waste water from the generic site was treated off site the PEC would be 14.2 mg/kg ww giving a PEC/PNEC ratio of 1.82. Nine sites using diantimony trioxide in textiles have reported information on releases making it possible to make a completely site-specific prediction of PEC. Of these all had PEC/PNEC ratios <1. Ten sites gave enough information on releases (to either water or air but not both) to make a site-specific prediction of PEC based partly on reported and partly on default values. All of these gave PEC/PNEC ratios <1. Six sites gave some information, but not enough to make a prediction of a site-specific PEC.

Conclusions to the risk assessment for the sediment compartment:

Conclusion (iii) There is a need for limiting the risks; risk reduction measures which are already being applied shall be taken into account.

This conclusion applies to the generic scenarios for formulation and application of flame-retardant textile back-coating and to one production site (site P1).

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those that are already being applied.

This conclusion applies to all other scenarios, including nineteen sites using diantimony trioxide in textile applications and three production sites, that all report releases.

3.3.1.3 Waste water treatment plants

$PNEC_{micro\text{-}organisms}$ = 2.55 mg Sb/l, which consists of a NOEC from a study performed according to the ISO 9509 protocol on inhibition of antimony on nitrification in activated sludge.
The risk characterisation ratios for WWTP are shown in Table3-107 below. Predicted PEC:s have only been included for the realistic worst cases, not for all sites giving emission information. Reported data for all sites are presented in the release section. The reason is that all realistic worst cases predicted PEC:s are below PNEC.

Table3-109 PEC/ PNEC ratios for waste water treatment plants.

	PEC_{STP} (µg Sb/l)	RCR (PEC/PNEC)

Production, P1	not relevant*	not relevant*
Production, P5	1.7	0.0007
Formulation as flame-retardant in plastics and rubber	42.5	0.02
Formulation as flame-retardant in plastics and rubber, assuming all releases to air	0	-
Processing as flame-retardant in plastics and rubber	14.9	0.006
Formulation as flame-retardant in textiles, generic site	148	0.06
Formulation as flame-retardant in textiles, site AMI19U	90	0.04
Formulation/processing as flame-retardant in textiles, site FT-2	15.5	0.006
Formulation as flame-retardant in textiles, site 96	25.0	0.01
Application of textile back-coating, generic site	733	0.29
Application of textile back-coating, site RC74U	17.7	0.007
Production of PET polymer – i.e. formulation	35	0.01
Production of PET articles – i.e. industrial use	0.33	0.0001-
Use in paint flame-retardant – formulation	not relevant	not relevant
Use in paint, pigment production	not relevant	not relevant
Use in paint, site AMI 3G - formulation,	not relevant	not relevant
Formulation in glass	0	-
Glass-manufacturing Proc Glass 2	not relevant	not relevant
Glass-manufacturing Proc Glass 3	not relevant	not relevant

* wastewater treatment on site, using mechanical separation

All risk characterisation ratios are below 1, which indicates that the risk to STP is low.

Conclusions to the risk assessment for waste water treatment plants:

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those that are already being applied.

This conclusion applies to all scenarios.

3.3.2 Terrestrial compartment

The $PNEC_{soil}$ is based on the porewater concentration measured at NOEC (9.7 mg Sb/l), divided with an assessment factor of 10 (cfr. Standard TGD-procedures) which is multiplied with the equilibrium solid:liquid distribution coefficient (K_d) for Sb in this soil. The K_d value for the soil used in the present studies by (Smolders et al., 2007 and Moser 2007) is 38 l/kg, which is the value observed for the Sb_2O_3 amended soil aged for five years and for the soluble $SbCl_3$ added to soil (Oorts et al., 2005).

The resulting $PNEC_{soil}$ after having performed this calculation is 37 mg Sb/kg dw corresponding to 32.6 mg Sb/kg ww.
The risk characterization ratios for agricultural soil are shown in Table3-110. Predicted PEC:s have only been included for the realistic worst cases, not for all sites giving emission information. The

reason is that all realistic worst case predicted PEC:s are below the PNEC. Reported emission data for all sites are presented in the release section.

Table3-110 PEC/ PNEC ratios for agricultural soil (30 d average).

	PEC (mg Sb/kg w w)	**RCR (PEC/PNEC)**
Production, site P1	1.51	0.046
Production, site P5	1.56	0.048
Formulation as flame-retardant in plastics and rubber	3.04	0.093
Formulation as flame-retardant in plastics and rubber, assuming releases to air	1.5	0.046
Processing as flame-retardant in plastics and rubber	2.04	0.063
Formulation as flame-retardant in textiles, generic site	1.5 (6.86)*	0.046 (0.21)*
Formulation as flame-retardant in textiles, site AMI19U	4.75	0.15
Formulation/application of back-coating as flame-retardant in textiles, site FT-2	2.06	0.06
Formulation as flame-retardant in textiles, site 96	2.4	0.074
Application of textile back-coating, generic site	28	0.86
Application of textile back-coating, site RC74U	2.14	0.07
Production of PET polymer – i.e. formulation	2.77	0.08
Production of PET articles – i.e. industrial use	1.51	0.046
Use in paint flame-retardant – formulation	1.62	0.050
Use in paint, pigment production	1.65	0.050
Use in paint, site AMI 3G - formulation,	5.12	0.16
Formulation in glass	1.5	0.046
Glass-manufacturing Proc Glass 2	1.5	0.046
Glass-manufacturing Proc Glass 3	1.5	0.046
Regional	1.5	0.046

*The PECsoil is calculated assuming that the waste water is not treated in a municipal STP and consequently no sewage sludge is spread on agricultural soil. The figures within brackets represent the PEC and RCR if STP treatment and spreading of sewage sludge on agricultural soil is assumed.

All risk characterisation ratios are below 1, which indicates that the risk to soil is low.

Conclusions to the risk assessment for the terrestrial compartment:

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those that are already being applied.

This conclusion applies to all scenarios.

3.3.3 Atmosphere

No PNEC can be derived for the atmosphere and so only a qualitative assessment can be made for this compartment. Neither biotic nor abiotic effects are considered likely due to the atmospheric release of antimony resulting from production and use of products containing diantimony trioxide, nor are any effects considered likely due to releases of antimony from unintentional sources.
Available information indicates a potential for antimony to be transported over long distances via the atmosphere.

Conclusions to the risk assessment for the atmosphere:

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

3.3.4 Secondary poisoning

The PNEC for secondary poisoning has been determined to 374.8 mg Sb/kg food. The basis for this is the results from a 90 d repeated dose study on rats and the assessment factors suggested in the TGD. The NOAEL from this study was 1686 mg/kg bw/day for female rats. The resulting PEC/PNEC ratios for the fish food chain are shown in Table3-111 and for the earthworm food chain in Table3-112. Predicted PEC:s have only been included for the realistic worst cases, not for all sites giving emission information.. The reason is that all realistic worst case predicted PEC:s are below the PNEC. Reported data for all sites are presented in the release section

Table3-111. PEC/PNEC ratios for secondary poisoning via the fish food chain

	BCF=40	
	PEC (mg Sb/kg)	RCR (PEC/PNEC)
Production, site P1	0.18	0.0005
Production, site P5	0.031	0.00008
Formulation as flame-retardant in plastics and rubber	0.094	0.0003
Formulation as flame-retardant in plastics and rubber, assuming releases to air	0.029	0.0000827
Processing as flame-retardant in plastics and rubber	0.052	0.0001
Formulation as flame-retardant in textiles, generic site	0.46	0.001
Formulation as flame-retardant in textiles, site AMI19U	0.083	0.0002
Formulation/application of back-coating as flame-retardant in textiles, site FT-2	0.050	0.0001
Formulation as flame-retardant in textiles, site 96	0.057	0.0002
Application of textile back-coating, generic site	1.2	0.003
Application of textile back-coating, site RC74U	0.045	0.0001
Production of PET polymer – i.e. formulation	0.083	0.0002
Production of PET articles – i.e. industrial use	0.029	0.0001
Use in paint flame-retardant – formulation	0.039	0.0001

Use in paint, pigment production	0.041	0.0001
Use in paint, site AMI 3G - formulation,	0.076	0.0002
Formulation in glass	0.029	0.0001
Glass-manufacturing Proc Glass 2	0.045	0.0001
Glass-manufacturing Proc Glass 3	0.030	0.0001

Table3-112. PEC/PNEC ratios for secondary poisoning via the earthworm food chain

	PEC (mg Sb/kg)	RCR (PEC/PNEC)
Production, P1	0.17	0.0005
Production, P5	0.17	0.0005
Formulation as flame-retardant in plastics and rubber	0.25	0.0007
Formulation as flame-retardant in plastics and rubber, assuming releases to air	0.17	0.0005
Processing as flame-retardant in plastics and rubber	0.20	0.0005
Formulation as flame-retardant in textiles, generic site	0.47	0.001
Formulation as flame-retardant in textiles, site AMI19U	0.35	0.0009
Formulation/application of back-coating as flame-retardant in textiles, site FT-2	0.20	0.0005
Formulation as flame-retardant in textiles, site 96	0.22	0.0006
Application of textile back-coating, generic site	1.7	0.004
Application of textile back-coating, site RC74U	0.20	0.0005
Production of PET polymer – i.e. formulation	0.24	0.0006
Production of PET articles – i.e. industrial use	0.17	0.0005
Use in paint flame-retardant – formulation	0.17	0.0005
Use in paint, pigment production	0.18	0.0005
Use in paint, site AMI 3G - formulation,	0.37	0.001
Formulation in glass	0.17	0.0005
Glass-manufacturing Proc Glass 2	0.17	0.0005
Glass-manufacturing Proc Glass 3	0.17	0.0005

All risk characterization ratios are far below 1, for both the fish and the earthworm food chains. This indicates that the risk of secondary poisoning from the use of diantimony trioxide is low.

Conclusions to the risk assessment for secondary poisoning:

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

3.3.5 Marine risk assessment

3.3.5.1 PBT-assessment

There is currently no agreed approach to perform a PBT-assessment of a metal, therefore a PBT-assessment will not be performed.

3.3.5.2 Risk characterisation for the marine environment

3.3.5.2.1 Marine water

The PNEC for marine surface water has been estimated to 11.3 µg Sb/l based on a NOEC for freshwater fish of 1.13 mg/l and an assessment factor of 100 according to the TGD. The PEC/PNEC ratios for marine surface water are given in Table3-113. Predicted PEC:s have only been included for the realistic worst cases and if these give PEC > PNEC also for other sites in the same use pattern, but not for all sites giving emission information. In this case there are no reporting sites known to be located by the sea. Reported data for all sites are presented in the release part.

Table3-113 PEC/PNEC ratios for marine surface water

	PEC (µg Sb/l)	**RCR (PEC/PNEC)**
Production, P1**	-	-
Production, P5*	0.23	0.02
Formulation as flame-retardant in plastics and rubber	1.00	0.09
Formulation as flame-retardant in plastics and rubber, assuming releases to air	0.20	0.02
Processing as flame-retardant in plastics and rubber	0.48	0.04
Formulation as flame-retardant in textiles, generic site	2.98	0.26
Formulation as flame-retardant in textiles, site AMI19U**	-	-
Formulation/processing as flame-retardant in textiles, site FT-2**	-	-
Formulation as flame-retardant in textiles, site 96**	-	-
Application of textile back-coating, generic site	13.9	**1.23**
Application of textile back-coating, site RC74U**	-	-
Production of PET polymer – i.e. formulation	0.9	0.08
Production of PET articles – i.e. industrial use	0.20	0.02
Use in paint flame-retardant – formulation	0.26	0.02
Use in paint, pigment production	0.27	0.02
Use in paint, site AMI 3G - formulation**,	-	-
Formulation in glass	0.20	0.02
Glass-manufacturing Proc Glass 2***	0.44	0.04

Glass-manufacturing Proc Glass 3**	-	-
Regional (PEC based on monitoring data)	0.20	0.02

* not known whether the releases are to freshwater or marine water

**not located by the sea

*** not located by the sea, included as a theoretical realistic worst case

The risk characterization for marine water indicates concern locally for the application of textile back-coating generic site.

Conclusions to the risk assessment for marine surface water:

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those that are already being applied.

This conclusion applies to all scenarios.
Despite having RCR >1 conclusion iii) is not drawn for application of flame-retardant back-coating. The reason for this is that, according to information from IAOIA, none of the sites covered by the survey IAOIA performed to collect exposure data from all their customers is located by the sea. However, it has to be pointed out that the coverage of this survey regarding textile backcoating sites was rather low. Therefore, it cannot be ruled out that textile backcoating sites located at the sea having emissions to the marine environment may exist.

3.3.5.2.2 Marine sediment

As no studies on marine sediment organisms are available the $PNEC_{marine\ sediment}$ is derived from studies on freshwater sediment species. Reliable and relevant chronic NOEC values are available for three freshwater sediment species with different living and feeding conditions. The freshwater NOEC is 78 mg Sb/kg ww. According to the TGD, the assessment factor used should be 50, which results in the following $PNEC_{marine\ sediment}$:

$PNEC_{marine\ sediment}$ = 1.56 mg Sb/kg ww

The PEC/PNEC ratios for marine sediment are given in Table3-114. Predicted PEC:s have only been included for the realistic worst cases and if these give PEC > PNEC also for other sites in the same use pattern, but not for all sites giving emission information. Reported data for all sites are presented in the release section.

It is the PEC derived using the median of the available partition coefficients of suspended matter that has been used for the risk characterisation

Table3-114. PEC/PNEC ratios for marine sediment.

	$PEC_{local,\ sed}$	**RCR**

	(mg/kg wet wt.)	(PEC/PNEC)
Production, P1**	-	-
Production, P5*	0.68	*0.44*
Formulation as flame-retardant in plastics and rubber	1.42	*0.91*
Formulation as flame-retardant in plastics and rubber, assuming releases to air	*0.65*	*0.42*
Processing as flame-retardant in plastics and rubber	0.92	*0.59*
Formulation as flame-retardant in textiles, generic site	3.35	*2.15*
Formulation as flame-retardant in textiles, site AMI19U**	-	-
Formulation/application of back-coating as flame-retardant in textiles, site FT-2**	-	-
Formulation as flame-retardant in textiles, site 96**	-	-
Application of textile back-coating, generic site	13.95	*8.9*
Application of textile back-coating, site RC74U**	-	-
Production of PET polymer – i.e. formulation	1.29	*0.83*
Production of PET articles – i.e. industrial use	*0.66*	*0.42*
Use in paint flame-retardant – formulation	0.71	*0.45*
Use in paint, pigment production	0.72	*0.46*
Use in paint, site AMI 3G - formulation**	-	-
Formulation in glass	*0.65*	*0.42*
Glass-manufacturing ProcGlass 2***	*0.67*	*0.43*
Glass-manufacturing ProcGlass 3**	-	-
Regional (PEC based on monitoring data)	*0.65*	*0.42*

*not known whether the releases are to freshwater or marine water

** not located by the sea

*** not located by the sea, included as a theoretical realistic worst case

Similar to freshwater sediment, the PEC/PNEC ratio is below 1 for the regional risk characterization, but above 1 in the local scenarios for certain use areas. An assessment factor of 50 has been used to derive the PNEC. The assessment factor could be lowered to 10 if two additional tests on marine sediment living species are performed. Furthermore, the risk characterisation is conservative in the way that the bioavailability is assumed to be 100%. There is however, no information available that makes it possible to take bioavailability into account

Conclusions to the risk assessment for the marine sediment compartment:

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those that are already being applied.

This conclusion applies to all scenarios.
Despite having RCR >1 conclusion iii) is not drawn for formulation and application of flame-retardant back-coating. The reason for this is that, according to information from IAOIA, none of

the sites covered by the survey IAOIA performed to collect exposure data from all their customers is located by the sea. For the formulation of flame-retardant in textiles, the coverage of this survey is high and there is a high probability that for this use area the marine scenario may not be relevant. For application of textile back-coating on the other hand the coverage of the survey is lower and it cannot be ruled out that sites located at the sea having emissions to the marine environment may exist.

3.3.5.2.3 Secondary poisoning in the marine environment

The same PNEC as used in the risk characterization for secondary poisoning in fresh water (93.6 mg Sb/kg food) is used also for marine water.

The PEC/PNEC ratios for marine secondary poisoning are given in Table3-115, and Table3-116.

Predicted PEC:s have only been included for the realistic worst cases and if these give PEC > PNEC also for other sites in the same use pattern, but not for all sites giving emission information. Reported data for all sites are presented in the release section.

Table3-115. PEC/PNEC ratios in fish for secondary poisoning in the marine environment

	BCF=40	
	PEC (mg Sb/kg)	**RCR (PEC/PNEC)**
Production,P1**	-	-
Production,P5*	0.0084	0.00002
Formulation as flame-retardant in plastics and rubber	0.021	0.00006
Formulation as flame-retardant in plastics and rubber, assuming releases to air	0.0080	0.00002
Processing as flame-retardant in plastics and rubber	0.013	0.00003
Formulation as flame-retardant in textiles, generic site	0.051	0.0001
Formulation as flame-retardant in textiles, site AMI19U**	-	-
Formulation/application of back-coating as flame-retardant in textiles, site FT-2**	-	-
Formulation as flame-retardant in textiles, site 96**	-	-
Application of textile back-coating, generic site	0.23	0.0006
Application of textile back-coating, site RC74U**	-	-
Production of PET polymer – i.e. formulation	0.019	0.00005
Production of PET articles – i.e. industrial use	0.0080	0.00002
Use in paint flame-retardant – formulation	0.0090	0.00002
Use in paint, pigment production	0.0092	0.00002
Use in paint, site AMI 3G - formulation**	-	-
Formulation in glass	0.0080	0.00002
Glass-manufacturing Proc Glass 2***	0.0096	0.00003
Glass-manufacturing Proc Glass 3**	-	-

* not known whether the releases are to freshwater or marine water

** not located by the sea

*** not located by the sea, included as a theoretical realistic worst case

Table3-116 PEC/PNEC ratios in fish-eating marine top predators for secondary poisoning in the marine environment

	BCF=40	
	PEC (mg Sb/kg)	**RCR (PEC/PNEC)**
Production, P1**	-	-
Production, P5*	0.0081	0.00002
Formulation as flame-retardant in plastics and rubber	0.010	0.00003
Formulation as flame-retardant in plastics and rubber, assuming releases to air	0.0080	0.00002
Processing as flame-retardant in plastics and rubber	0.0089	0.00002
Formulation as flame-retardant in textiles, generic site	0.017	0.00004
Formulation as flame-retardant in textiles, site AMI19U**	-	-
Formulation/application of back-coating as flame-retardant in textiles, site FT-2**	-	-
Formulation as flame-retardant in textiles, site 96**	-	-
Application of textile back-coating, generic site	0.053	0.0001
Application of textile back-coating, site RC74U**	-	-
Production of PET polymer – i.e. formulation	0.010	0.00003
Production of PET articles – i.e. industrial use	0.0080	0.00002
Use in paint flame-retardant – formulation	0.0082	0.00002
Use in paint, pigment production	0.0083	0.00002
Use in paint, site AMI 3G - formulation**	-	-
Formulation in glass	0.0080	0.00002
Glass-manufacturing Proc Glass 2***	0.0083	0.00002
Formulation in glass Proc Glass 3**	-	-

* not known whether the releases are to freshwater or marine water

** not located by the sea

*** not located by the sea, included as a theoretical realistic worst case

All risk characterization ratios are far below 1, both for predators and top predators.

Conclusions to the risk assessment for marine secondary poisoning:

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

3.3.6 Areas of uncertainty in the environmental risk assessment

As with any "generic" risk assessment, there are uncertainties inherent in the approach taken.

For most of the scenarios considered, the best information available to the specific industries has been used in preference to the default values. There is however not a complete set of data. This necessarily introduces uncertainties into the estimates.

The RCW-ambient PEC for water, sediment and soil is based on 90th percentiles from the available data. However, not all of the original data points are available, i.e. some studies only report median or mean values, whereas others report ranges, etc. This makes it very difficult to calculate a true 90th percentile value (which would need all of the individual data points from each study).

A consequence of taking the regional concentration to be the median of the country specific 90th percentile values is that, for a number of countries, the country-specific 90th percentile is actually higher than the concentration used for the regional concentration (e.g. France, Germany and UK for surface water (Table3-58) and Germany, Spain and UK for soil (Table3-72).

The selection of a value for the partition coefficient suspended matter – water, has a major impact on the conclusions on risk for both freshwater and marine sediments. Furthermore the bioavailability has been set to 100 % in the absence of other information.

DRAFT

4 HUMAN HEALTH

4.1 HUMAN HEALTH (TOXICITY)

4.1.1 Exposure assessment

4.1.1.1 General introduction

The human population may be exposed to diantimony trioxide at the workplace, from use of consumer products containing diantimony trioxide and indirectly via the environment through contact with contaminated air. Humans may also be exposed indirectly via the environment through consumption of food, water and soil. However, the exposure will then be to the antimony ion. Since there is a continuous exposure to diantimony trioxide via the environment, humans may be exposed during their entire lifetime. In addition, there might be combined exposure to humans from more than one of the three sources above.

Diantimony trioxide is released to the environment through air effluents and waste water from manufacture, formulation, processing, use and disposal of diantimony trioxide. As summarized in Table3-37, the emissions to air and wastewater resulting from production and use (and disposal) of diantimony trioxide constitute about 39 % (52 %) and 76 % (97 %) of the regionally estimated emissions, respectively. On the continental scale, the corresponding proportions of estimated emissions are 2 % (23 %) and 24 % (90 %) for air and wastewater, respectively. The unintentional sources identified and included in the modelling are production of non-ferrous metals, coal combustion and road traffic. The presence of antimony in the environment is also due to production and use (and disposal) of other antimony compounds and natural processes, such as weathering of rocks, soil runoff, volcanoes and sea salt spray. Environmental releases of diantimony trioxide are described in detail in the Environmental section of the RAR (3.1.2). The major use of diantimony trioxide is as a flame-retardant. However, it does not itself have flame-retarding properties; instead it is a synergist for halogenated flame-retardants in plastics, paints, adhesives, sealants, rubber, and textile back coatings. Other uses of diantimony trioxide include: as polymerisation catalyst used in PET resin manufacture and as a clarifying aid in certain glasses, and in pigments.

Diantimony trioxide is a solid substance at room temperature (melting point ≈ 655 °C) and is as a substance most often handled as solid powder; dry or in wetted form, pellets, paste, or granules. The vapour pressure of solid diantimony trioxide is low (1 mm Hg = 0.13 kPa at 574 °C) and it is only very slightly soluble in most solvents. The content of antimony in diantimony trioxide is 83.5 % (w/w).

In most final products, the diantimony trioxide is encapsulated in a matrix, in which it is either physically bound, such as in flame retarded rubbers, plastics and textile back coatings, or chemically bound in a transformed state such as in PET, glass and pigments. Only by way of wear processes (dry abrasion) is any release as diantimony trioxide feasible. In contrast, release processes such as leaching from moist/wet surfaces or into vessels will yield antimony that has been transformed into tri- and pentavalent hydroxo complexed form.

The particle size of diantimony trioxide differs between different technical products. The physical particle size of diantimony trioxide powders may range from 0.3 to 10.5 μm, depending on the type

and resolution of method used for the determination. However, such particle sizes are most often assayed for the purpose of compliance with the customer specification, and do not correspond to aerodynamic diameters relevant for airway deposition modelling. The manufacturers have stated that the commercially most relevant particle size of diantimony trioxide is in the range of 0.8-1.5 μm. Information and discussion on more relevant parameters for predicting airway deposition of particles is presented in the section on toxicokinetics (4.1.2.1).

Based on information in Chapters 1 and 2 the following exposure routes were considered to be relevant for the different populations evaluated in this assessment:

- Workers (occupational exposure from production, formulation and industrial and professional uses of diantimony trioxide)
  - Inhalation of airborne dust
  - Dermal contact with powder, pellets, paste, granules or final products
- Consumers
  - Inhalation, ingestion and dermal exposure of diantimony trioxide emitted from articles and in domestic dust
- Man via the environment
  - Inhalation of particles in air
  - Ingestion via intake of contaminated food and water, including breast feeding

Occupational exposure scenarios where diantimony trioxide is formed but not used, such as in battery production, have not been included in this assessment.

Polyethylene terephthalate (PET) and other polyesters are used, typically in non-woven fibre design, in the white cell adsorption layer of filters in certain medical equipment. Biomaterials of the fibre media are surface modified to obtain an optimal selectivity for the different blood cells (Kirk-Othmer, 2007). A limited survey conducted by the Swedish Medical Products Agency (MPA, 2006) indicates that such filters may contain antimony (speciation of antimony not specified). Due to lack of exposure data, the expected limited exposure compared to other sources and that antimony is not in the form of diantimony trioxide in these products, exposure from this application has not been included in the exposure assessment.

For the exposure assessment it is assumed, if not stated otherwise, that the measured values available are expressed as the total concentration of antimony. To calculate the concentration of diantimony trioxide, the measured levels of antimony are adjusted with a factor of 1.2 (correction for molecular weight). It should be clearly noted that the exposure for consumers (except via indoor/outdoor air and house dust) and "man via the environment" is not to diantimony trioxide itself but rather to the antimony ion, in the trivalent or predominantly pentavalent forms. However, the approach to transform all antimony to diantimony trioxide is taken in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide, in the risk characterisation. In addition, this approach may be regarded as conservative.

4.1.1.2 Occupational exposure

4.1.1.2.1 Introduction

There are several industries in which diantimony trioxide is produced or used, and the life cycle stages where occupational exposure may occur are; production, formulation, i.e. industrial use of diantimony trioxide as an additive and processing, i.e. industrial and professional use, of semi- or end-products containing diantimony trioxide. At some sites, both formulation and processing may take place. In addition, exposure might occur during recycling and disposal of articles containing diantimony trioxide, but there is no available information about this. It is assumed that exposure during recycling and disposal is limited compared to the exposure from production, formulation and industrial and professional use, and therefore these scenarios are not considered further in this exposure assessment.

The following data were used for the occupational exposure assessments of diantimony trioxide:

- Measured workplace data from production and uses of diantimony trioxide
- Physico-chemical data of diantimony trioxide (see chapter 1)
- Qualitative data, such as process descriptions and use pattern of the product, and quantitative data regarding frequency and duration of handling of diantimony trioxide
- Concentrations of diantimony trioxide used in the different products

For most scenarios of occupational exposure to diantimony trioxide a default value of 2000 cm^2 for exposed skin area is used (corresponding to hands and forearms). For Production a larger skin area is used (3160 cm^2), corresponding to face and neck in addition to hands and forearms, and for Processing in the Textiles sector, skin exposure is assumed to occur only on the hands (840 cm^2). A default respiratory volume of 10 m^3/work day and a default body weight of 70 kg are assumed.

Industry exposure data

A number of "tailor-made" questionnaires were developed by Industry (International Antimony Oxide Industry Association (IAOIA) specifically for the collection of data relevant for occupational exposure at the producer and downstream users of diantimony trioxide. Whereas the questionnaire for the four producers was written in English, the questionnaire sent out to the downstream users was made available in different EU languages. All downstream users were identified from customer databases of the primary diantimony trioxide producers and information from a consultant's database. The results from this survey together with information from other sources have been presented in nine separate reports (EBRC, 2005a; EBRC, 2006c; EBRC, 2006d; EBRC, 2006e; EBRC, 2006f; EBRC, 2006g; EBRC, 2006h; EBRC, 2006i; EBRC, 2006b). The four sections of the questionnaire covered general site information, information relevant to the type of diantimony trioxide used and the process conditions including control measures involved, a description of any personal protective equipment (PPE) used and occupational exposure data and details of the sampling methodology.

The questionnaires were distributed within Europe to previously identified companies. A total coverage of 100 % for producers and 63 % for downstream users, respectively, of the annual

consumption of diantimony trioxide was obtained. Process descriptions are based on visits conducted in each sector on representative sites. These visits were conducted by the authors of the reports. Occupational exposure data were gathered from each of the sectors, where available. Relevant exposure descriptors were collected by a questionnaire in order to provide for a sector-wide analysis of:

- Frequency/duration of handling and amount of handled diantimony trioxide powder
- Process and exposure controls, and details of protective measures
- Similarity of relevant process conditions
- Used forms of diantimony trioxide
- Laboratory particle size determinations - and dustiness (not sector specific)

The following diantimony trioxide downstream user markets were identified;

- PET resin
- PET fibre and films
- Flame retarded plastics – PVC
- Flame retarded plastics – non PVC
- Flame retarded textiles
- Pigments/paints/coatings and ceramics
- Rubber manufacturing
- Crystal glass

According to the survey results, diantimony trioxide is used as an input material in the different market sectors in one or more of the five following forms;
Powder (POW), dry powder without any additives.
Pre-dosed powder (PRE), dry powder without additives in pre-weighed sealed foil bags for direct feeding into closed systems without opening.
Dispersed (DIS), aqueous, pre-formulated preparations containing 50-80 % diantimony trioxide; besides water, other liquids can be used as dispersion aid.
Wetted (WET), diantimony trioxide in powder form, to which a minor portion of liquids, such as liquid process agents, oils, plasticisers etc has been added usually at 2–10 % (w/w) to reduce dustiness.
Masterbatch (MB), diantimony trioxide in granulated form, tightly enclosed in a polymer matrix, with a polymer content of 10–90 %.

In the survey by EBRC (2005a) occupational exposure data were collected from all four production sites within Europe, yielding a database of 106 personal monitoring values. Data were also collected by a questionnaire, and a process description was derived based on a subjective exposure assessment conducted during a site visit to one of the European plants. Information on duration of sampling, sampling date, sampling method, sampled size fraction, details on analytical methods and discrimination between personal and static sampling was mandatory. The database was screened for quality and relevance, i.e. the full set of information was required from each company: only data from 1998 or later, sampling had to be at least 120 min to be representative for the shift exposure, measured fraction had to be inhalable and values should not represent aggregated values in the form of averages or ranges.

Also, other data on occupational exposure to diantimony trioxide by inhalation had previously been provided by the Industry (Vandenbroele et al., 2003; EBRC, 2005a; EBRC, 2005b). From 68 sites contacted by Industry, and referred to in the report by Vandenbroele et al., only half of the sites (49

%) submitted data, either a complete data set (questionnaire submission) or a limited data set. 21 % of the sites stated that they had no exposure data available and 31% of the sites did not respond. These data were not considered adequate for risk assessment purposes.

In addition the Berufsgenossenschaftliches Institut für Arbeitsschutz, BIA, in Germany measured levels of diantimony trioxide in 50 different companies, representing most of the relevant industry segments, during the period 1989 to 1992. The number of samples was 132 and the results were related to the occupational exposure limits in Germany. 90 % of the measurements gave a result below 0.208 mg/m^3 and 50 % of the measurements gave a result below 0.0035 mg/m^3 (Bock et al., 1995). Since these values originate from an organisation that in Germany has the formal role of monitoring compliance with OELs, the finding that the overwhelming majority of data are several orders of magnitude below the current OEL of 0.5 mg/m³ indicates generally low exposures to diantimony trioxide in German diantimony trioxide downstream user industries.

Analogous and surrogate data

Analogous or surrogate data are used for some life cycle stages when the collected data is not considered to be sufficient and/or representative and data from other uses with similar handling of the substance are available. When reading-across from one sector to another, all available information is used, e.g. process descriptions, amount of substance used and data on frequency and duration of handling. An assessment of the quality and quantity of the available information is also of importance. When estimating the dermal exposure in downstream use scenarios the analogues data used (full-shift measured data from Production (scenario 1) were modified using the 90th percentile (rounded-off) of the duration of handling/shift (obtained from questionnaires) except for in one case (PPCC, scenario 5), where a slightly longer duration was used. For inhalation exposure assessment, where measured data from Plastics (scenario 3) were used, the mean duration of handling diantimony trioxide in the various downstream sectors was used for modification of the measured values.

Modelled data

Results from the model RISKOFDERM are used for one scenario.

Route of exposure

The main routes of occupational exposure to diantimony trioxide are anticipated to be by inhalation of airborne solid dust and dermal exposure to solid diantimony trioxide, based on the physico-chemical information and descriptions of the manufacturing process and formulation and uses of products containing diantimony trioxide (see Chapter 2). Dermal exposure may occur during direct handling, either by contamination of skin surfaces or by dermal deposition of airborne dust.

In a survey by Hughson (2005) on dermal exposure during production of diantimony trioxide it was shown that it is not just the hands and forearms which may become exposed to diantimony trioxide. There is also potential for similar exposure levels on other skin areas such as the face and neck, especially during packing. For this reason a larger skin area (3160 cm^2) is assumed for the Production sector compared to the other six occupational scenarios (2000 cm^2).

Particle size

The particle size is important for exposure assessment of dust. The handling of powder is assumed to give more respirable and inhalable airborne dust than handling of the granule grade qualities. This affects the degree of both dermal and inhalation absorption. The estimation of uptake through inhalation may depend on the deposition pattern in the airways, and this deposition pattern in turn depends on the particle size distribution.

In a study on the derivation of a characteristic median physical particle size (d50) value for the ”total material”, eight samples of diantimony trioxide, representing the range of commercially available particle sizes, and relevant parameters for predicting airway deposition were investigated (the study is described in section 4.1.2.1 Toxicokinetics). Except for the two samples with the highest median physical particle size all other samples show bimodal distributions. According to the author this can only be explained by a high tendency of agglomeration. It was observed that the fraction of aggregates decreased with increasing particle size, and one apparent interpretation would be that with decreasing particle size, the surface of particles in relation to their volume increases, and thus may enhance aggregation via adhesion. In this laboratory investigation the material with a physical particle size between 1.18 and 1.26 μm, corresponding to a MMAD (mass median aerodynamic diameter) of 4.12-4.33 μm, yielded the finest airborne particles after mechanical agitation.

In a workplace measurement survey of airborne dust, particle size distributions were estimated during conversion, refining and packaging (final handling) in a diantimony trioxide producing plant using personal cascade impactors (Hughson, 2005). Five or six samples from each work task were obtained. The particle size distributions were quite variable, both within and between job titles. For example, data from the packaging workers showed that between 7 % and 35 % of the particles were equal to or smaller than 3.5 μm (aerodynamic diameter). When using a GRIMM aerosol spectrometer (an instrument that is biased towards smaller particle size ranges compared to the impactors) in the packaging area, up to 60 % of the particles were smaller than 4 μm. It was indicated that the finished diantimony trioxide product in the plant had a physical diameter of approximately 1 μm.

Frequency and duration

Information on the frequency and duration of handling/task has been collected by Industry using questionnaires. The coverage and quality of the information varies between the different industry sectors, but data have been provided from all sectors, except from Production, where diantimony trioxide handling is considered to be continuous/full shift. This information may be used for recalculation of the exposure using modelled data or analogous/ surrogate data.

Although rather substantial, the provided data on frequency and duration of handling of diantimony trioxide powder for the reported scenarios can not in all cases be regarded as adequate and representative. There are large variations both in reported values for frequency and for duration within the same sector. Furthermore, there appears to be a poor correlation between total duration of handling and the amount of diantimony trioxide handled, which is not explained by different handling techniques. In addition, the information on the actual exposure situation is somewhat limited. Therefore, the representativity of the provided data is sometimes difficult to assess and it cannot be excluded that the duration of handling can possibly be longer than the reported maximum values at certain sites and at certain occasions. In addition, the duration of exposure may in some

cases be longer than the duration of handling diantimony trioxide powder, taking e.g. cleaning into account. Using the 90th percentile of the reported values for duration of handling, as suggested by Industry, may lead to an underestimation of the exposure and for a few scenarios a slightly longer duration has been used in order to achieve a reasonable worst case value. For measured data, the time during which the sampling has been performed is given. The duration of sampling is used (when appropriate) to recalculate measured values to time-weighted average full shift values assuming that the exposure outside the duration of sampling is negligible.

Personal Protective Equipment (PPE)

At workplaces where the substance is handled as the pure substance, the premises and equipment is expected to be more or less contaminated with the substance. This causes exposure e.g. during cleaning and during the everyday presence and activity at the workplace. For all activities the exposure is strongly influenced by plant conditions and worker habits. Bad hygiene in a plant could lead to high background concentrations. Examples of this are broken bags, dusty pallets and dusty rooms. The presence of effective control measures can have a great influence on the exposure (Lansink et al., 1996).

If the risk assessment, as based on potential exposure, indicates that risks are to be expected, the use of PPE may be one of the methods to decrease exposure, although other approaches e.g. technical and organizational are to be preferred. PPE are primarily intended for use during work operations entailing risk for increased exposure for a limited time, such as repair work, service and maintenance. The efficiency of a PPE is largely dependent on site-specific aspects of management, procedures and training of workers. The use of PPE normally reduces the level of exposure, but incorrect or careless use may lead to unforeseen and unexpected exposure. Thus, the exposure is normally assessed without directly taking into account the possible influence of PPE. The only exceptions are exposure via inhalation during Production, where data is available from all four existing production sites in Europe and where use of respiratory protective equipment (RPE) is mandatory, and when modelling dermal exposure using RISKOFDERM, where use of gloves is assumed. It may also be pointed out that the measured dermal exposure during Production, also used for read-across in several downstream use scenarios, is assessed for more or less protected workers using gloves and protective clothing to a large extent.

Short-term exposure

Short-term exposure is not assessed in this report, because there is no endpoint of concern for any acute effect, as verified by guideline-conform acute toxicity assays by the oral, dermal and inhalation route.

4.1.1.2.2 Scenario 1: Production of diantimony trioxide

During production the generic scenarios considered are:

- **Conversion,** which refers to conversion of antimony metal to antimony oxides and covers work tasks like loading of furnace with antimony ingots, supervision of operating conditions and routine inspections and adjustments.
- **Refuming** (if crude antimony metal feedstock is used) is done in order to adjust the chemical and physical properties of the product. This is achieved by feeding the material through a furnace which has a free flow of air into the transfer ducting.
- **Final handling** covers all kind of work tasks at places where the final product (diantimony trioxide) is handled, like weighing, packaging etc.

Summary of Industry information

Inhalation exposure

From data submitted by Industry (Vandenbroele et al., 2003) it could be concluded that during production of diantimony trioxide workers are exposed during different manipulations of diantimony trioxide (packaging, blending, formulation, converting, refining, roasting, oxidation) to different forms of diantimony trioxide. In two out of five companies investigated workers were continuously exposed to diantimony trioxide; in the other three, exposure was intermittent and/or occasional. From the five companies considered only one company (P-1) could provide recent information (1998-2003) concerning air sampling (Static area Sampling **SS** and Personal Sampling **PS**) and biological monitoring. Company P-4 provided measured personal and static air sampling data for different work areas (1987, 1991 and 1996). Air exposure levels between 0.3-9.8 mg Sb/m^3 (SS) and 1.5-9 mg Sb/m^3 (PS) are reported for company P-3 (roasting processes, packaging, further processing, 1991). However, there is no indication to which specific process the exposure data are related. Company P-5 provided limited personal air sampling data for the year 2002. It should be noted that this company started diantimony trioxide production in 2000. Company P-2 did not provide any workplace monitoring data.

For the crushing metal process, exposure levels of 2.3 mg Sb/m^3 (SS) and 1.4 mg Sb/m^3 (PS), respectively, were measured (1996). During the oxidation process, i.e. **conversion**, workers were exposed to air levels between 1.8 and 23 mg Sb/m^3 (PS). The 8-hr static air exposure levels for this process varied between 1 and 2 mg Sb/m^3.

4-hr atmospheric exposure levels (SS) during refining, i.e. **refuming,** varied between 0.05 and 4 mg Sb/m^3. Personal sampling data for the same process were around 2.3 mg Sb/m^3 (1998 and 2001). Exposure during making of slags/scories is lower; mean exposure values of 0.13 mg Sb/m^3 (SS) and 0.24 mg Sb/m^3 (PS) were noted (2000).

Depending on type of packaging, i.e. **final handling,** equipment used, mean static air exposure data varied between 0.12 and 4.24 mg Sb/m^3 for both SS and PS. Final handling also includes mixing (mean PS 3.2 mg Sb/m^3, range 2.4-3.8) and masterbatching processes (mean PS 4.3 mg Sb/m^3, range 1.9-6.7). Static air exposure levels during these processes were as low as 0.8 (0.5-1.3) and 0.33 (0.2-0.5) mg Sb/m^3, respectively, showing that static measurements do not reflect the actual exposure at least not during mixing and masterbatching. During production of sodium antimonate personal air exposure levels and static exposure levels ranged from 6.6 to 10 mg Sb/m^3 and from 1.7 to 1.9 mg Sb/m^3 (1996).

In the later survey by EBRC (2005a) occupational exposure data were collected from all four production sites in use in Europe, yielding a database of 106 personal monitoring values. It is not entirely clear from the report whether these exposure values represent Sb or Sb_2O_3, however after discussion with Industry it is assumed that they represent Sb_2O_3. Further, this uncertainty does not affect the conclusion on risk. Data were also collected by a questionnaire, and a process description was derived based on a subjective assessment conducted during a site visit to one of the European plants. Information on duration of sampling, sampling date, sampling method, sampled size fraction, details on analytical methods and discrimination between personal and static sampling was mandatory. The database was screened for quality and relevance, i.e. the full set of information was required from each company; only data from 1998 or later, sampling had to be at least 120 min to be representative for the shift exposure, measured fraction had to be inhalable and values should not represent aggregated values in the form of averages or ranges.

According to this survey roasting is no longer conducted within the European Union, and three specific tasks were identified for production of diantimony trioxide: oxidation (conversion), refuming and packaging, i.e. final handling. Job-specific typical (median) and reasonable worst case (90^{th} percentile) inhalation exposures of 0.54 and 2.9 mg/m^3, respectively, during oxidation, i.e. **conversion**, 0.23 and 0.94 mg/m^3, respectively, during **refuming** and 0.79 and 2.1 mg/m^3, respectively, during packaging, i.e. **final handling,** were derived for exposure without taking the use of respiratory protective equipment (RPE) into consideration (see Table 4-1).

It was reported by the author that all workers at all four production sites use respiratory protective equipment on a mandatory basis. In addition, according to a study on dermal exposure (Hughson, 2005), all workers at the production site investigated (1 site) were observed to obey this rule at the time of the survey. Based on the information on use of RPE a refined exposure assessment was conducted in the report by EBRC (2005a) reflecting the specific type of filter and face-piece combination used in each company by applying the corresponding assigned protection factor (APF) to each individual measured data point. The APF for a specific type and class of RPE is published in British Standards: BS4275:97 Guide to implementing an effective respiratory protective device programme (British standard, 1997). It relates to the likely performance of the whole RPE when worn correctly and used in accordance with the manufacturer's instructions.

Table 4-1 Air concentrations of diantimony trioxide (mg/m^3) from production of diantimony trioxide and values reflecting the use of respiratory protection equipment -RPE (EBRC, 2005a)

Workplace	Sampling method	No of samples	Concentration of diantimony trioxide in air				Concentration of diantimony trioxide in air reflecting use of RPE			
			Min	Median	P90	Max	Min	Median	P90	Max
Roasting	PS	No longer conducted within EU								
Oxidation	PS	47	0.00	0.54	2.9	11	0.000	0.027	0.15	0.56
Refuming	PS	21	0.05	0.23	0.94	1.1	0.003	0.012	0.047	0.054
Packaging	PS	38	0.08	0.79	2.1	4.7	0.006	0.040	0.11	0.23
All workplaces	PS	106	0.00	0.43	2.3	11	0.000	0.024	0.12	0.56
Roasting	SS	No longer conducted within EU								
Oxidation	SS	31	0.00	0.46	1.4	3.4	-	-	-	-
Refuming	SS	18	0.00	0.13	2.4	3.3	-	-	-	-
Packaging	SS	22	0.00	0.57	1.6	3.6	-	-	-	-
All workplaces	SS	71	0.00	0.32	1.7	3.6	-	-	-	-

The job-specific inhalation exposures were in this latter case 0.027 (median) and 0.15 mg/m^3(90th percentile) during oxidation, 0.012 and 0.047 mg/m^3 during refuming and 0.040 and 0.11 mg/m^3 during packaging, for typical and reasonable worst-case inhalation exposures, respectively (see Table 4-1).

Use of RPE in the production of diantimony trioxide and assigned protection factors for a specific type and class of RPE as published in BS4275 (British standard, 1997) are given in Table 4-2 below. It relates to the likely performance of the whole device when worn correctly and used in accordance with the manufacturer's instruction (EBRC, 2005a).

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Table 4-2 Use of RPE in the production of diantimony trioxide and the assigned protection factors for a specific type and class of RPE

Plant	Workplace	Use of RPE	Type of RPE	Assigned protection factor
A	Oxidation	Yes, mandatory	P3-quarter mask	20
	Refuming	Yes, mandatory	P3-quarter mask	20
	Packaging	Yes, mandatory	P3-quarter mask	20
C	Oxidation	Yes, mandatory	P2-quarter mask	10
	Refuming	na	na	na
	Packaging	Yes, mandatory	P2-quarter mask	10
D	Oxidation	Yes, mandatory	P3-quarter mask	20
	Refuming	na	na	na
	Packaging	Yes, mandatory	P3-quarter mask	20
E	Oxidation	Yes, mandatory	P2-full face mask	10
	Refuming	na	na	na
	Packaging	Yes, mandatory	P2-quarter mask	10

na=not applicable since refuming is not performed at this site

Particle analysis of airborne dust, composition analysis of inhalable dust samples and measurements of dermal exposure.

On behalf of Industry a workplace measurement survey was designed to provide information about occupational exposures to diantimony trioxide in the primary production industry at one European production site (Hughson, 2005). The work involved measurements of dermal exposure, particle analysis of airborne dust and composition analysis of inhalable dust samples. The inhalation exposure data is included in the survey by EBRC (2005a).

The work programme was designed to include developmental and laboratory validation of sampling and analytical methods, a workplace measurement survey of airborne dust, determining the airborne concentration, composition and size distribution of diantimony trioxide aerosols and finally work place measurements survey of dermal exposure in parallel with the second element above. A similar exercise with practically the same sampling methodology has previously been conducted in the zinc industry (Hughson and Cherrie, 2001), enabling the two sets of measurements to be compared.

Composition analysis

Since the substance of interest was diantimony trioxide, other dust which may be present in the workplace could cause positive interference in the gravimetric analysis. Samples of inhalable dust were therefore collected over a full or representative portion of the working shift and analysed for diantimony trioxide in order to aid the interpretation of the results from the personal cascade impactor measurements. The percentage of diantimony trioxide in the various dust samples ranged from 13 % to 69 %.

Particle size distribution

For particle size distribution assessment, five or six samples from each work task were obtained. The particle size distributions were quite variable, both within and between job titles. For example, data from the packaging workers showed that between 7 % and 35 % of the particles were equal to or smaller than 3.5 μm (aerodynamic diameter). When using a GRIMM aerosol spectrometer (an instrument that is biased towards smaller particle size ranges compared to the impactors) in the packaging area, up to 60 % of the particles were smaller than 4 μm. It was indicated that the finished diantimony trioxide product in the plant had a physical diameter of approximately 1 μm. Furthermore, it was recognised that individual particles tend to agglomerate thereby creating airborne particles with higher aerodynamic diameters compared to the physical size of the individual diantimony trioxide particles.

In addition, in a study performed by EBRC on behalf of IAOIA (EBRC, 2005c) the "total dustiness" was provided, indicating the propensity of a material to become airborne. According to the report "total dustiness" may serve as an indicator of the mobility under workplace conditions that may be utilised in selecting suitable analogies to other chemical substances with respect to their dermal loading. The data on dustiness from this study is not used further in this risk assessment report since there is a recent study on dermal exposure to diantimony trioxide available.

Dermal exposure measurements

A total of 35 sets of dermal exposure measurements were collected and distributed as follows: 12 sets of dermal exposure measurements were collected from the converting area, 6 sets of dermal exposure measurements were collected from the refuming area, and 17 sets of dermal exposure measurements were collected from the packing area. The work processes investigated and described in this survey are described below.

There were two **conversion** operators per shift involved with feeding the convertor furnace manually with antimony ingots. Workers in this area were equipped with disposable dust respirators, cotton overalls and wore protective rigger-type gloves for the majority of the working shift. The process depends on the transfer of diantimony trioxide powder from the furnaces and in-line cooling tubes to the storage silos and there were inevitable blockages in the system. Therefore, the operators are required to carry out regular checks on all of these transfer points. There may be a potential for direct contact with antimony metal and diantimony trioxide due to emissions from the various items of packing equipment located in the general vicinity of the convertors, even though the convertor operators are not involved in the packing. There may also be significant surface deposits of diantimony trioxide in the general area, and direct contact with these surfaces is possible.

There was one operator per shift involved in controlling the **refuming** process, although this was largely automatic while in operation. The workers were equipped with disposable dust respirators and thermal protective gloves for work in this area and the worker was observed to wear these for the majority of the shift. The operator is required to monitor the process conditions by checking the readings on the control panel located immediately adjacent to the refuming plant. Loading of the refumer is done via the pipeline system or via "big bags" (1000 kg capacity). There may be a potential for direct contact with the diantimony trioxide dust while manipulating the big-bags during the loading and unloading procedures and during removal of the empty bags.

There were two or three packers per shift depending on production demands. The workers in the packing area (**final handling**) wore disposable dust respirators, cotton overalls and protective rigger-type gloves for the majority of the working shift. Different grades of diantimony trioxide were held in various high-level storage silos and fed to a number of packing stations. The diantimony trioxide could be packed into 25-kg paper sacks using conventional semi-automatic bag packing. Alternatively, the product was packed into 20-kg polyethylene bags, which were heat-sealed at the packing station, or into 1000-kg capacity big-bags. There were visible dust emissions from the packing process and deposits of diantimony trioxide powder could be observed on the outer clothing and exposed areas of skin of all workers engaged in this work. Direct contact with diantimony trioxide is possible when attaching the big-bags to the filling points of the packing stations, when handling and stacking 25-kg paper sacks and also when filling and stacking the polyethylene bags.

The dermal exposure measurement method was repeated wet-wiping of the skin at a number of places considered representative of the skin area. Samples of skin contamination were collected at three different intervals over the working day (before rest breaks and at the end of shift) in order to assess the level of contamination over the working shift. The sampling was done by wet wipes from the back and the palm of each hand and from both forearms. Additional samples were collected from the neck, face and chest of each worker at the end of the shift. The neck and face samples were intended to provide an estimate of exposure for the head. The sample from the chest was intended to assess the degree of contamination under work clothes. A field blank sample was obtained for every subject sampled. This was done in order to check for contamination introduced during the sampling procedure. The field blanks comprised a series of three wipes, which were handled in the same way as the exposed samples but without being wiped over the workers' skin. The mass of antimony in each sample was used to calculate equivalent dermal surface loading of diantimony trioxide for each anatomical area, expressed in terms of mass per unit area ($\mu g/cm^2$/day).

Background levels of antimony on human skin were determined by collecting wipe samples from a number of human volunteers not occupationally exposed to antimony. This was limited to hand and forearm samples only. Out of a total of 30 samples collected all were below the limit of detection for the method, which was calculated as 0.5 $\mu g/cm^2$/day.

The overall average recovery efficiencies were 94, 89 and 95 %, for the low, medium and high spike levels, respectively, and broadly comparable with previous studies using zinc oxide and lead oxide. Thus, it was concluded by the authors that the sampling recovery generally showed an acceptable level of recovery across each of the surface contamination levels tested, and the results for diantimony trioxide were considered similar to those obtained for zinc oxide (Hughson and Cherrie, 2005), nickel powder (Hughson, 2004a) and lead oxide (Hughson, 2004b).

A total of 420 individual dermal samples, excluding field blanks, were collected for analysis of antimony and converted to an equivalent mass of diantimony trioxide by applying a factor of 1.197 in each case. There were 35 complete sets of measurements from 18 different workers. Out of 420 samples 25 were less than the limit of detection (LOD) (1 $\mu g/cm^2$) and for the purpose of statistical analysis these were set to ½ of the LOD, i.e. 0.5 $\mu g/cm^2$. All samples were corrected for blank levels, field blank and for analytical recovery from the sampling media. The summary exposure data were analysed for each anatomical area sampled by job title/task (Table 4-3). Dermal diantimony trioxide exposures for the hands and forearms (approximately 2000 cm^2) combined were within the range 1.8-20 $\mu g/cm^2$/day (median 5.2 and 90th percentile 16 $\mu g/cm^2$/day) for the workers involved in the conversion process. The corresponding measurements for the refuming and

packaging workers were 2.1-25 μg/cm^2/day (median 12 and 90th percentile 22 μg/cm^2/day) and 4.2-68 μg/cm^2/day (median 18 and 90th percentile 31 μg/cm^2/day), respectively.

The dermal exposure for the hand and forearms were strongly correlated, but there was no significant correlation between the hands and the other anatomical areas. However, there was a strong correlation between the dermal exposure for neck and face and for neck and chest.

The results in this kind of study are heavily dependant on whether the workers wear gloves or not and whether the gloves are designed to provide the correct type of protection. It was noted that some of the highest exposure measurements collected were obtained from a worker who did not routinely wear gloves. Furthermore, workers who did wear gloves still recorded relatively high dermal exposure, indicating that the gloves were not highly efficient. It was proposed that it would be appropriate to use the median and the 90th percentile as "typical" and "worst case scenarios" values, respectively.

Table 4-3 Summary of dermal diantimony trioxide exposure by process and anatomical area (Hughson, 2005)

			Dermal Sb_2O_3 exposure (μg/cm²/day)			
Process	**Anatomical area**	**N**	**Min**	**Median**	**P90**	**Max**
Convertor	Average hands	12	2.5	8.0	17	20
	Average forearms	12	0.8	3.2	18	22
	Hand and forearms	12	1.8	5.2	16	20
	Neck	12	<1.0	7.9	23	26
	Face	12	<1.0	8.4	23	24
	Chest	12	<1.0	2.7	7.4	12
Refuming	Average hands	6	1.0	8.8	14	17
	Average forearms	6	2.9	14	30	30
	Hand and forearms	6	2.1	12	22	25
	Neck	6	3.1	7.5	12	12
	Face	6	5.7	8.5	13	14
	Chest	6	1.1	3.9	7.3	9.3
Packing	Average hands	17	7.7	17	39	55
	Average forearms	17	1.0	21	29	81
	Hand and forearms	17	4.2	**18***	**31***	68
	Neck	17	<1.0	8.9	23	29
	Face	17	<1.0	13	22	42
	Chest	17	<1.0	4.7	9.2	11
All tasks	Average hands	35	1.0	12	21	55
	Average forearms	35	0.8	16	28	81
	Hand and forearms	35	1.8	15	25	68
	Neck	35	<1.0	8.7	23	29
	Face	35	<1.0	9.6	22	42
	Chest	35	<1.0	3.9	9.0	12

* Values chosen for calculations of dermal exposure in downstream use scenarios

Evaluation of exposure

Inhalation exposure

Measured data

Measured data on inhalation are available from production sites within Europe. The data provided by IAOIA in 2003 with the database compiled by (Vanderbroele et al., 2003) was limited e.g. on the frequency, duration, contact, and the particle size of the airborne diantimony trioxide. There was also limited data on the total number of exposed employees in the EU and a lack of information on the sex and age of the exposed workers. In addition, not all reported data included information on e.g. methods for sampling and chemical analysis, the duration of measurements or task of workers, date when samples were collected or the type of sampling conducted (personal or area measurements). Therefore, this report is not used for risk assessment purposes, especially considering that more recent and more qualified data exist.

In addition, the Berufsgenossenschaftliches Institut für Arbeitsschutz, BIA, in Germany measured levels of diantimony trioxide in 50 different companies during the period 1989 to 1992. Detailed information on individual values and exposure scenarios was not published. Values from this study have therefore not been included when the scenarios below are described.

The data provided by IAOIA in 2005 (EBRC, 2005a) is more recent and was considered having the best quality and should therefore be used for the exposure assessment.

Since measured data on inhalation were available from all four production sites within EU, and RPE is mandatory at all four sites and a good compliance of this order has been reported from the surveys conducted by EBRC (2005a) and IOM (Hughson, 2005), it was considered reasonable to take use of RPE into account when estimating the exposure during production of diantimony trioxide. Measured exposure data (median and 90th percentile values) were thus also recalculated to reflect the protection offered by the specific type of filter and face piece combination used in each company. It is, however, clear that RPE should not be used as a standard risk management measure but should only be applied where exposure cannot be prevented by other means (Council Directive 98/24/EC, Article 6). It should also be noted that the correction factors for RPE use relate to the likely performance of the whole RPE when worn correctly and used in accordance with the manufacturer's instruction and it can not be excluded that the protection offered by RPE is sometimes less effective or that some workers at some occasions do not use RPE at all. Therefore, two RWCs, with and without the use of RPE, are taken forward to the risk characterization.

Conversion

Conversion covers work tasks like loading of furnace with antimony ingots, supervision of operating conditions and routine inspections and adjustments and may cause risk of exposure.

Based on the above description and Industry exposure data (EBRC, 2005a), the median and 90th percentile inhalation exposures for conversion during production are 0.027 and 0.15 mg/m^3, respectively, when RPE is worn correctly, and 0.54 and 2.9 mg/m^3, respectively, without use of RPE.

Refuming

Refuming (if crude antimony metal feedstock) is done in order to adjust the chemical and physical properties of the product. This is achieved by feeding the material through a furnace, which has a free flow of air into the transfer ducting.

Based on the above description and Industry exposure data (EBRC, 2005a), the median and 90th percentile inhalation exposures during refuming are 0.012 and 0.047 mg/m^3, respectively, when RPE is worn correctly, and 0.23 and 0.94 mg/m^3, respectively, without use of RPE.

Final handling

The final handling of diantimony trioxide covers all kinds of work tasks at places where the final product (diantimony trioxide) is handled, like weighing, packaging etc., and is a common task in the production of diantimony trioxide.

Based on the above description and Industry exposure data (EBRC, 2005a), the median and 90th percentile inhalation exposures for final handling during production are 0.040 and 0.11 mg/m^3, respectively, when RPE is worn correctly, and 0.79 and 2.1 mg/m^3, respectively, without use of RPE.

Modelled data

Modelled data are not presented since measured data are considered sufficient.

Conclusions

Since measured data of sufficient quantity and quality are available for inhalation exposure during production of diantimony trioxide, these data are taken forward to risk characterisation. Median and 90th percentile values, with and without use of RPE, were calculated. The median value after correction for the use of RPE was considered as the typical exposure in the production plants presently in operation in the EU. When estimating reasonable worst-case values for production, two RWC scenarios, with and without the use of RPE, are taken forward to the risk characterisation. It is clear that RPE should not be used as a standard risk management measure but should only be applied where exposure cannot be prevented by other means (Council Directive 98/24/EC, Article 6). It should also be noted that the correction factors for RPE use relate to the likely performance of the whole RPE when worn correctly and used in accordance with the manufacturer's instruction and it can not be excluded that the protection offered by RPE is sometimes less effective or that some workers at some occasions do not use RPE at all.

Dermal exposure

Measured data

From the study by Hughson (2005) it can be concluded that the exposure per surface area is only slightly higher for hands and forearms compared to neck and face, with lower but measurable exposure of the chest. It was considered reasonable to use the values obtained for hands and forearms in the estimation of dermal exposure. The values relate to more or less protected workers and it cannot be assumed that these values are relevant if no PPE is used at all.

Conversion

Based on the above description the typical exposure for this scenario is estimated to 5.2 μg/cm^2/day, while the reasonable worst case is estimated to 16 μg/cm^2/day.

Refuming

Based on the above description the typical exposure for this scenario is estimated to 12 μg/cm^2/day, while the reasonable worst case is estimated to 22 μg/cm^2/day.

Final handling

Based on the above description the typical exposure for this scenario is estimated to 18 μg/cm^2/day, while the reasonable worst case is estimated to 31 μg/cm^2/day.

Modelled data

Modelled data are not presented since measured data are considered sufficient.

Conclusions

Since measured data of sufficient quantity and quality are available for dermal exposure during production of diantimony trioxide, these data are taken forward to risk characterisation. It was observed that the skin area exposed during production was larger than only hands and forearms. Thus, for calculation of the exposure during production a larger area of 3160 cm^2 (1140 cm^2 for forearms + 840 cm^2 for hands + 1180 cm^2 for head (face), all according to TGD) was used. Accordingly, the skin area considered to be relevant for dermal exposure during production is 3160 cm^2.

Values taken forward to risk characterisation

All values taken forward to risk characterisation are based on measured data. The body weight of the worker is 70 kg and the exposed dermal area is 3160 cm^2. For inhalation exposure, two RWCs, with and without use of RPE, are considered.

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Inhalation exposure
Conversion
With RPE: 0.027 (median, typical exposure) and 0.15 (90th percentile, RWC1) mg/m^3
Without RPE: 0.54 (median) and 2.9 (90th percentile, RWC2) mg/m^3

Refuming
With RPE: 0.012 (median, typical exposure) and 0.047 (90th percentile, RWC1) mg/m^3
Without RPE: 0.23 (median) and 0.94 (90th percentile, RWC2) mg/m^3

Final handling
With RPE: 0.040 (median, typical exposure) and 0.11 (90th percentile, RWC1) mg/m^3
Without RPE: 0.79 (median) and 2.1 (90th percentile, RWC2) mg/m^3

Dermal exposure
Conversion
Typical: 5.2 µg/cm^2 which corresponds to 0.23 mg/kg/day (5.2 µg/cm^2 x 3160 cm^2 / 70 kg /1000)
RWC: 16 µg/cm^2 which corresponds to 0.72 mg/kg/day

Refuming
Typical: 12 µg/cm^2 which corresponds to 0.54 mg/kg/day
RWC: 22 µg/cm^2 which corresponds to 0.99 mg/kg/day

Final handling
Typical: 18 µg/cm^2 which corresponds to 0.81 mg/kg/day
RWC: 31 µg/cm^2 which corresponds to 1.4 mg/kg/day

4.1.1.2.3 Scenario 2: Use as catalyst in production of PET

A survey was conducted by EBRC for IAOIA to collect data relevant for the assessment of occupational exposure to diantimony trioxide in this industry sector, see chapter 4.1.1.2.1. The use of diantimony trioxide in production of PET has been divided into two surveys, i.e. PET resins and PET fibres and films. The results from these surveys together with information from other sources have been presented in two reports (EBRC, 2006g; EBRC, 2006h).

Because of the similarities between the two sectors, the assessment of the two reports is presented in the same chapter; separate process descriptions are, however, presented. The reports submitted by Industry have been summarised in the following sections, and the conclusions proposed by Industry are also presented. The exposure information is further evaluated before conclusions on values for risk characterisation are presented.

Summary of Industry information

<u>PET resin</u>

The production capacities of European PET plants may vary between 70.000 and 280.000 tons per year. In the polymerisation process, diantimony trioxide or other antimony compounds are employed in small amounts as a catalyst in the production of PET, yielding final contents of antimony in the range 200-300 ppm in finished PET.

PET resins are produced commercially by two similar processes, using ethylene glycol (EG) and either (i) dimethyl terephthalate or (ii) terephthalic acid. In both cases, the bis-(2-hydroxyethyl)-terephthalate monomer is first produced as an intermediate, yielding either methanol or water as a by-product. The monomer is then polymerised at low pressure under heating and the presence of the antimony catalyst to the PET resin. Either diantimony trioxide or other antimony compounds, such as triacetate, are used. However, in both cases, the antimony compound in question is added to mono ethylene glycol in a reaction vessel, and the reaction to an antimony glycolate complex is conducted at elevated temperatures of approx. 100-150 °C for 2-10 hours for diantimony trioxide, and approx. 40 °C for antimony triacetate, until all antimony has completely dissolved to a clear solution, usually designated as the "catalyst". The ratio of diantimony trioxide to mono ethylene glycol varies only slightly between producers, ranging from 14.0-21.0 kg per ton. As such, this glycolate complex solution in monoethylene glycol no longer contains any free diantimony trioxide, and antimony is finally bonded covalently into the matrix of the PET product (Biros et al., 2002).

Seven companies have been identified by EBRC as PET producers in the EU. Each company was approached with a detailed questionnaire. Based on the information made available through the industry association Plastics Europe in Brussels, there are 16 PET production sites in Europe, which operate a total of 22 production plants. However, it was possible to identify 5 sites, involving 9 plants, which exclusively use other forms of antimony than the trioxide, and these were thus omitted from the evaluation. All companies in Europe that use diantimony trioxide responded with completed questionnaires to EBRC directly, reporting, among others, detailed data on the tonnage of diantimony trioxide used, frequency and duration of exposure, nature of the task involved, amount of material handled, description of handling procedures and control measures. Personal inhalation exposure data for diantimony trioxide were submitted by 7 of these 11 sites, yielding coverage of around 60 % based both on number of sites and diantimony trioxide consumption. Based on an inspection at a production site identified as representative by "Plastics Europe", a generic process description and a process flow chart were derived, which are considered descriptive of the entire sector and were mutually agreed between all participating companies.

PET fibres and films

Diantimony trioxide is used as a catalyst in the production of polyester fibres with a final antimony content of about 300 ppm on average. According to the CIRFS website, about 1.3 million tons of polyester fibres were produced by a total of 40 plants in Europe in 2005. Since other antimony catalysts such as antimony triacetate are also used, the total tonnages of diantimony trioxide used for polyester fibre production should be less than 468 tons, expressed as diantimony trioxide. In addition, diantimony trioxide is also used to a lesser extent (approx. 80 tons per year) in the production of PET films. As a direct follow-up to the PET resin survey, but outside the scope of the separate diantimony trioxide downstream user survey conducted in 2006, a data gathering exercise was initiated among European polyester fibre producers. A site visit was conducted at a plant identified by the German man-made fibres association (IVC e.V.) as being representative and covering all sub-processes with quantitatively relevant diantimony trioxide exposure.

Six major companies responded to the Industry data gathering exercise by submitting a completed questionnaire. The total consumption of five of these companies amounted to 106 tons of diantimony trioxide per year, and assuming an average of 21 tons per year for one company that did

not state the annual consumption for confidentiality reasons, these companies then represent a diantimony trioxide consumption of 127 tons per year, corresponding to a sector coverage of approx. 27 %, assuming 300 ppm antimony in the final product and only diantimony trioxide as catalyst used.

Used forms of diantimony trioxide: According to the feedback from the responding diantimony trioxide users, the following forms are used in PET fibre production: Wetted (WET): diantimony trioxide in powder form, to which a minor portion of liquid has been added; in this case mono ethylene glycol usually at 2 – 5 % (w/w) to reduce dustiness. Masterbatch (MB): diantimony trioxide in granulated form; tightly enclosed in a polymer matrix (polymer content of 10-90 %). Powder (POW): dry powder (without any additives), used only to a minor extent. The overwhelming majority (99 %) of diantimony trioxide consumption in this sector is in the form of wetted powder, whereas masterbatches (1 %) and dry diantimony trioxide powder (< 0.001 %) are consumed only in minor quantities.

Process descriptions

PET Resin - Initial powder handling

The first stage of the process is inevitably related to unloading and emptying of diantimony trioxide powder into reaction vessels. Depending on the technical layout of a particular plant, diantimony trioxide may be handled in several different ways:

(i) Dry powder (used for pneumatic feeding systems):

Diantimony trioxide in dry powder form is received in 800-kg closed stainless steel containers. The containers are positioned above the mixing vessel and connected to form a closed system, and the diantimony trioxide is discharged into the vessel though a cone valve under control of airflow.

These containers are changed once or twice a week, and the change-over takes approx. 20 minutes.

(ii) Wetted powder:

Most sites acquire diantimony trioxide as a "wetted", most commonly with 3 % monoethylene glycol, powder with a particle size range of approx. 0.4 – 1 µm. The wetted diantimony trioxide is obtained mostly in either 12-21-kg re-usable drums or in 10-25-kg plastic bags, but in one case also in 300-kg big bags. Between plants using wetted diantimony trioxide, there is some variety of handling systems. The following modifications exist: - a bucket is placed into a drum-like closed discharge unit, to which monoethylene glycol is fed through a flexible dispensing system. By this, the diantimony trioxide is swirled into the sealed reaction vessel, which has a ventilated air in/outlet levelling system, so that no escape of any material is possible. According to the Industry survey, workers mandatorily wear quarter masks and disposable (latex) gloves during this procedure. A maximum of six 21-kg drums and a total amount of approx. 120 kg are handled in this way in a single shift, once per day, seven days a week. The duration of this loading/feeding operation is approx. 30 minutes.

Buckets or bags may also be unloaded into a hopper or a similar feeding device under LEV from where the diantimony trioxide is discharged directly into the reaction vessel, or they are transferred into a sealed hood with gloves attached ("glove box") which is directly connected via a funnel to the ventilated reaction vessel.

Big bags are directly connected to the feedings system of the reaction vessel, with a dust collector fitted to the inlet to remove any dust formation.

In all cases, only 1-2 employees per shift will be involved in the handling of diantimony trioxide in the way described above.

The duration of the feeding operation varies between 5 minutes with five batches per day being prepared to 60 minutes with one batch per week, depending on the modification involved. Most plants produce continuously over the year, i.e. close to 365 production days per year. Overall, it is reasonable to say that on a daily basis, the cumulative duration of the diantimony trioxide handling procedure will under no circumstance exceed 60 minutes.

PET Resin - Further processing -polycondensation and extrusion

The catalyst solution generated in the first process step as described above is first reacted, within a completely closed system, with monomeric terephthalic acid and monoethylene glycol at temperatures in the range 200-280 °C and the esterification to increasing chain lengths is promoted by removing water and ethylene glycol by distillation under vacuum. The resulting molten polyester is then cooled under running water into strands, which are subsequently cut immersed under cooling water into small (3-4 mm diameter) chips. These are subjected to drying, automatic sieving and intermediate storage in silos, from where they are fed pneumatically directly to approx. 20-ton silo trucks for delivery to customers.

For some applications, a higher degree of polymerisation may be required, such as for PET bottles. For such purposes, the "chips" mentioned above are subjected to the so-called "SSP" (solid state polymerisation) process. For this, they are transferred pneumatically to a closed system in which they are heated up to temperatures around 160-210 °C under an inert atmosphere. This process initially involves crystallisation of the PET, followed by further removal of water and ethylene glycol, by which the continued esterification increases the chain length of the polymer. After reaction to the glycolate, the reaction mix is maintained in a closed system, to which only three potential "outlets" exist:

(i) During vacuum distillation of ethylene glycol in the polycondensation process, some small amounts of droplets containing minor amounts of oligomeric reaction products, containing also the antimony glycolate complex bound in the oligomer, may be physically removed from the reactor, which finally end up in the "sump" of the recycling distil. However, these are present in liquid form and are discharged discontinuously into steel drums, where they solidify and are shipped to incineration. No manual handling is required at this stage.

(ii) The continuous chipping under water generates a small amount of fine material. However, since the cooling water is re-circulated, these fines are collected on a continuous fibre filtering system, where the fines are retained on the fabric and are discharged into collection bins together with the moist filter material for disposal.

(iii) During conveying of the chips, a small portion of fine dust may be released which is collected in air filter systems and which are discharged into collection bins approx. once per shift, involving only a few kilograms of material representing fully polymerised PET with low amounts of covalently bound antimony.

According to Industry, the only potential for exposure to diantimony trioxide is in the area in which it is added to the glycolate reaction vessel. Further downstream in the process, any antimony is

covalently bound to either oligomeric material or within the final PET product. Further, the handling of antimony triacetate as input material, of antimony glycolate as reaction intermediate and of any further downstream minor waste products is not considered relevant by Industry and has not been included in the assessment, since antimony is no longer present as the trioxide.

PET Resin - Final processing

For the same reasons as given above, dust-free PET resin chips are at this stage merely subjected to thermal reforming processes such as moulding. Since the PET resin chips are transported in bulk containers and are loaded automatically into process shoppers, direct handling is not involved. Further, since antimony is no longer present as diantimony trioxide, the exposure to diantimony trioxide may be regarded as negligible.

PET Fibre - Initial powder handling

Diantimony trioxide as wetted powder (5 % mono ethylene glycol) is obtained in 200-kg plastic drums. These drums are wheeled to a specially designed loading dock where the drum is firmly locked and the opening of the drum is covered with a tight-fitting seal connected to a flexible funnel. Empty drums are returned to the provider for re-filling. When the unloading system is sealed, the entire system is closed. The drum is then inverted, so that the entire content drops into a reaction vessel containing approx. 6 m^3 of mono ethylene glycol. The duration of this task is less than 30 minutes and is performed every three to four days, thus no more than once per shift. This reaction vessel is stirred and maintained under light over-pressure under a nitrogen atmosphere. Diantimony trioxide is reacted to the glycolate over a period of 3-4 hours by gradual steam heating the vessel up to 170 °C. The reaction is complete when the diantimony trioxide is completely dissolved, after which it is cooled to approx. 50 °C and transferred via closed pipe systems to an interim storage tank. From there, the glycolate catalyst solution is continuously fed to the PET polymerisation plant. The continuous polymerisation to PET can be summarised briefly as follows:

- In the initial transesterification, dimethyl terephthalate and mono ethylene glycol are used as starting materials and first transformed to the ethylene glycol ester under formation of methanol.

- Alternatively, the reaction may also be conducted using monomeric terephthalic acid and monoethylene glycol.

- In the subsequent pre- and poly-condensation stages, the monomeric ethylene glycol ester is condensated to the polymer, whereby ethylene glycol is released; the reaction is facilitated by vacuum (removal of ethylene glycol) and temperature (300 °C).

Once the diantimony trioxide has been placed in the reaction vessel and converted to the glycolate, dermal and inhalation exposure is considered precluded by Industry because the entire process is conducted at elevated temperatures in a closed system.

PET fibre - Further processing (spinning process)

In the subsequent melt spinning process, the resulting liquid/molten polymer is then fed with the aid of metering pumps though “spinerets”, which represent disc-like steel plates with several hundred to several thousand minute nozzles. From these, individual thin threads emerge, which pass over a roll wetted with water to avoid adhesion, after which they are bundled to strands (spinfinish) and layered into large steel spin cans. The entire process is highly automated, requiring manual intervention only for maintenance purposes.

PET fibre - Final product manufacturing

From there, the bundled threads are drawn to "stretch" the fibre, followed by crimping (optional) and a drying/relaxation process. Subsequently the fibres are transferred to a cut box. From this, they are continuously drawn over a cutter wheel and cut into lengths according to their final destination, i.e.

- 10 mm length (non-woven, fleece materials),

- 30 mm length (apparel, textile clothing),

- 70 mm length (fibre-fill, padding, quilts, furniture).

Finally, the resulting fibres are formed into bales for final shipping to their destination, and are subject to the mechanical manufacturing processes in subsequent industries, which do not involve further chemical or physical transformation. Again, the entire process is highly automated, requiring manual intervention only for maintenance purposes.

Personal protection equipment

Workers require only gloves and normal overalls during handling of diantimony trioxide drums, since exposure is minimised by the use of a closed loading system. Protective clothing and RPE (half mask, filer P2) are mandatory during routine cleaning/maintenance operations occurring only in intervals of several months.

Throughout the subsequent process, no particular protection is required, since the esterification and polycondensation processes are run in completely closed systems, and the subsequent spinning process starts with a liquid /molten polymer, from which the polymeric fibres are continuously woven, processed and cut without the need for manual contact. Due to the nature of the individual processes and the handled product, it is considered by Industry that there is no relevant inhalation exposure to process dusts.

PET film production

In view of the similarity of process descriptions between the PET film and the PET fibre industry and the basic requirements of antimony catalyst preparation, a separate assessment was not considered to be required.

Inhalation exposure

Data on the frequency and duration of tasks involving exposure to diantimony trioxide have been collected in a questionnaire and the results are summarised in ranges as follows:

Unloading: 1 time per shift with duration of 5 minutes up to 0.5 times per month for 240 minutes.

Emptying: 3 times per shift, 30 minutes to 8 times per week, 20 minutes.

Supervision, maintenance: 2 times per shift, 60 minutes to 2 times per year, 240 minutes.

The entire database contained in October 2005 a total of 79 measured data from seven EU PET production sites. From these, all data points remained for subsequent statistical analysis after quality

screening performed by Industry. Further, only personal sampling data were taken into account. The number of these samples was 60. The static monitoring data (n=19) were evaluated to support the assumption of negligible exposure in periods in which no direct handling of diantimony trioxide occurs, and the available data are according to Industry supportive of this assumption. The number of personal samples was between 2 and 18 per site from the 7 sites that had submitted measured data.

According to Industry the data reported and analysed represent data that may be considered as short term personal exposure reflective of the exposure occurring for the duration of the handling of diantimony trioxide at the workplace described. All data represent the results of sampling for the "total inhalable" fraction.

The production of PET resin involves three different process stages: (i) initial handling of diantimony trioxide powder, (ii) further processing via antimony glycolate to PET resin "chips" via extrusion, and (iii) final processing of PET resin to containers such as bottles and films. For these three process steps, the exposure has been assessed according to the relevant routes of exposure, i.e. inhalation and dermal contact.

Initial powder handling

These data are analysed both as short term exposure and as 8-hour TWA reflective of an equivalent full shift exposure.

Table4-4 Inhalation exposure data (personal and static) according to workplace category [mg Sb_2O_3/m^3]

	St / P	Counts	Min	Median	P90	Max
Measured values	p	60	<0.001	0.042	0.16	2.1
8-hour TWA	p	60	<0.001	0.002	0.026	0.062
Only short-term values (t ≤ 65 min)	p	54	<0.001	0.048	0.16	2.1
Static monitoring results (t > 120 min)	st	19	0.001	0.002	0.024	0.074

p: personal sampling; st: static sampling (presented as supportive information)

The personal exposure data (based on TWA transformed values) can be summarised as follows:

- On an overall basis, typical personal inhalation exposure levels for diantimony trioxide in the entire sector according to Industry calculations are 0.002 mg Sb_2O_3/m^3.

- The overall RWC is 0.026 mg Sb_2O_3/m^3.

In addition, the static monitoring results performed for extended periods (t > 120 min) show that inhalation exposure in the production area during periods that extend well beyond the limited duration of diantimony trioxide handling are at the same level 0.002 mg Sb_2O_3/m^3.

Based on detailed feedback on frequency/duration of handling as well as other exposure modifiers and a set of sector-specific measured data, it is proposed by Industry to take forward the following 8-hour TWA values to risk characterisation for the handling of dry diantimony trioxide powder.

RWC: 0.026 mg Sb_2O_3/m^3

Typical: 0.002 mg Sb_2O_3/m^3

Processing

It is proposed by Industry to consider the inhalation exposure at this stage as negligible because no dust is formed during the extrusion process and the subsequent further treatment leading to PET resin chips. In addition, diantimony trioxide has been chemically transformed into antimony glycolate.

Final product manufacturing

Similar to above, the processes at this stage involve mere thermal reforming such as moulding. Inhalation exposure at this stage is considered negligible by Industry because no dust is formed, and diantimony trioxide has been chemically transformed into antimony glycolate.

Dermal exposure

Powder handling

The powder handling of diantimony trioxide involves unloading and emptying. The total duration of the unloading process can range from 0 to 240 minutes, whereas the total duration of the emptying process ranges from 0 to 90 minutes. As a result, the duration of the entire diantimony trioxide powder handling is typically 40 minutes (median) and as RWC 90 minutes (90^{th} percentile), as calculated by Industry.

Measured data on dermal exposure to diantimony trioxide during powder handling in the PET resin industry do not exist. However, an extrapolation from data measured during bag filling at a diantimony trioxide production (Hughson, 2005) site is considered feasible by Industry due to similar handling patterns. In this sector, occupational dermal exposure to Sb_2O_3 during production and subsequent handling was measured with the aid of the wipe sampling method. Workers were involved either in packaging or in other activities for a full shift.

Considering that diantimony trioxide powder handling in the PET resin sector does not occur full-shift and that for the remainder of a shift workers will perform tasks in other areas of the plant not involving diantimony trioxide exposure, it is proposed by Industry as a worst-case to carry forward the highest levels of dermal exposure measured for diantimony trioxide packaging, but modified by the duration of exposure, i.e. max. 90 minutes according to Industry calculations. Thus, the values for dermal exposure proposed by Industry are:

RWC: 7 µg Sb_2O_3/cm^2, and typical case: 3 µg Sb_2O_3/cm^2.

Processing

The dermal exposure at this stage is considered negligible by Industry because no dust is formed during the extrusion process and the subsequent further treatment leading to PET resin chips. In addition, diantimony trioxide has been chemically transformed into antimony glycolate.

Final product manufacturing

Similar to above, the processes at this stage involve mere thermal reforming such as moulding. Dermal exposure at this stage is considered negligible by Industry because no dust is formed, and diantimony trioxide has been chemically transformed into antimony glycolate. Thus, it is proposed by Industry to consider dermal exposure at this level as negligible.

In view of the close similarity of relevant process conditions between the three sectors, data from the PET resin sector are also considered representative of fibre and film production.

Evaluation of exposure

Inhalation exposure

Measured data

The data set collected by Industry contains 60 personal sampling measurements from powder handling in seven PET production sites in the EU. These data have been used to calculate 8-hour TWA exposure values and the median (typical) and 90th percentile (RWC) exposures are 0.002 mg/m^3 and 0.026 mg/m^3, respectively. The data set is considered robust enough to reflect powder handling in the EU PET industry.

No measured data exist for subsequent steps, i.e. processing and final product manufacturing, in this industry sector. However, it is proposed by Industry to consider exposure to diantimony trioxide in these steps as negligible. This assumption is considered justified by information given in the process description; i) diantimony trioxide is covalently bound to the polymer matrix, i.e. no longer present as diantimony trioxide, ii) dust formation is very limited since the material is in liquid form or in the form of chips, iii) the process is highly automated and closed systems and control measures are frequently used.

Modelled data

No modelling has been performed since measured data are considered sufficient.

Conclusions

As proposed by Industry the measured data (8-hour TWA values) for powder handling are taken forward to risk characterisation. The exposure during the subsequent stages of the process (processing and final product manufacturing) is considered negligible.

Dermal exposure

Measured data

Measured data on dermal exposure do not exist.

Analogous/surrogate data

As there are no measured data it is instead proposed by Industry to extrapolate from data measured during bag filling at a diantimony trioxide production plant (2005); a summary of these data is given in Table 4-3. This is considered justified as both scenarios involve handling of diantimony trioxide powder and the handling patterns are similar. It is further proposed by Industry to use the 90th percentile of the duration values reported, i.e. 90 minutes, which is considered justified. Data on duration of handling were provided from 50 % of the sites with a range of 15-255 min (median 40 min). The 90th percentile value is used for recalculation of the dermal exposure values from production of diantimony trioxide. Accordingly, this gives the following values for dermal exposure:

Typical: 0.018 mg/cm^2/day * (90/480) = 0.0034 mg/cm^2/day

RWC: 0.031 mg/cm^2/day * (90/480) = 0.0058 mg/cm^2/day

Modelled data

No modelling has been performed since analogous measured data from production of diantimony trioxide are considered sufficient for read-across.

Conclusions

For dermal exposure analogous data from final handling during production of diantimony trioxide modified for duration (90 minutes) are taken forward to risk characterisation. The exposure during the subsequent stages of the process (processing and final product manufacturing) is considered negligible.

Values taken forward to risk characterisation

The bodyweight of the worker is 70 kg and the exposed dermal area is 2000 cm^2.

Inhalation exposure
Powder handling
Typical: 0.002 mg/m^3
RWC: 0.026 mg/m^3

Dermal exposure
Powder handling
Typical: 0.0034 mg/cm^2 which corresponds to 0.10 mg/kg/day
RWC: 0.0058 mg/cm^2 which corresponds 0.17 mg/kg/day

4.1.1.2.4 Scenario 3: Use as flame-retardant in production of plastics

A survey was conducted by EBRC for IAOIA to collect data relevant for the assessment of occupational exposure to diantimony trioxide in this industry sector, see chapter 4.1.1.2.1. The survey on the use of diantimony trioxide as flame-retardant in plastics was divided by Industry into two distinct sectors, i.e. PVC and non-PVC plastics. The results from these surveys together with information from other sources have been presented in two reports. (EBRC, 2006d; EBRC, 2006c).

Because of the similarities between the two sectors, the assessment of the two reports is presented in the same chapter; separate process descriptions are, however, presented. The reports submitted by Industry have been summarised in the following sections and the conclusions proposed by Industry are also presented. The exposure information is further evaluated before conclusions on values for risk characterisation are presented.

Summary of Industry information

PVC
Diantimony trioxide is used as a flame-retardant at final concentrations of 1-4 % in a wide range of PVC products, e.g. wires, cables, sheets, foils, textiles and artificial leather.

Based on the client data base a total of 200 companies were identified by Industry in EU15 (EBRC, 2006d). Of these, 17 companies consumed more than 125 tons diantimony trioxide per year. 48 companies, representing 56 % of the annual consumption in this sector, replied to the questionnaire.

In this industry sector diantimony trioxide is mainly used as dry powder. Low-dusting forms of diantimony trioxide, e.g. masterbatch, are more frequently used in small-scale operations.

Non-PVC
Diantimony trioxide is used as a synergistic flame-retardant in the non-PVC plastics sector, for example in the production of flame-retardant or ignition resistant engineering polymers, natural high impact polystyrene (HIPS), acrylonitrile-butadiene-styrene copolymer (ABS), and polyethylene products.

In total 224 companies using diantimony trioxide for this end-use were identified in EU. Of these 47 companies (21 %) responded by returning a questionnaire in the survey, corresponding to 68 % of annual diantimony trioxide consumption in this sector. The majority of companies responding to the questionnaire identified themselves as producers or intermediate processors in the non-PVC plastics sector. However, it should be noted that it cannot be ruled out that some larger formulating sites may also serve as a supplier to other sectors than the non-PVC plastics, e.g. delivering dispersions to the flame-retardant textile sector.

In this industry sector, diantimony trioxide is consumed almost exclusively either as dry powder or as granulated master-batches with a polymer content of 10 to 90 %; in addition a few companies also use ‘wetted’ powder. Sub-processes identified include formulation of ready to use master-batches; processing into polymer-specific ready to use granules by extrusion technology; and manufacture of finished products. The way these sub-processes are combined at what site varies from case-to-case. A processor may for example buy a master-batch with diantimony trioxide and compound them with other additives, which is then used at another site moulding parts; or, as another example, a manufacturer of finished products may buy diantimony trioxide powder and

compound it into a master-batch on site before using it together with polymer (and other additives) to mould a finished product.

Process description

PVC

Based on the process descriptions received in the feedback to the questionnaire survey the production processes can be described by three common process stages:

- Raw material handling during formulation
- Processing
- Final product manufacturing

In the initial stages of PVC production, handling of diantimony trioxide powder can occur at a variety of loading operations. Handling can occur for example during direct addition of powders to mixers or extruders, or during the production of masterbatches.

PVC - Raw material handling during formulation

Masterbatch producers obtain diantimony trioxide exclusively as dry powder either in 25-kg sacks or in big bags. In the initial loading process, a wide range of exposure controls are reported, such as discharge of 25-kg bags into loading funnels or closed boxes with extraction in both cases, or big bag unloading stations, from where the material is directly transferred to storage hoppers/silos. During handling of diantimony trioxide, the majority of sites report the use of RPE and gloves.

Diantimony trioxide may be formulated either into “dry blends” or into granular masterbatches:

- For dry blends, diantimony trioxide is transferred with other components into specialised, closed mixing apparatuses (turbomixers, Banbury mixers), where it is mixed batch-wise for approximately 15 minutes. Because of the friction during the agitation, a temperature of about 90-120 ˚C can be reached in the mixer. When this temperature is reached, the compound is unloaded and subsequently dry blended in a powder cooler. Dry blend products do not require compounding, but do need to be screened to remove any oversize material before packing. Transfer of dry blends to the screeners is by means of Spiro flow evaluators. Thereafter, transfer is either in-house for finished products or to packaging units for sale as dry blend material.
- For masterbatches, the diantimony trioxide is first weighed and then added to a dry mixing apparatus where it is mixed with the other agents and PVC, from where it is downloaded to an extruder, where it is molten in a closed system and subsequently granulated, cooled and packed.

PVC - Processing

As far as direct handling of diantimony trioxide is concerned, this can be described as above under masterbatch production. However, once the powder is loaded into hoppers or silos, the transfer to the processing units is via exposure-minimising systems. Alternatively, other forms of diantimony trioxide (wetted powders, masterbatch or pre-weighed packages) are used, which similarly minimise exposure to dust.

Extrusion: Previously generated "dry blends" are discharged to interim storage hoppers/silos, from which they are transferred without the need for manual handling to extruders, in which the finished PVC product is homogenously molten and extruded either into semi-finished shapes or into PVC granules, which are processed into the final product.

Dry blended pre-mixes are loaded together with any other components and PVC into the mixing zone of the extruder. Any material leaves the extruder as a molten stream, from which it is chipped and then passed through an air-cooling system.

Gelled products require the pre-mix to be compounded by feeding the pre-mix into a continuous extruder that melts (120 ˚C), shears and transports the pre-mix to produce a gelled material, which is chopped to produce pellets of the PVC compound.

Calendering: Here, either "dry blends" or the solid raw materials are mixed batch-wise with plasticisers and PVC, after which the resulting liquid paste mix is discharged into a filter system, after which it is stored prior to further processing. This plastisol PVC may then be coated onto textile fabric by "gelling" at 180 ˚C or may be processed into foil or sheet material or coated onto a textile carrier.

In the calender, the elements of a formula are blended, heated and gelled by passing between several successive cylinders, which will finally produce flat sheet-like material.

PVC - Final product manufacturing

PVC is used in a wide range of products, and accordingly, the range of finished products is diverse. Nevertheless, the manufacturing of finished PVC products will require predominantly thermal reforming such as moulding, thermal joining and film blowing.

Since diantimony trioxide is homogenously contained at a low content (1-4 %) in a solid and robust product matrix, low releases can be assumed.

Non-PVC

Based on a site visit and the feedback to the questionnaire survey, the following scenario descriptions of this industry sector were generated. Not all scenarios are relevant for a specific site.

i) Unloading diantimony trioxide powder.

ii) Production of dispersed diantimony trioxide intermediate formulations.

iii) Production of granulated masterbatch materials by extrusion.

iv) Intermediate processing.

v) Production of finished products (non-PVC plastics).

Non-PVC - Unloading

Diantimony trioxide is predominantly delivered to formulating plants in large transport containers, i.e. large metal bins, 1000-kg intermediate bulk containers or big bags. Only a minority of

companies reported usage of dry or wetted powder in 25-kg bags. Specially designed docking stations are used to unload big bags and larger containers. These are designed to limit the exposure, for example by opening the big bag manually through a closed glove box. Transportation is then via pneumatic or screw transportation systems. Unloading large volumes from big bags may be performed from once to eight times per shift; when smaller packaging is employed as many as 20 handlings per shift may occur.

All sites reported LEV, hoods and other extraction devices being in place and/or the use of closed systems when handling diantimony trioxide powder. Most sites reported use of RPE either on a mandatory or voluntary basis.

Non-PVC - Production of dispersed diantimony trioxide intermediate formulations

Diantimony trioxide in dry or wetted form is dosed batch-wise from enclosed emptying stations or intermediate storage silos via closed screw transportation systems directly to a mixing vessel containing the (usually) aqueous dispersion of synthetic resins and other process additives. From this stirring vessel, the dispersion is pumped via a closed pipe system through a sieve to a bead mill to generate a homogenous dispersion. After milling, the product is finally pumped via a closed pipe system into 120-litre transport barrels. The diantimony trioxide content of the final product is approximately 50 %-80 %.

Exposure is likely to be primarily intermittent exposure during short intervals, for example during installation of the filling- or extraction nuzzles.

Non-PVC - Production of granulated master-batch materials by extrusion

Diantimony trioxide in the form of master-batch material, in a granulated form, is produced by an extrusion process. The master-batches are intermediates for use in further downstream processing. Diantimony trioxide and other additives are dosed into the pre-molten polymer, at a temperature typically 180 to 200 °C in the mixing zone of the extruder unit. The homogeneously molten material is then extruded in a thin stream and either first cooled in a water bath and then cut; or cut so that a droplet is formed, which solidifies upon dropping into a water bath; the droplets are then separated via centrifugation. In both cases the material is then dried in warm air and the water is re-circulated. Finally the material is packed in big bags. Potential exposure to diantimony trioxide is likely to be restricted to three points of this process: loading; at exit point from extruder; and during packing as part of polymer dust.

The friability of granulated master-batch material has been tested by mechanically agitating a representative polyethylene master-batch (< 30 % diantimony trioxide) for 96 hours. Only 0.1 % of the particles generated were less than 8 μm and less than 0.001 % was less than 0.2 μm.

Non-PVC - Intermediate processing

According to the process descriptions collected from the companies responding to the questionnaire, the majority of intermediate processors use non-dusting granular masterbatch as input materials. Thus, in practice, these materials will be unloaded from big bags or boxes either in special unloading devices or by pneumatic feeding, in both cases transferring the material into closed storage silos. Due to the low potential of dust formation upon mechanical agitation under these conditions, the potential for dermal and inhalation exposure is considered negligible by Industry and the conduct of a separate exposure assessment is not considered to be required.

Non-PVC - Production of finished products

Based on survey data, the following finished product types requiring flame retardant treatment and involving addition of diantimony trioxide (at levels ranging from 4-7 %) were identified:

- plastic foils or sheets,
- high voltage electric equipment/casings,
- glass fibre enhanced products,
- bitumen elastomers,
- wire and cable sheathing, heat-shrinking tubing.

The vast majority of responders to the questionnaire are involved in thermal processing of intermediate polymer granules to finished plastic products by extrusion or injection moulding. In this, pre-processed nondusting granular intermediates are used as input materials.

Inhalation exposure

Raw material handling during formulation

Measured data

Sector-specific inhalation exposure data for the handling of powder during the production of flame-retarded non-PVC plastics are available from five different companies handling “dry” powder and one company handling wetted powder, as summarised in Table 4-5 below (the individual raw data are available in (EBRC, 2006c).

Table 4-5 Summary of inhalation exposure levels for unloading of powdery diantimony trioxide in the non-PVC industry sector [mg Sb_2O_3/m^3]

Diantimony trioxide form	Measurement	Counts	Median	P90	Maximum
POW	Personal	28	0.24	0.61	4.2
POW	Static	2	0.013	0.019	0.020

It is proposed by Industry to take into account the 90th percentile and the median (150 and 30 minutes respectively) of the handling duration of diantimony trioxide, as stated from the responders to the questionnaire, to calculate the full-shift inhalation exposure values. The results are presented in Table 4-6 below:

Table 4-6 Inhalation exposure levels, non-PVC sector, unloading of powdery diantimony trioxide [mg Sb_2O_3/m^3]

Diantimony trioxide form	Measurement	Counts	Median	P90	Maximum
POW	Personal	28	0.073	0.19	1.3
POW	Static	2	0.004	0.006	0.006

A limited set of measured data, originating from 5 companies altogether, was also collected from the PVC-sector (EBRC, 2006d). The number of personal samples considered by Industry to be of sufficient quality is five. The database is considered too limited for a representative assessment, and in view of the similarities in powder handling and subsequent processes, it is proposed by Industry to use data from the non-PVC sector. The worst case (90^{th} percentile) handling duration proposed by Industry is 186 minutes in the PVC-sector and the following results can be derived accordingly.

Table 4-7 Inhalation exposure levels, PVC sector, unloading of powdery diantimony trioxide [mg Sb_2O_3/m^3]

Diantimony trioxide form	Measurement	Counts	Median	P90	Maximum
POW	Personal	28	0.091	0.23	1.6
POW	Static	2	0.005	0.007	0.008

Processing

Any exposure related to direct handling of diantimony trioxide powder in loading and mixing operations is covered by the scenario "raw material handling during formulation".

Measured data

Measured PVC data on inhalation exposure during processing are available from one company only and the representativity may therefore be questioned. Measured data from a single measurement is also available from processing in the non-PVC sector (EBRC, 2006d; EBRC, 2006c).

Analogous data

In consideration of the lack of scenario-specific data, it is proposed by Industry to adopt an alternative analogous approach for both PVC and non-PVC plastics, as follows: the process characteristics of this type of production are continuous, starting with automated dosing of masterbatch to the extruder, after which the process medium is contained in a closed system; after the extrusion process, the material is chipped under cooling with water, centrifuged (optionally) and dried to finally generate polymer chips with a low, homogeneously bound diantimony trioxide content, with negligible friability.

According to Industry the process conditions are very similar to the PET industry (EBRC, 2005b), which is why it is proposed to read-across from the (full-shift) 8-hour TWA values derived there, i.e. 0.002 mg diantimony trioxide /m^2 (typical) and 0.026 mg diantimony trioxide /m^3 (worst-case) exposure, respectively.

Final product manufacturing

For this process stage there are no specific exposure data available. In view of the lack of data, and in consideration of the arguments presented for processing above, it is proposed by Industry to take forward the (full-shift) 8-hour TWA values from the PET-industry, i.e. 0.002 mg diantimony trioxide /m² (typical) and 0.026 mg diantimony trioxide/m³ (worst-case) exposure, respectively.

Dermal exposure

Raw material handling during formulation

Measured data on dermal exposure to diantimony trioxide at the formulation stage in the flame retardant PVC industry do not exist. However, an extrapolation from data measured during bag filling at a diantimony trioxide production site (Hughson, 2005) is considered feasible by Industry. In this, occupational dermal exposure to diantimony trioxide during production and subsequent handling was measured with the aid of the wipe sampling method.

Considering that diantimony trioxide handling in this sector does not occur full-shift and that for the remainder of a shift workers will perform tasks in other areas of the plant not involving diantimony trioxide exposure, it is proposed by Industry as a reasonable worst case to carry forward the 90th percentile value (indicated as 35 µg/cm² in the Industry report (EBRC, 2006f) and differing slightly from the value indicated in Table 4-3, which is taken from the original report by Hughson (2005). It is further proposed to use as typical exposure the median (indicated as 16 µg/cm²) level of dermal exposures measured for diantimony trioxide during packaging, but modified by the 90th percentile of duration of handling (i.e. max 186 minutes for PVC and 150 minutes for non-PVC) during a shift. Thus, the values for dermal exposure under these conditions as proposed by Industry are:

PVC: RWC: 14µg/cm^2 (186 min/480 min x 35 = 14); typical: 6 µg/cm^2
Non-PVC: RWC: 11µg/cm^2; typical: 5 µg/cm^2

Processing

Any dermal exposure related to direct handling of diantimony trioxide powder in loading and mixing operations is covered by the scenario raw material handling during formulation above.

According to Industry any diantimony trioxide present during further processing is firmly contained in the liquid/molten matrix with no potential for dust formation. For lack of direct handling and because no dust is formed that could lead to contamination of surfaces due to any deposition, it is proposed by Industry to consider dermal exposure at this stage as negligible.

Final product manufacturing

No measured data specific for this sector on diantimony trioxide exposures exist. However, a detailed report on occupational exposure during handling of flame retandant treated textiles is available (Niven, 1993). It may be assumed that the release of diantimony trioxide from diantimony trioxide-treated (backcoated) textiles occurs at a rate similar or higher than from flame retandant-treated (homogenous, rigid) PVC; see EBRC (2006c, 2006d) for details. In conclusion, the following calculations were performed by Industry:

A typical (50th percentile) dermal exposure of 0.08 µg diantimony trioxide /cm² was calculated, with a corresponding reasonable worst-case (90th percentile) of 0.35 µg diantimony trioxide /cm².

Evaluation of exposure

The three scenarios, (i) raw material handling during formulation, ii) processing and iii) final product manufacturing, have been assessed. The scenario raw material handling has the highest exposure levels for both inhalation and dermal exposure. There are no measured data for the subsequent scenarios, but based on the process descriptions the exposure at these scenarios is assessed to be very low. Therefore the scenario "raw material handling during formulation" will be considered as representative of a realistic worst case for this industry sector and will be taken forward to risk characterisation.

Inhalation exposure

Measured data

It is proposed by Industry to use measured data for the handling of diantimony trioxide powder in the non-PVC plastics sector. Data are available from five companies producing PVC plastics and from five companies producing non-PVC plastics. The measured data provided by Industry are considered representative of these industry sectors. The calculations performed by Industry to obtain RWC and typical exposure values are, however, not considered correct as the measured values were wrongly corrected for duration of handling.

Instead 8-hr TWAs were calculated assuming that no exposure occurs outside the duration of sampling. The assumption is clearly justified in this case since the majority of the samples were collected during 480 minutes. In view of the similarities between the two plastics industry sectors, all measured values collected by personal sampling during powder handling in both PVC and non-PVC sectors are included in the assessment.

The sampling duration for each measured value has been used to calculate a corresponding 8-hr TWA. The 90th percentile of these recalculated values is taken forward as the RWC exposure value. The corresponding approach using the median value is used for typical exposure (see Table 4-8). Accordingly, the typical and RWC exposure values taken forward to risk characterisation are 0.13 mg/m^3 and 0.57 mg/m^3, respectively.

$$measured\_value * \frac{sampling\_duration}{480_{\min}} = corresponding\_8hrTWA$$

$$P90\_8hrTWA \Rightarrow RWC$$

Table 4-8 Measured inhalation exposure, PVC and non-PVC (mg/m³)

ID	Duration of sampling (min)	Value (mg/m³)	TWA (mg/m³)
21	480	0.08	0.08
21	480	0.17	0.17
21	480	0.43	0.43
21	480	0.12	0.12
21	480	0.126	0.126
21	480	0.509	0.509
21	480	0.412	0.412
21	480	0.343	0.343
21	480	0.265	0.265
21	480	0.584	0.584
21	480	0.653	0.653
21	480	0.549	0.549
21	480	0.241	0.241
21	480	0.119	0.119
21	480	0.36	0.36
60	480	0.365	0.365
60	480	0.088	0.088
60	15	3.84	0.12
60	480	0.005	0.005
60	480	0.011	0.011
60	480	0.361	0.361
60	480	0.035	0.035
60	11	1.061	0.024315
60	480	0.185	0.185
60	480	0.036	0.036
77	173	4.24	1.528167
77	218	1.04	0.472333
115	165	0.072	0.02475
115	191	0.228	0.090725
115	206	0.012	0.00515
8	122	0.47	0.119458
8	34	0.64	0.045333
54	480	0.067	0.067
54	480	2.376	2.376
105	240	0.013	0.0065
		P90	**0.57**
		Median	**0.13**

Modelled data

No modelling has been performed since measured data are considered sufficient.

Conclusions

The Industry proposal was not considered justified. Instead, the measured data provided by Industry have been used to calculate inhalation exposure levels as presented above. These values will be taken forward to risk characterisation.

Dermal exposure

Measured data

Measured data on dermal exposure do not exist.

Analogous/surrogate data

As there are no measured data it is instead proposed by Industry to extrapolate from data measured during bag filling at a diantimony trioxide production plant (2005); a summary of these data is given in Table 4-3. This is considered justified as both scenarios involve handling of diantimony trioxide powder and the handling patterns are similar. It is further proposed by Industry to use the 90th percentile of the handling duration values reported for the PVC and non-PVC sectors, i.e. 186 and 150 minutes, respectively. In view of the similarities between the two industry sectors, all reported values for duration of powder handling in both the PVC and the non-PVC sector should be used. The duration for powder handling is therefore determined to be 180 minutes (Table 4-9). This value is considered adequately conservative as data from 42 sites are available, with similar ranges of duration of handling in both subsectors, and considering that the median duration is considerably lower (30 min).

Table 4-9 Assumptions on the duration of handling in the non-PVC and PVC sectors

Exposure duration						
Sector	**Counts**	**Min**	**Median**	**Mean**	**P90**	**Max**
Non-PVC	16	2	30	49	150	200
PVC	26	5	33	68	186	300
All sectors	**42**	**2**	**30**	**61**	**180**	**300**

The duration of 180 min is used for recalculation of the dermal exposure values from production of diantimony trioxide. Accordingly, this gives the following values for dermal exposure:

Typical: 0.018 mg/cm^2/day * (180/480) = 0.0068 mg/cm^2/day

RWC: 0.031 mg/cm^2/day * (180/480) = 0.012 mg/cm^2/day

Modelled data

No modelling has been performed since analogous measured data from production of diantimony trioxide are considered sufficient for read-across.

Conclusions

For dermal exposure analogous data from final handling during production of diantimony trioxide, modified for duration (180 minutes), are taken forward to risk characterisation. The exposure in further processing and final product manufacturing is considered very low.

Values taken forward to risk characterization

The bodyweight of the worker is 70 kg and the exposed dermal area is 2000 cm^2.

Inhalation exposure
Raw material handling
Typical: 0.13 mg/m^3
RWC: 0.57 mg/m^3

Dermal exposure
Raw material handling
Typical: 6.8 $\mu g/cm^2$,day which corresponds to 0.19 mg/kg/day
RWC: 12 $\mu g/cm^2$,day which corresponds to 0.34 mg/kg/day

4.1.1.2.5 Scenario 4: Use as flame-retardant in textiles

A survey was conducted to collect data relevant for the assessment of occupational exposure to diantimony trioxide in the manufacturing of textiles treated with flame-retardant in Europe (EBRC, 2006f; EBRC, 2008); see chapter 4.1.1.2.1.

The report submitted by Industry has been summarised in the following sections, and the conclusions proposed by Industry are also presented. The exposure information is further evaluated before conclusions on values for risk characterisation are presented.

Summary of Industry information

Process description

Based on the client databases, 29 companies using 1757 tons diantimony trioxide per year were identified in the EU. Of these, only 6 companies consumed more than 125 tons of diantimony trioxide per year. In total, 15 companies responded in this survey, corresponding to 91 % (1603 tons per year) of the use of diantimony trioxide in this sector.

According to the survey results, diantimony trioxide is used as an input material for textile flame retarded backcoating in one of the following three forms: Powder (POW) - dry powder without any additives; Dispersed (DIS): aqueous, pre-formulated preparations containing 12-22 % diantimony

trioxide; Wetted (WET): Diantimony trioxide powder to which a minor portion of liquid has been added, usually at 3 % (w/w).

Small enterprises, that have low use rates, use predominantly wetted or dispersed forms of diantimony trioxide. These companies buy to a large extent pre-formulated semi-products and apply them with a minimum amount of "in-house" own handling as backcoatings. In contrast, higher consumption rates are related primarily to dry powder and these facilities still make up their own flame-retardant compounds based on a high proportion of dry powder and can thus be designated as "formulators".

For companies that only conduct backcoating application, diantimony trioxide is obtained from suppliers who provide a ready-to-use aqueous dispersion containing between 12 and 22 % diantimony trioxide.

A recipe of a textile backcoating, disclosed by one of the producers of flame-retardant dispersions gives the following composition: 22 % diantimony trioxide, 45 % organic brominated flame-retardant compounds, 1 % formulating agents and 32 % volatile components, predominantly water.

Diantimony trioxide is initially formulated into ready-to-use water-based dispersed formulations. This is not performed by all companies. These are then merely mixed on site with aqueous binding and other agents to yield the backcoating "compound", which is directly applied to textiles.

Formulation

Diantimony trioxide is received either in 25-kg paper or plastic bags on pallets, via big bags or containers on pallets. In the case of paper bags, the bags are cut open and the content of diantimony trioxide is poured into the mixing vessels via a manhole. The frequency and duration of handling as reported by responding companies can be summarised as between once per week to twice per shift with durations between 20 and 120 minutes per occasion.

These vessels are stated to be equipped with a filtered extraction device that creates airflow from outside to inside the vessel or there are suction devices in place to capture powders. The total number of responders was 9 and the number of used data was 8 (no data on duration was provided by one of the responders).

At some facilities, workers may continue to work in the same area, but without direct handling of diantimony trioxide. In the case of big bags or IBCs (Intermediate Bulk Containers), diantimony trioxide is first transferred to an interim storage vessel, involving extraction devices at the filling junctions, from which it is pneumatically discharged either in closed systems or under vacuum into the reaction/mixing vessel. Empty containers are then re-sealed and returned to the supplier. Prior to the addition of diantimony trioxide, the mixing vessel is filled partly with water, into which binders (acrylates), soaps and dispersion aids are mixed. Then, after addition of diantimony trioxide, the mixing is conducted by simple mechanical stirring at ambient temperature and normal pressure. Containers with emptied bags are collected and disposed. After the addition of diantimony trioxide, other powder formulation components are added to make up the final formulation. The sections of the plants where these compounds are made are segregated from the rest of the production. The reported cleaning procedures involve both wet removal and dry vacuum cleaning.

Processing

Diantimony trioxide is obtained at this process stage either via a supplier or from the on-site formulation unit as a viscous, liquid ready-to-use aqueous dispersion. This is pre-mixed with a liquid binder preparation and water at a ratio of 60:30:10 by slow stirring. All three materials are delivered in 120, 200 or 1000-litre sealed transport vessels. These are pumped into a 120-200-litre mixing drum with the aid of pumps and freshly mixed by slow stirring, after which the open drum is shifted immediately to the application machine. During mixing of the viscous flame-retardant formulation with the binder, the operator wears gloves and an apron and the preparation is made up in an area segregated from the rest of the process. Ventilation or other controls are not present in view of the viscous, aqueous nature of the compound. At the application machine, this viscous medium is pumped slowly into a vat with a rotating roll, over which the textile reverse layer is rolled. Alternatively, depending on the nature of the finished product, the flame-retardant compound may also be either foamed onto the textile or applied as a paste with the aid of knife coating machines. During the automated backcoating process, there is no manual handling. After completion of the process, the bulk of the remaining flame-retardant agent is transferred back to the transport vessel with the aid of a pump. Minor residual amounts of material are removed from the roll with a dry cloth. Workers wear disposable latex gloves during this terminal operation.

The backcoating is fixed to the textile in a drying process (130-145 °C) in an oven/stenting-frame, involving removal of air and moisture by extraction in the enclosed oven system, rendering air concentrations of the flame-retardant compound negligible. Any diantimony trioxide or diantimony trioxide-adhering to fibres, that may potentially be released, is removed by the extraction required for the drying process in the stent. The maximum duration of diantimony trioxide handling during a shift is considered to be approx. 6 hours. Prior to this, the machines will be prepared, the rolled textile will be mounted, and the diantimony trioxide flame-retardant compound will be mixed and loaded into the roller coating vat. There is no manual handling during this automated process. The remainder of the shift would be consumed by unloading, back-transfer of unconsumed flame-retardant compound and cleansing of the coating roll and vat. Finally, the roll is wiped clean with a dry cloth. The duration of these cleaning operations is less than 30 minutes.

Further handling

Diantimony trioxide may be contained in dry backcoatings at a level of up to 24 %; however, with respect to total portion of a treated textile, a range of 4-6 % can reasonably be assumed (EBRC, 2006f). In further processing of carpet or wall-coating textiles, a backing may be joined onto the treated textile with the aid of thermoplastic glue in a continuous rolling process, yielding the finished product. Other processes are not involved. There is no manual handling of treated carpet or textile during the process, apart from final loading/unloading.

In subsequent industries i.e., manufacture of furniture or other articles, it cannot be excluded that such back coated textiles may be subjected to operations such as cutting, sewing, upholstering and stuffing, where extensive dermal contact cannot be ruled out.

Inhalation exposure

Inhalation exposure to diantimony trioxide may occur during the loading and mixing of diantimony trioxide, but is minimised by a range of control measures. All of the responding users of diantimony trioxide powder report to have exposure control measures installed either in the form of engineering

controls and/or closed systems. In addition, some companies also use "wetted" forms of diantimony trioxide, with reduced dusting potential.

Formulation

Measured data

Four different companies reported some monitoring data from years 2004-2006 on workers involved in the formulation stage, where exposure to diantimony trioxide dust is possible (EBRC, 2006f). These data indicate generally low levels. With (n=11) or without (n=9) including two static sampling data points, exposures are at 0.024 mg/m³ (median), and 0.06 mg/m³ (90^{th} percentile). However, since the duration of handling varies between 20 and 120 minutes according to the responses of the companies involved in the survey, monitoring continued beyond the actual period of handling diantimony trioxide and as such reflects the overall low exposures when workers continue working in the same area. In consideration of the fact that the data do not differentiate between full-shift and short-term data, it is proposed by Industry to consider these as supportive data only.

Analogous data

For the assessment of short-term and full shift exposure, it is proposed by Industry to make reference to an "analogous" data set collected under conditions of very similar handling patterns and engineering controls, i.e. during operations in the PET industry.

Data that are relevant to a very similar handling of diantimony trioxide are available from a report on PET production (see chapter 4.1.1.2.3, (EBRC, 2005b) involving a monitoring data set with 60 individual personal samples from 11 different production sites. The results of this report can be summarised as follows:

Handling of diantimony trioxide occurred at all sites involved in this survey infrequently and only for a minor portion of the shift. The reported duration of handling of diantimony trioxide varied between 5 minutes, with five batches per day being prepared, to 60 minutes, with one batch per week, between sites/plants. In view of the nature of these operations, workers will move around during a shift to other areas and tasks not involved with exposure to diantimony trioxide, it was considered reasonable by Industry to assume negligible inhalation exposure to diantimony trioxide for these periods. For this reason, Industry calculated 8-hour TWA values for the PET sector, in addition to the measured short-term values. Typical (median, 0.002 mg/m^3) and realistic worst-case (90^{th} percentile, 0.026 mg/m^3) exposure levels were assessed.

Processing

Measured data

Some monitoring data during application of diantimony trioxide flame-retardant compounds to textiles are published. Two different sources of occupational exposure data from the textile flame-retardant industry were identified in the public domain, which are summarised briefly below:

(i) A monitoring survey involving inhalation exposure and blood biomonitoring was conducted in workers involved in the production of fireproof textiles for car upholstery (Cavallo et al., 2002; Iavicoli et al., 2002). Inhalation exposure was studied in 23 workers, with a control group of 23

non-exposed workers, by personal and static sampling. A sampler was used in order to collect the inhalable fraction. Sampling was performed during the entire working shift of 8 hours.

(I) Finishing operators; designated as "high" exposure group, 17 workers monitored, involved in the preparation of flame-retardant suspension based on diantimony trioxide, and subsequent impregnation of textiles

(II) Jet operators; designated as "low" exposure group, 6 workers monitored, performing dying operations on raw textiles; indirect exposure resulting from carry over from the area where diantimony trioxide is handled.

It is worth noticing that the total number of data points varies between both publications with 41 and 42 individual data points. Only the reported average values could be given for the separate exposure groups.

In conclusion, the actual application of flame-retardant compound to textiles in this industry sector has been reported to be associated with extremely low inhalation exposures, from personal sampling, in the range of 0.01 - 0.55 µg Sb/m³ (corresponding to 0.012 – 0.66 µg Sb_2O_3/m³). The static samples were in the range of 0.01 - 1.45 µg Sb/m³. These low levels can be explained by the use of aqueous dispersions and the process characteristics.

(ii) German occupational hygiene survey (BGAA, 2001).

In the period 1996-2000, personal sampling of inhalable antimony in air was performed at 65 different industrial sites in Germany consuming diantimony trioxide, accounting for a total of 131 measurements. The sampling duration was above 60 minutes for all measurements (BGAA, 2001). 23 samples were allocated by the authors to the sector of textile treatment, described as "formulation and application of backcoating".

Measurements were made both at sites with and without use of LEV. All 23 samples taken in this sector during the conduct of this sub-process were below the detection limit of 0.003 mg/m³, thus supporting the published survey reported under (i) above.

Analogous/surrogate data

It is proposed by Industry that in view of the similarity in handling patterns of diantimony trioxide exposure data may also be reasonably extrapolated from measurements made in a range of plants in the PET industry, as an "analogous", auxiliary approach.

Further handling

No data on inhalation exposure obtained during the processing of flame-retarded backcoated textiles have been submitted, but exposure can be considered to be low for the following reasons:

- Flame-retardant treatment is performed according to performance standards, and diantimony trioxide is bound within a polymer, usually acrylic, binder matrix from which release is precluded; any relevant release would render the final product sub-standard.

- Release of diantimony trioxide from textiles treated with flame-retardant has been measured both on "fresh" and aged back-coated textiles and was found to be minimal (Thomas and Stevens, 2006).

Dermal exposure

Dermal exposure to diantimony trioxide may occur during loading/mixing. It also occurs to a limited extent and for a short duration at final cleaning, during which gloves and overalls are worn.

Dermal exposure data for the formulation stage do not exist, but, proposed by Industry, an extrapolation from data measured during bag filling at a diantimony trioxide production site can be used with confidence. For the final processing stage, a detailed investigation of dermal exposures encountered during production of furniture upholstery is available.

Formulation

Analogous/surrogate data

Measured data on dermal exposure to diantimony trioxide at the formulation stage in the flame-retardant treated textile industry are not available. However, an extrapolation from data measured during bag filling at a diantimony trioxide production site (Hughson, 2005) is considered feasible by Industry. In this study, occupational dermal exposure to diantimony trioxide during production and subsequent handling was measured with the aid of the wipe sampling method. Workers were involved either in packaging or in other activities for a full shift.

Considering that handling of diantimony trioxide in the Textiles sector does not occur full-shift and that for the remainder of a shift workers will perform tasks in other areas of the plant not involving diantimony trioxide exposure, it is proposed by Industry as a worst-case to carry forward the 90th percentile value (indicated as 35 µg/cm² in the Industry report (EBRC, 2006f) and differing slightly from the value indicated in Table 4-3, which is taken from the original report by Hughson (2005). It is further proposed to use as typical exposure the median (indicated as 16 µg/cm²) level of dermal exposures measured for diantimony trioxide during packaging, but modified by the duration of handling. If the duration is set to be 120 minutes the exposures taken forward would be:

RWC: 9 µg/cm² (120/480 x 35), and typical case: 4 µg/cm² (120/480 x 16)

Processing

Application of flame retarded backcoating compound to textiles

Dermal exposure may theoretically occur during pre-mixing of the backcoating compound, but the use of pumps and the highly viscous nature of the media and the lack of need of any manual intervention involved preclude any relevant formation of splashes, which is why exposure at this stage may be considered minimal. During final cleaning of equipment after cessation of the backcoating process, dermal exposure cannot be ruled out. However, the duration of exposure is only for a short period of time (< 30 minutes) and gloves are worn during this task.

Modelled data

The estimation via RISKOFDERM was conducted with the following selections:

Used scenario: Model for handling of (potentially) contaminated objects (DEO unit 1)

Task or scenario: "Cleaning of spray guns or other tasks where contaminant is removed by rinsing and some wiping"

What is the quality of the ventilation related to the task done: "Normal or good ventilation"

What is the frequency of (skin) contact with the contaminant: "More than rare contact"

What kind of skin contact with the contaminant occurs: "Light contact"

What kind of product is handled: "Liquid"

Are significant amounts of aerosols or splashes generated in the task: "No"

What is the level of automation of the task done by the worker: "Manual task"

What is use rate of the product: "1"

What is the cumulative duration of the scenario during a shift: "30"

The resulting typical (median) and reasonable worst-case (90th percentile) exposures for hands are estimated by the model as shown in Table 4-10.

Table 4-10 Modelled potential and actual dermal exposure for the hands during cleaning of flame retardant backcoating application machines, task duration max. 30 minutes

Unit	Median	P90
Exposure rate (mg/min)	13.3	91.4
Duration of task during shift (minutes)	30	30
Estimated potential dermal exposure to flame-retardant compound, both hands (mg)	399	2740
Diantimony trioxide percentage in flame-retardant backcoating (%)	22	n.a.
Estimated potential dermal exposure to diantimony trioxide, both hands (mg)	87.8	602.8
Estimated potential dermal exposure to diantimony trioxide, per surface area[a)] (µg/cm²)	105	718
Estimated actual dermal exposure to diantimony trioxide (use of gloves)[b)] (µg/cm²)	10.5	71.8

a) assuming a total surface of 840 cm² for both hands;
b) all companies reported use of gloves as mandatory, use of gloves assumed to afford 90% reduction;
na: not applicable

It is proposed by Industry to take forward a typical value of 11 µg/cm² and a reasonable worst-case value of 72 µg/cm² to risk characterisation, in consideration that this work is performed with the aid of tools and gloves, and that cleaning is restricted to max. 30 minutes always conducted at the end-of-shift. Since after this task, washing of the hands is assumed to remove any contaminant, the limited duration of exposure should be accounted for in risk characterisation.

Further handling

A detailed report on occupational exposure during handling of textiles treated with flame-retardant is available (Niven, 1993), which can be summarised as follows: An investigation was conducted at

an upholstery firm in the UK in order to develop a method to assess the contamination of skin by antimony in the furniture upholstery industry. Automated working methods, such as production lines, were not used in this facility and each worker always performed the same duties. Occupational groups identified as having regular, extensive contact with diantimony trioxide-coated fabrics were cutters, sewing machine operators, upholsterers and cushion stuffers. However, it should be noted that also fabrics not containing diantimony trioxide were used to a small extent. These groups are briefly described below:

Cutters: The cutting area was located in a segregated room. Cutters received a roll of fabric directly upon delivery and spread it out onto large tables for cutting. Cardboard templates and pencils were used to mark cutting lines on the coated side. Large scissors were used to cut the fabric. Cutting through three or four layers of cloth at a time was common when cutting fabric for several cushion covers.

Sewing machine operators: No other tasks were carried out in the sewing room. Prepared fabric was brought from the cutting room in batches corresponding to items of furniture. These were sewn together with piping, zips and other fasteners fitted as required. Operators did not leave their machines at break times.

Upholsterers: Each working position in the large upholstery room was fitted with a compressed air line, which was used to operate a staple gun. Trestles were available to stand work pieces on if necessary.

Cushion stuffers: A single individual was employed to carry out general duties. One of these tasks involved stuffing pre-sewn cushion covers with foam pads or synthetic filling. On average, this individual could fill approximately 20 cushions per hour and would normally wait until there were enough cushion covers to last two hours; this occurs only once or twice a week. One single data of the exposure to the cushion stuffers gave a dermal exposure of 0.172 µg Sb_2O_3/cm² skin.

In conclusion, full-shift dermal exposure to diantimony trioxide during handling of flame-retardant treated textiles are generally very low, ranging from 0.02 to 1.4 µg Sb_2O_3/cm² skin. The statistical analysis of these data is summarised in Table 4-11 below.

Table 4-11 Summary statistics, dermal exposure to diantimony trioxide [µg Sb_2O_3/cm² skin]

Task description	No of data	Min	Median	P90	Max
Cutters	12	0.014	0.058	0.35	0.35
Sewers	11	0.029	0.17	0.58	1.4
Upholsterers	12	0.018	0.066	0.15	0.18
All tasks	**36**	**0.014**	**0.078**	**0.35**	**1.4**
Controls	12	0.000	0.006	0.014	0.016

In conclusion, based on the entire sample database, a typical (median) dermal exposure of 0.08 µg Sb_2O_3/cm² was calculated by Industry, with a corresponding worst-case (90^{th} percentile) of 0.35 µg Sb_2O_3/cm². It is proposed to take these values forward to risk characterisation.

Evaluation of exposure

Inhalation exposure

Measured data and analogous/surrogate data

Formulation

Measured occupational exposure data (n=11) during the process stage "formulation" were obtained from 4 companies. The median for these were 0.024 mg Sb_2O_3/m^3 and the 90th percentile 0.06 mg Sb_2O_3/m^3. These measured data are not considered sufficient for assessing the exposure during formulation in the textile industry.

For auxiliary purposes, references were also made by Industry to an "analogous" data set collected under similar handling operations in the PET industry. This is not considered to be applicable due to uncertainties concerning the similarity in handling of diantimony trioxide in the PET industry and the textile industry, which could support that the exposure is of the same magnitude in these two sectors. For example, manual handling of smaller sacks containing diantimony trioxide powder appears to be more common in the textiles sector compared to the PET sector, where diantimony trioxide is handled in a more automated manner using large closed containers.

Instead it is considered justified to read-across from the measured data collected during raw material handling in the plastics sector, which are considered to be reliable and representative of the real situation for workers during an 8-hr shift in that sector. The process descriptions of formulation in the textile industry and raw material handling in the plastics industry both indicate manual handling of 25-kg sacks containing diantimony trioxide, in contrast to the more automated process in the PET industry. Further, it is not considered feasible to adjust the measured exposure values in the plastics sector due to possible differences in duration of handling. The fact that handling of diantimony trioxide only occurs part of the day is already included in the exposure assessment in the plastics sector. The reported range for duration of handling in the plastics sector is quite large (2-300 min/shift) and values for duration of handling reported from the textile industry sector (30-120 min/shift) are all within this range. Further, the mean duration of handling diantimony trioxide powder is almost the same for both sectors (68 min for Textiles versus 61 min for Plastics). Information of duration of handling is available from a fraction of the two sector sites, i.e. 10 % (42 out of 424 sites in Plastics) and 31 % (9 out of 29 sites in Textiles). The values taken forward to risk characterization are therefore 0.13 mg/m^3 as a typical exposure and 0.57 mg/m^3 as a reasonable worst case. These values are higher than the values obtained from the limited measurements available and may be considered conservative.

Processing

A published survey at one site (car upholstery) reported extremely low inhalation exposures in the range of 0.000012 – 0.00066 mg Sb_2O_3/m^3. Although these values are based on measurements in two groups of workers at just one single site, it is proposed by Industry to take forward the average of 0.00014 mg Sb_2O_3/m^3, i.e. <0.001 mg/m^3, as a typical exposure value, and the maximum from both groups with 0.00066 mg Sb_2O_3/m^3, i.e. 0.001 mg/m^3 as a reasonable worst case value. Further, a German occupational hygiene survey of the textile backcoating sector provided 23 inhalable personal samples, all of which were below 0.003 mg Sb_2O_3/m^3, which support the conclusion that

taking forward the values given above are adequately conservative. This reasoning is considered justified.

Further handling

No data were available on inhalation exposure to diantimony trioxide during subsequent handling of flame-retardant treated textiles. However, it is proposed by Industry, and considered justified, to consider these as negligible for the following reasons:

- Inhalation exposure is already low and well-controlled in the two sub-process assessed above, where the relative content of diantimony trioxide in handed products is much higher.

- The finished textile product contains diantimony trioxide in a polymer matrix which is subject to stringent performance standards and any relevant release would render the final product sub-standard.

- Release of diantimony trioxide from textiles treated with flame-retardant has been measured both on "fresh" and aged back-coated textiles and was found to be minimal (Thomas and Stevens, 2006).

Modelled data

Since measured data are available, either from the textile sector during processing or from another sector that is suitable for read-across, no modelling of the inhalation exposure has been performed.

Conclusions

Industry's proposal for inhalation exposure during formulation is not considered justified. Instead, read-across from the Plastics sector is performed and the measured data from that sector are taken forward to the risk characterisation. Industry's proposal for inhalation exposure during processing and further handling is considered justified and is taken forward to risk characterisation.

Dermal exposure

Measured data and analogous/surrogate data

Formulation

Considering that diantimony trioxide handling in this sector does not occur full-shift and that for the remainder of a shift workers will perform tasks in other areas of the plant not involving diantimony trioxide exposure, it is proposed by Industry, and considered justified, as a reasonable worst-case to carry forward the highest levels of dermal exposure measured for final handling (packaging) during production of diantimony trioxide, but modified by the duration of handling during a shift (120 minutes). This is considered justified and this value for duration is used for recalculation of the dermal exposure values from production of diantimony trioxide. A duration of 120 minutes is considered adequately conservative as data from one third of the sites are available and no higher values have been reported. Accordingly, this gives the following values for dermal exposure:

Typical: 0.018 mg/cm^2/day * (120/480) = 0.0045 mg/cm^2/day

RWC: 0.031 mg/cm^2/day * (120/480) = 0.0078 mg/cm^2/day

Processing

Measured data for this process stage are not available.

Further handling

An investigation on occupational exposure during handling of flame-retardant treated textiles in upholstery manufacturing is available (Niven, 1993), involving 7 workers from four different production areas, and reporting a total of 35 individual full-shift samples. The dermal exposures ranged from 0.00002 to 0.0014 mg Sb_2O_3/cm² skin, from which a 50th percentile of 0.00008 mg Sb_2O_3/cm² and a 90th percentile of 0.00035 mg Sb_2O_3/cm² can be calculated. These values are in line with the fact that diantimony trioxide in this product type is contained in a polymer matrix.

Modelled data

Formulation

No modelling has been performed since analogous measured data from production of diantimony trioxide are considered sufficient for read-across.

Processing

Measured data for this process stage are not available. An attempt was made by Industry to predict potential dermal exposure with the aid of the model RISKOFDERM. This modelling is considered justified and, as proposed by Industry, a typical value of 11 µg/cm² and a reasonable worst-case value of 72 µg/cm² for dermal exposure of the hands are taken forward to risk characterisation (see Table 4-10).

Further handling

No modelling has been performed since measured data are considered sufficient.

Conclusions

As proposed by Industry, for formulation, the highest levels of dermal exposure measured for final handling during production of diantimony trioxide are used. These values have been modified by the duration of handling (120 minutes) when taken forward to the risk characterisation. As also proposed by Industry, modelled data is used for dermal exposure during processing and measured data is used for further handling.

Values taken forward to risk characterisation

The bodyweight of the worker is 70 kg and the exposed dermal area is 2000 cm^2 for formulation and further handling (hands and forearms) and 840 cm^2 for processing (hands only).

Inhalation exposure
Formulation
Typical: 0.13 mg/m^3
RWC: 0.57 mg/m^3

Processing
Typical: <0.001 mg/m^3
RWC: 0.001 mg/m^3

Further handling
Typical: negligible
RWC: negligible

Dermal exposure
Formulation
Typical: 4.5 µg/cm^2 which corresponds to 0.13 mg/kg/day
RWC: 7.8 µg/cm^2 which corresponds to 0.22 mg/kg/day

Processing
Typical: 11 µg/cm^2 which corresponds to 0.13 mg/kg/day
RWC: 72 µg/cm^2 which corresponds to 0.86 mg/kg/day

Further handling
Typical: 0.08 µg/cm^2 which corresponds to 0.0023 mg/kg/day
RWC: 0.35 µg/cm^2 which corresponds to 0.010 mg/kg/day

4.1.1.2.6 Scenario 5: Use in pigments, paints, coatings and ceramics (PPCC)

A survey was conducted by EBRC for IAOIA to collect data relevant for the assessment of occupational exposure to diantimony trioxide in this industry sector, see chapter 4.1.1.2.1. The results from this survey, together with information from other sources, have been presented in a report (EBRC, 2006i).

The report submitted by Industry has been summarised in the following sections and the conclusions proposed by Industry are also presented. The exposure information is further evaluated before conclusions on values for risk characterisation are presented.

Summary of Industry information

Diantimony trioxide is used in the manufacturing of "Complex Inorganic Coloured Pigments" (CICP), which are further used in subsequent industries such as plastics (50 %), coatings (35 %), enamels and ceramics (10 %) and building materials (5 %). The pigment production process involves chemical transformation of the input materials into a crystal (rutile) host matrix, in which various metal cations (e.g., titanium, nickel, chromium) apart from antimony are incorporated. Because of their light fastness and resistance to high temperatures, these pigments are ideally suited for coatings, plastic colouring and ceramics. Once incorporated into these rutile structures,

antimony is no longer present as diantimony trioxide, since it is converted to antimony(V)oxide in the calcination process. For this reason, the focus of the Industry report lies on the process stages involving exposure to diantimony trioxide itself. Apart from the use in pigment production, only two other (low level) uses have been reported; as pigment in the ceramics industry and as flame-retardant in special paints, for which, however, the available data on exposure during diantimony trioxide powder handling in pigment production are assumed to also apply.

Process description

A total of 29 companies were identified and contacted in EU15. Of these, 9 companies, representing more than 80 % of the total quantity of diantimony trioxide used in this application, replied. In this industry sector diantimony trioxide is mainly used as a dry powder and only to a lesser extent in other forms (wetted forms or aqueous dispersions).

The following process description is presented in the EBRC report (EBRC, 2006i). The description is based on a site visit at one European pigment producer's site and the feedback to the questionnaire survey. The production of CICP by using diantimony trioxide occurs by a batch-wise process, divided into the following steps (Table 4-12):

Table 4-12 Overview of tasks in pigment production

Workplace	Workplace description	Tasks performed
1	Loading and mixing	Emptying of big bags, mixing
2	Calcination	Preparation of crucibles, calcination, emptying of crucibles
3	Milling and final bagging	Quality control
4	Others	Repair, cleaning*, maintenance, etc.

* Routine cleaning is part of the jobs/task assigned to workplace 1-3; cleaning operations under workplace 4 refer to staff that perform only such cleaning and maintenance work.

According to the responses to the Industry survey, the duration for handling of diantimony trioxide in this industry sector ranges from 5 minutes per shift to 33 minutes per shift, with a mean value of 17 minutes (based on 6 data points). The maximum amount handled is 1000 kg per shift and minimum is 9 kg per shift.

Engineering exposure controls that are currently in place as reported by this industry sector are hood and extraction, extraction, or LEV. RPE (P1, P2 or P3) and protective gloves are reported as mandatory by 7 downstream users.

Loading and mixing

Antimony trioxide is delivered exclusively in dry powder form and in big bags according to production needs. The big bags containing diantimony trioxide are emptied directly into an enclosed dry pre-mixer, where dust exposure is minimised through a low-pressure extraction system. Whereas the entire loading procedure for all ingredients takes approx. 1-1.5 hours, the duration of the unloading of diantimony trioxide from big bags is typically in the range of approx. 10 minutes (worst-case approx. 30 minutes), usually only once per day. Total amounts of diantimony trioxide handled are typically 80 kg/shift or, as a reasonable worst case, 650 kg/shift. Any spills are removed

immediately by dry vacuum cleaning. During this task (performed by one worker per shift), a facemask (P3 filter), gloves and a chemical protection overall are worn as PPE.

Subsequently, the pre-mix is transferred into a second mixer in a closed pipe system. In this second mixer, the pigment pre-mixture is mixed with further raw materials according to product requirements and subsequently wetted with water to a level of 20-40 %. The resulting blend is a granular-type intermediate (approx. 10 % diantimony trioxide), which is then transferred for short-term storage to intermediate silos via a closed pipe system. Only one worker per shift is responsible for the task of loading and mixing.

Calcination

From these silos, the granulated blend is filled into small crucibles via an automated weighing and transport system, which does not require any manual interference. Due to the fully automated system, dermal exposure does not occur, and inhalation exposure to diantimony trioxide at this point is negligible due to the granulated nature and high water content of this blend. The filled crucibles are then manually loaded onto a trolley in several layers (one worker per one-half of a working shift), where, according to Industry, dermal exposure is also negligible since the handling occurs with gloves to avoid direct contact with the material.

After loading, the trolley is moved to a kiln in the same production unit. The hood of the kiln can be pneumatically raised and lowered to totally encompass the trolley, thus creating a closed system with extraction systems with a scrubber for the removal of steam and any particulates. The calcination takes place at approx. 1000 °C for a period of 24 – 48 hours, until the reaction to the pigment is finished. During this calcination process, neither dermal nor inhalation exposure can occur. When the calcination is completed, the trolley is moved to a cooling station fitted with an extraction hood to collect any material that may be suspended by convection during the entire cooling phase. After the calcination, the trolley is then moved back to its loading port, where the crucibles are manually unloaded into a closed transportation system, where a low-pressure extraction system effectively minimises dust formation.

At this stage of production, diantimony trioxide is no longer present in the crude pigment obtained from the calcination process. As above, there is little potential for dermal exposure, since the handling occurs with gloves and no direct contact with the material occurs. Nevertheless, normal overalls are worn, dermal exposure is controlled by using gloves and RPE devices are provided, but not mandatory, to protect workers from contact with fine pigment particles or nuisance dust for reasons of good industrial hygiene.

Milling and final bagging

The previously calcinated material is subsequently transferred via a closed pipe system to a mill in a closed system. After milling, the CIPC pigment powder is again stored in silos, from which it is manually filled into 25-kg bags for shipping. Whereas the milling and all associated transport processes are in an entirely closed system, dermal and inhalation exposure are possible during the final bagging step. Here, inhalation exposure is minimised by a local extraction system using special ventilated paper bags, and dermal exposure is controlled by using gloves. Only one worker per shift is associated with the task of milling and packaging of finished product.

Inhalation exposure

One company provided a consistent set of measurements (n=80) of occupational inhalation exposure extending over the sampling years 2002 (n=22), 2004 (n=29) and 2005 (n=29). All measurements were made according to the national standard (country not stated for reasons of confidentiality) for measuring antimony exposure. A stationary sampler (Gravikon VC25) was used to collect the inhalable fraction at different workplaces within the pigment plant for a minimum of 120 minutes each. All measurements were made under normal working conditions. The dust was collected on a glass fibre filter with 150 mm diameter and subsequently analysed for antimony (Sb) by inductively coupled plasma mass spectrometry (ICP). The results are summarised in Table 4-13 below:

Table 4-13 Static inhalation exposure monitoring in a pigment production site

Workplace	Concentration [mg Sb/m³]				No of data
	Min	Median	P90	Max	
1	0.001	0.011	0.033	0.068	21
2	0.001	0.004	0.024	0.074	31
3*	0.001	0.003	0.005	0.007	22
4*	0.001	0.001	0.004	0.007	6

Detection limit = 0.001

*Note: whereas exposures measured after the calcination stage (i. e. workplace 3 and 4) may relate to Sb, they do not, according to Industry,represent diantimony trioxide in view of the chemical conversion

These values represent static measurements, and thus may not be reflective of personal exposure. In the report submitted by Industry it is proposed, in view of the consistency of the measuring strategy and the continuously low levels of exposure, to use these values to extrapolate to personal inhalation exposure. In support of this, Industry notes that Cherrie (2004) based on a review of published data on static and personal inhalation exposure over a 10-year period, stated that *"The median ratio between personal and static concentrations was 1.5, although the individual data points ranged from 0.4 to 10"*. It is further noted by Industry that personal exposures are widely considered to be higher than static exposures, and it is proposed to use an extrapolation factor of 1.5 according to Cherrie to derive the dust exposure levels considered to approximate personal exposures, presented inTable 4-14. In addition, a conversion factor of 1.2 was used to convert mg Sb/m^3 to mg Sb_2O_3/m^3.

Table 4-14 Personal inhalation exposure extrapolated from static measurements in a pigment production site [mg/m³]

Workplace	Min	Median	P90	Max
1	0.002 (Sb) 0.002 (Sb_2O_3)	0.017 (Sb) 0.020 (Sb_2O_3)	0.050 (Sb) 0.060 (Sb_2O_3)	0.102 (Sb) 0.122 (Sb_2O_3)
2	0.002 (Sb) 0.002 (Sb_2O_3)	0.006 (Sb) 0.007 (Sb_2O_3)	0.036 (Sb) 0.043 (Sb_2O_3)	0.111 (Sb) 0.133 (Sb_2O_3)

3*	0.002 (Sb) 0.002 (Sb_2O_3)	0.004 (Sb) 0.005 (Sb_2O_3)	0.008 (Sb) 0.010 (Sb_2O_3)	0.011 (Sb) 0.013 (Sb_2O_3)
4*	0.002 (Sb) 0.002 (Sb_2O_3)	0.002 (Sb) 0.002 (Sb_2O_3)	0.006 (Sb) 0.007 (Sb_2O_3)	0.011 (Sb) 0.013 (Sb_2O_3)

Detection limit = 0.001 mg Sb/m³

* Note: whereas exposures measured after the calcination stage (i. e. workplace 3 and 4) may relate to Sb, they do not, according to Industry, represent diantimony trioxide in view of the chemical conversion

Whereas workers will be involved at workplace 1 for only max. 1.5 hours, all other workplaces involve full-shift operations.

As the duration of sampling is 120 minutes, it is proposed by Industry to see these data as reflective of full-shift exposure and further to take these values forward to risk characterisation, i.e. 0.060 mg/m³ and 0.020 mg/m³ as the reasonable worst case and typical case, respectively.

Dermal exposure

Sector-specific measured data on dermal exposure to diantimony trioxide at the formulation stage in pigment production do not exist. It is proposed by Industry to extrapolate from data measured during antimony production (Hughson, 2005) under the assumption that diantimony trioxide powder bagging and powder unloading are associated with similar exposure levels. In this, occupational dermal exposure to diantimony trioxide during production and subsequent handling was measured with the aid of the wipe sampling method.

Considering that diantimony trioxide handling in this sector does not occur full-shift and that for the remainder of a shift workers will perform tasks in other areas of the plant not involving diantimony trioxide exposure, it is proposed by Industry as a reasonable worst case to carry forward the levels of dermal exposure measured for diantimony trioxide packaging, but modified by the duration of handling during a shift. The duration ranges from 5 to 33 minutes per shift and it is proposed by Industry to use the 90th percentile and median of the duration, i.e. 23 and 8 minutes, respectively.

Thus, the values for dermal exposure for loading of diantimony trioxide under these conditions as proposed by Industry are:

RWC: 2 µg/cm², and typical case: 1 µg/cm²

It should be noted that these calculations performed by Industry are based on slightly different values for dermal exposure compared to the values gives in Table 4-3.

An estimate for the other four workplaces is not considered required by Industry, in consideration of the already low levels predicted for powder loading and the fact that the wetted (30 % water) diantimony trioxide pre-mixture is not handled directly. According to Industry, after the calcination stage, exposure to diantimony trioxide is no longer possible because a chemical conversion has taken place, so that dermal exposure to diantimony trioxide at this stage can be assumed to be negligible.

Evaluation of exposure

The three scenarios, (i) loading and mixing, ii) calcination and iii) milling and final bagging, have been assessed. Only the scenario loading and mixing has potential for any relevant inhalation and dermal exposure, for the following reasons:

- Diantimony trioxide is handled as a dry powder during initial loading and mixing.

- Prior to calcination, the material has already been extensively wetted and therefore no longer emits any dust.

- After calcination, diantimony trioxide is no longer present, since all antimony has been chemically converted into rutile pigments, from which antimony is not bioavailable.

Inhalation exposure

Measured data

Measured data on exposure, using a stationary sampler, are only available from one site in this industry sector. No details on this site have been provided and it is therefore not possible to assess how representative the site is of the total 29 sites in this industry sector. Other industry sectors with similar handling have reported considerably higher levels of exposure than the measured data from this site. Cherrie (2004) stated that *"The median ratio between personal and static concentrations was 1.5, although the individual data points ranged from 0.4 to 10"*. In the same paper, it is however pointed out that *"Key factors in determining the relation between personal and static measurements in any situation will include the volume of the room, the quantity of general ventilation, the time the person spends in the proximity of sources of hazardous substances (that is, with a source in their breathing zone), the presence of other internal or environmental sources of the contaminant, and others. In most circumstances, without knowing something about each of these factors it is impossible to predict what the relation between personal and static concentrations might be"*. It is therefore not considered appropriate in this case to use the factor 1.5 to extrapolate between static and personal measurements without further justification.

If the minimum and maximum values for the ratio between personal and static concentrations, 0.4 and 10, are used, the exposure will be 0.016 – 0.396 mg Sb_2O_3/m^3. This range can only be used as auxillary information on the inhalation exposure in the PPCC sector.

Analogous/surrogate data

Instead of using the stationary exposure data, read-across from the Plastics industry sector is considered appropriate using the measured data collected during raw material handling in that sector. These data are considered to be reliable and representative of the real situation for workers during an 8-hr shift in the Plastics sector. The process descriptions of formulation in the PPCC sector and raw material handling in the Plastics industry are similar in that dry powder of diantimony trioxide are handled in similar ways, such as emptying of bags and mixing. However, in the PPCC sector slightly lower amounts of diantimony trioxide powder are handled (typically 80 kg/shift in the PPCC industry, with a reasonable worst case of 650 kg versus 150 kg/shift and 1000 kg in the Plastics industry). Furthermore, the duration of handling is shorter in the PPCC sector (5-33 min; mean 17 min) compared to the Plastics sector (2-300 min; mean 61 min). Therefore it is

considered appropriate to modify the exposure data from Plastics and use the various mean duration times as a basis for this correction. This would lead to the following values for PPCC:

Typical: 17 min/61 min x 0.13 mg/m^3 = 0.036 mg/m^3

RWC: 17 min/61 min x 0.57 mg/m^3 = 0.16 mg/m^3

The values taken forward to risk characterisation are thus 0.036 mg/m^3 as a typical exposure and 0.16 mg/m^3 as a reasonable worst case.

Modelled data

Since measured data from another sector that is suitable for read-across is available, no modelling of the inhalation exposure has been performed.

Conclusions

The Industry's proposal for inhalation exposure during "loading and mixing" is not considered justified. Instead, read-across from the Plastics sector is performed and the measured data, corrected for duration of handling, from that sector are taken forward to the risk characterisation.

Dermal exposure

Measured data

Measured data on dermal exposure do not exist.

Analogous/surrogate data

As there are no measured data it is instead proposed by Industry, and considered justified, to use analogous measured data from final handling during production of diantimony trioxide. It is further proposed by Industry to use the 90th percentile of the duration values reported i.e. 23 minutes. This is however not considered justified as information on actual durations is only available from six work places from this diverse scenario comprising four different industry sectors. Instead the duration of exposure is assumed to be 60 minutes as a conservative approach. This value is used for recalculation of the dermal exposure values from production of diantimony trioxide. Accordingly, this gives the following values for dermal exposure:

Typical: 0.018 mg/cm^2/day * (60/480) = 0.0023 mg/cm^2/day

RWC: 0.031 mg/cm^2/day * (60/480) = 0.0039 mg/cm^2/day

Modelled data

No modelling has been performed since analogous measured data from production of diantimony trioxide are considered sufficient for read-across.

Conclusions

For dermal exposure analogous data from final handling during production of diantimony trioxide, modified for duration (60 minutes), are taken forward to risk characterisation.

Values taken forward to risk characterisation

The bodyweight of the worker is 70 kg and the exposed dermal area is 2000 cm^2.

Inhalation exposure
Loading and mixing
Typical: 0.036 mg/m^3
RWC: 0.16 mg/m^3

Dermal exposure
Loading and mixing
Typical: 2.3 $\mu g/cm^2$ which corresponds to 0.066 mg/kg/day
RWC: 3.9 $\mu g/cm^2$ which corresponds to 0.11 mg/kg/day

4.1.1.2.7 Scenario 6: Use as flame-retardant in production of rubber

Summary of Industry information

Diantimony trioxide is used as a flame-retardant synergist in the rubber industry for the fabrication of conveyor belts, rubber hoses, air springs and rubber-coated fabrics. The quantities added to rubber mixtures range from 2 to 10 % according to (BLIC and van der Velde, 2006), whereas the typical use is stated by the OECD (OECD, 2004) to be 3 %.

A survey was conducted by EBRC for IAOIA to collect data relevant for the assessment of occupational exposure to diantimony trioxide in this industry sector. The results from this survey, together with information from other sources, including a site visit, have been presented in a report (EBRC, 2006e).

In total, 41 companies were identified and contacted. Of these only three consumed more than 125 tons of diantimony trioxide per year. One third of the companies (n=13) replied to the questionnaire, corresponding to half of the diantimony trioxide consumption in this sector.

According to the survey results, diantimony trioxide is used in the following forms in the production of rubber: dry powder without any additives, dry powder without any additives in pre-weighed closed bags for direct feeding into closed systems without opening, aqueous, pre-formulated preparations containing 50 to 80 % diantimony trioxide, diantimony trioxide powder to which a minor portion of liquids has been added, usually at 2 to 10 % (w/w) to reduce dustiness or

as a masterbatch with diantimony trioxide in a granulated form tightly bound in a polymer matrix (polymer content of 10 %).

Laboratory particle size distribution investigations (DMT, 2005) have shown that diantimony trioxide powder wetted with 3 % of a phthalate gives a drastic reduction in dustiness. The dry powder had > 60 % of particles with aerodynamic diameters < 5 μm, whereas the wetted powder had < 5 % of particles with this size.

The results from the survey were compared to sales data supplied by the IAOIA members. General conclusions from this information are that: Large and medium-sized (using more than 10 tons/year) consumers in this sector mainly use diantimony trioxide in pre-weighed bags fed directly into closed systems and to a much lesser extent diantimony trioxide in the form of a dry powder, whereas small consumers (using less than 10 tons/year) use diantimony trioxide in a variety of forms, with dry powder corresponding to slightly less than 50 % of the total consumption. The small quantities involved indicate infrequent use of diantimony trioxide at these sites.

Process description

The production of rubber containing diantimony trioxide can be split into:

a) the initial formulation stage, consisting of weighing and mixing, which yields either rubber pre-mixes (pellets or sheets) or diantimony trioxide 'damped' by the addition of an oil, and

b) processing, consisting of shaping and curing (vulcanisation), in which the rubber premixes are thermally processed into the finished product.

In the initial **formulation** stage three process conditions can be identified:

i) **Formulation of diantimony trioxide** ('damped' with oil): The diantimony trioxide is loaded from 25-kg bags (max 1 000 kg per shift) into a ribbon blender. The bags are split over a grid within a closed extraction booth. The empty bags are contained within a "sausage", the entrance to which is inside the extraction booth. The powder is then mixed with typically < 5 % oil. The blended product is discharged into a bin and is then transferred via one of several bag lines into bags of 25 kg or smaller sealed sachets. The bags and sachets are stitched or sealed by hand. This 'damped' material will then be fed into either of the two processes below. All bagging lines are equipped with LEV and the operators wear dust masks, dust-proof overalls and gloves of leather and cotton, "cotton chrome gloves". An operator may typically handle as much as 5 tons of undamped diantimony trioxide per shift.

ii) **Conversion of dry diantimony trioxide powder into rubber pre-mix pellets**: The diantimony trioxide as a raw material is loaded manually from 25-kg bags (typically 5 bags per batch) into a mill tray opening that has a continually running extraction. The bags are slit, emptied and then placed into a bag baler that is also extracted. Inside the closed mill system the diantimony trioxide powder is incorporated into a binder. Operators wear filtered air flow visors, dust-proof overalls and gloves. Regular glove changing and hand-cleaning take place. During the course of an 8-hour shift, the mill operator will handle between 1 and 2 tons of diantimony trioxide. Due to operating practice and market needs it is unlikely that any given operator would spend more than two shifts per month at the mill in mixing such materials.

iii) **Conversion of dry diantimony trioxide powder into rubber pre-mix sheets**: In this process diantimony trioxide is used in the form of pre-dosed, ready to use plastic bags of 0.5 to 5 kg. The other compound materials are weighed on a balance and placed together with the diantimony trioxide bag on a conveyor belt feeding the mixer. Due to the use of sealed diantimony trioxide bags which do not require any manual handling, any dermal or inhalation exposure to diantimony trioxide is effectively minimised. Theoretically, approximately 0.5 to maximally 2.5 tons of diantimony trioxide-containing rubber compound may be mixed during a shift in batches of approx. 240 kg, so that approx. 2-10 such batches of diantimony trioxide-containing rubber compound may be produced during any single shift (note: not all rubber types will contain diantimony trioxide). The maximum amount of diantimony trioxide handled per shift is 300 kg (=10 kg diantimony trioxide per mixing incident). The manufacturing site that was visited stated that on average, diantimony trioxide is handled only 15 times per month. After loading, the mixer is closed with a ram. The mixing procedure is best described as "kneading" in a Banbury mixer. The high energy input to the kneading generates elevated temperatures in the mixer. The resulting homogenous rubber compound is subsequently ejected out of the closed system. Next the rubber pre-mix is loaded onto two successive mills from which the resulting milled rubber compound sheet emerges, and is mechanically cut to a pre-defined width, and is finally immersed continuously in an aqueous stearate bath to avoid sticking of the layers to each other, after which the sheet is dried and folded. The only contact with diantimony trioxide during this process is when the rubber sheet is manually loaded onto the immersion chamber, which occurs once per batch of sheet, requiring approximately one minute of contact, during which the worker wears leather gloves. Given the maximum number of sheet batches, this adds up to a maximum of 10 minutes of exposure per 8-hour shift.

Based on the replies of the questionnaire from 10 responders, the maximum duration of contact with diantimony trioxide as powder or in pre-dosed form, and the maximum amount handled by a single worker is presented in Table 4-15:

Table 4-15 Duration of handling and amount of diantimony trioxide powder

Maximum per shift for a single worker	Median	Mean	P90*
Duration [min]	18	24	53
Amount [kg]	200	560	1160

*90th percentile

Inhalation exposure

A study was conducted to assess current patterns and levels of exposure to rubber fume and dust from the rubber process in the British rubber industry (Dost et al., 2000). The data obtained from the general rubber goods, retread tyre (RT) and new tyre (NT) sectors were compared. A total of 179 rubber companies were visited and data were obtained from 52 general rubber goods, 29 RT and 7 NT manufacturers. The survey was conducted using a questionnaire and included a walk-through inspection of the workplace to assess the extent of use of control measures and the nature of work practices being employed. Exposure monitoring data, predominantly from the years 1995-1997, for rubber fume and rubber dust were obtained from these companies. Personal exposure to inhalable rubber process dust was measured, but not analysed for its composition. The authors

stated that dust samples reported for moulding and other post-extrusion operations represent exposure to nuisance dust.

Since diantimony trioxide is not used in tyre production, the dust exposure data focuses on production of general rubber goods. Data from post-extrusion processes presented by Dost et al. (2000) clearly indicate that exposures in subsequent process stages are almost an order of magnitude lower and Dost stated that dust measured during these processes does not represent "rubber process" dust but instead what he designated as "nuisance" (background) dust. For this reason, only the processes "weighing", "mixing", "milling", and "extrusion" are displayed in Table 4-16 below:

Table 4-16 Reported "rubber process dust" in general rubber goods companies, full shift [mg/m³]

Process	Min	P25	Median	Mean	P90[b)]	Max	No of data
Formulation							
Weighing (1)[a)]	0.4	-	4.2	4.9	-	14.4	18
Mixing (2)[a)]	0.2	-	1.2	2.6	-	18.6	18
Milling (3)[a)]	<0.1	-	0.8	1.3	-	9.2	46
Extrusion (4)[a)]	0.2	-	0.8	2.3	-	7.7	10
Processes 1-4	<0.1	-	1.0	2.3	6.0	18.6	92
All results [c)]	<0.1	-	0.7	1.5	-	29.5	335
Processing	**Min**	**P25**	**Median**	**Mean**	**P75**	**Max**	**No of data**
Shaping	0.02	0.4	0.7	0.9	0.9	14.3	116

a) Samples taken at processes other than 1-4 are considered not to be rubber process dust (see text above).
b) Since the raw data was not available it was not possible to calculate the 90th percentile, the pooled 90th percentile for the processes 1-4 was taken from (ECB, 2004).

c) The summary statistics of the pooled data set including "post-extrusion" processes in maintained for comparative reasons.

The data presented in the Table above have already been used in the European Risk Assessment Report on zinc oxide (ECB, 2004) in order to extrapolate to occupational inhalation exposure to zinc oxide in the rubber industry. In order to do so, a conversion factor of 0.25 from "rubber dust" to "zinc oxide" was derived by relating total dust (5.9 mg/m³) to zinc oxide dust (1.5 mg/m³) from measured values in the formulation stage in general rubber goods companies. The result is shown in Table 4-17. It should be noted that in the processing stage (shaping), the diantimony trioxide will be tightly bound within the rubber matrix and all emitted rubber dust particles therefore reflect the actual composition of the rubber mixture. Therefore the conversion factor has not been used to extrapolate exposure to zinc oxide for that stage.

Table 4-17 Extrapolated inhalation exposure to zinc oxide in general rubber goods companies, full shift [mg/m³]

Process	Min	Median	Mean	P90[b)]	Max
Weighing (1)[a)]	0.10	1.07	1.25	-	3.66
Mixing (2)[a)]	0.05	0.31	0.66	-	4.73
Milling (3)[a)]	0.01	0.20	0.33	-	2.34
Extrusion (4)[a)]	0.05	0.20	0.58	-	1.96

All results c)	0.01	0.25	0.58	1.53	4.73

a) Samples taken at processes other than 1-4 are considered not to be rubber process dust (see text above).
b) Since the raw data was not available it was not possible to calculate the 90th percentile, the pooled 90th percentile for the processes 1-4 was taken from (ECB, 2004).

c) The summary statistics of the pooled data set including "post-extrusion" processes in maintained for comparative reasons.

No similar measured data on diantimony trioxide concentration compared to total dust is available, nor is data available to compare the 'dustiness' of the components of total dust to that of diantimony trioxide. Diantimony trioxide is used in the same processes and in a similar way as zinc oxide. It has a similar particle size distribution. Diantimony trioxide is used in higher concentrations (2 to 10 % compared to 1 to 1.5 % of zinc oxide). It is classified as hazardous to health, which zinc oxide is not, which should mean that more care is taken to avoid exposure.

To account for the actual handling time, the full-shift values from Table 4-17 above have to be modified. The handling duration is 18 (median/typical) or 53 minutes (90th percentile/worst-case), respectively. Based on the process description and the above mentioned reported durations of handling, which correspond to only a minor portion of the shift, the data from Table 4-17 were converted to 8-hour time-weighted averages, as shown below (

Table 4-18):

Table 4-18 Extrapolated inhalation exposure to diantimony trioxide in general rubber goods companies, 8-hour TWA values [mg/m³]

Process	Typical handling duration (18 min)	Typical handling duration (18 min)	Worst case handling duration (53 min)	Worst case handling duration (53 min)
	Median	P90	Median	P90
Weighing	0.040	-	0.118	-
Mixing	0.012	-	0.034	-
Milling	0.008	-	0.022	-
Extrusion	0.008	-	0.022	-
All Processes	0.009	0.057	0.028	0.17

TWA: time-weighted average

As expected, inhalation exposures, extrapolated from the data set of Dost et al. (2000), are highest during weighing and mixing. The reasonable worst case for all processes in which "rubber process dust" is generated corresponds to 0.17 mg/m³ and 0.028 mg/m³ in the typical case, respectively. However, it is noted that Dost et al. (2000) does not differentiate the raw materials used and merely addresses "rubber process dust" as a whole. Beyond this, Vermeulen et al. (2000) mention the use

of several control measures, such as the use of “dust-free” chemicals, which reduced overall exposures considerably during the 9-year period between 1998-1997. Subsequent to this investigation period, it is specifically noted that in the meantime companies have adopted dust-reducing techniques on a wide scale. Finally, it should be noted that this extrapolation is based on a "total dust/zinc oxide dust" ratio which is very likely to overestimate the ratio of "total dust/diantimony trioxide dust" due to the current classification of diantimony trioxide as "Cat 3, carcinogen", which enforces stricter exposure control measures as employed for zinc oxide.

Vermeulen and coworkers evaluated exposures to inhalable particulates as well as dermal exposure to dust designated as “cyclohexane soluble matter” (CSM) in seven rubber manufacturing companies in 1988 and 1997 (Vermeulen et al., 2000). The identified exposure trends were used to study the effectiveness of control measures implemented over this nine year period. Sampling and analytical methodologies were identical in both surveys. Inhalable particulate matter was measured with a personal sampling head (n=486). The comparison of the exposure levels between 1988 and 1997 indicated a reduction by 5.7 % and 6.7 % per year for inhalable particulate and dermal exposure, respectively. The authors concluded that *"these results indicate that efforts taken to improve work conditions in the rubber manufacturing industry in The Netherlands over this decade have been successful in reducing both inhalable particulate and dermal contamination".* Personal exposure to inhalable particulates was reduced by 60 % over a nine year period at the workplace "compounding/mixing" which can be seen as an equivalent of the combination of "weighing" and "mixing" from the Dost et al. (2000) study. Whilst diantimony trioxide would most likely not be part of the evaluated “CSM” particulate, the general trend is likely also to apply to diantimony trioxide.

In the study by Dost et al. (2000) on exposure to rubber fume and rubber process dust in the British rubber industry, data relevant to the shaping and vulcanisation stage (i.e. curing) were reported, which similarly will be used as an analogous data base for the assessment of exposure to diantimony trioxide.

Shaping

Since only summary statistics (mean, median, range, and quartiles) were given in the publication, the 90th percentile was estimated by fitting a lognormal distribution to the 25th percentile, median, and 75th percentile. The fitted distribution is considered reasonable for use in the prediction of conservative inhalation exposures.

It is noted here that, after the formulation stage diantimony trioxide is tightly bound within the rubber matrix and all emitted rubber dust particles therefore reflect the actual composition of the rubber mixture. According to BLIC (2006) the used diantimony trioxide percentages in rubber mixtures range between 2 % and 10 %, whereas the typical use is stated by the OECD (2004) to be 3 %. Taking into account these percentages, the following inhalation exposures for shaping can be extrapolated from the Dost et al. (2000) data: If the concentration of diantimony trioxide in the rubber mixture is 3 % the inhalation exposure is 0.043 mg/m^3 (90th percentile) and 0.019 mg/m^3 (median), and if the concentration is 10 % the inhalation exposure is 0.14 mg/m^3 (90th percentile) and 0.064 mg/m^3 (median).

Curing

Typical (median) 8-hour TWA values for exposure to “rubber process dust” were measured (n=2) at 0.25 mg/m³ for curing (Dost et al., 2000). Applying the same conversion as for shaping above, then

exposures to diantimony trioxide can be estimated to 0.008 mg/m³ (typical diantimony trioxide percentage) and 0.025 mg/m³ (worst case diantimony trioxide percentage) accordingly. However, in view of the limited data set, it is proposed to base the assessment on the more conservative and also more extensive data set obtained for shaping (moulding).

Dermal exposure

No sector-specific data on dermal exposure to diantimony trioxide is available for either the formulation or processing stages.

An extrapolation from data measured during bag filling at a diantimony trioxide production site (Hughson, 2005) is considered feasible, assuming that bag filling and loading/unloading yield similar exposures, as has been shown by Hughson and Cherrie (2005) for a similar substance (zinc oxide). In this investigation, occupational dermal exposure to diantimony trioxide during production and subsequent handling was measured with the aid of the wipe sampling method. Workers were involved either in packaging or in other activities for a full shift. These results can be summarised as follows (for details, please see the original report) in Table 4-19:

Table 4-19 Dermal exposure to diantimony trioxide during production of diantimony trioxide (µg/cm²)

Diantimony trioxide production	No of data	Min	Median	P90*	Max
Packaging, full shift	51	1	16	35	115
Refuming	18	1	13	24	41
Converter	36	1	5	18	30

* 90th percentile

These values represent full shift exposures and handling powder that is 100 % diantimony trioxide. To extrapolate to dermal exposure in the rubber industry; the 90th percentile duration of exposure (53 min) and dust of 100 % diantimony trioxide will be used for the formulation stage and a full shift, but dust with the maximum percentage diantimony trioxide used in the rubber (10 %) will be used for the processing stage. The results of this extrapolation are shown in Table 4-20.

Table 4-20 Dermal exposure to diantimony trioxide in the rubber industry, extrapolated data, [µg/cm²]

Rubber industry	Min	Median	P90*	Max
Formulation	< 1	2	4	13
Processing	<1	1.6	3.5	11.5

* 90th percentile

Evaluation of exposure

Inhalation exposure

Measured data

There are no measured data on inhalation exposure of diantimony trioxide in the rubber industry.

Analogous/surrogate data

Formulation

It is proposed by Industry to determine the exposure to diantimony trioxide in the rubber industry through extrapolation from reported exposure to "rubber process dust" in the "general rubber goods" industry in the UK (Dost et al., 2000). It is also proposed to extrapolate this further by modifying for the duration of handling of diantimony trioxide. Another assumption is that the level of diantimony trioxide in the "rubber dust" is equal to that of zinc oxide. Each step of the proposed extrapolation contains uncertainties and will in total contribute to a significant uncertainty in the estimation of the exposure to diantimony trioxide. The approach suggested by Industry is therefore not considered justified.

First of all, the assumption that diantimony trioxide is equal to zinc oxide is not well substantiated. The content of zinc oxide in rubber goods is approximately 1-1.5 %, compared to 3 % (maximum 10 %) of diantimony trioxide in flame retarded rubber. That lower exposure of diantimony trioxide compared to zinc oxide is to be expected because of current classification of diantimony trioxide as a carcinogen (Cat 3) is not well substantiated either.

Further, the suggested exposure value to total rubber dust is based on measured exposure during four different processes (Weighing, Mixing, Milling and Extrusion). However, the measured exposures during Milling and Extrusion, where diantimony trioxide is not handled directly, are lower than during Weighing and Mixing. As it is not clear whether the reported data on duration of handling of diantimony trioxide fully correspond to the exposure data of total rubber dust, i.e. covering all four process steps, we believe that the recalculation of the extrapolated 90th percentile value (for all processes) to 8-hour time-weighted averages may lead to an underestimation of the actual exposure. As only 25 % of the sites, consuming less than 50% of the total amount of diantimony trioxide, responded to the questionnaire, the reported data on duration is also somewhat limited. Furthermore, it is unclear what form of diantimony trioxide (powder or pre-dosed) that was used at the sites that provided data on duration. It should also be noted that there is a large variation in the reported maximum amount handled per shift, ranging from 10 to 3000 kg.

Instead of the procedure suggested by Industry it is considered justified to use analogous data from the Plastics sector, where measured data collected during raw material handling are available (seeTable 4-8 in section 4.1.1.2.4). These data are considered to be reliable and representative of the real situation for workers during an 8-hr shift in the Plastics sector (typical exposure 0.13 mg/m^3 and RWC 0.57 mg/m^3). The process descriptions of formulation in the Rubber industry and raw material handling in the Plastics industry both indicate manual handling of 25-kg sacks containing diantimony trioxide. However, the duration of handling is shorter in the Rubber sector (2-75 min; mean 24 min) compared to the Plastics sector (2-300 min; mean 61 min). Therefore it is considered

appropriate to modify the exposure data from Plastics and use the various mean duration times as a basis for this correction. This would lead to the following values for Rubber - formulation:

Typical: 24 min/61 min x 0.13 mg/m^3 = 0.051 mg/m^3

RWC: 24 min/61 min x 0.57 mg/m^3 = 0.22 mg/m^3

The values taken forward to risk characterisation are therefore 0.051 mg/m^3 as a typical exposure and 0.22 mg/m^3 as a reasonable worst case.

Processing

An extrapolation for determination of the exposure to diantimony trioxide based on measured exposure to rubber fume and rubber process dust in the British rubber industry is suggested by Industry. This is considered reasonable for this use, as also the extrapolation to values for typical and reasonable worst cases for diantimony trioxide. Accordingly, the median (typical) and 90th percentile (RWC) exposures are 0.064 and 0.14 mg/m^3, respectively.

Modelled data

No modelling for inhalation exposure to diantimony trioxide in the rubber industry has been performed.

Conclusions

The exposure assessment for formulation, proposed by Industry, is not considered justified for reasons given above. Instead analogues data from the plastics sector are used. For processing, the Industry proposal to use extrapolated data is used.

Dermal exposure

Measured data

Measured data on dermal exposure do not exist.

Analogous/surrogate data

Formulation

Considering that handling of diantimony trioxide in this sector does not occur full-shift and that for the remainder of a shift workers will perform tasks in other areas of the plant not involving diantimony trioxide exposure, it is proposed by Industry as a reasonable worst-case to carry forward the highest levels of dermal exposures measured for final handling (packaging) during production of diantimony trioxide, but modified by the duration of handling during a shift (53 minutes). This is considered justified, but the value for duration is rounded off to 60 minutes. This value is used for recalculation of the dermal exposure values from production of diantimony trioxide. However, original Hughson (2005) data is used instead of the slightly modified values in the Industry report. Accordingly, this gives the following values for dermal exposure:

Typical: 0.018 mg/cm^2/day * (60/480) = 0.0023 mg/cm^2/day

RWC: 0.031 mg/cm^2/day * (60/480) = 0.0039 mg/cm^2/day

Processing

Considering the final content of diantimony trioxide in rubber goods, it is proposed by Industry, and considered justified, as a reasonable worst-case to carry forward the highest levels of dermal exposures measured for final handling (packaging) during production of diantimony trioxide during a full-shift, but modified by the highest content in the goods (10 %). Thus, the values for dermal exposure under these conditions to be taken forward are:

Typical: 0.018 mg/cm^2/day * 0.10 = 0.0018 mg/cm^2/day

RWC: 0.031 mg/cm^2/day * 0.10 = 0.0031 mg/cm^2/day

Modelled data

No modelling for dermal exposure to diantimony trioxide in the rubber industry has been performed.

Conclusions

Dermal exposure measured for final handling during production of diantimony trioxide, but modified by the duration of handling during a shift (60 minutes) in the case of formulation, and modified by the diantimony trioxide content in rubber goods (10 %) in the case of processing, are taken forward to the risk characterisation.

Values taken forward to risk characterisation

The bodyweight of the worker is 70 kg and the exposed dermal area is 2000 cm^2.

DRAFT

Inhalation exposure
Formulation
Typical: 0.051 mg/m^3
RWC: 0.22 mg/m^3

Processing
Typical: 0.064 mg/m^3
RWC: 0.14 mg/m^3

Dermal exposure
Formulation
Typical: 2.3 µg/cm^2 which corresponds to 0.066 mg/kg/day
RWC: 3.9 µg/cm^2 which corresponds to 0.11 mg/kg/day

Processing
Typical: 1.8 µg/cm^2 which corresponds to 0.051 mg/kg/day
RWC: 3.1 µg/cm^2 which corresponds to 0.089 mg/kg/day

4.1.1.2.8 Scenario 7: Use in production of crystal glass

Diantimony trioxide is used in manufacturing of glass as a fining agent or as a degasser to remove bubbles from the glass. Typical concentrations of diantimony trioxide in finished glass are less than 1 %. A survey was conducted by EBRC for Industry to collect data relevant for the assessment of occupational exposure to diantimony trioxide in production of crystal glass (EBRC, 2006b), see chapter 4.1.1.2.1.

The report submitted by Industry has been summarised in the following sections, and the conclusions proposed by Industry are also presented. The exposure information is further evaluated before conclusions on values for risk characterisation are presented.

Summary of Industry information

The users were identified from the customer databases of primary diantimony trioxide producers. Five companies in Europe use diantimony trioxide in glass production and three of these responded to the questionnaire. The companies reported to use diantimony trioxide in powder form of medium particle size grades in the range 1.3 - 1.5 µm or approx 5-6 µm, physical particle size.

Five relevant workplace categories at sites producing crystal glass were identified and are shown in Table 4-21.

Table 4-21 Overview of tasks in crystal glass production

Workplace category	Workplace description	Tasks performed
Cry1	Raw material handling	Raw material delivery, batch formulation, pot filling, melting
Cry2	Forming processes	Operation of multi-pot/semi-automated cold-top furnace, blowing operations
Cry3	Cutting processes	Finishing, manual and automated cutting operations
Cry4	Polishing processes	Acid polishing

Cry5	Others	Storage, shipment, repair, cleaning / maintenance*, quality control, etc

*Routine cleaning and maintenance is part of the task assigned to workplace category 1-4; cleaning operations under workplace category 5 refer to staff that perform only such cleaning and maintenance work.

Process description

Raw material handling (Cry1):

The production process starts with the handling and storage of incoming raw materials. Raw materials are usually received separately at a facility often designated as batch plant. They may then be crushed to a granular size in order to facilitate the melting process and reduce dusting. From the storage bins, they are transferred, usually by an automated system, to a weighing balance, and subsequently mixed with recycled glass to ensure homogenous melting. The amounts handled at large facilities may be as much as 15 tons per day or more. Loading of dusting materials is normally done through a chute fitted with extraction devices to control dust formation.

There will be a certain degree of manual operations in some facilities. In these cases the exposure pattern will be discontinuous. LEV will be present and the use of gloves and RPE is reported to be mandatory. In contrast modern large facilities with continuous manufacturing have diantimony trioxide delivered in dry powder form packed in big bags of 1000 kg, which is then unloaded to the storage silos with the aid of pneumatic handling systems and/or elevators. Batch material is then usually prepared and transported using fully automated systems. Alternatively, in some plants the raw materials used for preparation of batches are supplied in pellet form or as pre-mixed batches, with subsequent mixing with silica and automatic feeding to the furnace.

Melting

The actual melting process generally lasts between 24 and 36 hours and takes place at temperatures in excess of 1400 °C. At the end of the melting process, there is a conditioning stage during which the molten glass is lowered to a temperature of 1100-1200 °C.

Two types of melting processes can be distinguished, traditional pot process and continuous tank melters. In the former, the ready-mixed batch will be loaded by shovel or semi-automatically into ceramic pots, each holding some 750 kg of molten glass, heated by gas or oil. This melting process is a batch-process. LEV is present and the use of gloves and RPE is reported to be mandatory. The latter process is continuous with the take-off of crystal in one end, being balanced with the raw-material charging at the other end, to maintain a constant level of molten glass within the system.

Forming processes (Cry2)

At this process stage the molten glass is drawn from the furnace and worked on in forming machines by a variety of methods. These include pressing, blowing, drawing or rolling to produce the desired product, or else an amount of glass is extracted with a rod after which it is hand-blown to its final appearance. After the forming process, the glass object is transferred to a kiln where it is annealed at temperatures above 500 °C and cooled to room temperature.

The traditional pot process requires considerably more manual handling than the continuous tank melters. The latter process generally yields lower emissions to the workplace.

Cutting and finishing processes (Cry3)

At this stage the crystal ware is usually cut from a stem, where applicable, and is further subjected to cutting operations designed to add ornamental features or a decorative cut pattern. This may be done automatically, but many producers maintain a large staff of manual cutters. The actual cutting is done with the aid of diamond cutting wheels including a water rinsing system. Although local extraction devices are used, this process nevertheless creates a fine mist of very fine glass particles. The amount of material handled per worker is difficult to assess, but could vary between 10 and 300 kg per shift. The exposure is during full shifts and the contact level is extensive.

Polishing processes (Cry4)

The surface of the glass object may be improved by flame polishing, mechanical polishing using a series of wheels, or chemically polished by immersing the glass in a mixture of hydrofluoric acid (3 %) and sulphuric acid (70 %) at 60 °C for a period of 40 minutes. The latter is normally a semi-automated process, but involves manual loading and unloading of glass articles to the acid baths. Approximately 10,000 items may be handled per shift in large facilities.

The exposure is discontinuous. LEV is present and the use of gloves and goggles is reported to be mandatory.

Other (Cry5)

Storage and shipment of raw materials and finished goods which are usually in sealed packaging, is unlikely to give rise to high levels of exposure. The same applies to quality control. However engineering and cleaning and maintenance work are known to give rise to considerable exposure levels in practice. This type of work should be discriminated from minor routine cleaning and maintenance work done by each worker as part of his tasks. The latter is included in Cry1 to Cry4.

For cleaning, either wet or dry vacuum cleaning apparatus is used, largely excluding dermal contact. Gloves and RPE are reported to be mandatory.

Inhalation exposure

Measured data

Only one company reported five single measurements of occupational inhalation exposure to diantimony trioxide. National occupational health authorities conducted these measurements in June 2004.

During the sampling period the tasks of the personnel involved handling of different raw materials, i.e. dosing of the various materials to silos for further automatic transportation in the plant. The sampled personnel carried a test device for 8 hours per day on two different days. The measured particulate fraction was the inhalable fraction. The result from these measurements (n=5) was a median diantimony trioxide inhalation exposure of 0.007 mg/m^3. The 90th percentile and maximum inhalation exposures were 0.023 mg/m^3 and 0.033 mg/m^3, respectively.

Analogous/surrogate data

As only one set of recent exposure data for diantimony trioxide is available for this industry sector it is proposed by Industry to also use analogous data from occupational exposure reported in the draft voluntary risk assessment on lead (LDAI, 2006) taking into consideration the percentages of the respective substances. This approach is deemed to be justified by Industry because:

- Lead oxide and diantimony trioxide are received predominantly as powders and are compounded into the glass 'batch-matrix' containing at least 24 % lead oxide and less than 1 % diantimony trioxide, after which the same ratio is maintained homogenously throughout the entire production process.
- The particle size and dustiness properties of diantimony trioxide and lead oxides are similar.

The aerodynamic diameter and dustiness of a range of diantimony trioxide products relevant for the glass industry have been compared to the lead oxides Litharge and red lead oxide used in the same industry. The results for diantimony trioxide mostly fall within the range defined by the two lead oxides. It should be noted that once the diantimony trioxide and lead oxides have been compounded into the 'batch-matrix' this comparison is no longer relevant; particles formed in later stages e.g. cutting will contain both substances. Exposure data on lead from the draft voluntary risk assessment is used to estimate the exposure to diantimony trioxide at the same workplaces. It is based on data collected from the crystal glass industry in the period 1998 to 2001.

To extrapolate these values from lead to diantimony trioxide two factors are applied:

- 1.090 to convert milligram lead to milligram lead oxide, assumed equal mix of PbO with factor = 1.077 and Pb_3O_4 with factor = 1.103.
- 1/24 = 0.042 to take into account that lead oxide is used in min 24 %, and diantimony trioxide in max 1 % in the glass

Applying these factors gives the values for exposure to diantimony trioxide shown in Table4-22.

Table4-22 Inhalation exposure to diantimony trioxide, extrapolated from lead exposure, [mg Sb_2O_3/m^3]

Workplace category*	Median	P90**	Maximum
Cry1	0.001	0.011	0.10
Cry2	0.001	0.003	0.041
Cry3	0.003	0.015	0.050
Cry4	0.001	0.001	0.002
Cry5	0.001	0.005	0.005
Not allocated	0.000	n a	0.000
All data	0.002	0.011	0.10

*For explanation of workplace category, see above

** 90th percentile

The exposure is marginally higher at the work place where "cutting" is performed (Cry3). Given that antimony and lead are contained in a glass matrix at a fixed ratio at this process stage, read-across from the data extrapolated from lead to diantimony trioxide is considered justified by

Industry. The median value of 0.003 mg/m^3 and 90th percentile of 0.015 mg/m^3 are proposed to be taken forward to risk characterisation.

Dermal exposure

Measured data

No dermal exposure data for diantimony trioxide is available for this industry sector.

Analogous/surrogate data

A study reporting dermal monitoring data of lead and corresponding blood lead values is available from a UK HSE report (Wheeler and Sams, 1999). In this study, two different production sites for lead crystal glass were visited. The areas that were studied were mixing and processing (Cry1), glasshouse (Cry2), cutting shop (Cry3), and marking shop (Cry5). A total of 25 workers were monitored from all facilities together, which were selected in order to cover a range of tasks and correspondingly different levels of exposure. Dermal contamination was determined with a previously validated "bag-wash" method, in which workers were requested to insert their hands into a bag with ultra-pure distilled water, followed by agitation for two minutes. Four such samples were taken from each volunteer, one pre-shift and three prior to each break during their shift. The three last samples were pooled to form a "shift hand exposure". To extrapolate these values from lead to diantimony trioxide the same conversion as for inhalation exposure (see above) is applied. The resulting values are shown in Table 4-23. It is proposed by Industry to take forward exposure levels reported for the highest exposure group, i.e. workers involved in "cutting" (Cry3). Accordingly, a typical value of 3 µg/cm^2 (median) and a worst case value of 11 µg/cm^2 (90th percentile) are proposed to be taken forward to risk characterisation.

Table 4-23 Dermal exposure to diantimony trioxide, extrapolated from lead exposure

Workplace	Hand exposure [µg Sb_2O_3 / cm^2]		
	Median	P90**	Max
Cry1 (mixing)	1	n.a.*	1
Cry1 (processing)	<1	<1	<1
Cry2 (glasshouse)	<1	1	1
Cry3 (cutting shop)	3	11	12
Cry5 (marking shop)	<1	<1	<1
All data	<1	7	12

** not applicable (P90 not calculable)

** 90th percentile

Evaluation of exposure

Inhalation exposure

Measured data

The available measured data on inhalation exposure to diantimony trioxide in this industry sector is limited. Sector-specific measurements were only provided from one company. The data set consists of only five measurements in one workplace category (raw material handling, Cry1). Accordingly, the data can not be regarded as representative for this industry sector.

Analogous/surrogate data

Industry proposes to use analogous data from the lead Risk Assessment Report (LDAI, 2006). This should be justified as powders of lead oxide (at least 24 %) as well as diantimony trioxide (less than 1 %) are compounded into the glass "batch-matrix" and the same ratio is maintained homogenously throughout the entire production process. In addition, the particle size and dustiness properties of diantimony trioxide and lead oxides are similar.

The comparison with lead exposure in glass industry indicates that the exposure is marginally higher in the workplace category where cutting is performed, Cry3. Given that antimony and lead are contained in a glass matrix at a fixed ratio at this process stage, read-across from the data extrapolated from lead to diantimony trioxide is regarded as justified. Also, the typical and reasonable worst case values which are proposed to be taken forward to risk characterisation correspond reasonably well with the measured values on diantimony trioxide exposure. Therefore, the Industry proposal to use analogous data from the lead Risk Assessment Report (LDAI, 2006) is considered appropriate. Accordingly, the typical (median) and RWC (90^{th} percentile) exposures are 0.003 and 0.015 mg/m^3, respectively.

Conclusions

The values proposed by Industry, based on analogous data from the lead Risk assessment report, are considered appropriate and will be taken forward to risk characterisation.

Dermal exposure

Measured data

No dermal exposure data for diantimony trioxide is available for this industry sector.

Analogous/surrogate data

Similarly to inhalation exposure (see above), Industry proposes to use analogous data on lead exposure (Wheeler and Sams, 1999) for estimation of dermal exposure to diantimony trioxide in this industry sector. As for inhalation exposure, the Industry proposal to use these analogous data is considered appropriate. Accordingly, the typical (median) and RWC (90^{th} percentile) exposures are 3 and 11 $\mu g/cm^2$, respectively.

Conclusions

The values proposed by Industry, based on analogous data on lead exposure, are considered appropriate and will be taken forward to risk characterisation.

Values taken forward to risk characterisation

The bodyweight of the worker is 70 kg and the exposed dermal area is 2000 cm^2.

Inhalation exposure
Cutting
Typical: 0.003 mg/m^3
RWC: 0.015 mg/m^3

Dermal exposure
Cutting
Typical: 3 $\mu g/cm^2$ which corresponds to 0.086 mg/kg/day
RWC: 11 $\mu g/cm^2$ which corresponds to 0.31 mg/kg/day

4.1.1.3 Consumer exposure

4.1.1.3.1 Introduction

There is no known direct private use of diantimony trioxide as the substance as such. However, diantimony trioxide is used in several products, some of which are available to consumers. Some examples of end products containing diantimony trioxide and/or antimony derived from diantimony trioxide are:

- PET
- flat and pile upholstered furniture (residential and commercial furniture)
- cuddly toys
- upholstery seatings and automobile interior textiles in private and public transportation, draperies, and wall coverings
- electrical and electronic equipment e.g. distribution boxes for electrical lines
- polyvinyl chloride wire, cable and textile coating

Diantimony trioxide is chemically bound within the polymer matrix of PET. During the final extrusion process of PET in bottle manufacturing from granular PET resin decomposition of the diantimony trioxide catalysts can occur resulting in formation of metallic antimony (Biros et al., 2002). It can be anticipated that the presence of antimony in PET-bottled drinking water (Shotyk and Krachler, 2007) is due to leaching of antimony from the surface of the PET bottle matrix.

The release of diantimony trioxide from the surface of products to atmospheres may be a potential way of exposure. Due to negligible volatility of diantimony trioxide, vapour release is not relevant. Instead, diantimony trioxide may be released as dust due to wear or abrasion. Direct dermal contact with products containing diantimony trioxide may give dermal exposure.

No measured data on consumers' exposure were submitted by the Industry. The available mathematical models for consumer exposure are not appropriate for modelling intake of metals and metal compounds such as diantimony trioxide. The information on the release of diantimony trioxide from the materials and on the actual use of the materials, giving support for calculations on the exposure, is very limited. Four scenarios for consumer's exposure to diantimony trioxide are presented here; "drinking from a PET-bottle" and "sucking on cuddly toys" (oral exposure), "indoor air" (inhalation and oral exposure) and "sitting on upholstery fabric" (dermal exposure).

It should however be noted that although originating from the use of diantimony trioxide the actual oral exposure via PET-bottles and cuddly toys is not to diantimony trioxide itself but to the antimony ions. However, to calculate the concentration of diantimony trioxide, the measured levels of antimony are adjusted with a factor of 1.2 (correction for molecular weight). This approach is taken in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide.

4.1.1.3.2 Scenario No. 1: PET-bottle

According to EU legislation (Commission directive 2002/72/EC) there is a limit on the amount of antimony that is allowed to migrate from PET into food (EC, 2002). This migration limit for diantimony trioxide is set to 40 µg/kg food, expressed as antimony (migration limit amended by directive 2005/79/EC). The concentration of antimony in water intended for human consumption is stipulated as maximum 5 µg/L in Council directive 98/83/EC (EC, 1998). In WHO's Guideline values for chemicals that are of health significance in drinking-water, the value for antimony is 20 µg/L (WHO, 2006).

The Food Standards Agency (UK) has made a study of levels of antimony and other elements in bottled drinking water (UK Food Standards Agency, 2002). 161 samples, covering a total of 101 brands, of bottled water sold in the UK in 2001/02 were collected and analysed (104 mineral water, 44 spring water and 13 bottled drinking water). The bottle material is not stated in the report, but based on the market-share of PET in this segment, it is highly likely that a fairly high proportion would be bottles of PET. All samples were analysed by two independent laboratories. The different laboratories reported limits of detections (LOD) of 0.03 µg/L and 0.1 µg/L, respectively. Antimony was detected in 87 of the samples. As a conservative approach, all samples below LOD were assigned a value of 0.1 µg/L, i.e. the higher of the two LODs. The median and 90th percentile of all samples were then calculated to 0.2 µg/L and 0.5 µg/L, respectively. The source and form of antimony was not investigated. The measured antimony may have leached from the PET-bottles or may come from other sources.

(Shotyk et al. 2006) performed a study in order to determine whether the elevated concentrations of antimony in bottled water were a reflection of geologically and mineralogically diversity of the source region or whether they have been contaminated by the bottles used to contain them. 15 popular brands of water from Canada as well as 48 brands from Europe (Germany (13), France (9), Switzerland (4), Finland (4), Czech Republic (4), Denmark (3), Spain (3), Poland (2), Belgium (2), The Netherlands (2), and Italy (2) were analysed. In addition, water from one of the German sources prior to filtration and bottling was available. For comparison natural water from Ontario, Canada bottled commercially in polypropylene was analysed. The German samples contained 253-546 ng/L. Water collected from one of the German sources contained 3.8±0.9 ng/L prior to bottling compared with 359±54 ng/L from water purchased in PET bottles. This same brand of water

contained 626±15 ng/L after 3 months of storage at room temperature. The median concentration of antimony from the other EU countries was 343 ng/L. Natural water from Canada bottled in PET contained 156±86 ng/L, while water bottled in polypropylene contained 8.2±0.9 ng/L. The pristine groundwater in Ontario, Canada contained 2.2±1.2 ng/L. When the pristine water was stored in PET bottles from Germany the concentrations were elevated to 50±17 ng/L after 37 days in refrigerator, and 566 ng/L after six months storage in room temperature. The results show that there is a profound leaching of antimony from the PET container into the water. Moreover the data suggest that the antimony concentration in the waters bottled in PET is independent of the natural abundance in the source water, but rather dependent on the duration of storage. Other factors mentioned that might affect the rate of release are pH of the water, temperature, presence of cations and anions, and recycling and reuse.

In a follow-up study the effect of storage on leaching of antimony from PET bottles was further studied (Shotyk and Krachler, 2007). The bottles were stored at room temperature and directly exposed to light from both ambient daylight and artificial sources during daytime. As reported by the authors, the increase of antimony concentrations, after 6 months of storage, were 19 % in 14 brands of bottled water from Canada and 90 % in the 48 brands of water from 11 European countries. According to a Swedish producer of bottled water (Carlsberg) the guaranteed keeping quality for carbonated and still water in 150 cl PET-bottles is 7 and 8 months, respectively. Considering the fact that bottled water may be stored for several months before it is actually bought and consumed, the effect of storage on the concentrations of antimony in the water can not be neglected.

After personal communication with the authors, raw data on the concentrations of antimony in separate samples of bottled waters was provided. Thirteen data sets, from both before and after 6 months of storage, on the Canadian bottled waters and 41 data sets on the European bottled waters were available for evaluation. The results of this evaluation showed that the median concentration of antimony in the Canadian water samples increased from 156 to 167 ng/L after 6 months of storage and the 90th percentile concentrations increased from 303 ng/L to 359 ng/L. In the European water samples the median concentration increased from 343 ng/L to 641 ng/L and the 90th percentile concentrations increased from 535 ng/L to 879 ng/L. The maximum concentrations were found in one European sample which increased from 725 ng/L to 1513 ng/L after 6 months. In Shotyk and Krachler (2007), it was also reported that the concentration of antimony in pristine groundwater (n=3) increased from 1.7±0.4 ng/L to 26.6±2.3 ng/L and to 281±38 ng/L, after 6 months storage in PET bottles bought in Canada and Germany, respectively. To summarise, more antimony seems to be released from the European water bottles than the Canadian bottles, but the degree of change in antimony concentrations upon storage is variable.

In the same study, antimony concentrations in 69 brands of bottled water from 16 additional countries were determined. The reported median concentration was 216 ng/L, ranging from 8.9 to 2570 ng/L. The bottled water containing 2570 ng/L was from Peru. Two brands of bottled water from Europe, but purchased in Hong Kong, yielded 1650 and 1990 ng/L. According to the authors, the latter sample was a brand of bottled water from France which previously also had been bought in Germany and found to contain 725 ng/L and 1510 ng/L, respectively before and after storage for 6 months.

The migration of antimony from PET into food simulants has been studied by Fordham et al, (1995). The study was made on two different samples of food contact grade PET-polymer from European manufacturers. The polymer in the form of pellets was submerged in three food simulants (3 % acetic acid in water, 15 % ethanol in water and olive oil) and subjected to two experimental

conditions of temperature and time (40 °C for 10 days or 100 °C for 2 hours), following which the antimony content in the food simulant was analysed using ICP-MS. In the experiment, 3 g of PET was submerged in 20 ml of food simulant in each case. The content of antimony in the PET-polymer (as delivered) was also determined. The results are shown in Table 4-24. It seems that the temperature is an important factor and that the concentration is about the same in the solutions of 3 % acetic acid and 15 % ethanol.

Table 4-24 Content of antimony in polymer and concentration of antimony in food stimulant (Fordham et al., 1995)

Antimony in polymer (mg/kg)	3% (w/v) Acetic acid in water (µg Sb/kg)		15% (v/v) Ethanol in water (µg Sb/kg)	Olive oil (µg Sb/kg)	
	40°C / 10 days	100°C / 2 h	40°C / 10 days	40°C / 10 days	100°C / 2 h
160	2.7	3.9	2.3	< 10	< 10
230	1.2	2.6	1.1	< 10	< 10

Conclusions

Antimony concentrations in PET-bottled water from 11 European countries, before and after 6 months of storage (Shotyk and Krachler, 2007) is the most recent and representative data and therefore used to estimate a typical and a realistic worst-case exposure to diantimony trioxide from drinking a soft-drink or water that has been contained in a PET-bottle. The median concentration of antimony (0.343 µg/L) found in PET-bottled waters before storage for 6 months is estimated as the typical concentration of antimony. This value is supported by the concentration of antimony in bottled water from the UK (median: 0.2 µg/L and 90th percentile: 0.5 µg/L) in which the effect of storage is not considered. Due to the fact that bottled water may be stored for several months before it is bought and consumed the effect of storage can not be neglected when considering a realistic worst-case exposure. Therefore, the reasonable worst case concentration is estimated to 0.879 µg/L which is the 90th percentile of the concentration of antimony in water stored in PET bottles for 6 months. It is assumed that an adult is drinking 2 litres (V) per day giving a typical and a reasonable worst-case exposure of 0.686 and 1.758 µg antimony/day, respectively.

After conversion from antimony to diantimony trioxide this corresponds to 0.823 and 2.11 µg/day, respectively. With a body weight of 60 kg this implies a typical consumer exposure of 0.014 µg/kg/day (0.686 µg/day·1.2)/60 kg) and a reasonable worst-case consumer exposure of 0.035 µg/kg/day (1.758 µg/day·1.2)/60 kg). These values are taken forward to the risk characterisation.

It must however be considered that the actual exposure is not in the form of diantimony trioxide. The conversion to diantimony trioxide is made in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide.

It should also be noticed that liquids other than water might increase the leakage from the PET bottle. However, there is no data available on this. It was not considered relevant to make an exposure assessment for exposure via inhalation and dermal exposure for this scenario.

4.1.1.3.3 Scenario No. 2: Fabrics

Polyester fibres

The Danish environmental protection agency has made a survey of chemical compounds in textile fabrics (Danish Environmental Protection Agency, 2003a). The fabrics were purchased from different shops in Copenhagen and environs and tested for a number of chemical compounds and metals, including antimony. Furthermore, measurements of extractable antimony in these fabrics by artificial saliva and perspiration were made.

The test of extractable antimony was made as follows: 2 g of samples were extracted with 100 ml of solution at 40 °C for 1 hour. The solution was either artificial saliva or artificial perspiration. The results are presented in Table 4-25. Both results are listed if the standard deviation is higher than 20 % of the average of the determination in duplicate. Standard deviation of this magnitude is probably a result of non-homogeneity in the textile sample.

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Table 4-25 Test results for total antimony compared with extractable antimony, all results in mg/kg. The number after "±" states the statistical standard deviation. <LOD = below limit of detection for extractable antimony (0.5 for artificial saliva and 1.0 for artificial perspiration).

Sample	Total antimony	Extractable antimony (artificial saliva)	Extractable antimony (artificial perspiration)
100 % PET (colourful*)	110 ± 8.7	<LOD	<LOD
Cotton/PET (napkins)	35 ± 4.5	<LOD	not measured
100 % PET (trousers)**	8.3	not measured	<LOD
	10.3		
100 % PET (blouse)**	35	not measured	3.5 ± 0.1
	48		
100 % PET (underwear)	7.0 ±0.5	<LOD	<LOD
100 % PET (fleece)	27 ± 1.6	<LOD	<LOD

* a textile intended for use in apparel
** both analytical results are listed because standard deviation is higher than 20 % of the average of the determination in duplicate

The type of fabrics and the level of total antimony indicate that it is antimony from the production of polyester (as catalyst), and not antimony as flame-retardant, that is present in the fabrics. Extractable antimony was only found in one sample of artificial perspiration, and the amount extracted was 10 % of the total antimony in the sample.

Flame retardant treated textiles

The University of Surrey has made a study on flame-retardant release from textiles for BSEF and IAOIA (BSEF and IAOIA, 2006). As part of this study, a test to simulate the dermal exposure to diantimony trioxide from contact with flame-retarded textiles was performed. Two representative textile samples were used, HBCD-ATO and DBDE-ATO. The theoretical content of diantimony trioxide in the HBCD-ATO textile was 3.6 % (m/m) with a flame-retardant area density of 0.99 mg/cm^2, the corresponding numbers for the DBDE-ATO textile was 4.3 % (m/m) and 1.2 mg/cm^2.

The samples were aged thermally and with UV-radiation. The thermal ageing was made in an oven at 90 °C for five weeks with ambient humidity. The UV-ageing was done with UVA (340 nm) according to BS5782 Part 5 for 30 days at ambient conditions.

The Contact blotting test was used to simulate the dermal exposure to diantimony trioxide from contact with flame-retarded textiles. This test has been adapted by the US Consumer Safety Commission (US Consumer Safety Commission, 1994). The textile was immersed in the fluid with a wetted filter paper in contact with the upper face of the textile (back-coating at the rear). The amount of liquid is just sufficient to cover the textile and wet the filter paper. The combined fabric and filter paper was allowed to dry after which the filter paper was analysed for the uptake of diantimony trioxide.

Table 4-26 Summary of diantimony trioxide released in the Contact blotting test, expressed as average release (m/m) ± RSD %, with respect to original diantimony trioxide content

	HBCD-ATO			DBDE-ATO		
	Unaged	**Thermally aged**	**UV aged**	**Unaged**	**Thermally aged**	**UV aged**
Distilled water	$1.3 \pm 0.03 \cdot 10^{-4}$	$6.3 \pm 0.5 \cdot 10^{-4}$	$0.89 \cdot 10^{-3}$	$6.6 \pm 0.5 \cdot 10^{-4}$	$1.1 \pm 0.5 \cdot 10^{-3}$	$2.1 \cdot 10^{-3}$
Citric acid (5% in water)	$1.4 \pm 0.07 \cdot 10^{-3}$	$2.3 \pm 2.0 \cdot 10^{-3}$	$2.60 \cdot 10^{-3}$	$1.1 \pm 0.03 \cdot 10^{-3}$	$4.7 \pm 0.2 \cdot 10^{-3}$	$8.8 \cdot 10^{-3}$
Artificial saliva	$1.5 \pm 0.02 \cdot 10^{-3}$	-	-	$7.5 \pm 0.30 \cdot 10^{-4}$	-	-

The worst case dermal exposure scenario is based on the exposure assessment made by the National Research Council (2000). The assessment has been made with default values of skin area and weight of a woman. The following assumptions were made: The dermal exposure assessment assumes that a person weighing 60 kg (W_a) spends $1/4^{th}$ of his or her time sitting on furniture upholstery treated with diantimony trioxide (f_c =0.25), and that $1/4^{th}$ of the upper torso is in contact with the upholstery. The dermal exposure level was calculated using the highest expected application rate for diantimony trioxide (S_a=2.5 mg Sb_2O_3/cm^2/day). The area of the body in contact with the couch (A_b) is estimated to 1 934 cm^2 (1/4 of (4 957 + 2 779) for an adult woman, based on one-quarter of the trunk and upper extremities (according to TGD). As a conservative estimate, the fractional release (μ_w) of antimony was assumed to be the highest of the releases from the contact blotting test, i.e. 0.0088 % (to citric acid from UV-aged DBDE-ATO textile).

$$Exposure = S_a \cdot A_b \cdot \mu_w \cdot fc$$

From these assumptions the dermal exposure level is calculated to 0.11 mg/day, corresponding to $0.057 \cdot 10^{-3}$ mg/cm^2/day (assuming an exposure area of 1934 cm^2). For a person weighing 60 kg this would imply an exposure of 1.8 µg/kg/day.

This exposure assessment assumes that clothing and the skin presents no barrier to movement of diantimony trioxide. Furthermore the release fraction from the contact-blotting test represents dermal contact with only liquid between the skin and the layer containing diantimony trioxide, i.e. where a person is in contact with wet (from perspiration or spill) furniture. This is unlikely to occur six hours per day, therefore the derived exposure should be seen as a worst case, with potential for refining in case of a conclusion (iii). It should however be noticed that when diantimony trioxide is dissolved and leaches from the textile matrix, it will not be present in the form of diantimony trioxide.

4.1.1.3.4 Scenario No. 3: Cuddly toys

The Danish environmental protection agency has made a survey where they bought five different cuddly toys made of minimum 95 % natural materials and without electrical or electronic parts. These were analysed for content of various chemical compounds and elements. In two of these toys antimony was present above the detection limit (5 mg/kg), the result is shown in Table 4-27 below (Danish Environmental Protection Agency, 2003b). Antimony is found in concentrations typical for antimony used as catalyst in polyester production. This together with the low contents of bromine

and chlorine in the same material clearly indicates that the antimony found is not added as a flame-retardant.

The independent German testing institute Oekotest regularly tests toys on the market. In a survey in 2002, two out of 20 of the stuffed cuddly toys (teddy bear type) contained antimony, one of the toys which contained antimony also contained organohalogen compounds. The levels of antimony and halogen compound were not given. (Oekotest, 2002)

A maximum level of antimony (compound) that may migrate from toys is stipulated in Council directive 88/378/EEC of 3 May 1988 on the approximation of the laws of the member states concerning the safety of toys and associated standards. In a survey made by the Swedish National Chemical Inspectorate all toy-companies contacted could show certification that their toys complied with this regulation (KEMI, National Chemical Inspectorate, 2002).

Table 4-27 Analytical results from elemental analysis of toys (only those chemical elements relevant to this report are included)

Sample	Part	Antimony, mg/kg	Bromine, mg/kg	Chlorine, mg/kg
Pig	Body	< 5	< 5	240
	Filling	260	< 5	21
	Handkerchief	< 5	< 5	75
Rabbit	Trousers	36	< 5	130
	Body	< 5	< 5	70
	Filling	120	< 5	25

Oral exposure

Oral exposure was estimated for a child sucking/chewing on cuddly toys. Several ways to predict the exposure have been considered. It was found that there is lack of information on critical parameters such as the time which a child sucks on a cuddly toy and the migration rate of antimony in the toy. In this scenario the route of antimony will be via migration in saliva from the toy to the mouth or via ingestion of toy particles. The results from available mouthing studies such as the DTI study "Research into mouthing behaviour of children up to 5 years old" (DTI and Research commisioned by the consumer and Competition Policy Directorate, 2002) are generally not focused on cuddly toys and this route, and were therefore not deemed to be appropriate to use in this scenario.

An initial estimation has been made based on the following:

a) the exposure comes from ingestion of toy particles,

b) the content of antimony in the toy particles is the maximum found in toys in the studies presented above.

This approach is simplistic and contains uncertainties, these uncertainties are however not greater in this model than in a more complex one, including assumptions on several more parameters, and is more transparent.

The amount of toy material ingested per day has been set to 8 mg/day based on the convention adopted for the directive on safety of toys (Directive 88/378/EEC). This number is an assumed mean value and has been criticised as being too low (EUROPEAN COMMISSION, Health and consumer protection directorate-general, 2004). However, at least part of the criticism comes from the possibility of ingestion of liquid toy material, which is not the case here, and the Bundesinstitut fur Risikobewertung has in an expert opinion concluded that new information would be necessary to amend this number (BfR Expert Opinion, 2003).

The exposure is then:

8 [mg/day] * 260 [mg Sb/kg] = **2.1 µg Sb/day**

After conversion from the content of antimony to diantimony trioxide this corresponds to 2.52 µg/day. For a child of 10 kg this corresponds to **0.25 µg/kg/day**. This is a value with a high degree of uncertainty.

This value on exposure from sucking on cuddly toys is taken forward to the risk characterisation. It should however be noticed that when diantimony trioxide is dissolved and leaches from the textile matrix, it will not be present in the form of diantimony trioxide.

Dermal exposure from sleeping with cuddly toys

The dermal exposure for a child sleeping with cuddly toys is predicted to be significantly lower than the exposure in the scenario no 2 Fabrics. The reasoning for this is based on the following:

- The highest concentration of antimony found in cuddly toys is significantly lower than in the back-coating of the furniture.
- A reasonable worst case would be for a child to rest its head on a cuddly toy (as a pillow), with less than half the area of the face in contact with the toy, i e less than in scenario 2.

An exposure estimation will therefore not be made for this scenario, since scenario no 2 Fabrics give no cause for concern.

Exposure via inhalation was not considered relevant for this specific exposure.

4.1.1.3.5 Scenario No. 4: Indoor air

In a study by University of Surrey (Thomas and Stevens, 2006) the effect of wear on the potential release of flame-retardants (as debris and as volatiles) from backcoated textiles was investigated. In the fabric wear test it was shown that debris, containing short and long fibres from the fabric, was produced. Particulates were present in the debris with the largest quantity (both number and weight of particles) in the 10-90 µm size ranges with a low quantity by weight of smaller particles. The fabric wear test also revealed that emission of airborne particles in the size range 30 nm to 6.5 µm occurs. In this study, no diantimony trioxide was detected as volatiles and low levels of diantimony trioxide were detected in debris. No chemical analyses were made of the airborne particles.

Inhalation exposure

In one study (Thompson and Thornton, 1997) antimony was detected in house dust showing that consumers can be exposed to diantimony trioxide in indoor air. Compared to naturally occurring amounts in soil of around 0.2 µg/g, house-dust contained relatively high amounts of antimony with median values of 13 µg/g, corresponding to 15.6 µg Sb_2O_3/g. The 90th percentile in the same publication was close to 50 µg/g, corresponding to 60 µg Sb_2O_3/g. When taking the CSOIL (parameter set for human exposure modelling) estimate for particulate matter (dust) in indoor air of 52.5 µg/m^3 into consideration (Otte et al., 2001), 15.6 µg Sb_2O_3/g dust corresponds to 0.819 ng Sb_2O_3/m^3. This is considered a typical value. A reasonable worst case scenario of 60 µg Sb_2O_3/g corresponds to 3.15 ng Sb_2O_3/m^3. Since house dust consists to a certain extent also of dust and soil carried into the house from the external environment, part of the antimony present will also be in the form of pentavalent and trivalent compounds. Therefore, assuming in this scenario that all antimony is present as the trioxide is a conservative assumption.

Oral exposure

The specific hand-to-mouth behaviour of small children may play a particular role for their exposure. Therefore oral exposure to diantimony trioxide for children via ingestion of house dust has been estimated.

Children ingest between 50 and 100 mg of dust/day (Butte and Heinzow, 2002). In children with pica behaviour the daily ingestion of soil may be much higher (range up to 10 g/day).

Assuming that a child of 10 kg would ingest 100 mg dust/day this corresponds to a typical exposure of 0.156 µg Sb_2O_3/kg bw/day (0.1 g/d·15.6µg/g)/10 kg) and a reasonable worst case exposure of 0.600 µg Sb_2O_3/kg bw/day (0.1 g/d·60µg/g)/10 kg).

Values taken forward to risk characterisation

The actual exposure in Scenario No. 1-3 is not to diantimony trioxide but instead to antimony in hydro-complexed form.
The exposure in Scenario No. 4 is largely to diantimony trioxide.

Scenario No. 1: PET-bottle
The typical and reasonable worst-case oral exposure is estimated to 0.82 and 2.1 µg/day, respectively (corresponding to 0.014 and 0.035 µg/kg/day, respectively) for a person drinking from a PET bottle. This is based on measured values.

Scenario No. 2: Fabrics
The reasonable worst-case dermal exposure is calculated to 0.11 mg/day, corresponding to 1.8 µg/kg/day for a person sitting on upholstery fabric.

Scenario No. 3: Cuddly toys
The reasonable worst-case oral exposure for children sucking on cuddly toys is 2.5 µg/day (0.25 µg/kg/day).

Scenario No. 4: Indoor air

For exposure via indoor air to the general population the typical and reasonable worst case exposures of diantimony trioxide are $0.82 \cdot 10^{-6}$ mg/m^3 and $3.2 \cdot 10^{-6}$ mg/m^3, respectively.

Since small children may be at risk due to their specific hand-to-mouth behaviour oral exposure via indoor air was estimated for this group. The typical and reasonable worst-case oral exposure is 0.16 and 0.60 µg/kg/day, respectively.

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4.1.1.4 Indirect exposure via the environment

4.1.1.4.1 Introduction

The sources of human exposure to diantimony trioxide handled in this chapter are food, water and air. Diantimony trioxide may be released to the environment through air effluents from manufacture, formulation, processing, use and disposal of diantimony trioxide containing products. Other, more "unintentional" sources are production of non-ferrous metals, combustion of fossil fuels and road traffic. In the environment, diantimony trioxide will dissolve and the resulting trivalent antimony ion will hydrolyse to the neutral species $Sb(OH)_3$ which in an oxic environment will oxidize to the pentavalent $Sb(OH)_6^-$. In natural waters antimony exists almost exclusively in the dissolved phase in the two valency states +3 and +5. Even though the dominant species in oxic waters is the pentavalent $Sb(OH)_6^-$, the trivalent $Sb(OH)_3$ has been detected in concentrations much above what is expected (see the Environmental section 3.1.3). As a consequence, antimony present in drinking water and foods will not be present as diantimony trioxide.

Antimony is also a naturally occurring element. Therefore, its presence in the environment, and thereby also indirectly in water and in food and beverages produced from agricultural goods, may also be attributed to natural sources.

However, as previously described in the general introduction for the Exposure assessment, an approach in this risk assessment report is to adjust all measured levels of antimony with a factor of 1.2 to achieve the concentration of diantimony trioxide (correction for molecular weight). This approach is taken in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide. However, when data are taken forward to risk characterisation it must be recognised that the actual exposure for man indirectly via the environment is not to diantimony trioxide but rather to the antimony ions and part of the antimony may also originate from sources other than diantimony trioxide and its use.

Biomagnification of antimony is not likely to occur considering the finding of low antimony levels in higher animals (section 3.1.3.3).

In food or beverages antimony is present in trace amounts. According to a study by Moll and Moll, (2000) the main contributors to antimony intake are cereals, sweeteners, fish and crustaceans, fruits and vegetables, and alcoholic beverages.

4.1.1.4.2 Scenario No. 1: Exposure from food

Measured data on antimony in food

In the UK, antimony in adult diet was estimated in 1976 to be 29 µg/day, and more recently to around 3 µg/day (median) up to 4 µg/day (97.5^{th} percentile), in a total diet study (UK Food Standards Agency, 2003).

In another total diet study, the antimony concentrations in French food were investigated (INRA, 2004). Based on the concentrations in food samples and on the average consumption pattern, a distribution of the daily intake of antimony for adults (>15 y) was calculated to range from 0.5 µg/day (2.5^{th} percentile) to 2 µg/day (97.5^{th} percentile) with a median of 0.9 µg/day.

In a recent duplicate diet study, the dietary intake of antimony was investigated among nursing mothers from Germany, Poland and the Czech Republic, providing in total 2779 food samples (Wappelhorst et al., 2002). Food duplicates (parts of the original food consumed by the mothers) were collected daily over 2 to 8 weeks (on average 27 samples; range 13-56 samples) and the actual amounts of each food consumed were recorded. The period of the study should provide a representative average reflecting the eating habits of the mother. The age of the infants was on average 5 months (range 1-16 months).

Only the mean (10.9±28.1 µg/day) and median (4.1 µg/day) dietary intake of antimony was reported in the study. However, after personal communication with the authors, raw data of antimony concentrations in food and of antimony dietary intake in the individual women was provided which enabled further statistical evaluation of the data. The median and the 90^{th} percentile of the dietary intake of antimony were calculated from the median intake for each of the 19 individual women. This analysis gave a median dietary intake of 3.7 µg/day and a 90^{th} percentile of 4.8 µg/day. The total range was 1.2-108 µg/day, with one woman having far higher dietary intake of antimony than the rest of the women.

Also based on duplicate diet studies, a dietary intake of 4.1-14 µg/day was estimated in Turkey (Aras and Kumulainen, 1995) and somewhat lower levels were found in Brazil; 1.1-2.3 µg/day (Maihara et al., 1998).

A corresponding level of 4.6 µg/day was determined using a mixed diet composite to represent the intake of 25-30-year old males in the U.S (Iyengar et al., 1987).

In older studies, the reported amount of ingested antimony in Europe ranged from about 10 µg/day in a Swedish balance study (Wester, 1974; neutron activation) to an average of 23 µg/day in four normal German diets (Schelenz, 1977; neutron activation).

In Table 4-28 the concentration of antimony in food items and beverages from several studies are presented. In a recent study the antimony concentrations in French food was investigated (INRA, 2004). Highest values were observed in cured meats and fruit (2.4 µg/kg), sea food, vegetables and dried fruits (1.8 µg/kg) and meat (1.7 µg/kg), while most other values were around 0.3 µg/kg.

A study of levels of antimony and other elements in infant foods was made by the Food Standards Agency (UK) (UK Food Standards Agency, 2003). A total of 189 samples of infant food and formulae were purchased from retail outlets in the UK in 2001/02. The mean concentration of antimony of all foods was 1.7 µg/kg of food as sold. In a study by Murthy et al. (1971) the average antimony content in the diets of children from different institutions in the U.S. varied between 0.279 and 0.693 mg/kg of food for the different institutions, with a maximum value of 0.825 mg/kg. The consumption at the different institutions varied from 1.18 to 2.55 kg/day and the milk content of diet varied from 9.8 to 63.8%. Further, there were significant seasonal and geographical variations, with a peak during summer. The antimony levels in food from this study are high compared to more recent studies. One possible

explanation might be that storing of food has changed. It is mentioned in the study that enamel vessels and tin cans may contain appreciable amounts of antimony.

Table 4-28 Concentrations (mean value) of antimony (Sb) in foods and beverages.

Food group	Concentration µg / kg food	Reference	Food group	Concentration µg / kg food	Reference
Bread	1	(UK, 1997)	Green vegetables	< 1	(UK, 1997)
Misc cereals	4	(UK, 1997)	Potatoes	1	(UK, 1997)
Carcase meat	2	(UK, 1997)	Other vegetables	1	(UK, 1997)
Offal	1	(UK, 1997)	Canned vegetables	2	(UK, 1997)
Meat products	4	(UK, 1997)	Fresh fruit	1	(UK, 1997)
Poultry	1	(UK, 1997)	Fruit products	1	(UK, 1997)
Fish	3	(UK, 1997)	Beverages	< 1	(UK, 1997)
Oils & fats	2	(UK, 1997)	Milk	< 1	(UK, 1997)
Eggs	1	(UK, 1997)	Dairy products	1	(UK, 1997)
Sugars & preserves	2	(UK, 1997)	Nuts	1	(UK, 1997)
Freshwater fish (µg/kg wet weight)	3	(Uthe and Bligh, 1971)	Milk (µg/kg wet weight)	3	(Schelenz, 1977)
Potato powder (µg/kg wet weight)	8	(Schelenz, 1977)	Beer (µg/kg wet weight)	0.8 (0.2 – 17.6)	(Kallischnigg et al., 1982)
Vegetables (µg/kg wet weight)	0.002 – 0.052	(Bognar et al., 1981)	Vegetables (µg/kg dry weight)	0.4 – 2.5	(Furr et al., 1976)
Cereals	0.5±0.2	(Eriksson, 2001)	Chuck Roast	0.46±0.21	(Cunningham, 1987)
Cereals	1.0±0.2	(Eriksson, 2001)	Meat Loaf	0.79±0.22	(Cunningham, 1987)
Carrots	0.22±0.03	(Cunningham, 1987)	Beef & Vegetables	0.50±0.18	(Cunningham, 1987)
Green Peas	0.70±0.35	(Cunningham, 1987)	Chicken whole	0.657±0.85	(Cunningham, 1987)
Baked potatoes	1.81±0.18	(Cunningham, 1987)	Chicken drumstick/breast	1.15±0.141	(Cunningham, 1987)
Pork Chow Mein	0.88±0.09	(Cunningham, 1987)	Clams	1.10±0.15	(Cunningham, 1987)
Schrimp	2.81±0.12	(Cunningham, 1987)	Oysters	1.09±0.56	(Cunningham, 1987)

Conclusions on exposure via food

The total diet studies in the UK (UK Ministry of Agriculture, Fisheries and Food, 1998) and in France (INRA, 2004) and the duplicate diet study among nursing mothers from Germany, Poland and the Czech Republic (Wappelhorst et al., 2002) are all considered to be extensive and representative studies for the average EU population.

The exposure assessment of antimony via the diet is based on the duplicate diet study among nursing mothers by (Wappelhorst et al., 2002). Nursing mothers have somewhat higher nutrient intake, however it can be expected that also other groups of the population (e.g. young men and athletes) have a high food intake. Duplicate diet studies are considered to be the most appropriate for the purpose of determining dietary intake. In addition, Wappelhorst et al. (2002) is a well-performed study and the analysis of antimony concentrations was performed by ICP-MS which is a well-established sensitive method for the determination of trace elements. Raw data on the antimony intake was provided for statistical evaluation. The data reported from the total diet studies in UK and France are slightly lower than the Wappelhorst et al. data, but in the same order of magnitude. In addition, data from Turkey indicate that also higher exposure may occur.

Accordingly, the typical exposure via food is estimated to 3.7 µg/day (median) (Wappelhorst et al., 2002), which is similar to that reported in UK (median 3 µg/day; MMAF, Ministry of Agriculture Fishery and Food, UK, 1998) but somewhat higher than that reported in France (median 0.9 µg/day; INRA, 2004). The reasonable worst case exposure is estimated to 4.8 µg/day (90^{th} percentile; Wappelhorst et al., 2002), which also is similar to that reported in UK (97.5^{th} percentile 4 µg/day; MMAF, Ministry of Agriculture Fishery and Food, UK, 1998) but somewhat higher than that reported in France (97.5^{th} percentile 2 µg/day; INRA, 2004).

After conversion from the content of antimony to diantimony trioxide, the typical and reasonable worst-case exposure correspond to 4.44 and 5.76 µg/day, respectively. For a person weighing 60 kg this corresponds to 0.074 µg/kg/day (3.7 µg/d·1.2)/60 kg) and 0.096 µg/kg/day (4.8 µg/d·1.2)/60 kg) and these values are taken forward to the risk characterisation.

It must, however, be considered that antimony in food is not present in the form diantimony trioxide. The conversion to the content of diantimony trioxide is made in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide.

4.1.1.4.3 Scenario No. 2: Exposure via mothers' milk

Measured data on antimony in mothers' milk

Several available studies show that antimony is present in human breast milk in the population not occupationally exposed to diantimony trioxide, as well as in breast milk from occupationally exposed mothers. These studies are also described in the Toxicokinetics section (4.1.2.1). In a Russian study by Belyaeva (1967) antimony was measured in blood, urine, mother's milk, placenta, amniotic fluid and umbilical cord blood in female workers exposed to antimony aerosol containing metallic Sb, Sb_2S_5 and Sb_2O_3 (exposure concentrations were not reported). In the high-dose exposed women the mean concentration of antimony in mother's milk was 3.3 ± 2.2 µg/mL. This data is not considered to be representative for the general population.

Clemente and co-workers (Clemente et al., 1982) measured the concentration of antimony in more than 130 human milk samples (the exact number was not stated) collected from 21

different Italian subjects. The exposure of the subjects was not given. The samples were mature milk, having been collected 15 or more days after the childbirth. The analytical method was instrumental neutron activation analysis with the detection limit for antimony given as 0.05 ng/g wet weight. 49 samples from 16 subjects were above the detection limit and the mean concentration of antimony was 3.0 ± 0.4 ng/g wet weight (range < 0.05-12.9).

The concentration of antimony was also measured in a single pooled human milk reference sample (Iyengar et al., 1982) using instrumental neutron activation analysis. Nine samples were taken from this pooled milk sample with double determinations. The resulting mean concentration of antimony was 13 ± 5 ng/g dry weight. The wet-to-dry ratio for this freeze dried sample is reported to be 0.119. Therefore, the given mean concentration corresponds to 1.55 ± 0.6 ng/g wet weight.

Parr et al. (1991) summarised the results of a WHO/IAEA joint project on breast feeding, initiated in 1973. Within this project, the concentrations of several trace elements were determined in human milk samples taken at one particular time at about 3 months after birth from a large number of mothers in several countries during the late 1970s. Antimony was determined by instrumental neutron activation analysis. The analytical procedure and the development of a reference material are described in the paper by Ivengar et al. (see above). The following values are reported for several countries. Country (number of samples): median in µg/L ± total estimate uncertainty: Guatemala (84): 1.0±0.1; Hungary (71): 1.6±0.2; Nigeria (18): 4.1±2.4; Philippines (65): 11.0±1.5; Sweden (32): 3.0±0.8; Zaire (69): 3.6±0.4.

18 trace elements were determined using ICP-MS in 55 samples of human milk from 46 mothers in Austria, collected in 1995-1996 during various periods of lactation (13 colostrum, 18 transitional milk, 24 mature milk; Krachler et al., 1998). The authors report that almost 73 % of the samples (i.e. 40 out of 55 samples) were below the detection limit of 0.36 µg/L. The highest value of 6.9 µg/L was found in colostrum milk. However, this colostrum milk was collected within 3 days of birth of the child. The median concentration for colostrum milk was 0.7 µg/L. The overall median concentration in milk (from 1 to 293 days post partum) was <0.36 µg/L.

In the most recent and comprehensive study, antimony in the milk of nursing mothers was investigated (Wappelhorst et al., 2002). Nineteen mothers from Germany, Poland and the Czech Republic participated in the study, providing 2779 food and 536 breast milk samples. The age of their infants was on average 5 months (range 1-16 months). Breast milk samples were taken daily over 2 to 8 weeks (on average 27 samples; range 13-56) from each mother. The reported mean antimony concentration in breast milk was 0.14 µg/L (range between 0.06 and 0.57 µg/L). Since the data by Wappelhorst et al. (2002) represents the most recent available measurements of antimony concentrations in human milk and because samples from Germany, Poland and the Czech Republic were measured, this data may be considered to be representative of the current situation in Europe. The data may also be considered to be representative for the concentration of antimony found in human milk on average during the course of breast-feeding. Therefore, this data set was considered for further calculations of the exposure of breast-fed children. After personal communication, raw data of antimony concentrations in breast milk from the individual women was provided by the authors, summarised in Table 4-29, which enabled further evaluation of the data.

Table 4-29 Antimony concentrations (µg/kg) in human breast milk and total number of samples (N) from 19 nursing mothers (Wappelhorst et al., 2002); raw data provided by the author.

Mother	N	Min	Mean	Median	P90	Max
A	56	0.033	0.224	0.073	0.374	4.477
B	14	0.013	2.242	0.562	7.558	12.181
C	14	0.040	0.585	0.075	0.462	5.953
D	49	0.029	0.088	0.079	0.132	0.333
E	35	0.059	0.532	0.151	0.413	5.439
F	56	0.059	0.208	0.136	0.326	2.450
G	17	0.102	0.169	0.149	0.269	0.356
H	56	0.047	0.125	0.115	0.189	0.271
I	42	0.054	0.112	0.093	0.164	0.278
K	14	0.079	0.265	0.162	0.269	1.527
L	14	0.045	0.133	0.111	0.215	0.453
M	14	0.092	0.146	0.138	0.203	0.251
N	28	0.068	0.134	0.114	0.196	0.399
O	14	0.074	0.108	0.096	0.157	0.186
P	28	0.055	0.221	0.116	0.361	1.784
Q	14	0.055	0.120	0.098	0.214	0.229
R	13	0.083	0.309	0.130	0.806	1.519
S	14	0.050	0.080	0.069	0.129	0.152
U	16	0.034	0.075	0.061	0.084	0.310
Median	16	0.055	0.146	0.114	0.215	0.399
Mean	27	0.056	0.309	0.133	0.659	2.03
P90	56	0.085	0.543	0.154	0.531	5.54
Range	13-56	0.013-0.102	0.075-2.243	0.061-0.562	0.084-7.558	0.152-12.181

As shown in the table above, there was a wide range in antimony concentrations in breast milk both within and between women. In an attempt to condense the data, the medians, means and the 90th percentiles were calculated from the minimum, mean, median, 90th percentile and maximum concentrations found for each of the 19 individual mothers. The mean concentration from each mother reflects the average concentration of antimony which will appear in the breast milk during the length of exposure. Therefore, the median concentration, of the mean breast milk concentrations from all women, of 0.146 µg/L of this analysis is taken forward as a typical concentration. The 90th percentile concentration, of the mean breast milk concentrations from all women, of 0.543 µg/L is taken forward as a reasonable worst case. After conversion from the content of antimony to diantimony trioxide, the typical and worst case concentrations correspond to 0.175 µg/L and 0.652 µg/L, respectively.

It must, however, be considered that antimony in breast milk is not present as diantimony trioxide and the use of diantimony trioxide will only contribute to the antimony

concentrations in breast milk to a small extent. The conversion to diantimony trioxide is made in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide.

Calculation of exposure of babies via mothers' milk

The exposure to babies via mothers' milk was calculated according to (Darnerud et al., 1998), however slightly modified. It was assumed that an infant breast-feeds for 1 year, and that this year of life is subdivided into two periods (0 to 3 months and 3 to 12 months), reflecting the changing feeding demands of the infant. It is assumed that over the first 3 months the infant has an average weight of 6 kg and that the infant ingests 0.8 kg of milk per day. From 3 to 12 months it is assumed that the infant has an average weight of 10 kg and that the infant ingests 0.5 kg of milk per day. Further, it is assumed that the content of antimony in breast milk remains constant during the breast-feeding period.

Using the following equation and the assumptions, as detailed above, the average daily uptake (U_{milk}) of the breast-feeding infant is estimated for both the 0-3 month and 3-12 month periods of infant life.

$$U_{milk} = \frac{C_{milk} \times B_{ing} \times IR_{milk}}{BW_{infant}}$$

where:

C_{milk}-the concentration of diantimony trioxide in µg per kg wet weight mothers' milk

B_{ing} - the bioavailability of the ingested antimony

IR_{milk}-the ingested amount of milk by the infant (kg/day)

BW_{infant} -the average infant body weight over the exposure period (kg)

When calculating the external exposure the bioavailability factor (B_{ing}) is not included in the equation. Based on the above, the following exposure levels can be calculated:

Typical exposure:

0-3 months: 0.175 µg/kg milk x 0.8 = 0.140 µg/day

Assuming a body-weight of 6 kg, this exposure corresponds to 0.023 µg/kg/day

3-12 months: 0.175 µg/kg milk x 0.5 = 0.0875 µg/day

Assuming a body-weight of 10 kg this exposure corresponds to 0.009 µg/kg/day

Reasonable worst-case exposure:

0-3 months: 0.652 µg/kg milk x 0.8 = 0.522 µg/day

Assuming a body-weight of 6 kg, this exposure corresponds to 0.087 µg/kg/day

3-12 months: 0.652 µg/kg milk x 0.5 = 0.326 µg/day

Assuming a body-weight of 10 kg, this exposure corresponds to 0.033 µg/kg/day

Conclusions on exposure via mothers' milk

The typical values for exposure via mothers' milk taken forward to the risk characterisation are based on data from Wappelhorst et al. (2002) which is the most recent and representative data set for exposure to antimony via breast milk in Europe as of today and during the course of breast-feeding. Because of a higher milk consumption and lower body weight, children of 0-3 months of age have a higher exposure to antimony than older children. Thus, as an additional conservative measure, the exposure estimates for children of 0-3 months of age are taken forward to risk characterisation. The typical and reasonable worst-case exposure values are 0.140 and 0.522 µg/day (corresponding to 0.023 and 0.087 µg/kg/day).

4.1.1.4.4 Scenario No. 3: Exposure from drinking water

As described in the Environmental section (3.1.4.4), the reasonable worst case ambient antimony concentration in freshwater (dissolved) is 0.72 µg/L.

EU drinking water surveys with measured data on antimony concentrations in drinking water have recently been compiled and submitted in a report by Industry (EBRC, 2007). The result from this report show that the extent of these surveys is highly variable, ranging from only a few data points in most studies to approximately one thousand in one case (Finnish study). As most surveys only report mean/median and/or total range and individual raw data were not available, reasonable worst case concentrations were not estimated. However, the Finnish study reports a 98th percentile close to 0.2 µg/L (maximum 1.46 µg/L).

The reasonable worst case ambient antimony concentration in freshwater (dissolved) of 0.72 µg/L is taken forward to the risk characterisation. This value is supported by measured data of European drinking water. All median and mean values reported from these surveys are well below 0.72 µg/L.

Assuming a daily intake of 2 L, the reasonable worst case exposure of antimony from drinking water is 1.44 µg/day. After conversion from the content of antimony to diantimony trioxide this corresponds to 1.73 µg/day and for a person weighing 60 kg this corresponds to 0.029 µg/kg/day (1.44 µg/d·1.2)/60 kg) and this value is taken forward to the risk characterisation.

It must, however, be considered that the actual exposure is not in the form of diantimony trioxide. The conversion to diantimony trioxide is made in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide.

It should also be noticed that the emissions of diantimony trioxide from industrial production or the use of diantimony trioxide do not generally affect the antimony levels in drinking water. Antimony concentrations in water will also be influenced by the mineral composition of the area from which the water is collected (Kelepertsis et al., 2006) and also from other antimony sources other than emission of diantimony trioxide.

4.1.1.4.5 Scenario No. 4: Exposure via outdoor air

Based on measured data (see Environmental section 3) the reasonable worst case ambient background concentration of antimony in air is 2.6 ng Sb/m^3.

The conversion from the content of antimony to diantimony trioxide (corresponding to 3.12 ng Sb_2O_3/m^3) when taking forward this value to risk characterisation is made in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide. However, it should be borne in mind that the antimony present is not only derived from emissions of diantimony trioxide from industrial production or the use of diantimony trioxide but also from geogenic emissions and combustion of mineral fuels.

4.1.1.4.6 Local exposure

The reasonable worst case exposures on a regional level have been estimated for man via the environment based on available measured data. Local exposure via the environment was not modelled as no partition coefficients are available.

However, in the Environmental section of the RAR local predicted environmental concentrations have been estimated. Maximum local antimony concentration is estimated to 69 µg/L in freshwater (a generic site for the industrial use of diantimony trioxide as a flame-retardant in textiles (application to textiles; section 3.1.4). As the reported measured data is limited in this industry sector, this value represents a modelled worst case value for a generic site. Predicted environmental concentrations (water) for other scenarios range between 0.72 and 39.5 µg/L. The maximum local antimony concentration in air is 986 ng/m^3 (production of diantimony trioxide;section 3.1.4.6). This value is reported from one production site. Predicted environmental concentrations (air) for other scenarios range between 2.6 and 41 ng/m^3.

Comparing the maximum local concentrations to the regional concentrations, we find that the local concentrations are 96 times higher for freshwater and 379 times higher for air.

The maximum local concentration in air of 1 µg/m^3 is taken forward to the risk characterisation. Assuming a daily intake of 2 L the maximum local exposure of antimony from water is 138 µg/day. After conversion to diantimony trioxide this corresponds to 166 µg/day and for a person weighing 60 kg this corresponds to 2.8 µg/kg/day (138 µg/d·1.2)/60 kg) and this value is taken forward to the risk characterisation. It should however be noticed that this exposure is a gross overestimate as it is based on untreated surface water which is not representative for drinking water in the EU.

Values taken forward to risk characterisation

Scenario No. 1: Food
The typical and reasonable worst case exposures taken forward to the risk characterisation are 4.44 and 5.76 µg/day, or 0.074 and 0.096 µg/kg/day. These values are based on measured data on antimony concentrations in foodstuff from a recent duplicate diet study and they are supported by data from two extensive total diet studies.

Scenario No. 2: Mothers' milk
The values for exposure via mothers' milk taken forward to the risk characterisation are based on the most recent data since they are believed to be most representative for exposure to antimony in Europe as of today and during the course of breastfeeding. The typical and reasonable worst case exposure values are 0.140 and 0.522 µg/day, corresponding to 0.023 and 0.087 µg/kg/day, respectively.

Scenario No. 3: Drinking water
The reasonable worst case ambient antimony concentration in freshwater of 0.72 µg/L is taken forward. This value is supported by measured data of European drinking water. The exposure via drinking water was estimated to 1.73 µg/day (0.029 µg/kg/day).

Scenario No. 4: Outdoor air
The reasonable worst case ambient antimony concentration in air of 3.12 ng Sb_2O_3/m^3 is taken forward. This value is bases on measured data.

Local exposure
The maximum local concentration in air of 1 µg Sb_2O_3/m^3 and the maximum local exposure of 2.8 µg/kg/day via water are taken forward.

4.1.1.5 Combined exposure

Due to the use of diantimony trioxide in the society and the diffuse emissions from products, humans may be exposed from different sources. The total exposure (body burden) is the summary of all the specific exposures. The most important sources of human exposure are probably identified. Additions of individual scenarios are not considered to change any of the conclusions, and no calculation on combined exposure has therefore been performed.

4.1.1.6 Summary of human exposure

The external exposure of diantimony trioxide for the different populations taken forward to the risk characterisation are summarised in Table 4-30 and Table 4-31. For occupational exposure, the typical and reasonable worst-case scenarios are estimated from measured data or from analogous/surrogate data. For consumers and humans exposed via the environment, the exposure is estimated from measured data and calculations.

The following assumptions were made:

- **Bodyweight**
  - infant = 6 kg,
  - young child = 10 kg
  - consumer = 60 kg
  - worker = 70 kg

- **Skin area**
  - worker in production: hands and forearms + head (neck and face) = 840+1140+1180= 3160 cm^2
  - worker in downstream use scenarios = 2000 cm^2 (840 cm^2 for "Textiles-processing")
  - all front of a child = 1500 cm^2
  - $1/4^{th}$ of the upper torso (woman) = 1934 cm^2

- **Breathing rate**
  - worker during light activity = 10 m^3/ working day, i.e. 1.25 m^3/h, 8 h working day
  - adults 20-75 y (general population) indoor = 18 m^3/day

- **Time spent indoors (non-occupational) by general population**
  - adults 25-69 y = 15.3 h/day

Table 4-30. Summary of external exposure of diantimony trioxide for workers.

Inhalation exposure	**Typical** (mg/m³)	**RWC** (mg/m³)	**Remarks**
1. Production of diantimony trioxide			
- Conversion	0.027 with RPE	2.9 without RPE 0.15 with RPE	Measured data
- Refuming	0.012 with RPE	0.94 without RPE 0.047 with RPE	Measured data
- Final handing	0.040 with RPE	2.1 without RPE 0.11 with RPE	Measured data
2. Use as a catalyst in production of PET			
- Powder handling	0.002	0.026	Measured data
3. Use as a flame-retardant in production of plastics			
- Raw material handling	0.13	0.57	Measured data
4. Use as a flame-retardant in treated textiles			
- Formulation	0.13	0.57	Analogous data
- Processing	<0.001	0.001	Measured data
- Further handling	negl	negl	Industry info
5. Use in pigments, paints, coatings and ceramics			
- Loading and mixing	0.036	0.16	Analogous data
6. Use as a flame-retardant in production of rubber			
- Formulation	0.051	0.22	Analogous data

- Processing	0.064	0.14	Analogous data
7. Use in production of crystal glass			
- Cutting	0.003	0.015	Analogous data
Dermal exposure	**Typical** (mg/kg/day)	**RWC** (mg/kg/day)	**Remarks**
1. Production of diantimony trioxide			
- Conversion	0.23	0.72	Measured data
- Refuming	0.54	0.99	Measured data
- Final handing	0.81	1.4	Measured data
2. Use as a catalyst in production of PET			
- Powder handling	0.10	0.17	Analogous data
3. Use as a flame-retardant in production of plastics			
- Raw material handling	0.19	0.34	Analogous data
4. Use as a flame-retardant in treated textiles			
- Formulation	0.13	0.22	Analogous data
- Processing	0.13	0. 86	Modelled data
- Further handling	0.0023	0.010	Measured data
5. Use in pigments, paints, coatings and ceramics			
- Loading and mixing	0.066	0.11	Analogous data
6. Use as a flame-retardant in production of rubber			
- Formulation	0.066	0.11	Analogous data
- Processing	0.051	0.089	Analogous data
7. Use in production of crystal glass			
- Cutting	0.086	0.31	Analogous data

Table 4-31. Summary of external exposure of diantimony trioxide for consumers and humans exposed indirectly via the environment.

Inhalation exposure	**Typical** (mg/m³)	**RWC** (mg/m³)	**Remarks**
Consumers			
4. Indoor air	$0.82 \cdot 10^{-6}$	$3.2 \cdot 10^{-6}$	Measured data
Man exposed indirectly via the environment			
4. Outdoor air		$3.12 \cdot 10^{-6}$	Measured data
Local exposure – air		0.001	Measured data
Dermal exposure	**Typical**	**RWC** (µg/kg/day)	**Remarks**

	(mg/kg/day)		
Consumers			
2. Fabrics		1.8	Calculations
Oral exposure	**Typical** (µg/kg/day)	**RWC** (µg/kg/day)	**Remarks**
Consumers			
1. PET-bottle	0.014	0.035	Measured data
3. Cuddly toys		0.25	Measured data/ Calculations
4. Indoor air – dust (children)	0.16	0.60	Measuread data/ Calculations
Man exposed indirectly via the environment			
1. Food	0.074	0.096	Measured data
2. Infants exposed via mothers milk	0.023	0.087	Measured data
3. Drinking water		0.029	Measured data
Local exposure – water		2.8	Modelled data

4.1.2 Effects assessment: Hazard identification and dose (concentration)- response (effect) assessment

A number of studies on the toxicity of diantimony trioxide has been performed and reported by Industrial Bio-Test Laboratories (IBTL). Although the studies were performed in the beginning of the 1970[ies], before the time when several of IBTL employees were convicted for manipulating with study reports, the reliability of the data could be questioned (see http://pubs.acs.org/subscribe/journals/tcaw/10/i11/html/11regs.html). Therefore, these studies will not be used in this risk assessment.

4.1.2.1 Toxicokinetics, metabolism and distribution

The toxicokinetics of diantimony trioxide has not been completely investigated. With one exception, most of the studies are old and do not comply with today's standards.

4.1.2.1.1 Studies in animals

Inhalation

The lung retention of antimony and arsenic was investigated in 4 groups of male adult Syrian golden hamsters after single intratracheal instillations of different size fractions of either diantimony trioxide or arsenic trioxide particles (Leffler et al., 1984). The volume median diameters of the diantimony trioxide fractions were 7.0µm, 13.3µm and 19.5µm and of the arsenic trioxide, 14 µm. The fractions were neutron activated to obtain the radionuclides ^{122}Sb, ^{124}Sb and ^{76}As and suspended in 0.9% saline before instillation of 1.52, 0.23 and 0.33 mg/kg bw respectively of the diantimony trioxide fractions and 1.43 mg/kg bw of arsenic trioxide in each of 4-5 animals. One animal from each group was used as a reference. The reference animals were sacrificed immediately after the instillation with an overdose of Brietal Sodium intraperitoneally. To fix the instilled material, 0.2-0.5 ml of formalin solution (10%) was instilled to inhibit ciliary activity of the bronchial mucosa. The animals were then frozen at -20°C.

The animals were kept separately in plastic cages and fed with rat and mouse pellets and tap water *ad libitum* with a weekly addition of cabbage. The lung clearance of diantimony trioxide and arsenic trioxide was monitored externally for 190 and 43 h, respectively, and then the animals were sacrificed. Each animal was dissected and the tissue concentrations of antimony and arsenic were measured in the lung lobes, liver, kidneys, trachea, and stomach by gamma counting.

The elimination of diantimony trioxide from lung consisted of two phases. The calculated biological half-life ($t_{1/2}$) for the initial (0-20h) and second (20-200h) phase was 40 hours and 20-40 days, respectively. The $t_{1/2}$ for arsenic trioxide was 13 h. The lung tissue retention of the different size fractions (7, 13.3 and 19.5 µm) of diantimony trioxide after 190 h was 59.8, 48.9 and 44.5% respectively, and that of arsenic trioxide after 43 h was 5.8% of the instilled dose. The amount antimony in the liver was 7.2, 9.2 and 12.6 % of the instilled dose, respectively for the different size fractions. For the trachea, stomach and kidney the values were 0.6% or less for all fractions. The total value for kidney, stomach and trachea was 0.5, 1.1 and 0.9% of the instilled dose respectively for the three size fractions. This study has demonstrated a tendency for smaller particles to have longer lung retention than larger particles. Lower lung retention was accompanied by higher values of antimony in the liver and therefore a higher flux of the element through the body was indicated. This might have been due to mucociliary transport and gastrointestinal absorption. When the clearance of antimony and arsenic was compared it was obvious that the particle size had less influence on the clearance pattern than the solubility. A long biological half-life of diantimony trioxide was also observed. 190 h after exposure 44-60 % were retained in the lung and around 10 % of the instilled dose was retained in liver, kidney, stomach and trachea for the different size fractions, and implies a risk for accumulation after repeated exposure.

The lung retention of diantimony trioxide was studied in a sub-chronic and chronic inhalation toxicity study (Newton et al., 1994). The study is more thoroughly described in section 4.1.2.6 repeated dose toxicity and it is also reported in section 4.1.2.8 Carcinogenicity. In the sub-chronic study, Fischer 344 rats, 50 of each sex per group, were exposed to diantimony

trioxide at exposure levels of 0, 0.25, 1.08, 4.92, or 23.46 mg/m^3 for 13 weeks followed by a 27-week observation period. The particle mass medium aerodynamic diameter (MMAD) was 3.05 ± 0.21 µm with a geometric standard deviation (GSD) of 1.57±0.06. In the chronic study, Fisher 344 rats, 65 of each sex per group were exposed to diantimony trioxide at 0, 0.06, 0.51 or 4.50 mg/m^3 for 12 months followed by a 12-month observation period. The particle MMAD was 3.76±0.84 µm. The flow rate in both studies was 18-25 complete air changes per hour (the recommended flow rate in OECD guideline 412, 413 and 453 is 12-15 air changes per hour, Rapporteur comment). In both studies interim sacrifices were performed (at test week 1, 2, 4, 8, 13, 14, 16, 23, 31 and 41 in the sub-chronic study and at test month 6, 12, 18 and 24 in the chronic study) and the left lung lobe was placed into a scintillation vial and frozen for later diantimony trioxide analyses. In the chronic study, blood samples were obtained at all termination intervals. Faecal samples were collected at the 18- and 24-month terminations.

In the sub-chronic study, there was an initial rapid accumulation of diantimony trioxide in the lung, which then slowed after 2 weeks in the males and 4 weeks in the females to a second slower accumulation phase. There were no apparent differences in the rate of accumulation during this second phase among the groups. There was no indication of the lungs reaching a steady-state level of diantimony trioxide (Table 4-32). In the chronic study, steady-state lung burden levels appeared to have been reached only in the lowest dose group by 6 month of exposure (Table 4-33).

In both the sub-chronic and chronic study, the lung clearance half-lives increased with higher lung burdens. With a lung containing approximately 2 mg of diantimony trioxide per lung, pulmonary clearance was decreased by 80 % compared to a lung containing 0.01-0.02 mg diantimony trioxide per lung, giving an increase in the clearance half-life from 2 months at a lung burden of 0.01-0.02 mg diantimony trioxide per lung to 10 months at a lung burden of 2 mg diantimony trioxide per lung. Substantial amounts of antimony were found in the lungs of these animals also after the recovery period.

Table 4-32 Antimony trioxide lung burdens (µg/g tissue) in F344 rats during a 13-week antimony exposure period followed by a 27-week recovery period (mean ±SD; n=5)

Test week	Dose							
	0.25 mg/m^3		1.08 mg/m^3		4.92 mg/m^3		23.46 mg/m^3	
	Males	Females	Males	Females	Males	Females	Males	Females
1	8 ± 3	4 ± 1	21 ± 6	16 ± 2	44 ± 15	36 ± 11	199 ± 95	156 ± 35
2	8 ± 3	9 ± 1	37 ± 8	28 ± 4	123 ± 80	98 ± 23	477 ± 225	345 ± 63
4	17 ± 2	17 ± 1	45 ± 5	44 ± 5	158 ± 18	175 ± 10	696 ± 44	708 ± 94
8	19 ± 2	15 ± 2	72 ± 9	70 ± 11	321 ± 53	274 ± 20	966 ± 97	825 ± 75
13	40 ± 13	30 ± 5	143±20	108 ± 24	572 ± 43	489 ± 67	1830 ± 160	1901 ± 410
13 + 1R	43 ± 9	34 ± 3	133± 14	107 ± 5	513 ± 77	475 ± 66	2204 ± 172	2124 ± 117
13 + 3R	39 ± 6	31 ± 7	94 ±18	76 ± 27	514 ± 78	524 ± 59	2084 ± 76	1932 ± 181
13 + 10R	20 ± 3	16 ± 2	67 ± 9	60 ± 16	412 ± 63	360 ± 50	1669 ± 145	1535 ± 62
13 + 18R	9 ± 3	6 ± 1	36 ± 4	33 ± 6	405 ± 19	305 ± 35	1575 ± 245	1431 ± 124

Test week	Dose							
	0.25 mg/m³		1.08 mg/m³		4.92 mg/m³		23.46 mg/m³	
	Males	Females	Males	Females	Males	Females	Males	Females
13 + 28R	5 ± 1	5 ± 1	20 ± 7	25 ± 5	210 ± 49	198 ± 39	1267 ± 221	1249 ± 135

R: Number of weeks of recovery

Table 4-33. Diantimony trioxide lung burdens (µg/g tissue) in F344 rats during a 1-year diantimony trioxide exposure period followed by a 1-year recovery period (mean ± SD; n=5)

Test month	Dose					
	0.06 mg/m³		0.51 mg/m³		4.5 mg/m³	
	Males	Females	Males	Females	Males	Females
6	19.6 ± 4.9	15.1 ± 4.0	75.4 ± 10.1	76.9 ± 10.6	1190±167	1100 ± 332
12	11.5 ± 1.6	9.6 ± 1.1	132.0 ±35.1	107.0 ± 28.3	1420±238	1500 ± 183
12+6R	1.4 ± 1.3	2.2 ± 0.6	28.9 ± 5.1	33.2 ± 9.9	991 ± 194	757 ± 59
12+12R	0.4 ± 0.6	0.2 ± 0.5	8.1 ± 3.2	14.7 ± 8.2	554 ± 189	663 ± 54

The concentration of antimony in red blood cells was monitored in the rats exposed for 52 weeks (Table 4-34). A clear dose dependent accumulation of antimony occurred in red blood cells, while essentially none was found in the plasma. The accumulation in red blood cells was similar in males and females at all observations in the two lowest exposure groups but approximately two times higher in females than males at 6, 12 and 18 months in the high dose group. At 24 months the levels were similar in males and females also in the highest dose group. Small, but dose-related amounts of diantimony trioxide were found in the faeces at 18 months but not at 24 months.

Table 4-34 Diantimony trioxide concentration in red blood cells (µg/g) in F344 rats during a 1-year diantimony trioxide exposure period followed by a 1-year recovery period (mean ± SD; n=5)

Test month	Dose							
	Control		0.06 mg/m3		0.51 mg/m3		4.5 mg/m3	
	Males	Females	Males	Females	Males	Females	Males	Females
6	ND	ND	0.53 ± 0.31	0.74 ± 0.06	5.07 ± 0.29	5.69 ± 0.62	34.5 ± 3.83	75.6 ± 8.39
12	ND	ND	1.09 ± 0.21	1.48 ± 0.10	7.55 ± 0.60	9.94 ± 1.32	70.7 ± 6.31	121 ± 8.39
12+6R	0.17 ± 0.39	ND	0.8 6± 0.68	0.81 ± 0.30	3.93 ± 0.24	6.53 ± 0.90	38.6 ± 4.75	74.6 ± 18.3
12+12R	ND	ND	ND	ND	2.53 ± 0.27	3.39 ± 0.28	30.5 ± 7.53	36.6 ± 15.5

ND: none detected

This study shows that diantimony trioxide remains in lung for long periods of time after inhalation exposure. Absorbed antimony is retained for long periods of time in red blood cells, and antimony can be excreted in the faeces.

Dermal

No animal data on toxicokinetics after dermal exposure to diantimony trioxide could be located.

Oral

The oral bioavailability and the rate and route of excretion after oral (PO), intravenous (IV) ($SbCl_3$) or intraperitoneal (IP) administration of Sb_2O_3 was determined in 16 female and 24 male Sprague Dawley Crl:CD (SD) rats, 7 weeks of age (TNO, 2005). The study was also designed to contrast absorption kinetics and the ability to reach bone marrow after acute PO and IP exposure and to determine pattern of tissue distribution and excretion following acute and repeated exposure. It was performed according to OECD Guideline 417 and EU Guideline B36 on Toxicokinetics, and GLP. The average particle size was 0.91 μm determined by the Fisher test method. The different exposure groups are indicated below (Table 4-35).

Table 4-35 Exposure groups

Group	Route	Dosing frequency	Dose level	In-life period	Number of animals
A	IV	single	1.57 mg/kg $SbCl_3$	3 days	4□
B	IP	single	100 mg/kg Sb_2O_3	3 days	4□
C	PO	single	100 mg/kg Sb_2O_3	3 days	4□ + 4□
D	PO	single	1000 mg/kg Sb_2O_3	3 days	4□ + 4□
E	PO	14 day repeated	1000 mg/kg Sb_2O_3	14 + 3 days	4□ + 4□
F	Control	non-dosed (background)		1 days	4□ + 4□

No study- or test substance-related signs of toxicity or unusual behaviour were observed. Body weight gain was normal.

Blood, plasma, urine, faeces and several organs (bone marrow, liver, brain, femur, kidneys, thyroid, lungs testes, uterus, muscle, heart, prostate, ovaries, skin, spleen), residual carcass and gastrointestinal tract including content were collected for analysis of the antimony concentration by ICP-AES (faeces and dose formulations) or by ICP-MS (blood, urine, cage wash and tissue), both after sample destruction with aqua regia.

Blood kinetics. The blood concentration of antimony versus time curve after intravenous injection (group A) can be divided into two phases; first a rapid distribution phase from the blood into an extravascular compartment leading to the lowest concentration of antimony in blood, during the 72 h of observation, at 4 hours after injection. After this time point a second, slow redistribution phase of antimony back into blood is observed with a maximal level of antimony being reached at 48 h after injection. A slight elimination from blood was observed at 72 h. After oral administration (groups C and D) the absorption from the gastrointestinal tract into the vascular system was slow with a Cmax at approximately 24 hours after dosing for both dose groups. Despite a ten times difference in dosing, the Cmax was only two times higher in group D (1000 mg/kg) than group C (100 mg/kg). The difference between both dose

groups for the calculated AUC (0-72h) was in the same range with a ratio between the oral low and the high dose of 2.09 and 1.37 in males and females, respectively.

After C_{max} was reached, elimination from blood was slow and a decrease in blood concentration was only observed at 72 h after both oral doses and IV administration. The half-life calculations were based on two time points only (48 and 72h) and therefore, no reliable half-life could be calculated from the obtained kinetics curves. However, a bioavailability factor was calculated from the AUC (0-72h) of the intravenous and oral dose groups, which showed that the oral bioavailability of antimony is low in both the oral low dose (C) and the high dose (D) groups at 0.3% and 0.05%, respectively.

During the 14 days repeated oral high dosing, antimony blood concentration increased almost linearly with time up to 24 h after the last dose and then decreased slowly (<5% per day) during the next two days. This indicates that a steady state level was not reached after 14 days exposure at the dose level tested.

The blood concentration versus time curve observed after IP injection (group B) presented no clear peak and no elimination phase for the mean of all animals. Individual data show that the blood concentration increased steadily in time for three animals but reached a maximum value at 72h for only two of the 4 animals. At necropsy it was observed that animals that received the IP dose still had visible residues of the test substance in the abdominal cavity indicating a continuous release of antimony from the dosing site in the IP-dose group (B). The dosed diantimony trioxide may only, due to its low water solubility, be slowly removed from the peritoneal cavity.

Absorption and excretion. In the rats intravenously dosed with $SbCl_3$ (group A), around 30% of the total antimony was excreted in the faeces within the first 24h and around 12% was recovered from the urine 24 h after dosing (Table 4-36). Therefore, it can be concluded that at 24 h, the biliary excretion (ending in feces) of absorbed antimony is higher than the amount of urinary excretion. In the IP-dosed animals (group B), 36% of the antimony was recovered from the faeces 72h after dosing. In the orally dosed animals around 80% and 100% (low and high dose, respectively) of the Sb was excreted via the faeces 72h after dosing. Only very low amounts of antimony were found in the urine, the tissue and carcass in both orally dosed groups. The urinary excretion of antimony in male and female rats after repeated oral dosing was monitored during the 14 days of the dosing period and for 3 days afterwards. The excretion to urine was relatively stable with some fluctuation during the exposure period. Naturally, excretion decreased after the last day of exposure.

Table 4-36 Recovery of antimony in excreta and tissues in the various single dose groups, expressed as % of the dose.

		Dose group					
		A	B	C	C	D	D
Dose				male	female	male	female
(mg Sb/kg bw)		0.752	86.1	87.3	87.2	865	852
Urine	0-24h	11.62	0.22	0.040	0.025	0.0066	0.0061
	24-48h	1.57	0.10	0.003	0.006	0.0009	0.0013
	48-72h	0.67	0.06	0.006	0.003	0.0004	0.0007

	Subtotal	**13.76**	**0.39**	**0.050**	**0.034**	**0.0079**	**0.0080**
Feces	0-24h	31.8	34.13	71.70	73.37	98.38	90.83
	24-48h	<7.8	1.96	7.45	8.21	1.68	7.83
	48-72h	<9.1	0.18	0.09	1.75	0.01	0.07
	Subtotal	**n.a.**	**36.26**	**79.23**	**83.33**	**100.07**	**98.74**
	Cage wash	<0-61	0.01	<0.01	<0.01	0.002	<0.001
	Total excreted	**n.a**	**36.67**	**79.29**	**83.37**	**100.08**	**98.75**
	GI-tract (including content)	**n.a**	**n.a**	**0.004**	**0.019**	**0.002**	**0.003**
	Tissues residues						
	Excised*	n.a	n.a	0.007	0.008	0.0043	0.0023
	Organs	n.a	n.a	0.002	0.002	0.0003	0.0004
	Carcass	n.a	n.a	0.023	0.021	0.0017	0.0023
	Total retained	**n.a**	**n.a**	**0.033**	**0.031**	**0.0063**	**0.0051**
	Total recovery	**n.a**	**n.a**	**79.33**	**83.42**	**100.09**	**98.76**

Tissue distribution. The ability of antimony to reach bone marrow after a single oral low dose was around 8 times lower than in the intravenously dosed rats (Table 4-37). In the IP-treated animals, however, the Sb content in bone marrow was approximately 40 times higher than in the bone marrow of the IV-dosed and oral high dose rats and approximately 300 times higher than in the bone marrow of the oral low dose rats.

In the control group, although very low amounts of Sb was found in whole blood (2.8 and 3.0 ng antimony/g tissue, males and females respectively), significant and comparable concentrations of antimony were measured in the bone marrow (80 and 142 ng antimony/ g tissue, males and females respectively) and thyroid (98 and 195 ng antimony/ g tissue, males and females respectively). These concentrations were also comparable with the antimony concentrations found in the same organs of the oral low dose group and were also similar to the blood concentration in these animals.

In the oral high dose group the antimony concentrations were again comparable in the bone marrow (1192 and 1996 ng antimony/g tissue, males and females respectively) and thyroid (1507 and 2103 ng antimony/g tissue, males and females respectively) and 2-3 times higher than in whole blood (708 and 640 ng antimony/g tissue, males and females respectively). After repeated oral dosing the antimony concentration in bone marrow was 2 times higher and in whole blood 10 times higher than the concentrations seen in the respective organs of the single oral high dose group.

The antimony concentration was, after either a single oral low or high dose in the other measured organs, in femur around 3 times higher, in kidney around 5 times higher and in lung, liver and skin around 10 times higher than controls. In heart, spleen and ovaries the antimony concentration was increased around 6, 5 and 2 times, respectively after a single oral low dose and around 10-15 times after a single oral high dose, compared with controls. The concentration in femur, kidney, lung, liver, skin, heart, spleen and ovaries increased further with a factor of ten after repeated oral high dosing compared with the single oral high dose.

In conclusion, the absorption of diantimony trioxide after acute oral exposure to diantimony trioxide particle suspension in this study was low; only 0.3% after administration of 100 mg/ kg bw and 0.05% after administration of 1000 mg/ kg bw. It should however, be noted that the exposure levels in this study in rats are at least 2-3 orders of magnitude higher than the exposure levels for humans (For details on human exposure, see section 4.1.1, exposure assessment). Since the absorption of diantimony trioxide is clearly dose dependent we are hesitant to use 0.3% oral absorption derived from this study also for oral absorption for humans.

From this study it can also be concluded that the absorption of antimony after oral dosing of diantimony trioxide is a slow process, with a C_{max} at approximately 24h, followed by an even slower elimination phase from blood. However, antimony undergoes significant distribution to most tissues as it binds to red blood cells. Highest levels were found in bone marrow and thyroidea, followed by ovaries, spleen, liver, lung, heart, femur and skin.

The major part of diantimony trioxide is excreted via faeces but it is also excreted via urine.

In the IP-dose group, residues of the test substance were found in the abdominal cavity indicating a continuous release of antimony to the blood. This exposure group also showed the highest antimony concentrations in blood and bone marrow.

Table 4-37 Tissue distribution of antimony (ng Sb/g tissue) after single or repeated dosing of diantimony trioxide in rats.

	F **Control**		**A** Single IV 1.57 mg/kg $SbCl_3$	**B** Single IP 100 mg/kg	**C** Single PO 100 mg/kg		**D** Single PO 1000 mg/kg		**E** 14 d rep 1000 mg/kg	
	male	**female**	**male**	**male**	**male**	**female**	**male**	**female**	**male**	**female**
Whole blood	2.8	3.0	1111	11383	181	189	708	640	8278	6886
Plasma	1.8	2.8	3.1	18	3.1	3.2	2.8	2.6	21	10
Bone marrow	80	142	1083	40517	141	89	1192	1996	2486	3517
Femur	18.9	10.0	214	6081	56	38	48	32	254	265
Liver	3.9	3.0	n.d.	n.d.	36	25	41	64	823	675
Kidney	2.7	2.4	n.d.	n.d.	15	8	12	23	323	261
Lung	3.6	2.2	n.d.	n.d.	36	27	41	61	746	882
Heart	3.6	3.1	n.d.	n.d.	22	14	42	41	643	356

Spleen	10	32	n.d.	n.d.	83	50	197	113	1485	1386
Brain	<1.4	<1.4	n.d.	n.d.	1.4	<1.4	2.2	<1.4	30	17
Thyroid	98	195	n.d.	n.d.	158	120	1507	2103	2639	2280
Testes	<0.8	n.a.	n.d.	n.d.	2.3	n.a.	2.8	n.a.	39	n.a.
Prostate	9.5	n.a.	n.d.	n.d.	11.4	n.a.	8.5	n.a.	80	n.a.
Uterus	n.a.	15.2	n.d.	n.d.	n.a.	12.8	n.a.	11.4	n.a.	116
Ovaries	n.a.	17.3	n.d.	n.d.	n.a.	29.4	n.a.	262	n.a.	665
Muscle	2.7	3.3	n.d.	n.d.	18	4.4	5.0	4.7	39	44
Skin	2.3	1.5	n.d.	n.d.	41	11	9.6	16	90	103
Residual Carcass	10.4	5.7	n.d.	n.d.	27	22	19	26	303	221

n.a.: not applicable
n.d.: not determined

In a very briefly reported study a single dose of 200 mg of diantimony trioxide suspended in 5 ml of water was administrated by stomach tube to each of 3 rats (strain and sex not stated) and the total daily collected urine was analysed for its antimony content by colorimetric microdetermination of antimony with rhodamine B (Gross et al., 1955a). The average daily excretion of antimony increased until the fifth day where after it dropped (Table 4-38).

Although only a minor part of the antimony administered could have been dissolved and the total absorbed fraction was not determined in this study it indicates absorption from the gastrointestinal tract of at least 3.24 %.

Table 4-38 Average daily urinary excretion of antimony, following a single oral dose of Sb_2O_3 of 200 mg to each of three rats.

Days after dosage	Average excretion, mg (as Sb_2O_3)	Average excretion, %
0-1	0.300	0.15
2-3	2.200	1.10
4-5	3.957	1.96
6-8	0.069	0.03
Total	6.526	3.24

In another excretion study, rats (number, strain and sex not stated) that had been receiving 2 % diantimony trioxide in their diet for eight months were shifted to diantimony trioxide free diet, and antimony determinations were made of the collected daily urine and faeces for 48 and 32 days, respectively (Gross et al., 1955a). Antimony determinations were made by colorimetric microdetermination of antimony with rhodamine B. A rapid decline in the faecal content of antimony that lasted for about 7 days was followed by a more gradual diminution of antimony excretion that continued for more than 30 days. The urinary antimony excretion was similarly biphasic, with a rapid decrease in the excretory rate during the first five to six days followed by a more gradual decline.

Within the first couple of days after the diet shift, the urine contained less than 1 % of the amount antimony recovered in the faeces, indicating that excretion of antimony occurred

mainly via faeces. After approximately 3 weeks, the excretion of antimony was higher in the urine than in the faeces.

The tissue distribution of diantimony trioxide was also determined. Some of the animals (number not stated) were killed immediately after removal from the 2 % diantimony trioxide diet and the tissues analysed for antimony. For comparison, analyses were also performed on the animals that had been removed from the antimony diet and held on antimony-free diet for 48 days. The tissue distribution of diantimony trioxide (Table 4-39) shows that the highest concentrations were found in the thyroid gland and liver, followed by spleen, kidney, heart, lungs and bone. The results of the tissue distribution study also indicate that considerable amounts of antimony remained in the body 40 days after diantimony trioxide administration had ceased.

Table 4-39. Antimony tissue distribution in rats after 8 months of feeding with 2 % Sb_2O_3 in the diet.

Organ	Immediately after removal from Sb_2O_3 diet, µg/ g (as Sb_2O_3)	40 days after removal from Sb_2O_3 diet, µg/ g (as Sb_2O_3)
Thyroid	156	75
Spleen	8.1	4.8
Kidney	6.0	2.1
Heart	5.1	1.8
Bone	2.5	0.9
Gastrointestinal cont.	1060	0.8
Muscle	0.3	0.6
Lungs	3.7	1.2
Liver	15.5	9.2
Gastrointestinal tissue	10.7	1.5

A chronic feeding toxicity study performed on male Wistar rats (number not stated) is briefly described in a summary (Hiraoka, 1986). This study is also described in chapter 4.1.2.6. Repeated dose toxicity. There were one control group and one group receiving 1 % (w/w) diantimony trioxide (corresponding approximately to 500 mg/kg bw/day) in the diet for 12 weeks. All the rats were allowed antimony-free diet for the following 12 weeks. Blood, liver, kidney, spleen, heart, lung, brain, testes, bone, hair and nail were taken from the rats 0, 4 and 12 weeks after cessation of antimony trioxide administration and analysed for antimony content.

Antimony was detected in all organs examined and at all time points. At the time of removal of antimony containing diet the highest concentrations of antimony were found in the blood, spleen, lungs, kidney, hair and bone (Table 4-40). Four weeks after removal of diantimony trioxide containing diet, the antimony concentrations in the organs were generally unchanged from those measured immediately after exposure, while after a further 8 weeks they had decreased by about 50%.

Table 4-40 Antimony concentrations (ppm) in organs after administration of 1% (w/w) diantimony trioxide containing diet for twelve weeks to rats (Hiraoka, 1986).

	Control at 12 weeks after removal	**Time of removal from antimony containing diet**	**4 weeks after removal**	**12 weeks after removal**
Blood	0.05+0.01	76.41±42.73	81.05±6.79	40.16±5.19
Liver	0.01	10.17±5.48	5.95±0.77	2.80±0.46*
Kidney	0.004+0.003	16.07±2.52	9.94±1.60**	8.77±0.83**
Spleen	0.01+0.004	47.17±8.92	39.06±4.73	35.63±5.18
Heart	0.004+0.001	9.45±1.94	8.50±0.86	6.29±0.50*
Lung	0.005+0.002	22.23±4.67	20.08±2.25	10.86±2.14**
Brain	0.002	1.79±0.56	1.29±0.21	1.34±0.13
Testis	0.002+0.002	1.67±0.32	1.04±0.16*	0.84±0.09**
Bone	ND	5.56±3.04	1.07±0.26*	2.32±0.39
Hair	ND	14.17±8.26	22.88±7.65	10.63±2.39
Nail	ND	35.77	7.00	5.34

*Statistically significant from the time of removal of antimony diet ($p<0.05$)

** Statistically significant from the time of removal of antimony diet ($p<0.01$)

ND: not detected

This study shows that diantimony trioxide is absorbed from the gastrointestinal tract and distributed to most, if not all, organs, including e.g. testes.

In another study, male Wistar rats were fed a diet containing 1.0 or 2.0 % (corresponding approximately to 500 mg/kg/day or 1000 mg/kg/day) diantimony trioxide for 24 weeks (Sunagawa, 1981). This study is also reported in chapter 4.1.2.6. Repeated dose toxicity. Antimony content in 5 animals/group was measured in liver, kidney, spleen, heart, brain, lung, stomach, testes and blood.

Antimony was detected in all organs examined. The highest contents were found in blood, spleen, lung, kidney liver and stomach (Table 4-41). There was no appreciable difference in organ contents between the two exposure groups.

Table 4-41 Antimony concentrations in organs and blood after administration of diantimony trioxide containing diet for 24 weeks to rats. Values are mean of five rats ± SD (Sunagawa, 1981).

	Control	**1.0% Sb_2O_3**	**2.0% Sb_2O_3**
	(ppb)	**(ppm)**	**(ppm)**
Liver	2.42±1.13	15.19±9.07	17.74±4.31
Kidney	7.12±3.19	19.99±2.81	16.64±5.6
Spleen	19.89±12.40	61.88±6.78	55.71±14.81
Heart	7.13±2.94	12.63±4.42	11.91±3.53
Brain	9.09±8.58	4.04±1.74	4.03±0.97

Lung	15.6±5.72	28.84±4.67	27.26±8.12
Stomach	5.92±3.79	16.56±9.56	24.09±10.01
Testis	4.22±4.06	3.21±1.06	3.16±0.53
Blood	59.64±14.71	148.11±25.91	114.87±35.94

This study supports what other studies have shown; that there is a high distribution of antimony to blood, spleen, lung, liver and kidney after oral exposure, but also to testes and brain.

Oral data on antimony compounds other than diantimony trioxide

Some additional data on oral absorption from studies on antimony compounds with higher water solubility than diantimony trioxide are available. However, these studies were generated before the establishment of standardised test guidelines and thus do not comply with current GLP and guideline requirements.

Waitz et al. (1965) measured excreted antimony in urine 25 hours after single oral doses of tartar emetic (8, 16, and 32 mg ^{124}Sb/kg bw) in mice. The excretion in the respective dose groups was 7.9, 5.3, and 4.6 %. After a single oral dose of 16 mg ^{124}Sb/kg to separate groups of mice the antimony absorption reached 16 %, 25 hours after administration when total radioactivity was measured and added up in urine, carcass and tissues. In the same study, an additional group of mice was daily exposed to tartar emetic, 16 mg ^{124}Sb/kg bw by gavage, for 10 consecutive days. Daily measurements of antimony in urine were conducted representing the percentage of each daily dose excreted in the subsequent 24-hour period. The mean urinary excretion during the whole study period of 10 days was 5 %, but varied over the period from 0.15 to 7.9 %. The results of this study support a sub linear relation between ingested antimony and antimony in urine, suggesting decreased absorption at higher doses.

In another study conducted by Chertok and Lake (1970) nuclear debris from a sub-surface nuclear detonation test was used as test substance. The aim of the study was to yield information regarding the identities and relative concentrations of radionuclides in nuclear debris and to further study the bioavailability of these radionuclides. The nuclear debris was sieved and the fraction containing particles in a range of 60-250 µm size was weighed and placed in gelatine capsules with 2.79 g/capsule (amount of antimony not stated). Two female dogs were given one capsule each, orally, and then analysed daily in a whole-body detector for ten different gamma-emitting radionuclides, including ^{122}Sb. Urine and faeces were collected daily and analysed in the same detector. The daily analyses were continued until very low levels were reached, approximately 5 days. The whole body percentages were based on first day's analysis as 100 %, and the urine and faeces values represent percentages of the initial dose based on a standard, not further explained so the meaning is unclear. Approximately 3 % of the oral ^{122}Sb dose was found in the urine within 3 days after exposure, representing only the fraction excreted via the kidneys after intestinal absorption. Most of it was excreted on the first day. Gerber et al. (1982) conducted a study where pregnant mice were exposed to ^{125}Sb (18.5Bq/kg feed) in the diet from the day of the vaginal plug until sacrificed on day 3, 5 and 7 of pregnancy. Radioactivity in whole body (without intestinal tract) and tissues was measured by a gamma spectrometer with no further details given. Equilibrium for antimony radioactivity in the body was attained after 4 days. Intestinal

absorption was roughly assessed to be 1.7 %, from the equilibrium level and on the basis of 3 g feed ingested daily. However, the average turnover time for ^{125}Sb was determined to be 6 h and the authors thus conclude that the true value for intestinal antimony absorption was 7 %. The calculations are not explained in detail.

A study on rats (Moskalev, 1964) orally exposed to ^{124}Sb tartar emetic, indicates that the absorption from the gastrointestinal tract was about 5 %.

Little data is available on the absorption of antimony in young individuals. However, the retention of antimony on the fifth day after administration of ^{125}Sb given in chloride form was 40 and 20 % of the dose for suckling and 15-day-old suckling rat pups, respectively (Inaba et al., 1983). The approach of the NEA expert group (NEA/OECD, 1988) is a 20 % fractional absorption value for 3-months-old infants.

Other studies on oral absorption of antimony compounds in experimental animals report absorption levels between 1 and 20 % (Van Bruwaene et al., 1982; Felicetti et al., 1974a; ICRP, 1981). However, the great majority of the available data base on oral antimony exposure is of limited use for a conclusive decision on the level of intestinal absorption of antimony. Many studies were performed before GLP and guideline standards were available and lack in reporting of dose levels, which is important in this context (e. g. radioactivity (Bq) rather than antimony/kg bw is given where radioactive compound is used). Moreover, the bioavailability of antimony is most likely affected by the administration matrix, which in these studies vary from aqueous solutions to corn oil to rodent feed containing fibres and phytate, probably reducing the uptake, to mixtures with other toxic compounds such as As. Nevertheless, most studies indicate an intestinal absorption of at least 3 to 8 % antimony despite the mentioned differences in study protocols.

Intravenous

To study the elimination and metabolism of trivalent antimony 5 adult, male Sprague Dawley rats were given an intravenous (i.v.) dose of 800 µg $SbCl_3$/ kg (Bailly et al., 1991). The bile from each rat was collected for seven hours and its content of antimony was determined. During this time, the biliary flow ranged from 0.8 to 0.5 ml/h and about 10 % (range 6-15 %) of the amount of antimony administered was recovered in the bile.

To further determine whether the antimony secreted into the bile could undergo an enterohepatic cycle, the bile of one rat pre-treated with $SbCl_3$ (800 µg antimony/kg) and containing 70 µg antimony/ml was administered intraduodenally at a dose of 1 ml/kg to each of 3 control anaesthetised rats whose bile ducts were cannulated for bile duct collection. The animals were killed five hours after treatment. The bile collected during this period contained a mean value of 22.4 % (SE 1.3) % of the administered antimony dose whereas the liver had accumulated 10.7 (2.0) % and the kidney 2.0 (0.3) % of the dose. This result indicates that antimony undergoes enterohepatic recirculation.

To also asses if glutathione (GSH) plays a role in the biliary excretion of antimony, 6 rats each were treated with either D-L-buthionine-(S-R)-sulfoximine (BSO) to depress the hepatic GSH content or with butylated hydroxytoluene (BHT) to increase its concentration, before administration of a single intraperitoneal dose of $SbCl_3$ (800 and 200 µg antimony/kg to BSO and BHT-treated rats, respectively). At the time of killing (48 h after administration of

antimony) the average GSH concentration in liver was 66 % (BSO) and 120 % (BHT) of that in control animals. GSH depletion before antimony exposures decreased antimony faecal excretion and increased antimony urinary excretion. Increased GSH concentrations in liver had the opposite effect. This indicates the role of GSH in the biliary excretion of antimony.

Attempts were also made to characterize the nature of the metabolites in bile and urine. Paper chromatography of bile collected from animals pre-treated with $SbCl_3$; 800 µg Sb/kg intraperitoneally showed that Sb can be detected at a spot with an Rf value similar to that of GS-Sb-GS complex (results not shown). Additional attempts (hydride formation, ion exchange chromatography, various mineralization procedures) to detect any organic form of Sb failed. This may indicate that Sb is not methylated in the organism and is excreted in the inorganic form in urine.

This study indicates that trivalent antimony can be excreted in the bile and undergoes enterohepatic recirculation. Sb can form a complex with GSH and may not be methylated in vivo.

4.1.2.1.2 Studies in humans

In vivo studies

Inhalation

The retention of ^{125}Sb in the lungs of seven male workers accidentally exposed to ^{125}Sb oxide aerosols has been measured (Garg et al., 2003).

A saw-cutting operation of a zircaloy pressure tube led to the generation and release of ^{125}Sb aerosols in the working environment and caused an accidental inhalation exposure to seven persons. The particle size was likely <2µm AMAD (Activity Median Aerodynamic Diameter), but the aerosol size could be about 5-6 µm AMAD when using band saw or reciprocating saw as in this study, and therefore a default value of 5 µm AMAD is suggested. There is no data on ^{125}Sb oxide concentration in the air at the accident, so the inhaled doses are unknown. The residence time in the active area for these subjects was expected to range from about 30 min to 5 h. After the incident, all the subjects worked in areas where the possibility of further exposures with ^{125}Sb was practically ruled out. Thus, there were no multiple exposures observed in any of the workers after the start of follow up measurements. The ages of the workers ranged from 33 to 45 years. Four of them were non-smokers, two were smokers, and one was a heavy smoker. All of them were healthy workers without any known history of chronic lung diseases. After 1 day four persons were monitored by whole body counting; two persons were monitored after 2 days; and one person was monitored 3 days after the incidence. Follow up studies was carried out on all subjects for different periods of time ranging from 1 to 3 days post intake to 199 to 2422 d post intake.

The results of measurements of total body radioactivity of ^{125}Sb shows that most of the radioactivity was found confined to the lungs only. Practically no radioactivity was detectable over the liver and the other parts of the body. The initial rapid clearance after inhalation exposure is considered to be mainly from the upper thorax/gastrointestinal tract and is referred

to as the fast clearance/rapid phase. Material not cleared during the rapid phase is considered to be mainly in the alveolar interstitial (AI) region, which is considered to be the main long-term retention site in the lung. Therefore, in this study the retained activity at 7 days after exposure was taken as initial alveolar deposition (IAD). It was seen that the biological elimination half times for the long-term component are more than 600 days in all cases. The reported data shows a considerable inter-subject variation, which arises due to natural biological variability among individuals exposed under similar conditions. It also shows that the retention in all the smokers is considerably greater than that in all the non-smokers. This study indicates a slow elimination half-life from the lung, ranging from 600 to 1,100 days for non-smokers and 1,700 to 3,700 days for smokers. The retention in the lungs after 180 days was found to be more than 51% of the estimated IAD in all the seven cases.

Concentrations of antimony (Sb) in lung, liver, and kidney tissue from a group of deceased smelter workers from northern Sweden were compared with a group of persons without occupational exposure from a nearby area (Gerhardsson et al., 1982). The subjects consisted of 40 deceased men who had worked at Rönnskärsverken, a smelter and refinery, and who died during the period 1976-1978. As referents, 11 age-matched men were selected from a rural area, approximately 50 km from the factory. The average age of the workers was 66.6 years and of the reference group 67.5 years. In the group of smelter workers 15 died from malignancies, 6 from respiratory cancer, a total of 17 workers died from cardiovascular diseases, and 8 from other causes. In the reference group 7 died from myocardial infarction and 3 from cerebrovascular diseases. The mean values of exposure time were in the different groups; 30.9 ± 7.1 years in the malignancy group, 32.5 ± 8.2 years in the cardiovascular diseases group and 28.9 ± 10 years in the group exposed workers with other diagnosis. Antimony concentrations in lung tissue were determined in all of the 40 deceased smelter and refinery workers and in the 11 referents. For 21 of the workers and 8 of the referents antimony concentrations were also determined in the liver and kidney cortex.

The antimony concentrations in the lung, liver and kidney tissues of the reference group were reported to agree well with other studies. The antimony concentration in lung tissue of exposed workers was 12-fold higher than that of the reference group (315 µg/kg wet weight compared to 26 µg/kg wet weight as mean values). The mean antimony concentration in liver tissue and kidney cortex of smelter workers did not differ from that of the reference group.

The time from the last exposure to date of death varied between 0 and 23 years. No significant differences of antimony concentration in lung tissue were found when death occurred <1, 1-5, 6-10 or 11-23 years after the cessation of exposure, indicating a long biological half-life in lung tissue.

The amount of inhaled diantimony trioxide dust retained in the lungs of 113 male workers employed in the antimony process was determined by X-ray spectrophotometry (McCallum et al., 1971). The men had been employed between 6 months and 43 years in the antimony process, however, the majority had worked at the factory for less than 20 years and in most cases this had involved the operation of various types of furnace for different periods of time, or work in other sections of the process such as the baghouse.

The amount of antimony in the lungs of these men ranged from zero to just over 11 mg/cm^2 and tended to rise as the period of employment in the antimony work increased. The individual levels of lung antimony showed a good deal of variation, which may be partly because men who had remained at the works for very long periods had not been continuously exposed to high levels of antimony in air. Still, there appeared to be an association between length of employment and lung antimony level (r= +0.552). If only the first 20 years of employment were considered the association between "period of employment" and "lung antimony level" was higher (r= +0.703). No statistical calculations were presented.

This study indicates that there might be an association between length of employment in antimony industry and antimony lung burden.

A group of 109 male workers from two glass-producing factories were examined for the contents of antimony and lead in blood and urine (Lüdersdorf et al., 1987b). This summary will only report the results from the antimony exposure. Blood and urine samples were collected at the end of the shift on days when antimony-containing glass was being produced. Determination of airborne Sb_2O_3-concentrations were done by personal and stationary air sampling at specific work places. The measured air values were between <5 and 840µg/m^3.

Because most antimony in blood is bound to the erythrocytes the analyses were done on whole blood. The resulting SbB-values were between 0.4 and 3.1µg/l. Measured blood values from 51 unexposed control persons resulted in SbB-values ranging from 0.3-1.7µg/l.The concentrations of antimony in urine were between 0.2 and 15.7 µg/l. Eight unexposed persons urine were determined to have SbU-values ranging from 0.2 to 0.7 µg/l. There were no measurements on antimony content in faeces. Because only two personal air samples were found above the detection limit of 50 µg Sb_2O_3/m^3, a correlation between the external exposure and SbU could not be calculated.

This study shows that antimony is partly excreted renally after inhalation of diantimony trioxide.

During a five-day survey the Sb_2O_3 exposure of workers in a textile industry was monitored (Iavicoli et al., 2002). The processing is carried out in different manufacturing departments while the preparation and application of Sb_2O_3 based flame retardant takes place in the dyeing plant. A group of 21 male workers, living in a rural area, were examined to evaluate personal Sb_2O_3 exposure and Sb urinary levels. Ambient air and urine samples were taken each day. 15 males from a rural area who were not occupationally exposed to Sb or its compounds were used as controls.

Airborne Sb_2O_3 was collected by personal and area samplers (n=66). In most cases, sampling was performed during the entire work shift, even if in a few instances, the particular nature of the manufacturing cycle required some of the personal samplings to be divided into two sub-samplings, each of which had duration of about 4 hours. Urine samples were collected at the beginning and end of each work-shift. On days 1, 2, 3 and 5 of the workweek only one shift was examined, while on day 4, due to the larger amounts in production than on the remaining days, the survey included two work-shifts.

The workers were divided into two sub-groups; "high exposure operators" and "less exposed operators". The mean exposure of the "high exposure operators" was 0.11 ± 0.07 µg Sb/m^3 and for the "less exposed operators" 0.05 ± 0.04 µg Sb/m^3. The mean urinary Sb levels at the beginning and end of the shift were 0.39 ± 0.26 µg/l and 0.46 ± 0.32 µg/l for high (n=24) and 0.18 ± 0.10 µg/l and 0.18 ± 0.06 µg/l for less (n=15) exposed operators respectively. The mean urinary Sb levels for the controls, measured each morning, were 0.1 ± 0.06 µg/l. There were no measurements on antimony content in faeces.

In conclusion, inhaled diantimony trioxide can be absorbed and excreted in the urine. Higher exposure levels leads to higher urinary Sb levels.

A group of 21 workers, working in the production of lead batteries, was examined with regard to the antimony concentration in blood (Sb-B) and urine (Sb-U); seven men from the casting area and 14 men from the formation area (Kentner et al., 1995). Personal air sampling assessed antimony (Sb_2O_3) and volatile SbH_3 in the air. The collection of urine samples was conducted after at least 3 workdays at the beginning (U1) and at the end (U2) of the fourth or fifth workday. During U2, venous blood samples were also collected. After the following weekend without Sb exposure the third urine sample was collected at the beginning of the first shift (U3). According to the change in shifts worked (early, normal or late), there was 56-104h between samples U2 and U3. Sb was determined by hydride atomic absorption spectrometry. Air samples and blood samples were analysed after wet oxidative digestion. The Sb concentrations from urine were adjusted to creatinine.

The results of the measurement are presented in Table 4-42

Table 4-42 Median and minimum/maximum concentrations of antimony in the workplace air (Sb-A), in urine (Sb-U) and in the blood (Sb-B) at the end of the last shift of the working week (U2)

	Casters			Formers		
	Min.	Median	Max.	Min.	Median	Max.
Sb-A (µg/m3)	1.2	4.5	6.6	0.6	12.4	41.5
Sb-U (µg/ g crea.)	2.8	3.9	5.6	3.5	15.2	23.4
Sb-B (µg/l)	0.5	2.6	3.4	0.5	10.1	17.9

The median Sb concentrations in blood and urine for both formers and casters were reported to be considerably above the median values for occupationally non-exposed individuals (<0.5 µg Sb/l in urine and <1 µg/l in blood) (Lüdersdorf et al., 1987a). Workers from the formation area (formers) had a 3 times higher median Sb exposure than those from the casting area. This higher exposure was also reflected in the 3-4 times higher Sb-U and Sb-B levels observed in the formers. The formers were exposed to both SbH_3 and Sb_2O_3, the casters only to Sb_2O_3. There were no measurements on antimony content in faeces.

The elimination profiles of the casters and formers indicated that during the weekend without Sb exposure complete excretion of the antimony absorbed during the preceding week took place. The half-life of renal elimination amounts to approximately 4 days for both groups.

This study shows a correlation between Sb concentrations in air and in blood/urine.

Female workers engaged in an antimony plant were exposed to antimony aerosol containing metallic Sb, SbS_5 and Sb_2O_3 (concentrations were not reported) (Belyaeva, 1967). The levels of antimony were measured in blood and urine from 161 women with high exposure, 157 women with lower exposure and from 115 control women. Antimony was also measured in breast milk, placenta, amniotic fluid and umbilical cord blood. The high exposure group worked in areas with high dust levels involving production of e.g. SbS_5, smelting and enrichment, the lower exposure group worked in the same plant in the chemical laboratory and control department. The control group had the same age and working conditions as the high exposure group but with no exposure of antimony. The mean values (mg %) of antimony in blood and urine were 5.3 ± 0.6 and 2.9 ±0.5 respectively in the high exposure group, 4.0 ± 0.5 and 2.1 ± 0.4 respectively in the low exposure group and 0.33 ± 0.06 in blood in the controls (the antimony level in urine from controls were not reported). In the workers, antimony was detected also in breast milk (3.3 ± 2.2 mg/l), placenta (3.2-12.6 mg %), amniotic fluid (6.2 ± 2.8 mg %) and umbilical cord blood.

Although the unit (mg %) used in this study to present the results is difficult to understand, this study shows that antimony can be distributed to the foetus and excreted in breast milk, thus exposure may occur both in utero and during breast-feeding.

Dermal

No study regarding the toxicokinetics of diantimony trioxide after dermal *in vivo* exposure has been located.

Oral

Using different spectrometric methods antimony has been detected in human breast milk, umbilical cord, placenta, amniotic fluid and foetal liver. The exposure route and source of antimony exposure in all of these studies is unknown but it may be assumed that the dominating exposure route is oral and some of it may come from uptake of diantimony trioxide.

Clemente and co-workers (1982) measured the concentration of Sb in more than 130 human milk samples (the exact number was not stated) collected from 21 different Italian subjects. The exposure of the subjects was not given: The samples of human milk are representative of mature milk having been collected 15 or more days after the childbirth. 49 samples from 16 subjects were above the detection limit and the mean concentration of antimony was 3.0 ± 0.4 ng/g wet weight (range < 0.05-12.9).

The concentration of antimony was also measured in a pooled human milk reference sample (not further specified) (Iyengar et al., 1982). Nine samples were taken from the pooled milk sample and 18 determinations were performed. The mean concentration of antimony was 13 ± 5 ng/g dry weight.

The transfer factor of antimony from food to milk in nursing mothers was measured by Wappelhorst and co-workers (2002). Nineteen mothers from Germany, Poland and the Czech

Republic participated in the study, providing 2779 food and 536 breast milk samples. Samples were taken daily over 2 to 8 weeks from each mother. Altogether, samples were taken over 73 weeks. To reach the detection limit, three separate digestions were made of each milk sample. For each digestion round (microwave assisted pressure digestion), 500 mg freeze-dried milk powder, 4 ml HNO_3 and 2 ml H_2O_2 were weighed and digested in teflon vessels. The three digestion solutions were then pooled and evaporated to approximately 5 ml before ICP-MS analysis of antimony content. The average antimony intake of the nursing mothers (0.154 ± 0.351 µg/kg/d) was calculated by analyzing food duplicates and recording the actual amounts of each food consumed. The mean antimony concentration in breast milk was 0.14 µg/L (range between 0.06 and 0.57 µg/L). The transfer factor, calculated as g Sb/kg food divided by g Sb/ L milk was 13.2.

These studies show that antimony is present in human breast milk in the normal population.

The concentration of numerous trace elements, including antimony, was measured in human fetal livers with gestation ages between 12 and 40 w (Shand et al., 1985). No details on the fetuses or the pregnant women were given. The results show that the mean antimony concentration in fetal livers was 0.02 µg/g dry weight, ranging between 0.01 and 0.06 µg/g dry weight. Published literature values for adult livers were presented for comparison, ranging from 0.01 to 0.42 µg/g dry weight. No data on maternal antimony levels were presented.

This study indicates an in utero exposure of human fetuses to antimony.

In vitro studies

An in vitro study on human skin has been performed in accordance with OECD Guideline for Testing of Chemicals, Guideline 428, and the OECD Environmental Health and Safety Publication Series on Testing and Assessment No.28 (Roper and Stupart, 2006). The aim with this study was to establish the likely dermal penetration of antimony resulting from topical exposure to diantimony trioxide. Ten samples of full-thickness human skin (originating from 3 abdomen and 7 breast samples) were obtained from six females (24-57 years old), attending to a plastic surgery unit. The women's state of health is not reported. The samples were cut into smaller pieces and stored at -20 °C until analysis. Split-thickness membranes were prepared (200-400 µm depth), and mounted into an automated flow-trough diffusion cell apparatus. The surface area of exposed skin (stratum corneum) within the cells was 0.64 cm^2, and the receptor chamber volume was 0.25 mL. The peristaltic pumps were adjusted to maintain a flow-rate of about 1.5 mL/h. A phosphate buffered saline solution containing streptomycin (0.1 mg/mL) and penicillin G (100 units/mL), with the pH adjusted to 7.4 was used as receptor fluid.

Diantimony trioxide was dissolved in a hydroxypropyl methylcellulose in water solution (1 % w/v). A solution of only hydroxypropyl methylcellulose in water (test preparation 3) was used as a control blank in all experiments to assess the background and endogenous levels of diantimony trioxide in the test samples. Four skin samples (from two donors) were rejected due to high background levels of diantimony trioxide (0.340-0.493 total µg). Six samples (originating from 1 abdomen and 5 breast samples) were further analysed in the study. A 'low' dose and a 'high' dose of diantimony trioxide; 100 µg/cm^2 and 300 µg/cm^2 were used

respectively. 6.4 µL of the test preparation were applied to the skin surface, and receptor fluid was collected every sixth hour for 24 hrs (0-6 h, 6-12 h, 12-18 h, and 18-24 h). At 6 h post dose, the exposed skin surface was washed with soap solution. The skin surface was then dried with tissue swabs. At 24 h post dose, the underside of the skin was rinsed with receptor fluid, and collected into the 18-24 h vial. The skin was then dried with 3 tissue swabs. These swabs were pooled into the 6 h tissue vial. The stratum corneum was removed with 20 successive tape strips. The blank, receptor fluid, skin wash, tissue swabs and tape strips were analysed for the amount of diantimony trioxide by inductively coupled plasma mass spectrometry (ICP-MS). The skin under the cell flange (unexposed skin) was cut away from the exposed skin. The mass balance for the two test concentrations, 100 µg/cm^2 and 300 µg/cm^2, were 89.77% and 106.41% respectively. The results are presented in Table 4-43.

Table 4-43 Corrected distribution (endogenous levels (control levels) subtracted) of mean values and (range) of diantimony trioxide at 24 h post dose following topical application on human split-thickness skin.

		Dose				
		Control	**100µg/cm²**	**100µg/cm² Normalised to 100% mass balance**	**300µg/cm²**	**300µg/cm² Normalised to 100% mass balance**
Dislodgeble dose[1]	µg/cm² % applied dose	0.37	71.89 (55.8-96.07) 89.35 (69.45-119.40)	99.56	242.07 (221.67-248.58) 106.23 (97.28-120.25	99.82
Stratum corneum[2]	µg/cm² % applied dose	0.02	0.28 (0-0.56) 0.34 (0.0.74)	0.38	0.17 (0-0.41) 0.07 (0-0.12)	0.07
Absorbed dose[3]	µg/cm² % applied dose	0.06	0.01(0-0.03) 0.01 (0-0.03)	0.01	0.05 (0-0.3) 0.02 (0-0.13)	0.02
Dermal delivery[4]	µg/cm² % applied dose	0.08	0.06 (0-0.32) 0.07 (0-0.4)	0.06	0.24 (0-0.92) 0.10 (0-0.4)	0.10
Total absorbed dose[5]	µg/cm² % applied dose		0.2 0.24	0.25	0.325 0.135	0.135
Mass balance	µg/cm² % applied dose	0.47	72.23 (55.9-96.95) 89.77 (69.48-120.49)	100	242.47 (226.23-274) 106.41 (97.39-120.25)	100

1: The mass of test items that is removable from the application site

2: Total amount of test items in stratum corneum

3: The mass of test item reaching the receptor fluid or systemic circulation within a specified period of time.

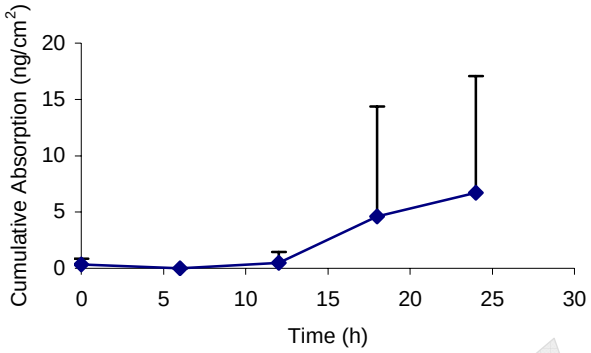
4: The sum of the applied dose found in treated skin (without the tape stripped stratum corneum) and the absorbed dose at the end of the experiment.

5: The sum of the applied dose found in treated skin , half of the amount found in stratum corneum and the absorbed dose at the end of the experiment.

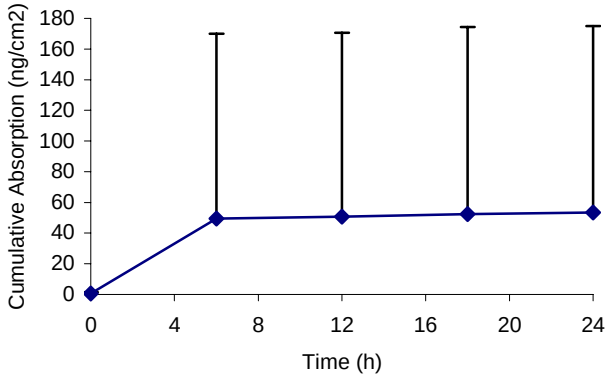
DRAFT

Figure 4-1 Absorption profile of diantimony trioxide (ng/cm^2) in receptor fluid following topical application of 100 µg/g or 300 µg/g to human split-thickness skin (mean + SD, n=6)

100 µg/g



300 µg/g



The cumulative absorption profiles in receptor fluid (

Figure 4-1) shows that the absorption of the "low"dose increases with time from 12h-24 h, while at the "high dose" the absorption seems to reach a plateau after 6 h. Endogenous diantimony trioxide was detected in all skin samples and the elevation in skin after topical exposure to diantimony trioxide was low, less than 1 %. All 20 strip tapes were pooled together to give an assumption of the total concentration in stratum corneum rather than dividing it into different skin layers. This may be a reasonable approach as the total concentration was low in stratum corneum but it gives no information about the concentration in different cell layers, and subsequently it affects the decision whether the amount in the stratum corneum should be included in the calculation of the absorption or not. Since no information is given of the presence of diantimony trioxide in different cell layers of stratum corneum, it seems reasonable to include half of the amount found in stratum corneum in the calculated total dermal absorbed dose, as was decided for the substance HBCDD at previous discussions at the Technical Committee On New And Existing Substances (TCNES).

For the two tested low and high concentrations of diantimony trioxide the concentration in stratum corneum was 0.38 % and 0.07 % and the dermal delivery was 0.07 % and 0.10 % respectively for the two tested concentrations. These values are normalised to a 100 % mass balance.

Under the present testing conditions the total dermal absorption is estimated to 0.26 % (0.07 + 0.38/2) and 0.135 % (0.1 + 0.07/2) respectively, for the two tested low and high dose., Therefore, based on this study a value of 0.26 % for dermal absorption is suggested. This calculated value is low. Still, the donor specific dermal absorption in the corrected vs. the non-corrected exposure groups is significantly different (Wilcoxon-signed rank test, $p < 0.01$). The large standard deviation in exposed samples in this test may be explained by the limited number of samples with possibly intra and inter individual differences.

In conclusion, even though the donor specific dermal absorption in the corrected vs. the non-corrected combined exposure groups is significantly different, the dermal absorbtion of 0.26 % is still low, and dermal absorption of diantimony trioxide is considered negligible.

4.1.2.1.3 Other studies

Systemic availability of diantimony trioxide is a function of regional deposition in the respiratory tract, which in turn depends foremost on particle size. However, product specific physical particle size distributions do not necessarily reflect the particle size of aerosols that may be formed under practically relevant workplace conditions, for example during manual operations such as filling and emptying of bags, or under mechanical agitation as in mixing and weighing operation. Therefore, physical particle size distribution and the median (d50) physical particle size was determined for the airborne fraction of 8 samples of diantimony trioxide, representing the range of commercially available particle sizes (EBRC, 2005c). The airborne fraction was generated during mechanical agitation in a rotating drum according to the method of Heubach (1991) and the particle size distribution was determined by laser diffraction according to OECD TG 110. The relative densities of the different samples were given by the different producers, the methods used was not stated. The data on particle size, relative density and calculated mass median aerodynamic diameter (MMAD), the relevant parameter for predicting airway deposition of particulate matter, is presented in Table 4-44

Table 4-44 Data on particle size, dustiness and relative density of eight diantimony trioxide samples (A-H) and zinc oxide (EBRC, 2005c).

Sample	Relative density (g/cm3)	d50* (μm)	MMAD# of airborne particles (μm)	Geometric standard deviation of MMAD
A	5.4-5.7	5.96	35.23	3.97
B	5.44	5.76	28.85	3.53
C	5.6	3.11	16.87	3.58
D	5.4-5.7	1.59	7.42	3.03
E	5.6	1.26	4.33	2.44
F	5.44	1.18	4.12	2.64
G	5.4-5.7	0.92	10.21	3.49
H	5.5	0.84	37.01	5.00
Zinc oxide	5.6	~1	36.04	3.70

*D50 = median physical particle size

#MMAD = mass median aerodynamic diameter

Except for the two samples with the highest median physical particle size the physical particle size distribution for all other samples show bimodal distributions (having two separated peaks) (data not shown here). According to the author this can only be explained by a high tendency of agglomeration. It was observed that the fraction of aggregates decreased with increasing particle size, and one interpretation would be that with decreasing particle size, the surface of particles in relation to their volume increases, and thus may enhance aggregation via adhesion. A comparison of the various diantimony trioxide samples shows that the material with a physical particle size between 1.18-1.26 μm, corresponding to a MMAD of 4.12-4.33 μm, yields the finest airborne particles after mechanical agitation. The other samples obviously undergo particle aggregation to a greater extent, thus rendering larger particle sizes in airborne matter. Therefore, the material with the largest proportion of small particles (MMAD of 4.12 μm) and hence the highest fraction of respirable material can be selected as a reasonable worst case for occupational inhalation exposure. In order to estimate the deposition in the respiratory tract (head, tracheobronchial and pulmonary region) of particles with a MMAD of 4.12 μm (representing the finest particles) the Multiple Path Particle Deposition (MPPD) model (CIIT, 2002-2006) was used with the following input data; The human five lobular lung model, a polydisperse particle distribution, oronasal (normal augmenter) mode, a full shift breathing rate of 10 m^3 - corresponding to a tidal volume of 1100 ml and a breathing frequency of 20/min, and an aerosol concentration of 500 $\mu g/m^3$.

The calculated deposition fractions were; 79 % in the head region, 2.6 % in the tracheobronchial region and 6.0 % in the pulmonary region.

The fate and uptake of deposited particles depends on the clearance mechanisms present in the different parts of the airway. In the head region, most material will be cleared rapidly, either by expulsion or by translocation to the gastrointestinal tract. A small fraction will be subjected to more prolonged retention, which can result in direct local absorption. More or less the same is true for the tracheobronchial region, where the largest part of the deposited

material will be cleared to the pharynx (mainly by mucociliary clearance) followed by clearance to the gastrointestinal tract, and only a small fraction will be retained (ICRP, 1994). Once translocated to the gastrointestinal tract, the uptake will be in accordance with oral uptake kinetics.

In consequence, the material deposited in the head (79 %) and tracheobronchial (2.6 %) regions would be translocated to the gastrointestinal tract without any relevant dissolution in view of the low water solubility of diantimony trioxide, where it would be subject to gastrointestinal uptake at a ratio of 1 % (for explanation of this oral absorption value see last paragraph in section 4.1.2.1.4 summary of toxicokinetics below) . Although histopathology reveals alveolar accumulation of inhaled diantimony trioxide the material that is deposited in the pulmonary region (6.0 %) may be assumed by default to be absorbed to 100 %. This absorbtion value is chosen in the absence of relevant scientific data regarding alveolar antimony absorption although knowing that this is a conservative choice. Thus total predicted inhalation absorption would be: 79% × 0.01 + 2.6% × 0.010 + 6.0% = 6.82 %

This way of calculating inhalation absorption figures has previously been used in the EU RARs on zinc compounds.

4.1.2.1.4 Summary of toxicokinetics

There are no quantitative data available, regarding the absorption, disposition and retention of diantimony trioxide in humans after inhalation, dermal or oral exposure. However, antimony has been detected in blood and urine of workers exposed to diantimony trioxide via inhalation indicating that trivalent antimony can be absorbed from the lungs and excreted in the urine. Furthermore, the absorption of diantimony trioxide through human skin has been measured in an *in vitro* percutaneous study. Based on the low dermal absorption value of 0.26 % calculated from this study, dermal absorption of diantimony trioxide is considered negligible.

Biomonitoring and autopsy data indicate that antimony is retained in the lungs for long periods of time (based on limited data from a study of seven men, the biological elimination half times has been estimated to 600-1100 d for non-smokers and 1700-3700 d for smokers) and that antimony is accumulating in lung tissue after repeated exposure to diantimony trioxide in the air. It has also been shown that the excretion of antimony from the lung occurs in two phases, one rapid initial phase followed by a slower second phase. Antimony has also been detected in human foetal liver as well as in human breast milk, placenta, amniotic fluid and umbilical cord blood, indicating that antimony can be distributed to the foetus and excreted in breast milk, thus exposure may occur both in utero and during breast-feeding.

The retention and distribution of diantimony trioxide following intratracheal instillation or inhalation exposure has been studied in hamster and rat, respectively. The results demonstrate that the extent of deposition and subsequent clearance of antimony from the lung is primarily dependant on solubility and particle size. Larger and more soluble trivalent antimony particles are more rapidly cleared than small and less soluble particles that tend to remain in the lungs for extended periods. The elimination of diantimony trioxide from lung seems to consist of two phases, one early rapid clearance followed by a slower clearance phase. In hamsters the calculated biological half times ($t_{1/2}$) for the initial (0-20 h) was 40 hours and for the second

(20-200 h) phase it was approximately 20-40 days. The rapid initial phase of elimination from the lungs is considered to be mediated by mucociliary transport.

After oral exposure of diantimony trioxide to rats, the absorption of diantimony trioxide is slow. Also the elimination of antimony from blood is a slow process. However, Sb undergoes significant distribution to most tissues as it binds to red blood cells. Highest concentrations have been found in bone marrow and thyroidea, followed by, spleen, lung, liver, ovaries heart, kidney, femur and skin. Levels of antimony can also be detected in testes and brain.

Also after oral exposure the elimination occurs in two phases; a rapid elimination of antimony primarily via faeces but also via urine lasting for about a week followed by a slower decrease lasting for more than 30 days. Trivalent antimony may form a complex with GSH followed by excretion via the bile, release of antimony and enterohepatic recycling. Antimony does not seem to be methylated in vivo. There are no quantitative animal data reflecting the total absorption of diantimony trioxide from lung or skin. However, in the intratracheal instillation study by Leffler and coworkers (1984), in which hamsters were exposed, it was shown that the retention of diantimony trioxide 190 h after exposure was 44-60 % for the lung and 7.2-12.6 % for the liver. In this study the retention of diantimony trioxide was not measured in e.g. bone marrow or thyroid, which are known to contain relatively high amounts of antimony after exposure. Neither was the excretion in urine or faeces determined. It is clear that the method used, intratracheal instillation instead of inhalation, may have had an impact on the lung clearance and subsequently on the absorption of antimony from the lung. However, based on the levels of antimony found in the liver and taking into account that antimony was not measured in e.g. bone marrow, thyroid or in urine in the Leffler study it can be concluded that the absorption must have been > 12.6 %.

Based on data on physical particle size and density for 8 different samples of diantimony trioxide and with the use of the MPPD model and values on gastrointestinal tract absorption a value of 6.82 % have been calculated for inhalation absorption. This way of calculating inhalation absorption figures has previously been used in the EU RARs on zinc compounds, therefore 6.82 % inhalation absorption is proposed.

There is one study which has measured the absorption of diantimony trioxide after oral exposure to diantimony trioxide particle suspension. In this study, absorption after oral dosing was low; only 0.3 % after administration of 100 mg/kg bw and 0.05 % after administration of 1000 mg/kg bw. It should, however, be noted that the exposure levels in this study in rats are at least 2-3 orders of magnitude higher than the exposure levels for humans (for details on human exposure, see section 4.1.1 exposure assessment). Since the absorption of diantimony trioxide particle suspensions is clearly dose dependent we are hesitant to use 0.3 % oral absorption derived from this study also for oral absorption for humans. In addition, for “consumers” and “man via the environment” the source of antimony is mainly food and water and therefore, these groups are not likely orally exposed to particles of diantimony trioxide, but to the antimony ion. As described in the environmental section of the RAR, antimony released as Sb_2O_3 into the environment will not remain as Sb_2O_3 forever. Instead it will dissolve and a large fraction will also oxidize to pentavalent antimony in oxic environments. Therefore it may not be relevant to use only absorption data from high concentrations of Sb_2O_3 particle suspensions when assessing the human absorption from the gastrointestinal tract. Instead, we also propose to consider the data published on intestinal

absorption of more soluble antimony compounds than diantimony trioxide. Although all these studies have been performed with different study protocols which do not meet current standards they indicate an average intestinal absorption of 3-8 % (0.15-40 %). Considering the above data and the poor solubility of diantimony trioxide an oral absorbtion of 1 % is proposed for diantimony trioxide.

4.1.2.2 Acute toxicity

There is one study on acute inhalation toxicity of diantimony trioxide performed according to current OECD test guideline standards and one reliable animal study on dermal acute toxicity. The animal studies on oral acute toxicity are old and they are not carried out or reported in compliance with current test standards.

4.1.2.2.1 Studies in animals

In vivo studies

Inhalation

In an OECD 403 study with one modification (sampling duration reduced from 20 to 1 minute to avoid overloading of the sampling filters) five male and five female rats (8.5 and 7 weeks old, respectively) were exposed, nose-only to diantimony trioxide (5.20 mg/L) for a single 4-hour period (LPT and IAOIA, 2006). In addition, three male and three female satellite animals were used under the same treatment conditions for histopathological examination. The particles had a mass median aerodynamic diameter (MMAD) of 1.664 µm the geometric standard deviation of the MMAD was calculated as 4.27. LC_{50} was assigned to a level greater than the limit concentration (5.20 mg/L air) tested since no mortality occurred. After completion of exposure, the animals were observed for a period of 14 days. No clinical signs of toxicity or respiratory irritation were observed. The satellite animals were subjected to examination upon necropsy at 24 hours after cessation of exposure and the main study animals at the end of the 14-day observation period. Nose, larynx, trachea and lungs (five levels) were fixed from all animals for histopathological examination. Multiple red-grey foci (0.1-0.2 mm diameter) were observed as a macroscopic change in the lung of one satellite animal. No other abnormalities were detected at necropsy. The following microscopic changes were noted in the lung (five levels) 24 hours and 14 days after end of administration. All animals revealed an activation of macrophages in the lungs (five levels) 24 hours and 14 days after exposure. The alveolar lumen contained aggregations of foamy alveolar macrophages and macrophages with phagocytic material. In addition, inflammatory reactions were noted with granulocytic infiltrations and secretion. The changes observed are considered to be test item-related but can be explained with physiological clearance mechanisms to be expected after inhalation exposure to such a high concentration of poorly soluble dust material. The findings were minimal to mild in severity, and clearly subsided to a large extent after the 14-day observation period. No test item-related changes were noted in the nose (five levels), larynx and trachea 24 hours and 14 days after exposure. In conclusion, no signs of

acute toxicity or respiratory irritation were observed in rats exposed to a single, nose only dose of diantimony trioxide. Consequently LC_{50} for acute inhalation toxicity is higher than 5.20 mg/ L (5200 mg/ m^3).

In a briefly reported acute inhalation toxicity test four rats (strain and sex not stated) were exposed to Mica flake pigments coated with TiO_2 and Sb_2O_3 for 4 hours by the Wright-Dust-Feed Mechanism (Haskell Laboratory for Toxicology and Industrial Medicine, 1962). The particle size was not stated. Dry airflow of 10 L/min and 10 lbs/sq. in. was used. The nominal concentration was 18.6 mg/L, the highest that can be achieved by the Wright-Dust-Feed Mechanism, however, the concentration of Sb_2O_3 was not specified. No rats died. According to the clinical observations the rats were normal during and after exposure, though dusty. The histopathological examination revealed slight to moderately severe residual pulmonary response. Particles characteristic of mica were found in all the phagocytes; and pulmonary edema, hyperemia, petechial hemorrhages and focal emphysema were noted. Changes (not further specified) were also found in the brain and liver. A comparison is made with another study where TiO_2 coated flakes was used showing no phagocytised mica flakes in the lung, but the exposure to these flakes were at approximately one third of the concentration of the exposure to mica flakes with TiO_2 and Sb_2O_3.

The data from this study suggest that Sb_2O_3 may cause changes in lungs, brain and liver. It is not, however, possible to conclude at what concentration and if this was a route specific inhalation effect or due to oral and/or dermal exposure as well. No conclusions on acute toxicity properties of diantimony trioxide are drawn from this study.

Because of the lung reactions observed in rats inhaling mica flakes coated with TiO_2 and Sb_2O_3 (described above) the contribution to this reaction made by Sb_2O_3 was investigated (Haskell Laboratory, 1963). Two different TiO_2 pigments, containing either diantimony trioxide and Al_2O_3 or Al_2O_3 alone, were suspended in a physiological saline solution and 1 ml of a 5 % suspension (50 mg/dose) was injected intratracheally to each of 4 anesthetised rats (strain and sex not stated). The concentration of Sb_2O_3 was not stated. 4 control rats received 1 ml of saline alone intratracheally. All rats survived the administration and the 14 days observation period that followed. The control rats showed inconsequential weight losses but no signs of respiratory impairment. In the test groups there was initial weight loss and evidence of respiratory difficulty immediately after injection. The rats receiving TiO_2 pigment with Sb_2O_3 and Al_2O_3 were more adversely affected (not further described) with respect to respiratory effects than the rats receiving TiO_2 pigments with Al_2O_3 alone. Thereafter both test groups showed normal respiration and gained weight in the remainder of the observation period. 14 days after dosing the rats were sacrificed and examined. Most of the TiO_2 pigment with Al_2O_3 alone had been cleared from the lung and deposits were noted in the pulmonary lymph nodes in only one case. In contrast, the TiO_2 pigment with Sb_2O_3 and Al_2O_3 was detectable in the lungs and it was also evident in the lymph nodes of 3 rats. In rats receiving TiO_2 pigment with Al_2O_3 alone, the lungs showed changes that were regarded as minimal and probably reversible in nature. These were increase in lung weight and slight to moderate focal cellular changes. On the other hand, lungs of rats receiving TiO_2 pigment with Sb_2O_3 and Al_2O_3 evidenced the effects of longer retention of the pigment. There was microscopic evidence of greater alveolar and lymphoid deposition of pigment, an increase in lung weight

and cellular changes and collagen deposition. Changes in other organs were more difficult to assess. The author concludes that the Sb_2O_3-containing pigment is biologically more active and thus the Sb_2O_3 is the causative agent.

The data from this study suggest that diantimony trioxide may cause respiratory and histological changes in the lungs. It is, however, not possible to conclude at what concentration of diantimony trioxide the effects occurred. No conclusions on acute toxicity properties of diantimony trioxide are drawn from this study.

Six acute inhalation toxicity studies with thermal degradation and pyrolysis products of mixtures of diantimony trioxide and substances of which the identity has been deleted from the study reports and flame resistant nylon, polyester and polythene have been performed on ChR-CD male rats (Haskell Laboratory for Toxicology and Industrial Medicin, 1950; Haskell Laboratory for Toxicology and Industrial Medicin, 1972; Haskell Laboratory for Toxicology and Industrial Medicin, 1974; Haskell Laboratory for Toxicology and Industrial Medicin, 1970a; Haskell Laboratory for Toxicology and Industrial Medicine; Haskell Laboratory for Toxicology and Industrial Medicin).

These studies are not considered to be of adequate quality for use in this risk assessment since neither the composition nor the concentration of diantimony trioxide in the air used for exposure is reported.

Dermal

The health effects following dermal application of diantimony trioxide were studied in eight albino rabbits (strain and sex not stated) (Gross et al., 1955a). The method of application was adapted from the procedure of Draize with minor modifications. The day before dosing the animals were clipped over the entire trunk with an electric clipper, care being taken to avoid cutting or abrading the skin. 25 g of diantimony trioxide dust (geometric mean was 1.3 μm ± 1.65 μm, arsenic content was 0.2 %) was incorporated into an aqueous methylcellulose paste and lightly applied to the denuded skin, which comprised about two-thirds of the animal torso. The area was covered by an impervious membrane (Vinylite) and allowed to remain in contact for one week. No significant local reaction or any apparent sign of systemic toxicity was observed after this single application. According to the TGD a body weight of 3.0 kg should be applied for the rabbits which gives a dose of 8.3 g/kg bw.

In conclusion, no mortality or other clinical symptoms were reported in this study. Consequently, a LD_{50} > 8300 mg/kg bw can be derived for dermal exposure of rabbit.

Dermal application of diantimony trioxide suspended in corn oil at a concentration of 8 g/kg was given to 6 rabbits (Carnegie-Mellon Institute of Research, 1978). One rabbit died, severely autolysed, and there was one rabbit with unsteady gait at 24 h. No other toxic effects were observed.

In conclusion, this study is poorly documented; there is no more information than described here. Consequently, it is not possible to draw any conclusions on dermal toxicity of diantimony trioxide from this study.

Oral

The acute toxicity of diantimony trioxide was investigated in 4 groups of 10 rats each (strain and sex not stated) (Fleming, 1938). The rats were given single oral doses of diantimony trioxide (origin and purity not stated) mixed with water (concentration and volume not stated), the doses ranging from 160-225, 250-290, 350-450 and 470-600 mg/kg bw in the respective dose group. In the highest dose group food was withheld for 24 h before exposure. Control animals were used but number and treatment was not stated. The length of the observation period was 3-4 weeks.

In the 160-225 mg/ kg bw group none of the animals died. No effect on rectal temperature was observed 2 h after treatment. Five rats gained weight, two lost weight and three showed no change. None showed any distress or digestive upset. Three rats were killed for pathology and in two a diffuse green brown discoloration of the liver was found.

In the 250-290 mg/ kg bw group no mortality was observed. A slight drop (not further specified) in rectal temperature was observed 1.5 h after treatment but none developed any digestive disturbance or diarrhea. Nine of the animals gained weight and one lost weight in the 3 weeks following treatment. Three of the rats were killed for pathology and showed no gross abnormality in the gastrointestinal tract. In one rat a greyish brown colour of the liver was observed.

In the third dose group, 350-450 mg/kg bw, no death was reported. A slight drop (not further specified) in rectal temperatures 2 h after treatment was observed. No diarrhoea developed and there were no significant changes in weight. Three rats were killed for pathology and two showed a grey brown discoloration of the liver.

The 470-600 mg/kg bw group were kept for a month without exhibiting any signs of toxicity. All rats showed a good gain in weight. These animals were then used for repeated exposure with the same dose for 14-20 days and no information was therefore available on gross pathology.

This study indicates that the oral LD_{50} is higher than 600mg/kg bw. A slight drop in rectal temperatures in the 250-290 and the 350-450 mg/kg bw groups was observed but there were no effects on body weight gain or digestion. Discoloration of the liver was observed in animals from all dose groups examined. Discoloration of the liver is however an un-specific finding affected by multiple factors, including bile pigments and different stages of putrefaction. No conclusions can be drawn from this finding as it is not described in detail.

In an inadequately described single dose toxicity study, rats (strain, sex and number not stated) were fed diantimony trioxide, 20 g/kg, as a 50 % suspension in 25 % agar (Carnegie-Mellon Institute of Industrial Research, 1945). No mortality was observed and the animals gained weight well.

This study is regarded as invalid and cannot be used to derive a LD_{50} due to the lack of information on the number of animals exposed and more.

In a similar inadequately described single dose oral LD_{50} toxicity study, groups of six Sherman rats (sex not stated) were exposed to diantimony trioxide (Smyth and Carpenter,

1948). Doses and route of administration was not stated. No mortality was observed in the group exposed to the highest dose, which was 20 g/kg.

This study indicates that the single dose oral LD_{50} for diantimony trioxide in rats is >20000 mg/kg bw, despite information on all exposure doses was not given.

The LD_{50} of diantimony trioxide was also investigated in a briefly reported stomach intubation study (Carnegie-Mellon Institute of Research, 1978). Five rats (strain and sex not stated) were administered diantimony trioxide, 20 g/kg, in a corn oil suspension (1ml = 0.3g). The study was according to the authors performed under "standard test procedures". No mortality was observed. Other signs of toxicity that were reported were; pilo-erection 1 h, diarrhoea 3.75 h, fur wet 1 day. No more information was given.

In conclusion, no control group was included in this study and no details on test procedures were given. Although the study can be regarded as inconclusive, we still suggest that a single dose oral LD_{50} for diantimony trioxide in rats >20000 mg/kg bw can be derived.

In another very briefly reported acute toxicity study 10 rats (strain and sex not stated) were administered 2.5 g diantimony trioxide (Gross et al., 1955a). The geometric mean was 1.3µm ± 1.65µm and the diantimony trioxide was contaminated with 0.2 % arsenic. The diantimony trioxide was suspended in 5 ml of water and introduced by catheter into the stomach of the animals, resulting, according to the author, in a dosage of about 16 g/kg bw. According to the authors, no toxic effects were noted during the subsequent 30 days of observation and the rate of growth as well as the food consumption was not altered in comparison with that of control animals (neither the number nor the treatment of control animals was stated).

In conclusion, no mortality and no other clinical symptoms were reported from this study. Consequently, a LD_{50} > 16000mg/ kg bw can be derived.

In a very briefly reported study the toxicity of a polythene composition containing diantimony trioxide (polythene + 0.2% "Akroflex" C/ diantimony trioxide/ "Chlorowax") was tested (Haskell Laboratory for Toxicology and Industrial Medicin, 1950). 10 g of the composition were mixed with 20 ml of 5 % citric acid and let stand 48 hours at room temperature. The solids were then removed by filtration and the filtrate neutralised to pH 6 with sodium bicarbonate. Single doses of the neutralised extract, varying from 1 to 4 ml, were fed to rats without producing any toxic effects.

This study is regarded as invalid and will not be used for the risk assessment of diantimony trioxide, due to the fact that the extract was not analysed and the amount of diantimony trioxide not known, the number of animals included in the study was not stated and no control animals was used.

In vitro studies

No in vitro study regarding the acute toxicity is available.

4.1.2.2.2 Studies in humans

In vivo studies

Inhalation

No human data regarding the acute toxicity after inhalation of diantimony trioxide could be located.

Dermal

No human data regarding the acute toxicity after dermal exposure to diantimony trioxide could be located.

Oral

In an old report, acute poisoning caused by drinking fruit lemonade contaminated with Sb_2O_3 was described (Dunn, 1928). For the staff in a store in Newcastle, lemonade was made by fruit crystals or lemonade powder and stored overnight in enamelled buckets provided with lids to keep the dust out. Shortly after taking the lemonade, during the following morning a number of the staff were "very sick" (symptoms were not described), and between fifty and sixty of them had to be taken away and treated at the Royal Infirmary. All recovered, although for some it took some days.

The lemonade powder contained approximately 80 % sugar, 18.5 % tartaric acid and 1.5 % sodium bicarbonate. Moreover, the made up lemonade in the buckets contained antimony compounds and a minute quantity (not further specified) of zinc and 0.12 % boron trioxide. The antimony compounds, expressed as metallic antimony, amounted to 0.013 %. This would, according to the author, be equivalent, in an ordinary 10 oz. tumblerful of the liquid, to 0.57 grain of metallic antimony, or over 1.5 grains of tartar emetic. Also, according to the author, the "Emetic dose" of tartar emetic is given in the B.P. as 0.5 to 1.0 grain, so that the cause of the sickness of those who drank even a third or half a tumblerful of this liquid is sufficiently clear.

The enamel of the buckets had obviously been acted on by the liquid, because the glazed surface was gone and the roughened coating could be removed as a fine powder. Full detail of the powder composition is not given but a content of 6.1 % of Sb_2O_3, nearly 60 % silica and less than 8 % of boron trioxide was reported.

This report is lacking in details. The symptoms of the sick staff are not described, however, although it is not totally clear it can be assumed they were vomiting. Although the content of Sb_2O_3 could possibly explain the sickness, as stated by the author, it cannot be excluded that there were other causes for the sickness, e.g. bacterial growth due to insufficient hygiene, or presence of other substances, such as boron. We find the report inconclusive due to the fact that there is no causal relationship with exposure to Sb_2O_3, and it will not be used for the risk assessment of Sb_2O_3.

In vitro studies

No in vitro studies regarding the acute toxicity of diantimony trioxide could be located.

4.1.2.2.3 Summary of acute toxicity

No reliable information is available on the effects of single exposure via any route in humans. An old report reports acute poisoning caused by drinking fruit lemonade contaminated with Sb_2O_3 (Dunn, 1928), but it is lacking in details and considered inconclusive due to the fact that there is no causal relationship with exposure to diantimony trioxide.

For acute inhalation toxicity there is one animal study which has been performed according to OECD TG 403 and which shows no signs of acute toxicity after inhalation exposure to diantimony trioxide, indicating a LC50 > 5.20 mg/ L (5200 mg/m3). The animal studies on acute oral exposure are all old, they do not comply with today standards and in most of them mortality was the only parameter investigated. Still, they indicate that oral LD50 is in excess of 20000 mg/kg bw in rats. There is one valid study on dermal exposure, which indicates that the LD50 for dermal exposure is higher than 8300 mg/kg bw. In conclusion, diantimony trioxide is of low acute inhalation, oral and dermal toxicity.

4.1.2.3 Irritation

4.1.2.3.1 Skin

Studies in animals

The skin-irritating or penetrating properties of diantimony trioxide were studied in eight albino rabbits (Gross et al., 1955a). The method of application was adapted from the procedure of Draize with minor modifications. The day before dosing the animals were clipped over the entire trunk with an electric clipper, care being taken to avoid cutting or abrading the skin. 25,000 mg of diantimony trioxide dust was incorporated into an aqueous methylcellulose paste and lightly applied to the denuded skin, which comprised about two-thirds of the animals' torso. The area was covered by an impervious membrane (Vinylite) and allowed to remain in contact for one week. No significant local reaction resulted from this single application, nor was there any apparent sign of systemic toxicity.

From this study it can be concluded that diantimony trioxide is not irritating to the skin of rabbits.

In a very briefly reported combined study, a 1:8 mixture of antimony oxide (Sb_2O_3) and perbromophenyl ether was tested for primary skin irritation and sensitisation (Haskell

Laboratory for Toxicology and Industrial Medicin, 1970b) and personal communication from Haskell scientist via IAOIA). In the test for irritation 0.05 ml of 50 %, 25 % and 10 % (w/v) suspensions of the mixture in a guinea pig fat/ acetone/ dioxane mix were applied to clipped intact shoulder skin of each of ten male, albino guinea pigs. It is not stated in the study report whether the application was occlusive or non-occlusive. In addition, the duration of application is not stated. The reaction after 24 h is stated as no evidence of irritation.

This study shows that 0.05 ml of a 50 % suspension of the mixture is not irritating to the skin of 10 male albino guinea pigs. However, the study is poorly described, the highest amount of diantimony trioxide tested is only 6.25 % and the volume tested is only 10 % of the volume recommended in OECD guideline 404. Thus, this study does not allow any conclusions to be drawn on the irritating potential of diantimony trioxide.

In a similar, combined dermal irritation and sensitisation study a 6:1 mixture of hexabromobenzene and antimony oxide (Sb_2O_3) was suspended in 1:1 acetone-dioxane containing 13 % guinea pig fat and applied to male albino guinea pigs (Haskell Laboratory for Toxicology and Industrial Medicine, 1970). In the test for irritation 0.05 ml each of 50 % and 25 % suspensions were lightly rubbed into intact shaved skin of 10 animals each. It is not stated in the study report whether the application was occlusive or non-occlusive. In addition, the duration of application is not stated. “Primary irritation reactions” were scored at 1 and 2 days. At the “50 % concentration” there were 5 animals with mild erythema and 5 with no evidence of irritation on day 1. At the 25 % concentration 2 animals showed mild erythema while 8 were negative on day 1. At 2 days all animals in both groups were negative.

This study shows that 0.05 ml of a 50 % suspension of the mixture cause mild erythema in 5 of the animals on day one. The 25 % concentration caused mild erythema in 2 animals on day one. At day 2 no irritation was observed in any of the animals. However, the study does not include any controls and thus it is impossible to know what agent is causing the irritation seen on day one. The small volume tested, which was only 10 % of the volume recommended in OECD guideline 404, could explain the lack of response on day 2. Thus, this study does not allow any conclusions to be drawn on the irritating potential of diantimony trioxide.

The potential of diantimony trioxide to cause skin irritation was tested in a “range-finding toxicity study” in rabbits. The full report reads as follows: “Skin Irritation, Rabbit, Uncovered. Conditions: Standard. Dosed in mineral oil. Conclusions: No irritation on 5 rabbits from a 50% suspension of diantimony trioxide in mineral oil” (Carnegie-Mellon Institute of Research, 1978). This study is inconclusive and does not allow any conclusions to be drawn.

Studies in humans

White and co-workers reported three cases of dermatitis in workers exposed to antimony in a melting process (White et al., 1993). Three men, between 28 and 33 years of age were employed at a brazing rod manufacturing plant. After changes at their workplace they were assigned the task of melting antimony metal and due to insufficient precautionary measures they were exposed to fumes from the melted antimony. Shortly after the process changes they noted the onset of skin lesions. Physical examination revealed crusted follicular papules and pustules of the arms (accentuated in the antecubital fossae), trunk and forehead in two of the workers. One of these also had a dry eczematous patch on the left trunk. In the third worker

erythematous follicular papules were noted on the ventral and dorsal aspects of both forearms and on the posterior legs and back. The urinary antimony level, which was measured in one worker, was 53.2 μg/L, which is in the range for exposed individuals (levels in unexposed persons are less than 1.0 μg/L). None of the workers had any history of skin disease or atopy. In all three workers the dermatitis resolved with the avoidance of antimony-related work.

The authors concluded that the three workers present strong evidence for antimony related dermatoses: these workers were exposed to other metal fumes for many years without skin manifestations; lesions appeared when antimony was introduced to the process, and resolved when antimony exposure was avoided. Two of the workers had exposure only to molten metal fume, and not to metallic dust. During site visit it was noted that the temperature in the work area was quite high, and the skin of employees was damp with perspiration.

This study indicates that fumes from melted antimony, presumably diantimony trioxide, may cause dermatitis in humans.

The occurrence of a skin eruption in 23 persons amongst a population of about 150 men employed in the manufacture of diantimony trioxide has also been reported and the morphology and histology of the rash known as ”antimony spots” is described (Stevenson, 1965). Intense itching preceded the skin eruption. A diffuse blotchy erythema may occur but most commonly the early lesions are small erythematous papules and may be associated with much excoriation. The papules enlarge and in some cases become frankly pustular. The sites most commonly involved in the 23 cases were antecubital area, shins, back of neck, forearms, trunk, back of knees and face. In general, the lesions were present on those dust-laden areas most exposed to heat and therefore to sweating. Two furnacemen who presented one side of their body to heat when working had lesions only on the limbs of that side. The rash subsides in from 3 to 14 days when the worker is transferred to a cooler part of the factory. The eruption occurs in the warm summer months and is rarely seen in the winter. 17 of the 23 men affected were furnace workers and 5 were doing a different job but also under hot conditions.

Histologically, the early lesions showed epidermal cellular necrosis with associated acute dermal inflammatory cellular reaction. The lesions appear to be closely associated with sweat ducts.

This study suggests that workers exposed to diantimony trioxide are liable to develop a transient skin eruption affecting areas most exposed to heat and where sweating occurs.

In another paper severe discomfort from skin irritation in warm weather was described in men working with the production of antimony oxide and the pure metal from sulphide ore by various smelting processes (McCallum, 1963). The rash consisted of papules and pustules around sweat and sebaccous glands and was compared in appearance to the lesions of chickenpox or smallpox. It affected particularly the fore arms and thighs and the flexures and did not appear on the face, hands or feet. The spots disappeared rapidly over a weekend or public holiday, but reappeared on return to work. Over hundred men were employed but the frequency of dermatitis was not stated.

This study indicates that work in various smelting processes in the production of diantimony trioxide and the pure metal is connected with dermatitis in warm weather.

The clinical examination of 51 male workers employed in an antimony smelting plant has been reported (Potkonjak and Pavlovich, 1983). The entire study is also reported in the 4.1.2.6. Repeated dose toxicity section. The subjects were aged between 31 and 54 years (mean 45.23), they were exposed to dust containing predominantly antimony oxide [Sb_2O_3 (38.73-88.86%), Sb_2O_5 (2.11-7.82%), SiO_2 (0.82-4.72), Fe_2O_3 (0.90-3.81%) and As_2O_3 (0.21-6.48%)], had worked in the factory from 9-31 years (mean 17.91) and had pneumoconiotic changes. Over a 25-year period they were examined 2-5 times; the evaluation included among other things a physical examination (specialist consultations were obtained when appropriate).

"Antimony dermatosis", characterised by vesicular or pustular lesions with residual hyperpigmentation, were present in 32 of 51 exposed workers (63%), especially during the summer season and when working near the furnace where temperatures were excessively high.

This study indicates that antimony related dermatosis may occur in humans exposed to diantimony trioxide at high temperatures.

A combined test was conducted to determine the irritation and sensitisation potential of a fibre treated with a mixture of antimony oxide (Sb_2O_3) and a substance of which the identity was deleted from the report (Haskell Laboratory for Toxicology and Industrial Medicine, 1970). The fibre contained 1 % antimony oxide (by weight). One-inch squares of the test fabric were applied to the arms of ten men and to the arms or legs of ten women and held in place with adhesive tape for six days. Two weeks after removal, new patches were applied for 48 hours. Skin under the patches was examined at two and six days and on final day at patch removal. No skin reactions were seen at any of the examinations.

This study shows that one-inch squares of a test fabric of unknown identity containing 1 % antimony oxide (by weight) was not irritating to the skin of 10 men and 10 women. However, the amount of diantimony trioxide applied was not given and there is no information on how much of the diantimony trioxide in the fibre that came into contact with the skin. Therefore, no conclusions on the irritation potential of diantimony trioxide can be drawn from this study.

A similar patch test was performed with fibre containing Sb_2O_3 and a substance of which the identity was deleted from the report (the concentration of antimony oxide was not specified) (Haskell Laboratory for toxicity and Industrial Medicin., 1970). One-inch squares of the test material were applied to the arms of 46 men and to the arms or legs of 127 women and held in place with adhesive tape for six days. Two weeks after removal, new patches were applied as a challenge for skin sensitisation and were removed after 48 hours. Skin under the patches was examined at two and six days and on final day at patch removal. After six days of occluded wear one subject had papules along the edge of patch area, however, similar papules were also seen under the tape area. Subjects had small indented areas under patch that appeared as red spots that coincided with the crimped pattern of this fibre. No conclusions on the irritation potential of diantimony trioxide can be drawn from this study.

This study shows that one-inch squares of a test fabric of unknown identity containing an unknown amount of diantimony trioxide were not irritating to the skin of 46 men and 127 women. Since the amount of diantimony trioxide applied was not given and there is no information on how much of the diantimony trioxide in the fibre that came into contact with

the skin no conclusions on the irritation potential of diantimony trioxide can be drawn from this study.

4.1.2.3.2 Eye

Studies in animals

An acute eye irritation/corrosion test of diantimony trioxide was performed according to OECD guideline 405 and in compliance with GLP regulations (LPT and IAOIA, 2005a). Three Himalayan rabbits were each administrated with a single installation of 100 mg diantimony trioxide powder (99.93 % pure) into the conjunctivae sac into one eye (right). The test item was placed after gently pulling the lower lid away from the eyeball. The lid was then gently held together for about one second in order to prevent loss of the material. The left eye which remained untreated, served as a control.

The eyes were examined ophthalmoscopically with a slit lamp prior to the administration and 1, 24, 48, 72 hours and 4 days after administration. The eye reactions were observed and registered. 24 hours after administration, fluorescein was applied to the eyes before being examined to aid evaluation of the cornea for possible lesions.

Conjunctival redness (grade 1) was observed in 2 animals at 24 hours and in one animal at the 48- and 72-hour time points after installation, respectively. The fluorescein test performed 24 hours after instillation did not reveal any corneal lesions. The iris was not affected by instillation of the item.

There were no systemic intolerance reactions.

In conclusion, mild conjunctival redness was observed in 2 animals. However, the effect does not fullfill the EU-criteria for classification as irritant.

In another study the eye irritation potential of antimony oxide was investigated in 5 male and 5 female New Zealand albino rabbits (Wil Research Laboratories, 1979). Both eyes of each animal were examined before testing by applying one drop of fluorescein ophthalmic solution directly on the cornea. All animals were without eye defects or irritation. 100 mg of antimony oxide (a fine, white powder) was instilled into the conjunctival sac of the right eye of the 10 rabbits. The left eye of each animal remained untreated and served as a scoring control. The treated eyes of two male and two female rabbits were washed by flushing with distilled water for one minute starting no sooner than 30 seconds after application of the test material. Examinations for eye irritation were made at 6, 24, 48 and 72 hours and 4, 7, 10 and 14 days following application. Fluorescein dye was applied to all eyes at 48 hours, 7 and 14-day readings. Observed injury or irritation was graded and scored according to the method of Draize in which corneal, iris and conjunctival effects were scored separately.

Slight corneal opacity was noted in one un-rinsed eye at the 24-h reading, increased to marked opacity through the 4-day reading, and then decreased to slight opacity by the 10-day reading. Slight iritis was also noted in this animal at the 24-h through the 7-day reading. Corneal

opacity and iritis were not noted in any other rinsed or un-rinsed eyes throughout the study. Irritation of the conjunctivae included slight to moderate erythema, chemosis and discharge in all un-rinsed eyes at the 6-h reading, increasing to slight to severe at the 24-h reading. Irritation of the conjunctivae had generally decreased to slight and moderate erythema, chemosis and discharge at the 48-h through 7-day reading. No irritation was observed at the 10 and 14-day readings. At 48 hours, necrosis of the lower conjunctivae (where the compound had pocketed) was noted in the un-rinsed eyes of three male and two female rabbits. At the same time necrosis of the nictitating membrane was noted in the un-rinsed eye of the third female animal. The necrosis in the un-rinsed eyes was only noted at 48 h for the conjunctivae and at 48h to 7 dayes for nictitating membrane. In the rinsed eyes, necrosis of both the lower conjuntivae and the nictitating membrane was observed in one male rabbit at 48 h. The necrosis of the nictitating membrane in the rinsed eye persisted until day 10.

The primary eye irritation index in unwashed eyes was 10.3, 12.2, 12.5, and 14.0, the maximum possible being 110 according to Draize weighted scoring system (Draize et al, 1944), at 6, 24, 48 and 72 hours, respectively. This rating was classified as mild irritation in the study. The primary eye irritation index in washed eyes was 4.5, 3.5, 7 and 3 at 6, 24, 48 and 72 hours respectively. If the data from this test are evaluated according to the eye irritation test in Annex V to dir. 67/546/EEC, then the mean values for the ocular lesions of the unwashed eyes at 24, 48 and 72 hours were as follows; cornea opacity 0.39, iris lesion 0.17, redness of the conjuntivae 1.56 and oedema of the conjunctivae 1.39. The lesions were not present at the end of the observation period, 14 days.

This study showed that antimony oxide caused necrosis of the lower conjunctivae and the nictitating membrane in 7 out of 10 animals. However, necrosis of the conjunctivae was only observed at 48 h post exposure and therefore the relevance of this finding is questionable. The study also indicates that antimony oxide is mildly irritating when applied to the eyes of New Zealand rabbits. However, according to current EU criteria the irritation effects are not severe enough for classification.

In another briefly described eye irritation test, 10 male albino rabbits were selected on the basis that the eyes were free of grossly visible stains following the installation of 5 % aqueous solution of flourescein sodium, according to the method of Smyth and Carpenter (1948). After a two-hour interval to allow the eye to return to normal, 1 ml, containing 100 mg of diantimony trioxide dust (the geometric mean of the particle size was 1.3µm and the geometric standard deviation 1.65µm, giving a 50% size by weight of 1.55µm) in aqueous suspension, was instilled into the right eyes, the left eyes serving as controls. The eyes were examined in strong diffuse light after one, two, and seven days following preliminary staining with flourescein sodium. There were no irritative effects of either the conjunctiva or the cornea that could be attributed to the test material. The results were not further specified.

The administration of diantimony trioxide as an aqueous suspension is not considered relevant, since most exposure of diantimony trioxide to the eye will be as dust, and it is not in accordance with the current OECD guidline 405. In addition, the volume used, 1 ml, is ten times larger than recommended in the OECD guidline 405. It can be assumed that most of the suspension that was applied was lost/removed immediately due to physiological mechanisms and therefore the exposure dose is un-clear. In conclusion, this study cannot be used to judge the eye irritating potential of diantimony trioxide.

The potential of diantimony trioxide to cause eye irritation was also assessed in another very briefly described “range-finding toxicity study” (Carnegie-Mellon Institute of Research, 1978). The full study-report reads: "Eye irritation, rabbit. Conditions - Standard. Dosed as a solid in mineral oil. Conclusions - No corneal injury on 5 eyes from 40 mg per eye of the powder or from 0.5 ml per eye of a 50% suspension in mineral oil".

Due to the poor documentation of this study it is regarded as invalid and will not be used in the risk characterisation.

Studies in humans

The clinical examination of 51 male workers employed in an antimony smelting plant has been reported (Potkonjak and Pavlovich, 1983). The entire study is also reported in the 4.1.2.6. Repeated dose toxicity section. The subjects were aged between 31 and 54 years (mean 45.23), they were exposed to dust containing predominantly antimony oxide [Sb_2O_3 (38.73-88.86%), Sb_2O_5 (2.11-7.82%), SiO_2 (0.82-4.72), Fe_2O_3 (0.90-3.81%) and As_2O_3 (0.21-6.48%)], had worked in the factory from 9-31 years (mean 17.91) and had pneumoconiotic changes. Over a 25-year period they were examined 2-5 times; the evaluation included among other things a physical examination (specialist consultations were obtained when appropriate).

Conjunctivitis was seen in 14 cases (27.5 %).

In conclusion, conjunctivitis was observed in workers employed in an antimony smelting plant. However, since the exposure contained not only diantimony trioxide it is unclear what caused the conjunctivitis and consequently whether diantimony trioxide has eye irritation potential or not.

Additional cases of conjunctivitis and irritation of the eyes observed in workers exposed to diantimony trioxide have been reported (Renes, 1953; Karajovic, 1957). The entire studies have been reported in the 4.1.2.6. Repeated dose toxicity section. Common for the two studies is that the exposure situations are not very well described. Consequently, it is unclear what caused the conjunctivitis and whether diantimony trioxide has an eye irritation potential or not.

4.1.2.3.3 Respiratory tract

Studies in animals

Although no validated animal test for respiratory tract irritation is currently available, the respiratory irritation potential of diantimony trioxide was investigated in an acute inhalation toxicity study of diantimony trioxide in rats (LPT and IAOIA, 2006). This study is also reported in the 4.1.2.2 Acute toxicity section. Following a 4 h exposure at a concentration of 5.20 ± 0.16 mg diantimony trioxide/L air, no test item-related changes were noted in the nose, larynx or trachea at macroscopic and microscopic examination. Neither were dyspnoea or

rhinitis observed. In conclusion, this study suggests that diantimony trioxide is not a respiratory irritant.

Studies in humans

The clinical examination of 51 male workers employed in an antimony smelting plant has been reported (Potkonjak and Pavlovich, 1983). The entire study is also reported in the 4.1.2.6. Repeated dose toxicity section. The subjects were aged between 31 and 54 years (mean 45.23), they were exposed to dust containing predominantly antimony oxide [Sb_2O_3 (38.73-88.86%), Sb_2O_5 (2.11-7.82%), SiO_2 (0.82-4.72), Fe_2O_3 (0.90-3.81%) and As_2O_3 (0.21-6.48%)], had worked in the factory from 9-31 years (mean 17.91) and had pneumoconiotic changes. Over a 25-year period they were examined 2-5 times; the evaluation included among other things a physical examination (specialist consultations were obtained when appropriate).

Chronic bronchitis, defined as “chronic persistent or recurrent coughing with expectoration” lasting 3 months each for 2 consecutive years, was found in 37.3 %. Chronic coughing without expectoration was found in 23.5 %. Upper airway inflammation was observed in 35.3 %. It is not clear from the report if the group with upper airway inflammation was included in the group with chronic bronchitis.

This study indicates that workers employed in an antimony smelting plant show effects that may be due to irritation in the respiratory tract. However, the exposure situation is not very well described and thus it is unclear what may have caused the irritation of the respiratory tract and consequently it is unclear whether diantimony trioxide has a respiratory irritation potential or not.

Additional case report studies on workers, occupationally exposed to diantimony trioxide, have described effects that could indicate irritation in the respiratory tract; nasal septal perforations, irritated and nosebleeds, rhinitis, laryngitis, tracheitis, bronchitis and catarrh in the upper respiratory tract (Renes, 1953; Karajovic, 1957; Cooper et al., 1968 and Klucik et al., 1962). The entire studies have been reported in the 4.1.2.6. Repeated dose toxicity section. Common for the studies is that the exposure situations are not very well described. Consequently, it is unclear what may have caused the irritation and whether diantimony trioxide has a respiratory irritation potential or not.

4.1.2.3.4 Summary of irritation

The only animal study which can be used for risk assessment of the skin irritation potential of antimony oxide shows that antimony oxide is not irritating to rabbit skin. However, several human case report studies indicate that diantimony trioxide may cause dermatitis on skin damp with perspiration and thus the lesions appear to be closely associated with sweat ducts. The lack of dermal irritation in rabbits may be explained by the fact that rabbits lack sweat glands (Brewer and Cruise, 1994). In conclusion, diantimony trioxide should be regarded as a skin irritant in humans under conditions that evoke sweating.

Mild conjunctival redness was observed in 2 out of 3 rabbits in an eye irritation study performed according to OECD guideline and GLP standards (LPT and IAOIA, 2005a) and mild irritation effects were also reported in another rabbit eye irritation study (Wil Research Laboratories, 1979). However, neither of these effects fulfil the EU criteria for classification as irritating to eyes. There are three case report studies on workers, occupationally exposed to diantimony trioxide, where conjunctivitis and irritation to the eyes have been described. However, there is little exposure data in these studies and therefore it is unclear whether diantimony trioxide was the causative agent or not. In conclusion, summing up both animal and human data, diantimony trioxide is not classified as an eye irritant .

In an acute inhalation toxicity study of diantimony trioxide in rat, where also irritation of the respiratory tract was evaluated, no signs of respiratory tract irritation was found. In five case report studies on workers, occupationally exposed to diantimony trioxide, effects that could indicate irritation in the respiratory tract has been described. However, there is very little exposure data in these studies and it is unclear whether diantimony trioxide was the causative agent. Based on available data, diantimony trioxide is not considered irritating to the respiratory system.

Classification proposal: Xi; 38 (Irritating to skin)

Rationale for the classification

The classification proposal is based on practical experience in humans.

4.1.2.4 Corrosivity

From the data presented in the preceding text it is evident that diantimony trioxide is not a corrosive agent.

4.1.2.5 Sensitisation

4.1.2.5.1 Studies in animals

Skin

The skin sensitisation potential of diantimony trioxide, 99.93 % pure, was investigated in female Dunkin.Hartley guinea pigs according to the Magnusson and Kligman method (OECD guideline 406) and GLP (LPT and IAOIA, 2005b). No positive control group was tested concurrently in this study. However, a positive historical background group from the same laboratory was used.

A preliminary study with 8 animals was performed to determine the appropriate dose level for intracutaneous and topical administration. For all test preparations diantimony trioxide was suspended in water. For the intracutaneous administration three different concentrations, 0.001, 0.01 and 0.1 %, of diantimony trioxide (0.1 ml of each) were injected into the shoulder region of one animal, three further concentrations, 1, 5 and 10 %, were injected into a second

animal. For the topical administration three animals each was shaved or shaved and depilated and 2 ml of the test preparation was spread over a filter paper (2 x 4 cm). Two concentrations each of 0.5, 1, 5, 10, 25 and 50 % were applied to the shaved or shaved and depilated flanks of 3 animals each and held in contact by an occlusive dressing. The filter papers were removed after 24 or 48 h and the application sites were assessed for erythema and oedema immediately, 24 and 48 (depilated) or immediately and 24 hours (non-depilated) after removal of the filter paper. No skin reactions were observed at any concentration or at any time point. Therefore, it was decided to use 10 and 50 % of diantimony trioxide for the intracutaneous and topical administrations respectively, since these were also the highest technically feasible concentrations.

In the main experiment 20 animals were given 3 pairs of intradermal injections (0.1 ml each); 1) a 1:1 mixture (v/v) of Freund's complete adjuvant (FCA):0.9 % NaCl. 2) 10 % suspension of diantimony trioxide in water and 3) a 1:1 mixture (v/v) of diantimony trioxide suspension: FCA/ 0.9 % NaCl resulting in a final concentration of diantimony trioxide of 10 %. The injections were given in the shoulder region which was cleared of hair so that one of each pair was applied on each side of the midline. As the diantimony trioxide suspension was not irritating to the skin in the preliminary study, the fur was shaved at the application area 6 days after the intracutaneous injection, and the exposed skin was coated with 0.5 ml sodium lauryl sulphate 10 % in vaseline in order to induce a local irritation. Seven days after the intracutaneous injection, the shoulder region of the same animals was shaved again and treated topically with 50 % suspension of diantimony trioxide in water, using the patch-test technique. The exposure time was 48 h. Two weeks after the topical application (21 days after the intracutaneous injection) the flanks of the same animals were shaved and depilated for a challenge with a topical application of 50 % suspension of diantimony trioxide in water on the left flank and the vehicle on the right flank, using the patch-test technique. The exposure time was 24 h. 21 h after the filter paper had been removed, the treated skin was cleaned.

During the induction period a vehicle control group of 10 animals was treated in the same way as the test group but received water instead of the test item. However, at challenge the control group received diantimony trioxide and the vehicle in the same way as the test group.

No skin reactions were observed in the test group animals after challenge, neither in animals of the vehicle control group.

In conclusion, this study shows that diantimony trioxide does not have sensitising properties.

In a poorly conducted and reported combined skin irritation and sensitisation study a 1:8 mixture of antimony oxide and perbromophenyl ether have been tested (Haskell Laboratory for Toxicology and Industrial Medicine, 1970). After the primary irritation test, described in chapter 4.1.2.3., sensitisation was tested as follows. Treatments were given over a 3-week interval and consisted of nine topical applications (probably 0.05 ml, communicated from Haskell scientist via IAOIA) of 25 % (w/v) antimony oxide mixture in 1:1 acetone-dioxane with 13% guinea pig fat (f.a.d.) to clipped abraded skin of 5 guinea pigs and four sacral intradermal injections (probably 0.1ml, communicated from Haskell scientist via IAOIA) of 1% mixture in acetone-dimethyl phtalate. After a 2-week rest period, the guinea pigs received challenge applications (volume not stated) of 2 5% mixture in f.a.d. on intact and abraded skin and 50% mixture in f.a.d. on intact skin. Reactions were evaluated after 24 hours. The

duration of applications were not stated in the report. In addition, it was unclear whether the applications were occlusive or non-occlusive.

At challenge, the 50 % mixture caused mild erythema in 2 animals of 10 exposed on intact skin; the 25% mixture caused mild erythema in 1 animal of 10 exposed on intact skin. No erythema was found after challenge on abraded skin. A guinea pig was according to the Haskell Laboratory sensitised when the reaction score in the challenge test showed a "two-step" increase over the primary irritation score. None of the ten guinea pigs was sensitised according to the Haskell laboratory.

This study is inconclusive and cannot be used to estimate the sensitising potential of diantimony trioxide. A mixture and not the pure substance have been tested without the inclusion of control animals. A Buehler test, conducted according to the current OECD guidelines, should use the highest dose to cause mild irritation for the induction phase. It has not been demonstrated in the study that 50 % corresponds to this dose. Thus, the highest dose used may be too low for detecting a sensitising effect. In addition, a minimum of 20 animals should be used in a Buehler test, conducted according to the current OECD guidelines. Only 10 animals were used which is not enough for reliably detecting positive response in at least 15 % of the animals. No conclusions can be drawn from this study.

In a similar combined skin irritation and sensitisation study a 1:6 mixture of antimony oxide and hexabromobenzene was tested (Haskell Laboratory for Toxicology and Industrial Medicine, 1970). To determine the sensitisation potential 10 guinea pigs, which also participated in the irritation test described in chapter 4.1.2.3., were given a series of exposures over a three-week period. 5 animals received nine applications (volume not stated) of 50 % suspensions in a fat/ acetone/ dioxane mixture on abraded skin. It is not clear if the application was occlusive or not occlusive. The remaining 5 animals received four sacral intradermal injections (0.1 ml each of 1 % suspensions in propylene glycol). A two-week rest period was followed by a challenge test consisting of 50 % and 25 % suspensions (volumes and duration not stated) on both intact and abraded skin. Treated sites were observed at 1 and 2 days.

At challenge, the 50 % mixture caused mild erythema on day 1 in 4 animals of 10 exposed on intact skin and in 6 animals of 10 exposed on abraded skin. All animals were negative on day 2. The 25 % mixture caused mild erythema on day 1 in 2 animals of 10 exposed on intact skin and in 4 animals of 10 exposed on abraded skin. All animals were negative on day 2. According to the Haskell laboratory there was no evidence of sensitisation in any of the test animals.

This study is inconclusive and cannot be used to estimate the sensitising potential of diantimony trioxide. A mixture and not the pure substance have been tested without the inclusion of control animals. The test protocol is very unclear. It is not clear whether all animals received epicutaneous as well as intradermal applications. The doses that have been used caused mild erythema after challenge with the same frequency as observed with the same doses in the irritation test. Therefore, it is not possible to conclude whether the mild erythema observed after challenge is caused by irritation or sensitisation. A minimum of 20 animals should be used in a Buehler test, conducted according to the current OECD guidelines. Only 10 animals were used which is not enough for reliably detecting positive response in at least 15 % of the animals. No conclusions can be drawn from this study.

Respiratory tract

No data regarding sensitisation after inhalation of diantimony trioxide could be located.

4.1.2.5.2 Studies in humans

Skin

None of the human studies, described below, are adequate for assessing the skin sensitisation potential of diantimony trioxide.

The potential of diantimony trioxide and other chemicals used in the ceramics industry to cause contact sensitisation was investigated in enamellers and decorators in the ceramics industry (Motolese et al., 1993). 190 workers (119 females and 71 males) from 5 ceramics factories underwent dermatological and allergological examination, using occupational test series. The chemicals tested included 7 chemicals from the GIRDCA (Italian research group on contact and environmental dermatitis) standard series, 15 substances specifically used by enamellers and decorators of which one was diantimony trioxide and 7 compounds that decorators are exposed to. Patch tests with pure powder of diantimony trioxide, obtained directly from a chemical industry supplying ceramic producers, were applied with Finn Chambers (Epitest, Finland) on Scanpor tape (Norgeplaster, Norway) to the healthy skin of the back, removed after 2 days and read 1 day later. Diantimony trioxide was also patch tested, at the same concentration as used for the ceramic workers, on 92 healthy volunteers (cadets of the Military Academy of Modena).

Two subjects from the ceramics industry were, in the results part, reported to show positive responses to diantimony trioxide whilst no positive reactions were discovered in the healthy volunteers. However, in the discussion it is stated, “only one patient showed a positive reaction to diantimony trioxide” and it is not clear if the reaction was of an allergic or irritating nature. The rapporteur has sent two e-mails to Dr. Motolese and asked for clarification on theses issues and we received the following answer: “as you have noticed reading my report I have made a mistake. The exact number of patient positive to diantimony trioxide was two and not three as written in the paper”.

This study might indicate that diantimony trioxide has a sensitising potential in humans. However, in the way the data have been presented it is not very clear if an allergic reaction to diantimony trioxide have been observed. In addition, the study do not contain any information on exposure of the workers to diantimony trioxide, neither on the levels of exposure nor on how many of the workers that had actually been exposed to the substance before the patch test was performed. In conclusion, this study cannot be used to judge the sensitising potential of diantimony trioxide.

Patch tests were carried out with dry diantimony trioxide powder and with a suspension of the powder in water in 20 control subjects (Stevenson, 1965). A 50 % mixture of the powder in soft yellow paraffin was tested in 10 subjects. The diantimony trioxide used was in three

grades of refinement commonly encountered in the diantimony trioxide manufacturing process. No further details were given. No positive reactions were obtained.

The amount of diantimony trioxide used was not stated and the number of subjects included in the study was low and their previous exposure to diantimony trioxide was not stated. Therefore, this study is inconclusive and cannot be used for risk assessment.

A combined test was conducted to determine the irritation and sensitisation potential of a Dacron fibre treated with a mixture of antimony oxide (Sb_2O_3) and perbromophenyl ether (Haskell Laboratory for Toxicology and Industrial Medicine, 1970). The fibre contained 1 % antimony oxide (by weight). One-inch squares of the test fabric were applied to the arms of ten men and to the arms or legs of ten women and held in place with adhesive tape for six days. Two weeks after removal, new patches were applied for 48 hours. Skin under the patches was examined at two and six days and on final day at patch removal. No skin reactions were seen at any of the examinations.

A similar patch test was performed with Dacron fibre containing 1 % Sb_2O_3 and 6 % octabromobiphenyl (Haskell Laboratory for toxicity and Industrial Medicin, 1970 and personal communication from Haskell scientist via IAOIA). One-inch squares of the test material were applied to the arms of 46 men and to the arms or legs of 127 women and held in place with adhesive tape for six days. Two weeks after removal, new patches were applied as a challenge for skin sensitisation and were removed after 48 hours. Skin under the patches was examined at two and six days and on final day at patch removal. After six days of occluded wear one subject had papules along the edge of patch area, however, similar papules were also seen under the tape area. Subjects had small indented areas under patch that appeared as red spots that coincided with the crimped pattern of this fibre. No evidence of compound-related sensitisation was seen.

The two above studies were conducted with a Dacron fibre containing 1 % diantimony trioxide according to Industry. There is no information on the distribution of diantimony trioxide from the fibre to the skin so the conclusion which can be drawn from these studies is that 1% diantimony trioxide in the fibre is not sensitising. However, no conclusions on the sensitising potential of diantimony trioxide can be drawn from these studies.

Respiratory tract

No human data regarding the sensitisation after inhalation of diantimony trioxide could be located.

4.1.2.5.3 Summary of sensitisation

There are no human studies of adequate quality that can be used for assessing the sensitising potential of diantimony trioxide. However, there is one reliable animal study, performed according to TG 406 and GLP, which shows that diantimony trioxide has no sensitising properties.

4.1.2.6 Repeated dose toxicity

4.1.2.6.1 Studies in animals

Inhalation

In a whole-body inhalation study, the sub-chronic toxicity of diantimony trioxide was evaluated (Newton et al., 1994). This study is also described in the 4.1.2.1 Toxicokinetic section. Fischer 344 rats, 50 males and 50 females per dose group, were exposed to diantimony trioxide in exposure chambers for 6 hours/day, 5 days/week for up to 13 weeks followed by a 27-week observation period. 30 rats/sex/dose group were exposed for the full 13 weeks, with 5 rats/sex/group killed after 1, 2, 4 and 8 weeks of exposure. The diantimony trioxide exposure concentrations were 0, 0.25, 1.08, 4.92, or 23.46 mg/m^3. Control animals were exposed to clean air only. The flow rate was 18-25 complete air changes per hour (the recommended flow rate in OECD guideline 412, 413 and 453 is 12-15 air changes per hour, Rapporteur comment). The test sample was a mix of equal-sized lots of diantimony trioxide obtained from nine different suppliers. The purity of the 9 lots was 99.68 ± 0.10 % and the mass medium aerodynamic diameter (MMAD) was 3.05 ± 0.21 micrometers with a geometric standard deviation (GSD) of 1.57±0.06.

Animals were observed twice daily for viability and overt signs of toxicity. Detailed observations were conducted weekly and body weights were measured weekly throughout the exposure and observation period. Ophtalmoscopic examinations were performed on all animals pretest and on the day before their scheduled sacrifice. Sacrifices were conducted on 5 animals/sex/group at exposure weeks 1, 2, 4, 8, and 13 and at recovery weeks 1, 3, 9, 18, and 27. Complete gross postmortem examinations of all major organs were performed in all animals. Histological examinations were performed on hematoxylin-eosin-stained tissue sections of heart, nasal turbinates, larynx, trachea, lung and peribronchial lymph node. Hematology and clinical chemistry analyses were conducted on five animals/sex/group at exposure weeks 1, 2, 4, 8, and 13.

No exposure-related mortalities were reported. Corneal irregularities were observed with about equal incidence, 30%, in all groups, including controls. According to the authors these effects were similar to a spontaneous degenerative condition reported in Fischer 344 rats and were, therefore, not considered to be treatment related. The irregularities appeared after about 2 weeks of exposure and did not abate during the 27-week observation period. Male body weight gains were significantly lower in the highest dose group compared to the controls. The difference was small, approximately 6% and statistically significant from week 3 to the end of the study, except for week 12. Female body weight gains were unaffected by diantimony trioxide exposure. No exposure-related changes in hematological parameters were noted. In both sexes, the mean absolute and relative lung weights were significantly increased at the two highest dose levels by week 13 of exposure but returned to normal after the 3rd week of the observation period. Microscopic changes observed in the lungs are shown in Table 4-45 and Table 4-46. Chronic interstitial inflammation (minimal to moderate severity) and interstitial fibrosis (minimal to slight severity) were seen in the lungs of both control and treated animals from the exposure and observation periods. During the observation period,

these effects were most frequent in the highest dose group. Also, granulomatous inflammation (minimal to moderate severity) in the lungs was most frequent in the highest dose group during the observation period. Bronchiolar/alveolar hyperplasia (minimal to mild severity) was seen in only two males from the highest dose group terminated following the exposure period of the study. Alveolar macrophages were more numerous (minimal to moderate severity) in the lungs of the treated animals than in their comparable controls. The exposed rats had scattered macrophages containing small particles of foreign material in the lungs and in the peribronchial lymph nodes (minimal to moderate severity). The incidence and severities of these findings were greater during the observation period than during the exposure period. For both periods, the animals in the highest two dose groups were most severely affected. No histopathologic findings were reported in any other tissues examined.

In conclusion, this study did not indicate any systemic toxic effects of diantimony trioxide after sub-chronic inhalation exposure in rats. For the local effects, chronic interstitial inflammation, granulomatous inflammation and fibrosis are observed in the lungs of the animals of the highest dose group. However, it should be noted that the incidence of interstitial chronic inflammation and interstitial fibrosis in controls (17 and 12 out of 25 males and 15 and 8 out of 25 females, for the respective conditions) after 13 weeks is high and too severe in degree as to be completely satisfying. Thus, the study might reflect sub-chronic disease of moderate severity due to other causative agent than diantimony trioxide.

Table 4-45. Microscopic findings after a 13-week inhalation exposure period to diantimony trioxide.

	Males					Females				
Dose (mg/m³)	0	0.25	1.08	4.92	23.46	0	0.25	1.08	4.92	23.46
Number of animals examined	25	25	25	25	25	25	25	25	25	25
Effect										
Interstitium: chronic inflammation	17	15	11	13	16	15	11	13	11	15
Granulomatous inflammation	0	0	0	0	1	0	1	0	0	1
Interstitium: fibrosis	12	13	8	10	10	8	7	6	4	12
Bronchiolar/alveolar hyperplasia	0	0	0	0	2	0	0	0	0	0
Alveolar/intraalveolar macrophages	3	1	5	11	9	2	0	4	10	11
Alveolar/intraalveolar macrophages: foreign particulate material	0	8	11	17	23	0	4	11	20	23
Peribronchial lymph node macrophages: foreign particulate material	0	1	0	1	3	0	0	0	3	3

Table 4-46. Microscopic findings related to the 13-week inhalation exposure period to diantimony trioxide seen during the 1- to 27-week observation period.

	Males					Females				
Dose (mg/m³)	0	0.25	1.08	4.92	23.46	0	0.25	1.08	4.92	23.46
Number of animals examined	25	25	25	25	25	25	25	25	25	25
Effect										

	Males					Females				
Interstitium: chronic inflammation	15	13	17	17	25	9	14	12	16	25
Granulomatous inflammation	2	0	4	1	6	1	0	0	5	7
Interstitium: fibrosis	8	12	6	9	21	7	5	4	11	20
Bronchiolar/alveolar hyperplasia	0	0	0	0	0	0	0	0	0	0
Alveolar/intraalveolar macrophages	6	10	5	21	24	1	3	11	21	25
Alveolar/intraalveolar macrophages: foreign particulate material	0	17	22	25	25	0	13	23	25	25
Macrophages in the perivascular/peribronchiolar aggregates of lymphoid cells: foreign particulate material	0	0	0	0	3	0	0	0	3	2
Peribronchial lymph node macrophages: foreign particulate material	0	0	2	15	15	0	0	4	16	18

In a subsequent whole-body inhalation study, performed by Bio/dynamics Inc. (Newton and Daly, 1990) and published by Newton and co-workers (1994), the oncogenicity of diantimony trioxide was evaluated (Newton and Daly, 1990; Newton et al., 1994). This study is also reported in the 4.1.2.1 Toxicokinetics and 4.1.2.8. Carcinogenicity sections. Fisher 344 rats, 65 males and 65 females per group, 8 weeks of age, were exposed to diantimony trioxide at 0, 0.06, 0.51 or 4.50 mg/m^3 for 6hr/d, 5d/wk for 12 months followed by a 12-month observation period. Control animals were exposed to clean air only. The flow rate was 18-25 complete air changes per hour (the recommended flow rate in OECD guideline 412, 413 and 453 is 12-15 air changes per hour, Rapporteur comment). Five animals per sex per group were sacrificed at 6 and 12 months of exposure and at 6 months post-exposure. All surviving animals were sacrificed at 24 months (12 months post-exposure). The purity of the diantimony trioxide was 99.68 % and the particle MMAD was 3.76±0.84 µm with a geometric standard deviation of 1.79±0.32 for all concentrations. The concentrations of diantimony trioxide were determined by atomic absorption.

Animals were observed twice daily for viability and overt signs of toxicity. Detailed observations were conducted weekly and body weights were measured twice pre-test, weekly for the first 13 weeks, monthly thereafter and at termination. Ophtalmoscopic examinations were performed on all animals pre-test and on the day before their scheduled sacrifice. Hematological effects were evaluated at 12, 18 and 24 months. Complete gross postmortem examinations of all major organs were performed in all animals. Histological examinations were performed on hematoxylin-eosin-stained tissue sections of heart, nasal turbinates, larynx, trachea, lung and peribronchial lymph node. At each sacrifice, the left lung lobe was frozen for later diantimony trioxide analyses and blood samples were collected. Fecal samples were collected at the 18- and 24-month sacrifices.

Survival was not affected by the exposures to diantimony trioxide. At termination, there was 56 % survival of the males and 48 % survival of the females in the control groups with 56-58 % (males) and 40-66 % (females) survival in the exposed groups.

Detailed weekly observations showed a dose related increase in chromodacryorrhea (shedding of bloody tears) in the males, which did not change during the post exposure recovery period (data not shown). This effect was also confirmed during the ophtalmoscopic evaluation. According to the full study report by Bio/dynamics Inc., chromodacryorrhea was not seen in the females. In contrast, Newton and co-workers reported that chromodacryorrhea was also found in the females. Chromodacryorrhea is a sign of discomfort caused e.g. by stress, aging and illness. Thus, in this particular study it is difficult to asses what might have caused the condition. An increase in corneal scars was observed in both the males and females. However, the definitive ophtalmoscopic evaluation found these scars to be nearly equally distributed among all groups, similar to a spontaneous degenerative condition reported in Fischer 344 rats and were therefore not considered to be treatment related. Ophthalmoscopic evaluation at 6 months found compound related ocular irritation but this was not indicated at 12 or 18 months. Ophthalmoscopic examination at 24 months revealed an increase in cataracts (including focal posterior polar cataract, posterior subcapsular cataract and complete cataract) with respective incidences in the control, low-, mid- and high-dose groups of 11, 15, 21 and 18 % in males and 13, 40, 36 and 47 % in females. A microscopic evaluation of the eyes was performed on the animals in the control and the highest dose group. Moderate or severe lens degeneration was observed in 14 and 11 % of the males and in 13 and 33 % of the females for the control and high dose group, respectively. No statistical calculations were presented, but the cataracts in high-exposure-level females was well above normal and judged, by the authors, to be diantimony trioxide exposure related.

No significant differences in body weight gains were observed among the dose groups.

Absolute and relative lung weights were also unaffected in all exposure groups.

No diantimony trioxide related hematologic effects were found. Elevated total leukocyte counts and atypical lymphocytes in some animals in all groups at the terminal euthanization indicated, according to the authors, the presence of leukemia a common finding in aged Fischer 344 rats (authors' comment).

Effects observed in the lungs and peribronchial lymph nodes are shown in Table 4-47 and

Table 4-48. Chronic interstitial inflammation (minimal to moderate severity) was observed in the lungs of several control and treated animals during both the exposure and the observation periods. Interstitial fibrosis, granulomatous inflammation and bronchiolar/alveolar hyperplasia (all of minimal to moderate severity) occurred in a small number of animals during the observation period and was most pronounced in the high-dose group. Increased numbers of alveolar/intraalveolar macrophages and particulate material in alveolar/intraalveolar macrophages were seen in all dose groups (but not in the control group) during both the exposure and the observation periods. However, the increase in alveolar macrophages may be regarded as a normal pulmonary response to the foreign particles entering the lung.

Other post mortem findings, not further specified, were by the authors considered not to be treatment related.

The diantimony trioxide lung burden data show (see subsection 4.1.2.1.1) a lung burden-dependent effect on the diantimony trioxide clearance rate in the high dose group. It was calculated that with a lung containing approximately 2 mg of diantimony trioxide after 52

weeks of exposure, pulmonary clearance was decreased by 80 % with an increase in the clearance halftime from 2 to 10 months. Thus, the clearance mechanism was significantly impaired at this exposure level and was interpreted by the authors as an intrinsic toxic effect of diantimony trioxide rather than a general effect due to particle overload. This was assumed since the rate of clearance from the lungs of deposited benign or slightly toxic insoluble particles has been reported to be reduced by 50 % at a dust volume of about 1000 nl/lung (Muhle et al., 1990). In the current study a 50 % inhibition of clearance was seen at 400 µg of diantimony trioxide/lung. Volumetrically, with a diantimony trioxide density of 5.5 g/cm^3, this is according to Newton and coworkers (1994) equal to a dust volume of about 270 nl/lung. However, according to the rapporteur, 400 µg of diantimony trioxide is equal to about 73 nl (calculated as 400 µg / 5.5 g/cm^3).

In conclusion, this study indicates that diantimony trioxide exposure causes impaired clearance and chronic interstitial inflammation in the lung. However, there was also a high (62% and 67% for males and females, respectively) frequency of chronic interstitial inflammation in the lung of control animals, indicating that the lung inflammation also had other cause than the diantimony trioxide exposure. However, Newton and co-workers stated that the lung inflammation observed in controls is commonly seen in rats of this age and strain and judged that it did not compromise the health status of these animals and consequently it did not affect the results of the study. An elevated total leukocyte count was also observed in both control and diantimony trioxide exposed animals. The elevated total leukocyte count was by the authors explained by the presence of leukaemia. However, no supportive data for this explanation was presented. A dose related increase in chromodacryorrhea was observed in the males throughout the whole study. Since no data was presented it is not clear if chromodacryorrhea occurred also in the control group. Due to the discrepancy between the two reports on the occurrence of chromodacryorrhea in the females, it is not clear if this effect was also observed in the females. According to the authors, chromodacryorrhea can be secondary to dental abnormality, infectious disease or xerosis in rats. Although not seen in all animals (the teeth were not specifically examined) microscopic periodontal disease was seen in some rats and could according to the authors explain the presence of chromodacryorrhea. However, considering the interstitial inflammation in the lung and the elevated total leukocyte count it cannot be excluded, that the chromodacryorrhea was a result of the pulmonary inflammation. However, the causative agents to chromodacryorrhea are multiple, thus it is not possible to conclude if these animals were suffering from infectious disease or stressful condition in general.

The cataract findings in this study are present in both controls as well as in treated animals. Although cataracts have been reported to occur after oral exposure to other metallic compounds, (Ginsburg and Buschke, 1923; Alagna and D´Aquino, 1956; Schaumberg et al., 2004) there is, in this study, no dose-response relationship and statistical evaluation is lacking for this observation. Therefore, the cataract finding is inconclusive and will not be forwarded to the risk characterisation.

Table 4-47. Non-neoplastic microscopic findings seen after a 1-year inhalation exposure period to diantimony trioxide.

	Males				Females			
Dose (mg/m^3)	0	0.06	0.51	4.5	0	0.06	0.51	4.5
Number of animals examined	13	13	12	13	16	13	11	14

	Males				Females			
Effect								
Interstitium: chronic inflammation	10	8	11	12	10	11	10	14
Granulomatous inflammation	0	0	0	1	1	0	0	0
Interstitium: fibrosis	0	0	0	0	0	0	0	0
Bronchiolar/alveolar hyperplasia	0	0	0	0	0	0	0	0
Alveolar/intraalveolar macrophages	6	11	9	13	6	10	8	14
Alveolar/intraalveolar macrophages: foreign particulate material	0	13	12	13	0	13	11	14
Macrophages in the perivascular/peribronchiolar aggregates of lymphoid cells: foreign particulate material	0	2	6	7	0	6	4	7
Peribronchial lymph node macrophages: foreign particulate material	0	3	5	13	0	0	6	13

Table 4-48. Non-neoplastic microscopic findings related to the 1-year inhalation exposure period to diantimony trioxide seen during the 1-year observation period.

	Males				Females			
Dose (mg/m³)	0	0.06	0.51	4.5	0	0.06	0.51	4.5
Number of animals examined	52	52	53	52	49	52	54	50
Effect								
Interstitium: chronic inflammation	32	37	36	48	33	40	48	48
Granulomatous inflammation	3	2	5	7	2	2	5	3
Interstitium: fibrosis	0	0	1	2	0	1	1	4
Bronchiolar/alveolar hyperplasia	3	1	2	4	1	0	0	6
Alveolar/intraalveolar macrophages	31	44	46	52	28	40	48	50
Alveolar/intraalveolar macrophages: foreign particulate material	0	15	38	51	0	24	49	48
Macrophages in the perivascular/peribronchiolar aggregates of lymphoid cells: foreign particulate material	0	22	46	47	0	31	47	47
Peribronchial lymph node macrophages: foreign particulate material	0	6	34	39	0	6	29	39

A NOAEC of 0.51 mg/m^3 is derived from this study based on impaired lung clearance observed at 4.50 mg/m^3. Although there might be some uncertainty regarding the accuracy of the LOAEC and NOAEC numerical values, as the study had a high background incidence of lung inflammation in control animals, the NOAEC of 0.51 mg/m^3 is brought forward to the risk characterisation.

The carcinogenic effects of one concentration of diantimony trioxide and antimony ore were evaluated in an inhalation study in Wistar rats, 90 males and 90 females per group (Groth et al., 1986a). This study is also described in section 4.1.2.8., Carcinogenicity. The animals (free of endemic respiratory disease stated by the supplier, Charles River Breeding Labs, Wilmington, Mass.), 8 months of age, were exposed to diantimony trioxide [time-weighted average (TWA) 45 mg/m^3 (range = 0-191.1)], antimony ore [TWA 36-40 mg/m^3 (range = 0-91.1)] or filtered air (controls) in exposure chambers, 7 h/day, 5 days/week for up to 52 weeks. The MMADs for diantimony trioxide and antimony ore were 2.80 and 4.78 µm, respectively (GSD not reported). The Sb content in the diantimony trioxide was 80 % and in the antimony ore it was 46 %. Major contaminants in the diantimony trioxide were lead (0.23 %), tin (0.21 %) and arsenic (0.004 %) and in the antimony ore they were aluminium (0.48 %), iron (0.33 %), lead (0.25 %), tin (0.16 %) and arsenic (0.079 %). At 6, 9, and 12 months after initiating exposures 5 animals/sex/group were sacrificed and autopsied, the remainder of the animals were sacrificed 18-20 weeks post-exposure. In addition, all animals that died or were sacrificed due to ill health were autopsied. At autopsy all organs were examined grossly and tissue sections from the lungs, liver, kidneys, pancreas, spleen, adrenal, thyroid, pituitary, bladder, brain, eye, bone marrow, skin, lymph nodes (mesenteric and tracheobronchial), stomach and colon (ascending and descending) from each rat were fixed in buffered 10 % formalin, embedded, sectioned and stained with hematoxylin and eosin for examination by light microscopy. Samples from the testes and prostate from males and mammary gland, ovary, uterus and cervix from females were also examined as well as any abnormal tissue. In addition, at the final sacrifice heart tissue was sampled and examined by light microscopy. At sacrifices, portions of liver, lungs, kidneys, brains, spleens and blood from five animals/sex/group were sampled for antimony concentration analysis.

The data indicate that there was no treatment related mortality. The mean body weight of the males exposed to diantimony trioxide and the females exposed to antimony ore was slightly but statistically significantly reduced (6.2 % and 6.4 %, respectively). Sporadic bleeding from eyes and hematuria occurred in all groups, but appeared to occur more frequently in the Sb_2O_3 and Sb ore groups (data not shown).

The concentration of antimony in the lungs of the male rats (38,300 µg Sb/g dry weight) after 9 months of exposure to diantimony trioxide was significantly higher than in the females (25,600 µg Sb/g dry weight). The concentrations of antimony in the lungs of these groups were considerably greater (males: 5.4 times; females 5.7 times) than in animals exposed to antimony ore. The concentration of arsenic in the lungs of animals exposed to diantimony trioxide was 213 µg/g dw for males and 150 µg/g dw for females. This was considerable more arsenic than measured in the lungs of males and females exposed to Sb ore (in males 21 times as much and in females 10.8 times as much, $p < 0.05$)

Gross pathology at the final sacrifice showed that the lungs of all animals from both exposure groups had slightly elevated, confluent, white and yellow foci on the pleural surfaces of all lobes.

Histopathological examination of the lungs at 6 months showed that the lungs from the female rats exposed to Sb_2O_3 contained particles evenly scattered throughout all lobes of the lung and in more than 90 % of the alveoli. Several dense particle aggregates about the size of macrophages were present in about 10 % of the alveoli. Individual macrophages were obscured by the particles. In some alveoli the particles were embedded in dense, pink, homogeneous protein. Alveolar-wall thickening, consisting of interstitial fibrosis, alveolar-

wall cell hypertrophy and hyperplasia appeared in about 50 % of the alveolar duct regions, affecting about 5-10 % of all alveoli. Cuboidal and columnar cell metaplasia occurred in some of these foci. All these effects increased with the time of exposure. In addition, foci containing cholesterol clefts were seen. At the sacrifice, 4-5 months postexposure, the density of particles and amount of protein in the alveoli had significantly decreased. However, the extent of interstitial fibrosis had increased. In some rats it affected over 80 % of the alveoli. The number of foci containing cholesterol clefts had also increased and in some rats dense scars that appeared to be confluent areas of interstitial fibrosis were present. At 6 months, the lungs of male rats exposed to Sb_2O_3 had the same amount of interstitial thickening as the female rats, however, there was less alveolar protein. At 12 months, the severity of the interstitial fibrosis was the same as in the females. In some interstitial areas, there was, in addition, dense eosinophilic material. The cuboidal, alveolar-wall cell metaplasia was, however, not as extensive as in the females and there were fewer foci with cholesterol clefts. At 4 months post-exposure a diminution in the amount of alveolar-wall metaplasia was observed and it was less severe than that observed in females. There were fewer foci with cholesterol clefts and less alveolar protein. In interstitial spaces there were more mononuclear cells, lymphocytes and plasma cells than observed in the female rat lungs. The extent and severity of the interstitial fibrosis was the same as in the females. The histopathology of the lungs in female rats exposed to Sb ore was similar to that seen in Sb_2O_3-exposed females, except that there were fewer particles and less alveolar protein visible at all sacrifice intervals. In addition, they contained mononuclear cell granulomas similar to those that are seen in the early stages of silicosis or sarcoidosis. In Sb ore exposed males the alterations in the lungs were similar to those seen in Sb_2O_3-exposed males. None of these effects were seen in the control animals. No other exposure-related non-neoplastic effects were observed.

According to the authors, no significant pathological alterations were seen in any of the control lungs. Occasional foci containing lymphocytes, typical of chronic pneumonia, were seen in a few rats.

In conclusion, even though all animals were free of endemic respiratory disease at the start of the study, occasional foci containing lymphocytes, typical of chronic pneumonia, were seen in a few control rats. In addition, sporadic bleeding from eyes and hematuria occurred in all groups, but appeared to occur more frequently in the diantimony trioxide and antimony ore groups. These findings may indicate that some animals had some non-treatment related sickness. However, the lung changes described after diantimony trioxide and antimony ore exposure was not observed in the control group and therefore it can be assumed that the lung effects observed were treatment related and a LOAEC of 45 mg/m^3 is suggested from this study based on pleural plaques, lung fibrosis and cholesterol clefts.

A chronic inhalation toxicity study of diantimony trioxide has also been performed in female rats and miniature swine (Watt, 1983). 148 female Fischer rats from the Charles River Laboratories, 14 weeks of age, were divided into three groups (the number per group was not specified) and exposed to 0, 1.9 ± 1.8 and 5.0 ± 3.8 mg diantimony trioxide/m^3 for 6 h/day, 5 days/week for one year in whole body exposure chambers. Similarly, eight female Sinclair S-1 miniature swine were divided into control (n = 2), low dose (n = 3) and high dose (n = 3) groups and exposed in the same way as the rats. The exposure concentrations have been recalculated from the values in the study report as they were reported as Sb, 1.6 ± 1.5 and 4.2 ± 3.2 mg Sb/m^3 respectively. Control animals were treated the same as exposure animals, that

is, moved to exposure chambers during each exposure period. Rats and swine of the same exposure group were exposed in the same chamber; the swine free on the floor and the rats in pairs in cages suspended from the walls of the chamber. Air samples were taken within the exposure chambers at the same level as the suspended rat cages. The diantimony trioxide used was 99.4 % pure with arsenic (0.02 %) and lead (0.2 %) as the major contaminants. Only particles with mean aerodynamic diameter of 15 µm or less would pass into the chamber. The particle size (Ferret's diameter) was 0.44 and 0.40 µm for the low and high concentrations (GSD 2.23 and 2.13, respectively). Surviving animals were kept up to 15 months post-exposure.

Prior to and after 3, 6, and 12 months of exposure animals were evaluated for evidence of toxicity and blood samples were taken. The rats were also evaluated at approximately 9 months exposure. The rats were anesthetized ip with Surital or V-Pento and sacrificed through exsanguinations. The swine were anesthetized with V-Pento and sacrificed at the end of the exposure period by anesthetizing with V-Pento and opening the chest cavity. Differential count, red and white blood cell count, hemoglobin, hematocrit, mean corpuscular volume, mean corpuscular hemoglobin and mean corpuscular hemoglobin volume, serum enzymes and chemistry were determined in both rats and swine. Animals were weighed periodically throughout the exposure. At sacrifice, the heart, lung, liver, spleen and kidney were weighed and fixed in 10 % buffered formalin for subsequent light microscopic examination. Prior to exposure, at 6 months and at the end of exposure the swine were subjected to roentgenograms and electrocardiograms.

No exposure-related effects on survival, haematology or clinical chemistry were noted in either rats or swine. Roentgenograms and electrocardiograms showed no abnormalities. T-waves generally had a negative deflection, but no change in amplitude was observed, throughout the study, which according to the authors is normal for miniature swine. The body weights of the exposed rats were significantly higher than the controls at pre-exposure and throughout the exposure period. At the end of exposure and post exposure there was no significant difference in body weights between exposed and control rats. No effect was observed on swine body weight. There was no effect on organ weights except for lung. For the rat, the lungs were significantly heavier than the controls at nine months in the high dose group and at 12 months in both exposure groups. No significant difference was observed at one year post exposure. The swine lungs show the same dose-response pattern after 12 months of exposure, however, the differences were not statistically significant. No exposure related histopathological alterations were observed in the swine. However, a number of effects were observed in the lungs of exposed rats (Table 4-49). These included focal fibrosis, adenomatous hyperplasia, multinucleated giant cells, cholesterol clefts, pneumonocyte hyperplasia, and pigmented macrophages. The severity of the effects increased with both time and exposure concentration. Postmortem findings that appeared to be treatment related included discoloration and increased pulmonary alveolar-intralveolar macrophages in both exposure groups and focal subacute-chronic interstitial inflammation and granulomatous inflammation in the high-exposure group.

Focal fibrosis was the earliest non-neoplastic effect, observed already after three months in the high-dose group. The effect was time- and dose-dependent. After one year of exposure, focal fibrosis was seen in 10/21 animals of the low dose and 17/20 animals in the high dose group.

Table 4-49 Non-neoplastic microscopic alteration seen after inhalation exposure to diantimony trioxide.

		Focal fibrosis			Pneumocyte hyperplasia			Adenomatous hyperplasia			Multinucleated giant cells			Cholesterol clefts		
Group	Death or sacrifice	Control	Low	High	Control	Low	High	Control	Low	High	Control	Low	High	Control	Low	High
A	Pre-exposure	0/3	0/1	0/0	0/3	1/1	0/0	0/4	0/1	0/0	0/4	0/1	0/0	0/4	0/1	0/0
B	From start through 5 months of exposure	0/2	0/4	1/3	0/2	3/4	3/3	0/2	0/4	0/3	0/2	0/4	0/3	0/2	0/4	0/3
C	From 6 through 9 months of exposure	0/4	0/3	3/3	0/4	3/3	3/3	0/4	0/3	0/3	0/4	0/3	2/3	0/4	1/3	3/3 *
D	From 9 through 12 months of exposure	0/4	1/5	5/5 ***	4/4	5/5	4/5	0/4	0/5	5/5 ***	0/4	0/5	5/5 ***	0/4	2/5	5/5 ***
E	At the end of exposure (12 months)	0/9	9/9 ***	8/9 ***	0/9	8/9 **	9/9 ****	0/9	1/9	6/9 ***	0/9	4/9 **	8/9 ****	0/9	9/9 ****	8/9 **
F	Between 2 and 12 months post-exposure	0/6	4/5 ***	7/7 ***	0/6	2/5	5/7 **	0/6	0/5	5/7 **	0/6	2/5	5/7 **	0/6	2/5	4/7
G	12 to 15 months post-exposure	1/13	12/17 ****	17/18 *****	0/13	12/17 ****	12/18 ****	0/13	0/17	13/18 ****	0/13	5/17 *	11/18 ****	0/13	9/17 ***	14/18 ****
Total A-G		0/41	26/44	41/45	4/41	34/44	36/45	0/42	1/44	29/45	0/42	11/44	31/45	0/42	23/44	34/45

Statistically different from control: *p<0.05; ** p< 0.02; *** p< 0.01; **** p< 0.001; *****p<0.0001

A variety of neoplastic and non-neoplastic changes were observed in other tissues, most notably mammary glands, but they were not considered treatment-related since their incidence and severity were comparable among all groups. No exposure-related effects were observed in the swine.

A re-evaluation of the histopathology tissue sections from the Watt- and the Newton-studies indicated higher lung deposition of antimony and more severe lung damage in exposed rats in the Watt-study than in the Newton-study, which allegedly were conducted at similar exposure levels (1.9-5.0 and 0.06-4.50 mg/m^3, respectively). This suggests that the exposure levels in the Watt study may have been above those reported, but the difference could also be due to

different particle generation techniques or different strains of rats. Although there is some uncertainty with regard to the actual exposure concentration, due to the occurrence of pulmonary effects in rats exposed 6 hours per day, 5 days a week for one year in this study, a LOAEC of 1.9 mg diantimony trioxide /m^3 is derived. This is based on findings indicative of endogenous lipoid pneumonia (focal fibrosis, pneumocyte hyperplasia, presence of giant cells and cholesterol clefts) observed in animals in this and the higher dose group.

A group of 24 guinea pigs were exposed to diantimony trioxide dust via inhalation for 2 h per day, seven days a week, during the first three weeks, thereafter for 3 h per day for additional 27 weeks (Dernehl et al., 1945). Eleven control animals were included. However, the way of treatment was not stated. Diantimony trioxide was generated by heating the metal at 700 °C in the presence of large quantities of air. The fume was reported to contain 99.8 % diantimony trioxide with a particle size of 1 μm or less. The average diantimony trioxide dust concentration in the exposure chambers was 45.5 mg/m^3, determined gravimetrically by means of an electrostatic precipitator. Fluctuations in dust concentration were reported to occur. No food was provided during the 2-3 hour diantimony trioxide exposure period. The animals were sacrificed, after a 12 h fasting period, at various time points, corresponding to exposures between 33 and 609 h. The laboratory studies included weight changes, blood changes and gross and microscopic pathology of the heart, lungs, liver, kidneys, spleen and gastro-intestinal tract.

Four of the treated animals died during the experiment (in week 10, 16, 17 and 21, respectively). These animals were exposed for 184, 298, 321 and 402 h. In each instance death occurred suddenly after a week or more of illness. A reduced weight gain was observed in treated animals. However, this was not related to diantimony trioxide exposure (data not shown). Blood cell counts in the six animals that were most heavily exposed showed a decrease in the overall number of white blood cells, a decrease in the relative amount of polymorphonuclear leucocytes and eosinophiles but a relative increase in lymphocytes and monocytes. No statistical details were provided. Red blood cell counts and haemoglobin was normal. An increase in the relative weight of liver and lung were observed in the treated animals. 95 % of the livers from treated animals were above the average control liver weight and 50 % were above the maximum control liver weight. Similarly, 84 % of the lungs from treated animals were above the average control lung weight and 79 % were above the maximum control lung weight. However, no exposure time-dependent relationship was evident. No statistical calculations were presented but from the data presented for the liver on "% increase over average control weight" the calculated average liver weight increase was 30.0 ± 17.1 %. The calculated average liver weight increase over maximal control liver weight was 14.6 ± 9.2.

Microscopy of all exposed animals revealed a thickening of the alveolar wall as found in interstitial pneumonitis, the severity ranged from areas of simple thickening to areas in which the process had practically obliterated the alveoli. According to the authors normal guinea pig lung often presents a mild pneumonitis, however, not as extensive or as severe as that found in the exposed animals. There was a hypertrophy of the pulmonary lymphoid tissue and particles of diantimony trioxide were visible in phagocytes in interstitial tissue spaces. Edema was also observed. Pneumonia was observed in 2 of the animals that died and 3 additional animals, of which one died, showed small areas of organizing pneumonia. The pneumonias were intensely hemorrhagic. Pneumonia was not noted in the controls. The lungs of the 15

animals exposed for 138 h or more were reported to show scattered sub-pleural petechial hemorrhages. Fatty degeneration of the liver occurred in 11 (45.8 % of all treated animals) of the 15 animals having exposure for 138 h or longer. Only one of the 11 control animals (9 %) showed any evidence of fat in the liver. In most cases the greatest concentration of fat was around the central vein with lesser quantities surrounding the portal vein and a relatively clear zone intervening the two. Two of the animals dying with pneumonia presented extensive fatty degeneration involving all three zones. The spleen showed hyperplasia of the lymph follicles in 50 % of the cases. There was an abnormal amount of blood pigment present in 62 % of the cases. In addition, phagocytes loaded with diantimony trioxide were present in small numbers. 50 % of the spleens examined showed a marked reduction or absence of polymorphonuclear leucocytes. No abnormalities in either heart or kidney were observed.

This study indicates that diantimony trioxide may cause pulmonary toxic effects, liver weight increase, fatty degeneration of the liver and death in guinea pigs after repeated exposure of 45.5 mg diantimony trioxide/m^3. However, since data for individual control liver and lung weights were not reported the study is regarded inconclusive and will not be used in the risk characterisation.

An inhalation toxicity study of diantimony trioxide was performed in rats and rabbits (strain and sex were not stated) (Gross et al., 1955b). Diantimony trioxide was suspended in water, dispersed by a compressed air atomizer and heat dried. The dust-laden air was passed through an impinger which trapped particles larger than 1 μm before being blown into inhalation chambers. The average particle size was 0.6 μm as determined by electron microscope. 50 rats were exposed to 100-125 mg/m^3 for 14.5 months (100 h/month) and 20 rabbits were exposed to 89 mg/m^3 for 10 months (100 h/month). No control animals were used. Beginning at two months and at varying intervals (not further specified) thereafter, groups of animals were killed. The lungs of these animals, as well as of those, which died of spontaneous pneumonia, were expanded with formalin.

Nine (18 %) rats and 17 (85 %) rabbits died spontaneously in the inhalation chambers. The predominant cause of death was bacterial pneumonia and it affected the rabbits to a higher degree than the rats. The lungs, in rabbits beginning with the fifth month and in rats from the ninth month, were mottled with chalk-white foci 1-2 mm in diameter upon the pleura and cut surface. The latter was also coarse-textured and dry. The white mottling of the lungs increased in intensity with longer exposure. Microscopically, the inhaled fine diantimony trioxide dust was early associated with swelling, proliferation and desquamation of alveolar macrophages. The dust was found within alveolar macrophages and as densely agglomerated deposits in the peribronchial and perivascular stroma. With longer exposure, fatty degeneration of macrophages became a prominent feature. This culminated in necrosis, cell rupture and a progressive increase in the alveolar lipid content. Finally, colourless, needle-shaped crystals formed in the air spaces. These crystals, soluble in fat solvents and giving a positive Lieberman-Burchard reaction for steroid, were found in sheaves and singly. Large numbers of coarse and fine sudanophilic droplets were demonstrable within macrophages and also in the alveolar debris. This debris contained finely dispersed diantimony trioxide. Collagenous thickening of alveolar walls and interstitial thickening occurred early and were preceded by increased tortuosity, arborescence and thickening of reticulum fibers. Such fibrosis, initially minimal became more extensive with increasing exposure. In the rabbit lungs the interstitial pneumonia was more pronounced than in rat lungs, and characterised by great widening of the

alveolar septa. Initially this widening was caused by edema fluid. Later, septal histiocytes increased in size and number and collagen fibers between the capillary and respiratory membrane made the septal thickening permanent. There was no indication of fibrosis in the pulmonary lymph nodes of neither rats nor rabbits, in spite of frequently considerable deposits of diantimony trioxide.

A high incidence (not further specified) of spontaneous deaths from bacterial pneumonia in the inhalation chambers was reported.

In conclusion, considering the reported high incidence of spontaneous deaths from bacterial pneumonia and the absence of control animals the study is considered inconclusive and it is not clear to what extent the diantimony trioxide exposure has contributed to the overall pathology of the lung. This study will not be used for the risk assessment of diantimony trioxide.

In a briefly described inhalation lung toxicity study, two groups of albino Sprague Dawley, C.D., rats were exposed to dust of diantimony trioxide or powdered antimony ore (Cooper et al., 1968). Only the results from the diantimony trioxide exposure will be presented here. The group exposed to diantimony trioxide consisted of 10 male and 10 female rats, approximately 84 days old and weighing from 342 to 425 g (males) and 227 to 276 g (females). A specially constructed exposure chamber was used whereby only the nose of the animal came in contact with the aerolised diantimony trioxide at a concentration of 1700 mg antimony/m^3 of air. This corresponds to an exposure of 2000 mg diantimony trioxide/m^3. The purity or the particle size of the substance was not stated. All animals received 1 to 6 exposures, of 1 h each, every 2 months, during a period of time ranging from 66 to 311 days. Animals were sacrificed periodically for study of gross and microscopic pathology.

Initially, at 66 days after the first series of 1 h exposures the pulmonary pathology consisted of a phagocytic response. The dust-laden phagocytes were generally lying free within the alveolar spaces or they were intermingled with cells of the septa. In some situations there was a tendency for the cells to focalise the dust into small deposits throughout the lung. At more prolonged intervals of time after the first and subsequent exposures, this focalisation became increasingly prominent. At 311 days the phagocytic response persisted, without any appreciable chronic pneumonitis, neither cellular, collagenous nor fibrotic.

Non-pulmonary tissue, such as the tracheobronchial lymph nodes remained soft and without abnormal enlargement. Microscopically, they exhibited scattered deposits of intracellular antimony, attended by mild hyperplasia but without evidence of chronic inflammation. The spleens disclosed microscopic presence of scattered dust particles accompanied by a moderate proliferation of reticuloendothelial elements. There was no hepatic and renal pathology of significance (no data shown).

This study showed that repeated inhalation exposure of 2000 mg diantimony trioxide/m^3, 1 to 6 exposures, of 1 h each, every 2 months, during a period of time ranging from 66 to 311 days in rats caused pneumoconiosis without signs of chronic inflammation. However, since the purity of the diantimony trioxide given was not stated and the dosing regime was very different from current standard the study will not be used to derive a LOAEC.

The developmental toxicity properties of diantimony trioxide were recently investigated in a repeated inhalation toxicity study in rats, which is also presented in section 4.1.2.9., Toxicity for reproduction (MPI, 2003). The study was conducted in accordance with Standard

Operating Procedures and was based on a draft guideline published in the US EPA Health Effects Test Guidelines and the OECD Guideline Number 414.

Mated females were exposed to diantimony trioxide via nose-only inhalation from Day 0 (fertilization) to Day 19 (one day prior to scheduled euthanasia and laparohysterectomy) of gestation (6 h/d) at concentrations of 2.6 (SD ± 2.43), 4.4 (SD ± 3.88), and 6.3 (SD ± 4.18) mg/m^3. A purity of Sb_2O_3 of 99.87 % is stated. Control females received clean air by the same procedure and dosing regime as the treated females. Dust aerosol atmospheres of the test article were generated into the breathing air of the treated animals using a Wright Dust feeder. A chamber airflow of at least 0.6 liter per minute per animal resulted in at least 10 chamber air changes per hour. The mass median aerodynamic diameters (MMAD) and geometric standard deviations (GSD) ranged from 1.59 to 1.82 μm and 1.713 to 1.744, respectively. Observations of the dams included clinical signs, conducted daily following exposures, gestation body weights and gestation food consumption. Blood samples were collected from 10 dams/treatment group and the Sb concentration in RBC was determined. A complete necropsy was performed on all dams. The lungs and brain were weighed and the lungs then infused via the trachea with 10 % neutral buffered formalin. Based on organ weight changes, the lungs of 10 females/group randomly selected were processed for histopathological examination.

All animals survived to scheduled euthanasia on Day 20 of gestation. The food consumption in the 4.4 and 6.3 mg/m^3 groups was statistically higher than controls over GD 15-18, 18-20 and 0-20. These increases in food consumption corresponded with slight increase in body weight gains over these same intervals, but the differences in weight gain compared to controls were not statistically significant and according to the author not considered toxicologically meaningful. The mean antimony level in RBC were (without a clear dose-response relationship) statistically higher than controls in each of the treated groups, 0.128 ± 0.0286, 3.275 ± 1.0391, 3.078 ± 0.5624 and 5.591 ± 1.3248 μg/g for the dose groups 0, 2.6, 4.4, 6.3 mg/m^3, respectively. A dose-related increase in lung weights, absolute and relative to brain weights, was seen in the diantimony trioxide-treated groups. These differences in lung weights from controls were statistically significant and considered indicative of a treatment-related response. The absolute lung weights were 24.2 %, 31.1 % and 38.6 % heavier than control in the 2.6, 4.4, and 6.3 mg/m^3 groups, respectively. Lung weights relatively to brain weights were 20.3 %, 26.3 %, and 34.8 % heavier, respectively. Test article-related microscopic findings were observed in the lungs of all animals evaluated at all exposure levels with a diffuse accumulation of pigmented alveolar macrophages, reflecting accumulation and phagocytosis of the test article particulate matter. Pulmonary alveoli contained variable numbers of macrophages with abundant eosinophilic cytoplasm with minimal to moderate quantities of brown granular pigment, as well as small to moderate quantities of extracellular eosinophilic proteinaceous material containing similar pigment. Throughout the lungs, scattered foci of acute inflammation (0/10, 7/10, 4/10 and 6/10 in control, 2.6, 4.4, and 6.3 mg/m^3 groups respectively) and type II cell hyperplasia (0/10, 5/10, 4/10 and 5/10 in control, 2.6, 4.4, and 6.3 mg/m^3 groups respectively), were observed. Accumulations of pigmented macrophages and associated inflammation were likely the cause of the increased lung weights of treated animals compared to controls. The microscopic finding of increased numbers of alveolar macrophages containing foreign material noted in the current study is similar to findings observed in previous subacute and chronic inhalation studies of diantimony trioxide in Fischer rats. However, as would be expected, the inflammation and type II cell hyperplasia noted in the current study was generally of acute to

subacute duration as opposed to the granulomatous inflammation and interstitial fibrosis observed in the previous studies by Watt, Groth et al., and Newton et al. (Watt, 1983; Groth et al., 1986a; Newton and Daly, 1990).

In conclusion, this study showed a dose related increase in lung weight and a diffuse accumulation of pigmented alveolar macrophages, probably reflecting accumulation and phagocytosis of the test article particulate matter in the dams. Also, scattered foci of acute inflammation and type II cell hyperplasia were often observed. It should also be noted that the exposure doses in this study were almost the same in the three dose groups. This is also reflected in the mean level of antimony in RBC, which did not show a dose dependent increase. The concentration of antimony in RBC was almost the same in the two lowest groups and barely twice as high in the highest dose group. This indicates that three different dose groups were not achieved in this study, which is the minimum number of dose groups recommended in The OECD guideline 414.

A LOAEC for local toxicity in the lungs of 2.6 mg/m^3 is suggested from this study based on acute pneumonia with significantly increased lung weight, 24.2 % and 20.3 % higher than control for absolute and relative weights, respectively.

Dermal

No studies on repeated dermal exposure on diantimony trioxidehave been located.

Oral

In a 90-day oral feeding study of diantimony trioxide groups of 12 male and 12 female Wistar rats of the Alpk:APSD strain were fed diets contaning 0, 1000, 5000 or 20000 ppm diantimony trioxide (Hext et al., 1999, also reported by CTL, 1997). Diantimony trioxide, supplied as a white solid with a purity of 99 %, was mixed in the diet and the homogeneity of the mixture was > 95%. No information is provided on the size of the diantimony trioxide particles in the diet, which is likely to affect the gastrointestinal uptake of the substance. The calculated mean daily doses of diantimony trioxide were 84, 421 and 1686 mg/kg bw in males and 97, 494 and 1879 mg/kg bw in females. Cage-side observations were made daily, which included recording changes in clinical condition or behaviour. The body weight of each rat was recorded before exposure started and then once a week until termination. At termination, blood samples were taken and analysed for haematology and clinical chemistry parameters. Complete necropsies were performed on all rats. The adrenal glands, brain, kidneys, liver, epididymides and testes were weighed. All organs and tissues were examined for macroscopic lesions and fixed in 10 % neutral buffered formalin or other appropriate fixative. Tissues from an extensive range of organs from the controls and the top dose group were examined under the light microscope, together with any macroscopically abnormal tissue from the intermediate groups.

There was no substance-related effect on food intake, body weight gain or clinical signs of toxicity. A 10 % increase in absolute and relative liver weight was observed in female and male animals in the high dose group along with changes in some clinical chemistry parameters. Male animals in the high dose group showed a 30 % increase in triglycerides ($P<0.01$) and a 12 % decrease in alkaline phosphatase ($P<0.05$). Alkaline phosphatase was

also decreased in female animals both at 5000 (24 %) and 20000 (37 %) ppm ($P<0.01$) and in a dose-dependent manner. Cholesterol and aspartate aminotransferase levels were significantly increased in females in the high dose group by 13 and 52 %, respectively. However, no histological changes in the liver were observed to support an adverse effect on liver, and the absence of any other evidence of antimony intoxication suggests that these findings are adaptive to treatment. There was a slight increase in cysts in the pituitary of both sexes in the high dose groups. This was not considered to be treatment related as this is a common spontaneous change in this strain of rats and all values were within the historical control range. Three of the twelve (25 %) males in the high dose group had slight (n=2) or moderate (n=1) plasma cell infiltration in the cervical lymph node. This was not observed in treated females or in any control animal, but according to the evaluation made by the laboratory it has been seen in historical controls but the frequency is not reported.

A NOAEL corresponding to 1686 mg/kg bw/d (males) and 1879 mg/kg bw/d (females) can be derived from this study.

In another study performed on male Wistar rats12-15 animals per group were fed with diet containing 1.0 or 2.0% (corresponding to approximately 500 mg/kg/day or 1000 mg/kg/day) diantimony trioxide for 24 weeks (Sunagawa, 1981). Apart from hematocrit and haemoglobin values, which were measured in 11 animals per group, all other parameters were analysed in 5 animals per exposure group. This study is also reported in chapter 4.1.2.1. Toxicokinetics. No changes in total body weight or organ weights were noted. The haematologic examination showed a small but significant decrease of RBC (control = $9.6 \pm 0.4 \times 10^6/mm^3$; 1.0 % Sb_2O_3 = $7.5 \pm 0.6 \times 10^6/mm^3$; 2.0 % Sb_2O_3 = $8.0 \pm 0.4 \times 10^6/mm^3$) in the groups exposed to diantimony trioxide. All values, however, are within normal range. Serum biochemical parameters showed significantly increased GOT [= ASAT (aspartateaminotransferase)] values in both exposure groups. ALP (alkaline phosphatase) was significantly elevated in the 2 % Sb_2O_3 treated group. The increases in ASAT were 35 % and 31 %, respectively, in the 1 % and 2 % Sb_2O_3-dose groups, respectively and for ALP the increase was 17 % in the 2 % Sb_2O_3-dose group. The histopathological examinations of the livers showed slight disorder and cloudy swelling in hepatic cords in 3 out of 5 rats in the 1 % Sb_2O_3 treated group and 2 out of 5 rats in the 2 % Sb_2O_3 treated group.

This study indicates that repeated exposure to diantimony trioxide via the oral route might have an adverse effect on RBC in Wistar rats. However, despite the small decrease of RBC reported in groups exposed to antimony, the levels of RBC is still within normal range for rats (Harkness and Wagner, 1989). Slight disorder and cloudy swellings of the hepatic cords was observed in the liver and a significant increase in ASAT and ALP, which indicate hepatic cell lesion and cholestasis, respectively, suggests that diantimony trioxide is toxic to the liver. However, the changes in the biochemistry parameters must be compared with normal ranges, for examples with historical control data of the laboratory in question, and also evaluated in context of other findings (clinical and histopathological) supporting the biochemistry findings. Since information on historical controls and adverse histological liver findings are lacking a LOAEL will not be derived from this study.

In a very briefly reported study 20 rats (strain and sex not stated) were given a basal diet containing 2% diantimony trioxide (corresponding approximately to a daily intake of 1000 mg/ kg bw) for 8 months (Gross et al., 1955a). 20 pair-fed controls were given the basal diet without diantimony trioxide. When the diantimony trioxide group gave some evidence of restriction in food intake, the food was restricted in a like amount for the control group. The

animals fed diantimony trioxide gained weight at a slower rate than the pair-fed controls. After 8 months of feeding with control or diantimony trioxide diet the body weights were 350 g in the control group and 300 g in the exposed group. No statistical information is given. No gross or microscopic abnormality of organs and tissues was noted in the animals examined at the end of the experiments although the extent of histopathological examination is not specified.

Due to the little information presented in this study, it is considered inconclusive and cannot be used to derive a NOAEL.

In a briefly described oral toxicity study groups of male albino rats, strain and number was not stated, were administered diantimony trioxide in the diet for 30 days (Carnegie-Mellon Institute of Industrial Research, 1945). The administered doses were 0, 60, 270 and 1070 mg/kg/d. It was not stated what pathological examinations that was performed.

The group receiving 1070 mg/kg bw/day ate significantly less than did control animals (data not shown), grew significantly less (average weight change per rat was +60.4 g for exposed animals versus +105.1 g for controls) and had significantly higher red blood cell count (7.84 x 10^6 rbc/mm^3 for exposed animals versus 6.52 x 10^6 rbc/mm^3 for controls). The p-value was not stated. One of these rats had minor cloudy swelling in the kidney but otherwise the pathology of all rats was normal.

Due to the lack of information on the number of animals used and which examinations that were performed, this study is considered inconclusive and cannot be used for risk assessment.

In a briefly described range finding study groups of 10 Sherman rats, 5 of each sex in one cage, were administered diantimony trioxide in the diet for 30 days (Smyth and Carpenter, 1948). 3-4 dose levels ranging from 60-1070 mg/kg bw/day (not all dose levels are given) and a control diet were administered. Effects on the increase in body weight from the approximate 120 g starting weight, reduction in appetite, death and micropathology of adrenal, upper intestine, kidney, liver and spleen were studied.

No deaths occurred at any dosage level. At 1070 mg/kg bw/day reduced growth, reduced appetite and micropatology was observed (no further details were given). The maximum dosage level having no effect was 270 mg/kg bw/day.

Due to the lack of information on the effects and their significance, this study it is considered inconclusive and cannot be used to derive a NOAEL.

A chronic feeding toxicity study performed on male Wistar rats (number not stated) is briefly described in a summary (Hiraoka, 1986). This study is also described in chapter 4.1.2.1. Toxicokinetics. There were one control group and one group receiving 1 % (w/w) diantimony trioxide (corresponding approximately to 500 mg/kg bw/day) in the diet for 12 weeks. All the rats were allowed antimony-free diet for the following 12 weeks. Blood and organ were taken from the rats at 0, 4 and 12 weeks after cessation of diantimony trioxide administration.

No effects on behaviour, general appearance, body weight, haematology or blood biochemistry was observed.

Due to the lack of information on the number of animals used and which examinations that were performed, this study is considered inconclusive and will not be used for the risk assessment of diantimony trioxide.

4.1.2.6.2 Studies in humans

In vivo studies

The human repeated dose studies reported below are case reports on workers employed in industries manufacturing diantimony trioxide. The workers were probably subjected to inhalation, dermal and oral exposure, although it can be assumed that exposure via inhalation was the dominating route.

Inhalation

In a case study, illness among workers of a mining company engaged in the mining, concentrating and smelting of an antimony sulphide ore was reported (Lucian and Renes, 1953). The illness was found among workers engaged in the smelting operations and among maintenance workers who spent a substantial part of their time in the smelter building. During the first five months of operation, there were 78 men who had worked two weeks or longer in the smelter building. Sixty-nine of these workers made 218 visits to the plant physician for reasons of occupational illness. Air samples were collected during the sixth months, when the conditions were substantially the same as during the previous five months, and analysed for the presence of antimony and arsenic. The smelter was divided into two zones, the electric furnace area and the cupel area. The average zone concentration of antimony was 10.07 and 11.81 mg per m^3 of air, respectively for the two zones. The average zone concentration of arsenic was 1.10 and 0.36 mg per m^3 of air, respectively for the two zones. From the nature of the operations in which the air-borne contaminants were evolved, it seemed logical to assume that the antimony and arsenic existed as oxides. The molecular structure of arsenic trioxide was identified in the fume by x-ray diffraction analysis. No analysis is reported for antimony trioxide.

The frequency of various types of symptoms and illnesses observed each month was relatively constant, however with increasing length of employment in the smelter, there was a progressive increase in the number of severe illnesses, like nasal septal perforations, laryngitis, tracheitis and pneumonitis. The following percentage distribution of the diagnoses made during the first five months was; bronchitis (7 %), conjunctivitis (4 %), dermatitis (20 %), gastritis (3 %), gastroenteritis (5.5 %), laryngitis (11 %), neuritis (1 %), pharyngitis (8 %), pneumonotis (5.5 %), rhinitis (20 %), secondary sinusitis (1.5 %), septal perforations (8.5 %) and tracheitis (10 %). Chest x-rays taken of six men who were acutely ill from heavy exposures to smelter fumes showed definite pneumonitis extending fanwise from each hilus. Among workers who had heavy exposures to smelter fumes seemingly systemic toxic effects, such as abdominal cramps, diarrhea, vomiting, dizziness, nerve tenderness and tingling, severe headaches and prostration. Antimony was detected in urine in seven out of nine workers and arsenic was found in the urine of one worker (the number of urine samples taken

were not stated). Laryngitis, with voice changes among affected workers, ranged from hoarseness to inability of speech. The dermatitis that developed among the workers was observed in sweaty, hairy, friction areas, such as axillae, groin and back of the neck. These were nodular ulcerative lesions. The occurrence of dermatitis was sporadic and most of the cases occurred during one week in the second month of operation, presumably after the workers had been very heavily exposed to fumes.

Of the subjective symptoms reported by the workers, soreness and bleeding of the nose were experienced by more than 70 % of the workers. Sore throat, hoarseness, burning or redness of the eyes, metallic taste in the mouth, pain in the chest, headache and shortness of breath were the second most frequent complaints and were noted by about 25 % of the workers. About 10 % of the workers complained about weight loss, nausea, vomiting, diarrhoea, inability to smell properly and tightness in the chest. Less frequent complaints were spitting of blood, abnormal urination, abdominal cramps, muscle soreness, insomnia and blurred vision.

This case study indicates that occupational exposure to antimony may result in upper respiratory irritations, pulmonary inflammation, skin lesions and systemic reactions. Due to the operations of antimony it can be assumed that the antimony existed as oxides. However, the precense of diantimony trioxide was not analysed. Although exposure to arsenic trioxide also occurred, no early signs of arsenic intoxication, such as increased pigmentation of certain skin areas, keratoses of the palms and soles, loss of hair and nails, garlic odour of the breath and perspiration or swelling of ankles were observed. No detailed exposure data were presented, therefore this study is inconclusive and cannot be used for quantitative risk assessment.

Examinations, including radiography, were performed at the work place on 101 workers employed for at least 3 years in an antimony sulphide ore smelting plant in Serbia (Karajovic, 1957). The report is in German and an English summary is available. Upon chemical analysis of produced dust it was shown to contain 35-90 % Sb_2O_3, 1-6 % Sb_2O_5, 1-4 % Fe_2O_3 and 0.3-9 % As_2O_3. The working conditions were reported to be “not at all ideal”. Subjective symptoms that were reported were light respiratory difficulty, tiredness, myalgia, light coughing, and light dyspepsia without pain or diarrhoea.

Lung radiography was performed on 62 workers. 31 workers were found to have lung changes. Reported were also 22 cases of emphysema with bronchitis, 51 cases of catarrh in the upper respiratory tract, 16 cases of deviation of the nasal septum, 12 cases of conjunctivitis and 16 cases of antimony dermatosis. The antimony dermatosis was characterised by vesicular or pustular lesions with residual scars and hyperpigmentation, especially during the summer season and 13 of the cases were working near the furnace where temperatures were high.

No effects were observed on the gastrointestinal tracts, the liver, the cardiovascular system or the central or peripheral nervous system, which according to the authors indicated that there were no systemic toxic effects.

20 of the examined workers who had definite or suspected pneumoconiotic symptoms were taken to the clinic for more detailed examinations. The same subjective and objective findings as found during the first examination were also found in the selected group. eight workers were reported to have pneumoconiosis simplex, four of these had lighter lung ventilation

insufficiency. No cases of progressive pneumoconiosis were found. There were no effects on the heart and the EKG was normal.

This study indicates that exposure to diantimony trioxide may cause pneumoconiosis, emphysema, irritation to the eye, respiratory tract and the skin. However, data on exposure are lacking and no control group was included in the study. Therefore, this study cannot be used for quantitative risk assessment.

In a briefly reported study, men working at a plant near Newcastle in United Kingdom with the production of antimony oxide and the pure metal from sulphide ore by various smelting processes were radiographed (McCallum, 1963). A number (not further specified) of the workers had radiographic lung changes, resembling the simple pneumoconiosis of coal workers. These changes appeared to be symptomless and were first noticed in men who were radiographed during another investigation. Observation of these men did not show any alteration in the radiological opacities, but two men developed tuberculosis lesions, which responded promptly to chemotherapy. The only man with pneumoconiosis who had respiratory symptoms had chronic bronchitis with respiratory obstruction. Duplicate air samples were taken in different areas of the factory and mean values of antimony in air ranged between 0.53 and 5.3 mg antimony/m^3 in most of the workplace areas. Concentrations as high as 36.7 mg antimony/m^3, on an intermittent basis, were recorded in the metal tapping area.

This study supports that lung changes resembling pneumoconiosis occurs in humans occupationally exposed to diantimony trioxide. However, due to lack of specific exposure data this study cannot be used for quantitative risk assessment.

In a more comprehensive radiological investigation of workers from the plant near Newcastle in United Kingdom, 274 men were examined by using macro-radiographs to show magnified areas of the lung fields (McCallum, 1967). 26 new cases of antimony pneumoconiosis were discovered. Another 18 men with antimony pneumoconiosis were already under clinical observation. All the antimony pneumoconiosis was of the simple type. Clinical examination and lung functioning tests (not further specified) did not reveal any other harmful effects of inhalation of diantimony trioxide. Histological examination of the lungs of antimony workers (number not stated, but probably few since it was indicated that such material was scarce) suggested that there was little or no reaction to antimony dust in the lungs. Histological sections of the lungs of an antimony worker who died from carcinoma of the lung showed an accumulation of dust particles and dust laden macrophages lying in alveolar septa and in perivascular tissues without fibrosis or an inflammatory reaction.

This study supports that lung changes resembling pneumoconiosis occurs in humans occupationally exposed to diantimony trioxide. However, no exposure data were reported and no control group was included, therefore this study cannot be used for quantitative risk assessment.

Male workers in Pennsylvania, USA exposed to the dust of antimony ore and diantimony trioxide during their job of converting crude ore into diantimony trioxide were examined

(Cooper et al., 1968). The reason for doing the investigation was that fine small opacities was observed throughout both lung fields in a 33 years old, male worker who had worked for 10 years converting crude ore into diantimony trioxide. This worker also reported that the refined powder irritated his nose and caused nosebleeds at times. The number of employees at the plant since 1960 had been 34 but 6 worked only for a few months and were excluded from the examination. The ages ranged from 25-61 years and the duration of exposure ranged between 1-15 years (not further specified). The antimony concentration in air, measured at different locations was 138 mg/m^3 in the bagging area, 11-75 mg/m^3 in ten other locations, 1.0-9.8 mg/m^3 in thirteen other locations and 0.081-0.95 mg/m^3 in another thirteen locations. Data on antimony in urine are presented for all except one worker and the range is 0-1020 µg antimony/1000 ml urine.

Chest x-ray was performed on 13 workers. Three cases with antimony pneumoconiosis and five with suspicious findings were observed. In the group as a whole, very little time had been lost for sickness. No tuberculosis had been observed in the antimony workers. Pulmonary function studies, including vital capacity, lung volumes, minute ventilation, tidal volume, mixing efficiency, maximum-mid expiratory flow rates, forced expiratory volume in one second, maximum breathing capacity and diffusing capacity, were performed on 14 subjects. Arterial blood oxygen, carbon dioxide pH and plasma bicarbonate were determined at rest and after exercise. No consistent pattern of abnormalities in lung function was observed but isolated findings were noted in some (not further specified). Of those with abnormalities of pulmonary function, one had definite small opacities, one had very early changes and two had no changes in the lungs. The remaining three subjects with either suspicious or definite pneumoconiosis all had normal pulmonary function. Electrocardiograms were done on seven workers, three of whom had antimony pneumoconiosis. six workers had normal tracings and one showed a slight bradycardia

This study indicates that occupational exposure to diantimony trioxide may give rise to pneumoconiosis and isolated cases of abnormal lung function. Bradycardia and irritated and nosebleeds was also reported. Due to the lack of individual exposure data the study cannot be used for quantitative risk assessment.

The lung function and radiological characteristics was investigated in 51 male workers employed in an antimony smelting plant in Serbia (Potkonjak and Pavlovich, 1983). This study is also described in the 4.1.2.3. Irritation section. The subjects were aged between 31 to 54 years old with an average age of 45 years and had worked in the plant for 9 to 31 years. All had experienced pneumoconiotic changes. They were exposed to airborne dust containing predominantly antimony oxide [Sb_2O_3 (38.73-88.86 %), Sb_2O_5 (2.11-7.82 %), SiO_2 (0.82-4.72 %), Fe_2O_3 (0.90-3.81 %) and As_2O_3 (0.21-6.48 %)]. No information on antimony air concentration is given. Over a 25-year period they were examined 2-5 times; the evaluation included a physical examination (specialist consultations were obtained when appropriate), laboratory analysis (erythrocyte sedimentation rate, blood differential count, hemoglobin, hematocrit and urin analysis), postero-anterior chest X-ray and pulmonary function studies. Arterial blood gases were measured at rest and after exercise.

The X-rays of the 51 men showed sporadic and singly disseminated small, dense, roundish or polygonal opacities of pinhead type, diameter usually less than 1.0 mm, in 34 workers (67 %); numerous small, pinhead type opacities densely distributed in the mid-lung regions of seven

workers (14 %); a markedly high profusion of pinhead type opacities in the entire lung field of 9 workers (17 %) and a markedly high profusion of larger (1-3 mm), often irregular, opacities in the entire lung field of one worker (2 %). No case of confluent massive fibrosis was observed. Other findings included enlarged, dense hilar shadows and emphysematous changes in the upper and lower regions (34.5 %); active tuberculous lesions (i.e. non-calcified) were seen in one case and their identity was confirmed by laboratory examination; inactive tuberculosis in 18.2 %; peribronchial changes were noted in 17 workers (33 %). The pneumoconiotic changes were only seen in smelters who were exposed to the antimony oxide dust for more than 9 years (data not shown).

Clinical respiratory symptoms and signs observed were permanent breathlessness in effort in seven cases (16 %), periodical breathlessness in effort in 26 cases (50 %), coughing with expectoration (chronic bronchitis) in 19 cases (37 %), coughing without expectoration in twelve cases (24 %), wheezing in twelve cases (24 %), generalised weakness in 13 cases (26 %), chest tightness or pains in 13 cases (26 %), whistling brhonchi in twelve (24 %) and snoring, coarse brhonchi in 14 (27 %). According to the authors the prevalence of chronic coughing was markedly high whereas the other pulmonary symptoms and signs showed no particularity - they were found as often in patients with other simple pneumoconioses. Pulmonary function tests showed obstructive changes of the forced expiration volume in 17.6 %, light abnormality of airway resistance in 17.2 % and moderate to severe airway resistance in 9.1 %. Small airway obstruction as manifested by forced expiratory flow rates was recorded in 16.7 %, bronchospasms was seen in 4.4 % and hyperinflation was noted in 34.5 %. Arterial blood gases were normal during rest and after exercise; hypoxia was noted in 2 subjects only. Expired gases had normal O_2 and CO_2 concentrations. Non-homogenous alveolar ventilation was found in 31 %. Abnormal CO transfer was observed in two cases.

Conjunctivitis was seen in 14 cases (27.5 %) and upper airway inflammation in 18 cases (35.3 %). It is not clear from the report if the group with upper airway inflammation was included in the group with chronic bronchitis. “Antimony dermatosis” characterised by vesicular or pustular lesions with residual hyperpigmentation were present in 32 workers (63 %), especially during the summer season and when working near the furnace where temperatures were excessively high.

No systemic toxicity with regard to the cardiovascular, hepatic, hematopoietic, renal or central or peripheral nervous system was noted, except for musculo-skeletal complaints which were noted without any objective signs of pathology.

This study shows that pneumoconiotic changes (the frequency was not stated) in smelters can be observed after 9 years of exposure and more frequently after 10 or more years. The pneumoconiosis, which was called antimoniosis, was characterised by numerous small opacities of pinhead type, densely distributed in the middle and lower lungfields. Emphysema was observed in 34.5 % but massive fibrosis was not seen in any case. Changes in lung function were observed, however, no consistent pattern could be ascertained. Chronic coughing, chronic bronchitis, conjunctivitis and dermatitis were clinical signs but no systemic toxicity was observed.

Lesions of the respiratory tract and the lungs caused by diantimony trioxide have been reported in workers involved in antimony processing metallurgical works in Slovakia (Klucik et al., 1962). The paper is in Czech but a summary in English is available. The workers were

exposed to smoke and diantimony trioxide dust and in a limited degree to a concentration of antimony trisulphide for periods ranging from a few years (not further specified) to 28 years. The number of workers examined and their age was not stated. 99.2 % of the diantimony trioxide dust particles and 95 % of the antimony trisulphide concentrate particles were smaller than 5 μm; the average size of the diantimony trioxide particles was 1.03 μm and that of the antimony trisulphide particles 1.84 μm.

Rhinitis (54.3 %), perforation of the septa (33.2 %), pharyngitis (76.5 %), bronchitis (54.3 %), pneumoconiosis (20.8 %) and symptoms of emphysema (41.9 %) was observed. The development of pneumoconiosis stopped at the micronodular size, the nodules did not tend to confluence. As analysis of the dust did not show any content of free SiO_2, the authors presumed that pneumoconiosis was caused by diantimony trioxide. Due to unclear exposure conditions of the workers, this study is considered inconclusive and will not be used for risk characterisation.

In a briefly reported study the degree of pneumoconiosis in 72 workers, employed in an antimony plant between 6 months and 43 years, was compared with their mean period of employment and their antimony lung level (McCallum et al., 1971). Data from this study is also presented in the 4.1.2.1. Toxicokinetics section. A postero-anterior radiograph of the lungs, which was categorised according to the international classification from 1958, was available for the men. In category 1 the opacities are sporadic and singly disseminated, category 2 shows opacities in the mid-lung regions and in category 3 a markedly high profusion in the entire lung-field is involved. Antimony lung levels were measured by X-ray spectrophotometry.

An increase in radiographic category of pneumoconiosis was associated with a rise in the mean period of employment, although there was a great deal of variation. An increase in radiograph category was also associated with an increase in antimony lung levels, although there was a wide variation in the amount of antimony in the lungs in each category. No statistical calculations were performed. Due to unclear exposure and lack of statistical calculations this study will not be used for risk assessment.

The mortality and the underlying causes were investigated in workers employed in an antimony plant in England where roasting of antimony ore started in the 1920s (Jones, 1994). This study is also described in more detail in the 4.1.2.8. Carcinogenicity section. Over the year's antimony metal, antimony alloys and diantimony trioxide were produced in the plant. Until the early 1970s considerable quantities of lead alloys were made, containing as much as 80 % lead and 10 % arsenic. All production of antimony metal and its alloys stopped in 1973 and after that only diantimony trioxide was manufactured. Since the 1960s the bulk of ore used was a sulphide ore containing about 60 % of antimony and up to 0.5 % of arsenic. Arsenic metal and its trioxide were also brought into the plant to make arsenic alloys.

No exposure data were presented but due to the description of the work place it can be assumed that variable occupational exposure to lead, metallic antimony, metallic arsenic, diantimony trioxide, arsenic trioxide and polycyclic aromatic hydrocarbons did occur. All men employed between 1961-1992 and with at least three months of employment were

recruited into the survey. Of the 1452 men that were recruited, 32 were not traced. Of the 1420 men that were traced 357 had died and 29 emigrated by 1992.

The workers were subdivided into four occupational groups: a) antimony workers, b) maintenance workers, c) zircon workers and d) others (including office workers and management staff).

Expected death rates were calculated based on local rates of Tyneside conurbation 1961-73 and Tyne and Wear 1974-83. Man-years at risk were calculated for the population in separate decennial age periods for each year from 1st of January 1961-31st of December 1983. Appropriate age specific death rates were then applied to calculate the number of deaths expected for each cause of death considered. The cause of deaths were divided as follows; lung cancer, stomach cancer, other neoplasms, circulatory disease, ischaemic heart disease, respiratory disease, genitourinary disease, accident and suicides and others. The observed numbers of deaths for each cause were compared with the expected figures calculated as above.

Except for increase in mortality from lung cancer no effects were observed in mortality from other causes.

No effect on mortality from circulatory disease, ischaemic heart disease, respiratory disease, genitourinary disease, accident and suicides and others was observed in this study. However, there were no exposure data reported and thus this study cannot be used for any quantitative risk assessment.

Dermal

No human data on repeated dermal exposure with diantimony trioxidehave been located.

Oral

No human data on repeated oral exposure with diantimony trioxide have been located.

In vitro studies

No data on repeated exposure with diantimony trioxidehave been located.

4.1.2.6.3 Summary of repeated dose toxicity

The majority (9 out of 14) of the repeated dose studies in animals cited above are considered inconclusive because they do not comply with current test guidelines, they lack essential information regarding exposure conditions and statistical evaluations of the results or both control and exposed animals showed signs of non-treatment related illness. Still, there are inhalation studies indicating that diantimony trioxide is toxic to lung (Newton et al., 1994;

Watt, 1983; Groth et al., 1986a; MPI, 2003). A NOAEC of 0.51 mg/m^3 is derived from the study by Newton et al. (1994), based on impaired lung clearance observed at 4.50 mg/m^3. Although there might be some uncertainty regarding the accuracy of the LOAEC and NOAEC numerical values, as the study had a high background incidence of lung inflammation in control animals, the NOAEC of 0.51 mg/m^3 is brought forward to the risk characterisation.. Furthermore, acute pneumonia has been reported in a reproductive toxicity inhalation study performed by (MPI, 2003) supporting adverse lung effects at exposure levels ≥ 2.6 mg/m^3.

Two repeated dose oral studies suggest that diantimony trioxide may be toxic to the liver. This is based on a 10 % increase in liver weight and significantly elevated ASAT values in one study (Hext et al., 1999), supported by significantly elevated ASAT levels observed in another study (Sunagawa, 1981). However, in the absence of histological change or any clinical signs of antimony intoxication to support that the liver findings are adverse, the findings are regarded as adaptive or incidental to treatment with diantimony trioxide. A NOAEL of 1686 mg/kg/d for liver toxicity is suggested.

No studies on repeated dermal exposure have been located.

In humans, all the data comes from case report studies on workers employed in industries manufacturing diantimony trioxide. The workers were probably subjected to inhalation, dermal and oral exposure, although it can be assumed that exposure via inhalation was the dominating route. Although no detailed data on exposure levels were presented and consequently no NOEC or LOEC values can be derived from these studies they indicate that repeated inhalation exposure to diantimony trioxide may cause pulmonary inflammation, lung emphysema and pneumoconiosis. Only isolated cases of changes in lung function were reported. The irritation observed on skin, in eyes and in the respiratory tract is considered in the 4.1.2.3 section of Irritation.

4.1.2.7 Mutagenicity

4.1.2.7.1 Studies *in vitro*

Tests in bacteria

The potential of antimony trioxide to induce point mutations was measured in the *Salmonella*/ microsome assay using both the pre-incubation (60 min) and plate incorporation protocol (Elliot et al., 1998). The study was performed according to OECD Guideline 471 and GLP. The tester strains used were *Salmonella* TA 1535, TA 1537, TA 100, TA 98 and two E. coli strains WP2PuvrA and WP2P. The antimony trioxide was 99.9 % pure and dissolved in DMSO. Appropriate positive controls were included. The doses tested were 100, 200, 500, 1000, 2500 and 5000 µg/plate, with and without metabolic activating system. No statistics were performed to evaluate the result. A 3.7 fold increase in the number of *E. coli* WP2PuvrA

colonies could be seen at the dose of 100 µg/plate with S9, compared to control. The increase was not commented on by the authors; however, the increase could not be repeated in the pre-incubation assay. With this exception, antimony trioxide did not induce any increases in the number of revertant colonies in the *S. typhimurium* or the *E. coli* strains in the two independent assays.

Antimony trioxide (99.999 % pure) was tested in the *Bacillus subtilis* Rec assay using strains H17 (Rec^+) and M45 (Rec^-) (Kuroda et al., 1991). The test substance was dissolved in distilled water and administered on paper disks with a diameter of 8 mm placed on *B. subtilis* pretreated agar petri plates. Antimony trioxide was tested at 0.3, 0.6 and 1.1 µg/disk. The treatment resulted in a difference in the diameter of the killing zones in the spores from B. subtilis larger than 4 mm. Antimony trioxide was classified as positive in the rec assay. In the same publication a *Salmonella* mutation assay was also conducted with strains TA 100 and TA98 using the preincubation protocol (20 min), in the presence and absence of rat liver S9 mix. Positive controls were included. The highest dose tested was 1.71 µg/plate for antimony trioxide. No statistical method was used to evaluate the result. There was no indication of mutagenicity in the *Salmonella* assay with strains TA100 and TA 98.

In a study where the genotoxicity of 127 metal compounds were evaluated, antimony trioxide was tested in the *Bacillus subtilis* Rec assay (Kanematsu et al., 1980). Cultures of the two strains H17 and M45 were streaked radially onto agarplates where 0.05 ml of metal solution (0.05 M) were dropped onto filter paper disks, resulting in a dose of 729 µg/disk for antimony trioxide. The antimony trioxide was dissolved in distilled water and the purity was of "the highest purity commersially available". No positive controls or metabolic activation system were included. A cold incubation protocol was used, where the treated plates were incubated at 4°C for 24 hours and then incubated at 37°C overnight. The difference in the inhibition length was measured between the recombination deficient and competent strain and a difference of more than 4 mm was judged as positive. The treatment with antimony trioxide resulted in a difference of 5 mm and was thereby classified as positive. In addition the same authors performed a spot test with the filterpaper technique described above whith five strains of *Salmonella* (TA 98, 100, 1535, 1537 and 1538) and two strains of *E. coli* (B/r WP2 try^- and WP2 hcr^- try^-). The same concentration as in the Rec assay was used. Antimony trioxide did not show any mutagenicity in the spot test.

Tests in mammalian cells

The genotoxicity of antimony trioxide has also been tested in mammalian cells (Elliot et al., 1998). The Mouse Lymphoma L5178Y TK(±) mutation assay was performed according to OECD guideline 476 and GLP. Antimony trioxide with a purity of 99.9 % was dissolved in DMSO. According to the publication referred to by the authors, the assay was performed with the microwell protocol, rather than the soft agar protocol. Duplicate cell cultures in medium were treated with test material for 4 hours in the presence and absence of S9-mix and were allowed to have an expression period of 48 hours. Negative and positive controls were included. The highest dose tested was limited by the solubility of antimony trioxide to 50 µg/ml (172 µM), where precipitation of the test substance could be seen. No significant toxicity could be seen at any dose level either with or without metabolic activation. The number of mutant colonies and the survival was determined on day 10-12 after growth in selective (trifluorothymidine, TFT) and non-selective medium, respectively. The mutant

colonies were counted by eye and small colonies (more likely associated with gross structural chromosome aberrations) and large colonies (resulting from gene mutations) were differentiated, but not separately shown, which makes it difficult to draw any conclusion concerning the possible chromosome aberration inducing potential of antimony trioxide in this test. The assay was conducted as two independent assays. The data were examined by the Study Statistician who, in colsultation with the Study Director, considered that a statistical analysis was not necessary, which is otherwise recommended by the UKEMS guidelines, when the microwell protocol is used (Cole et al., 1990). No increase in the number of mutant colonies could be seen with antimony trioxide compared to control in the Mouse Lymphoma L5178Y TK(±) mutation assay under any of the test conditions used.

Table 4-50 Summary of *in vitro* mutagenicity data on antimony

Assay	Cells/Strain	Dose/Concentration	Result	Ref.
Bacterial mutation / Ames test	*S. typhimurium* TA98, TA100, TA1535, TA1537; *E. coli* WP2, WP2P*uvr*A	100, 200, 500, 1000, 2500, 5000 µg/plate antimony trioxide	Negative	(Elliot et al., 1998)
Bacterial mutation / Ames test	*S. typhimurium* TA98, TA100	0.43, 0.86, 1.71 µg/plate antimony trioxide	Negative	(Kuroda et al., 1991)
Bacterial mutation / Spot test	*S. typhimurium* TA98, TA100, TA1535, TA1537, TA1538; *E. coli* WP2, B/r WP2	2.5 µmol (729 µg)/disk antimony trioxide	Negative	(Kanematsu et al., 1980)
Bacterial DNA repair / Rec assay	*B. subtilis* H17 and M45	0.3, 0.6, 1.1 µg/disk antimony trioxide	Positive	(Kuroda et al., 1991)
Bacterial DNA repair / Rec assay	*B. subtilis* H17 and M45	2.5 µmol (729 µg)/disk antimony trioxide	Positive	(Kanematsu et al., 1980)
Mammalian mutation	Mouse lymphoma L5178Y TK(±)	6, 13, 25, 50 µg/ml (21, 45, 86, 172 µM) antimony trioxide	Negative	(Elliot et al., 1998)
In vitro cytogenetic	Human lymphocytes	10, 50, 100 µg/ml antimony trioxide	Positive	(Elliot et al., 1998)

Assay	Cells/Strain	Dose/Concentration	Result	Ref.
In vitro SCE	Human lymphocytes	0.1, 0.5, 1, 2, 5 µM (0.03, 0.15, 0.29, 0.58, 1.5 µg/ml) antimony trioxide	Positive	(Gebel, 1997)
In vitro SCE	V79 Chinese hamster cells	0.09, 0.17, 0.34 µg/ml antimony trioxide	Positive	(Kuroda et al., 1991)

An *in vitro* cytogenetic assay was conducted according to OECD Guideline 473 in human lymphocytes isolated from two different donors with and without metabolic activation (Elliot et al., 1998). Antimony trioxide (99.9 % pure) was dissolved in DMSO and 10, 50, and 100 µg/ml (final concentrations) was added to duplicate cultures of cells, which had been stimulated with phytohemagglutinin (PHA) for 48 hours. The highest dose was limited to the solubility of the test compound. Positive controls were included, both with and without metabolic activation. For the cultures containing S9-mix the test substance was removed after 3 hours of treatment and the cells given fresh media. In the absence of S9-mix the test substance were left in the cultures until harvest, except for the cultures taken at 92 hours, which had a medium change after 72 hours. Colcemid was added to all the cultures 2 hours before harvest for preparation of metaphases on slides, which was performed at 68 and 92 hours after culture initiation. The slides were read blindly and 100 cells per culture were scored for metaphases. Chromosomal aberrations as well as polyploidy and endoreduplication were recorded. There was no evidence of cytotoxicity measured by the mean mitotic index for the cultures treated with antimony trioxide compared to control. The results were evaluated by statistical analysis using Fischer´s one-sided exact test. No increase in the number of polyploidy and endoreduplicated cells was noted. In lymphocytes from donor 1, a statistically significant dose dependent increase in the percentage aberrant cells (excluding cells with only gap-type aberrations) were seen at the 68 hour sampling time in cultures treated with antimony trioxide in the presence of S9-mix ($p < 0.05$ at 50 µg/ml and $p < 0.01$ at 100 µg/ml). No data from the 92-h sampling time is reported for this donor. In lymphocytes from donor 2, a statistically significant increase in the percentage aberrant cells (excluding cells with only gap-type aberrations) was seen at the 68 hour sampling time in cultures treated with 100 µg antimony trioxide/ml with and without S9-mix ($p < 0.01$). At the 92 hour sampling time, a statistically significant increase was seen at 100 µg/ml of antimony trioxide without S9-mix ($p < 0.05$).

The potential of diantimony trioxide to induce sister chromatid exchanges (SCE) in vitro has been evaluated in human lymphocytes, collected from healthy non-smoking donors, 25-35 years of age (Gebel, 1997). The cells were stimulated with PHA and after 24 h 5-bromo-2´-deoxyuridine (BrdU) was added for 24 h. Thereafter the cultures were treated with antimony trioxide at final concentrations of 0, 0.1, 0.5, 1, 2 and 5 µM, corresponding to 0, 0.03, 0.15, 0.29, 0.58 and 1.5 µg/ml, for 24 h. In total, lymphocytes were cultured for 72 h at 37 °C. Diantimony trioxide (p.a. grade) was dissolved in distilled water and tested in concentrations

up to a cytotoxic response in the culture, determined by the absence of dividing cells. No positive controls were included. Colcemid was added 2 hours prior to harvest. Slides were prepared and coded. A total of 30 metaphases from each culture were scored for SCE. One hundred metaphases per slide were scored to determine cell proliferation. The results were statistically analysed in the two-sided Student´s t-test. Diantimony trioxide induced a significant dose-dependent increase in the number of SCEs in lymphocytes *in vitro* from a minimum dose of 0.5 μM (0 μM = 8.6 SCE/ metaphase, 0.1 μM = 10.0, 0.5 μM = 11.5 ($p < 0.05$), 1 μM = 14.7 ($p < 0.001$), 2 μM = 25.2, 5 μM = cytotoxic). The potential of antimony trioxide (99.999 % pure) to induce SCE in V79 Chinese hamster cells was evaluated by Kuroda and co-workers (1991). Saturated solutions were made in distilled water. Cells were cultured in plastic petri dishes for 24 hours, and then diantimony trioxide (0.09, 0.17 or 0.34 μg/ml final concentration) was added together with BrdU and incubated for 28 hours in the dark. Colcemid was added for the last 2 hours. Positive controls were used. SCEs were scored in 20 stained methaphases/dose. The experiment was repeated once. The statistical evaluation was done by Student´s t-test. A statistically significant increase in the number of SCEs/metaphase was obtained. The significant increase could be seen from the concentration of 0.09 μg/ml. No cytotoxicity was reported at the final concentration of 0.34 μg/ml.

Conclusion – *In vitro* studies

Negative results were obtained with antimony trioxide in two Ames tests and in a further study using the "spot" technique. Negative results were also obtained in the Mouse Lymphoma TK± assay. Positive results were only obtained in two rec± assays for DNA damage, but it is difficult to draw conclusions on the reliability of this result on the basis of the details provided. One of the Ames assays and the Mouse Lymphoma TK± assay were performed according to OECD guidelines. Considering the results obtained in these studies it can be concluded that antimony trioxide does not induce gene mutations *in vitro.*

The induction of chromosome aberrations of antimony trioxide was investigated in one study, which was positive and performed according to OECD guidelines. The induction of SCE of antimony trioxide has been investigated in two studies, which were positive. It is concluded that antimony trioxide has the potential to induce structural chromosome aberrations in mammalian cells in vitro.

4.1.2.7.2 Studies *in vivo*

A bone marrow micronucleus assay was performed according to OECD Guideline 474 and GLP after both single and repeated dose exposure of CD-1 mice (Elliot et al., 1998). In the single dose study, 5 male and 5 female mice were each given a single oral gavage dose of 5000 mg/kg antimony trioxide (99.9 % pure) in hydroxypropylmethylcellulose/aqueous polysorbate 80. Cyclophosphamide was used as positive control. Sampling times were 24 and 48 hours post dosing. In the repeated dose study, oral gavage doses of 400, 667 and 1000 mg/kg were given daily to 5 males/ dose and time point for 7, 14 and 21 days. Sampling was performed the day after the respective last dose was given. Bone marrow was sampled from femurs and stained with polychrome methylene blue and eosin. Slides were scored blindly.

Two thousand polychromatic erythrocytes were examined for micronuclei per animal. No toxicity in animals treated with antimony trioxide was observed in the study. A significant decrease in the percent polychromatic erythrocytes was only seen in females at the 24 h sampling time in the single dose study. The statistical analysis consisted of a one-sided Student´s t-test on transformed data. No statistically significant increase in the incidence of micronuclei was observed in the single or repeated dose study.

A chromosomal aberrations test in mouse bone marrow was performed after both single and repeated dose exposure of antimony trioxide to Swiss albino mice (Gurnani et al., 1992). The purity of the antimony trioxide was not stated in the publication. However, according to personal communication with one of the authors the antimony trioxide was of analytical grade and purchased from Merck India. It has been verified by Merck that the only antimony trioxide that has been sold by Merck in India is of a minimum of 99% purity. Therefore it could not be verified, but anticipated that the antimony trioxide used in this study was of acceptable purity. In the single dose study, antimony trioxide was given orally by gavage as an aqueous suspension at 400, 667 or 1000 mg/kg bw to each of 5 male and 5 female mice per dose. In the repeated dose study, 5 male mice per dose were each administered the same doses of antimony trioxide daily for 21 days. Sampling times were 6, 12, 18 and 24 hours after exposure for the single dose study and on days 7, 14 and 21 of the repeated dose study. In the high dose group, 1000 mg/kg bw, the animals died on day 20 of treatment. This is in contrast with the Elliot et al. (1998) and the Kirkland et al. (2007) studies, where no lethality was seen at the same doses of antimony trioxide. Due to the lethality no chromosomal aberrations were analysed in the high dose group at day 21. No information on the lethality on day 20 was given in the paper and no further information has been obtained after personal communication with one of the authors. No positive control was used. All animals were injected intraperitoneally with colchicine, 4 mg/kg, 1.5 hours before sacrifice when bone marrow of femurs was removed, fixated and stained with Giemsa. A total of 100 methaphase plates from each animal were scored for chromosomal aberrations. Data from the single dose study was analyzed by t-test and a two-way analysis of variance. The results from the repeated dose study were analyzed by a one-tailed trend test, two-way ANOVA followed by Duncan´s multiple range test. In the single dose study, no statistically significant difference in the number of chromosomal aberrations could be recorded between the treated and the control mice of either sex. In the repeated dose study the frequency of aberrations without gaps increased proportionally with the dose to a significance of $P < 0.001$ in the trend test for the first 14 days (the highest dose was lethal on day 20 of treatment), giving a 6-7 fold increase in the number of aberrations at the highest dose. The frequencies of chromosomal aberrations were significantly increased also in the 400 and 667 mg/kg bw groups at the $p < 0.05$ level.

The same data was also published elsewhere (Gurnani et al., 1993). As only one chromosomal aberration study was performed, only the (Gurnani et al., 1992) study is referred to in this RAR. Strangely, the animals in the high-dose group which died on day 20 were evaluated for aberrations after sacrifice on day 21 in the 1993 publication. Due to the unexplained lethality in the high-dose group and the discrepancy between the two publications, the Gurnani study results are considered questionable and will not be used for risk assessment.

Both micronuclei and chromosome aberrations were evaluated in the bone marrow of Sprague Dawley rats following a 21 day oral repeated exposure to antimony trioxide (Covance Laboratories Ltd, 2005). The same data were also published by Kirkland et al. (2007). The dose levels were 250, 500 and 1000mg/kg/day and based on the previous performed work by Gurnani et al. (1992) and Elliot et al. (1998) and therefore no range-finder experiment was performed. The study was performed according to GLP and OECD guidelines 474 and 475. The antimony trioxide (purity 99.93 %) was formulated in 0.5 % (w/v) hydroxypropylmethylcellulose and 0.1 % (w/v) aqueous polysorbate (0.5 % HPMC and 0.1 % polysorbate) and administered by gavage approximately every 24 hours. The positive control, cyclophosphamide, was administered on a single occasion. Both male and female rats were used in groups of six. The same animals were used both for metaphase processing and micronucleus preparations. No clinical signs of toxicity were observed in any of the dose groups. Approximately two hours prior to the scheduled sample time, animals were injected with colchicine and the animals were sampled 24 hours after the final administration of antimony trioxide. The slides for the micronucleus preparation were stained with acridine orange and the slides for methaphase chromosome spreads were stained with Giemsa. The slides were scored blindly. For the micronucleus data and the chromosome aberration data a heterogeneity test was performed, but additional statistical analysis was not considered necessary. Antimony trioxide did not induce chromosome aberrations or micronuclei in the bone marrow cells of male or female rats at any of the doses tested. However, mitotic index data and data on the percentage of polychromatic erythrocytes showed no evidence of bone-marrow toxicity in antimony trioxide-treated animals. A toxicokinetic study conducted separately by TNO (2005), in which dosing was similar to that of the Covance study (Covance Laboratories Ltd, 2005; Kirkland et al., 2007) established that the oral absorption of antimony trioxide is low. Although this study showed that antimony trioxide reaches the bone marrow, and that the concentration of antimony trioxide was highest in the bone marrow compared to the concentrations in other tissues, it should be noted that the absorption of antimony trioxide, when administered as an oral suspension at a dose of 1000 mg/kg bw, as in the studies above, is only 0.05 %. This means that, despite the high doses that were administrated, the bone marrow may not have been sufficiently exposed.

An in vivo rat liver DNA repair, unscheduled DNA synthesis (UDS), assay, conducted according to the OECD Guideline 486 and GLP, have been performed following single oral administration of antimony trioxide to male Alderley Park AlPk:ApfSD rats (Elliot et al., 1998). Five rats per dose were each given a single oral gavage dose of antimony trioxide (99.9 % pure) at 3200 and 5000 mg/kg in hydroxypropylmethylcellulose/aqueous polysorbate 80. 1,2-dimethylhydrazine (DMH) served as positive control. At 2 or 16 hours after administration hepatocytes were isolated following collagenase perfusion and incubated with ^{3}H-thymindine for 4 hours followed by a cold chase overnight. The slides were coated in Ilford K2 emulsion and left for 14 days at 4 °C before developing. Slides were coded and scored blind. An image analysis system was used to score the nuclear and cytoplasmatic grain, assessing 60 cells per animal. The treated animals showed no signs of toxicity. There was no increase in net nuclear grains or percentage of cells in repair at either sampling time.

Table 4-51 Summary: genotoxicity in vivo.

Assay	Dose	Result	Ref.

Assay	Dose	Result	Ref.
Mouse bone marrow micronucleus	Single, gavage: 5000 mg/kg antimony trioxide	Negative	(Elliot et al., 1998)
Mouse bone marrow chromosomal aberration	Single, gavage: 400, 667, 1000 mg/kg antimony trioxide	Negative	(Gurnani et al., 1992)
Mouse bone marrow micronucleus	Repeated, gavage: 400, 667, 1000 mg/kg/day antimony trioxide; up to 21 days	Negative	(Elliot et al., 1998)
Mouse bone marrow chromosomal aberration	Repeated, gavage: 400, 667, 1000 mg/kg/day antimony trioxide; up to 21 days	Positive	(Gurnani et al., 1992)
Rat liver UDS	Single, gavage: 3200, 5000 mg/kg antimony trioxide	Negative	(Elliot et al., 1998)
Rat bone marrow chromosomal aberration	Repeated, gavage: 250, 500, 1000 mg/kg/day antimony trioxide; 21 days	Negative	(Covance Laboratories Ltd, 2005), (Kirkland et al., 2007)
Rat bone marrow micronucleus	Repeated, gavage: 250, 500, 1000 mg/kg/day antimony trioxide; 21 days	Negative	(Covance Laboratories Ltd, 2005), (Kirkland et al., 2007)

Conclusion – *In vivo* studies

Four *in vivo* mutagenicity studies are available for diantimony trioxide. One micronucleus study on mouse bone marrow, performed in agreement with OECD guidelines and GLP, gave negative results after both single and repeated exposure (Elliot et al., 1998). Another study, performed according to OECD guidelines and GLP, where both micronuclei and chromosome aberrations were evaluated in rat, males and females, negative results were obtained after repeated oral dosing with antimony trioxide (Covance Laboratories Ltd, 2005; Kirkland et al., 2007). A rat liver UDS performed according to OECD guidelines did not show any evidence of DNA damaging capacity (Elliot et al., 1998). A mouse bone marrow chromosomal aberration test using a single and a repeated oral dose protocol with antimony trioxide was negative with the single oral dose protocol, but positive with the repeated oral exposure using only male mice (Gurnani et al., 1992). However, due to lethality at the highest dose and unclear reporting of the study, this study is regarded questionable and will not be used for the risk assessment. Consequently, negative results were obtained using OECD test guideline protocols and according to GLP and using two different species – mouse and rat. Therefore, it can be concluded that diantimony trioxide does not cause systemic mutagenicity after oral administration . It should be noted that the absorption of antimony trioxide is only 0.05 % (see section 4.1.2.1), when administered as an oral suspension at a dose of 1000 mg/kg bw, as in the studies above. This means that, despite the high doses that were administrated, the bone marrow may only have been marginally exposed and thus, it is not possible to conclude whether the negative *in vivo* results are due to lack of mutagenic potential or due to inability of the test to detect a mutagenic effect at the low concentrations achieved in the bone marrow following oral exposure. Therefore, it is not possible to conclude whether the results are relevant also for the situation in the lung after inhalation exposure, which is the specific site of contact tissue and the site where tumours have been found in the carcinogenicity studies. However, the *in vivo* data might suggest that a possible mutagenic potency of diantimony trioxide would be low and it is believed that a possible local genotoxic effect of diantimony trioxide would only be biologically relevant at concentration levels that also cause particle overload. Therefore, there is also no concern for local mutagenicity of diantimony trioxide.

Mechanistic data

Toxic metal ions lead to diverse types of damage to different cellular constituents, including DNA, mainly due to competition with essential metal ions. Regarding the genotoxicity of metal ions, two mechanisms seem to be predominant, the generation of oxidative DNA damage and the interference with DNA repair and DNA replication processes (Hartwig, 1995). The mechanism behind the genotoxicity seen with antimony compounds is unknown, but a few studies have been performed to investigate the mechanism behind the possible genotoxicity of antimony compounds.

Schaumlöffel and Gebel (1998) conducted a single cell gel test (Comet assay) in vitro where a significant increase in DNA fragments could be seen from 5 to 50 μM (1.1 and 11 μg/ml, respectively) of antimony trichloride in isolated lymphocytes, indicating a capacity of antimony trichloride to cause DNA strandbreaks. The authors also investigated the effect of addition of proteinase K to antimony trichloride treated lymphocytes in the single cell gel test. Treatment with proteinase K results in the loss of DNA-protein cross-links. No difference in the number of DNA fragments from antimony trichloride treated lymphocytes could be seen with or without proteinase K, indicating that antimony trioxide might have a low capability to induce DNA-protein crosslinks. In addition, the number of micronuclei formed in human lymphocytes in vitro could not be suppressed by co-incubation with superoxide dismutase or catalase. This suggests that induction of oxidative stress may not be a crucial step in the mechanism of DNA damage induction by antimony.

An antimony (III) containing drug for the treatment of leishmaniasis, potassium antimonyl tartrate, was able to induce cell death associated with DNA fragmentation in axenic amastigotes, an intracellular stage of the parasite Leishmania infantum, at 10 μg/ml in vitro. This was shown by DNA fragmentation analysis such as cytofluorometry, electrophoresis and colorimetric detection in situ (Sereno et al., 2001). These results suggest that trivalent antimonials are able to induce fragmentation of the DNA of the parasite.

Conclusion – Mechanistic studies

The data is too inadequate to draw any conclusion about the mechanism behind the possible genotoxicity of antimony compounds.

Human data

The genotoxicity of antimony trioxide in occupationally exposed workers was assesed by the sister chromatid exchange (SCE) and micronucleus (MN) tests and the enzyme (Fpg)-modified comet assay (Cavallo et al., 2002). The latter method detects and converts oxidized bases (apurinic sites) to strand breaks. The study consisted of 23 male workers (age 41.7 ± 10.1 years) that were occupationally exposed to low doses of antimonytrioxide in an industrial plant producing fireproof textiles for car upholstery and a control group of 23 healthy non-exposed males (age 42.8 ± 9.8 years) were studied. The exposed workers were divided into two groups; one high exposure group, A (n = 17), comprised of finishing and intermediate

inspection operators who handled a mixture containing antimony trioxide while the low exposure group, B (n = 6), were jet operators not directly exposed to antimony trioxide.

Once a week, the finishing operators prepare the antimony trioxide-based flame/fire retardant suspension by mixing the fire-proofing and binding ingredients (not specified). This mixture is then pumped automatically into a tank and used to impregnate the fabric. On this day they are assisted by the intermediate inspection operators who check the product after finishing with the flame retardant and sometimes sample the textile to ensure that the correct amount of the retardant has been applied to the material. The jet operators (Group B) work approximately 20 meters away from the finishing plant. Their task is to dye the raw textiles. The control group was matched with the exposed study groups for age, smoking habits and the use of drugs, the latter was lately omitted from the statistical analysis. Airborne antimony trioxide was monitored by personal samplers (n = 41) in conformity with ISO 7708 from Monday through Friday. The amount of antimony was determined by atomic absorption spectrometry. The antimony trioxide-containing flame retardant was applied on Thursday. The mean air concentration of antimony, expressed as elemental antimony/m^3 of air inhaled was 0.12 ± 0.11 µg/m^3 for Group A and 0.052 ± 0.038 µg/m^3 for Group B.The difference was statistically significant. (Student`s t-test, P=0.02). The mean concentration is lower than the occupational limit values (generally 0.5 mg/m^3). Air sampling was not performed in the control group.

Blood was obtained for the different analysis by venipuncture from which lymphocyte cultures were prepared. For the SCE assay the lymphocytes were cultured in the dark for 48 hours with BrdU and metaphases were blocked during the last 2 hours with colcemid. After hypotonic treatment the cells were fixed and stained with Giemsa. The slides were coded and 52 metaphases were red. The cells for the micronucleus assay were treated in the same way except for a 28 hour incubation with cytochalasin B instead of BudR and no addition of colcemid. About 1000 binucleated cells were examined for micronuclei. The mean SCE and MN did not differ significantly beween the three groups (A, B and control), analysed by one-way ANOVA and Dunnett´s test.

In the Fpg enzyme modified comet assay whole blood were mixed with melting agarose (with and without Fpg enzyme) and administered on blond gel films and allowed to solidify with coverslips on top. The glass slides were put in lysis solution and washed. Electrophoresis was performed to allow the DNA fragments to migrate which were later stained with ethidium bromide and examined by eye under a fluorescent microscope. Images of 50 randomly selected cells were examined from each sample. The tail moments (TM) in the presence and absence of enzyme values were compared using Student´s t-test.The TM for enzyme treated and non treated cells were compared subject by subject and considered to have an oxidative damage when the two values differed significantly (P<0.01). In the high dose (group A), 11 subjects had significant levels of oxidative damage versus 5.5 expected. After adjustment of confounding factors the measure of risk for Group A subjects compared to controls was 14.2 (P=0.002). No other significant findings could be seen. In this study the “high exposure group” is exposed to air concentrations that is almost equivalent to ambient air concentrations and no monitoring of the non-exposed control workers were performed for comparison. In addition, the workers are likely exposed to a series of chemicals which has not been identified.

Conclusion – Human study

In the human study the genotoxicity of antimony trioxide in lymphocytes from occupationally exposed workers was assessed. No induction of micronuclei or sister chromatid exchanges could be seen between the two exposed groups and the unexposed control. In an enzyme-modified comet assay a significantly higher proportion of the workers in the "high exposure group" showed oxidative DNA damage in their lymphocytes compared to control. However, the workers were exposed to diverse chemicals and no monitoring was performed on the control group, therefore a correlation between the oxidative DNA damage and air concentration of antimony trioxide is uncertain.

4.1.2.7.3 Summary of mutagenicity

Considering the available genotoxicity data, antimony trioxide does not induce gene mutations *in vitro,* but do induce structural chromosome aberrations in cultured mammalian cells *in vitro.* Negative *in vivo* results on chromosome abberations and micronuclei were obtained in two different species – mouse (Elliot et al., 1998) and rat (Covance Laboratories Ltd, 2005), (Kirkland et al., 2007). An *in vivo* UDS assay in rats was also negative (Elliot et al., 1998). The tests were performed according to GLP and using OECD test guideline protocols and oral administration. However, according to toxicokinetik studies, the absorption of a particle suspension of diantimony trioxide after oral exposure is only 0.05 % at the dose of 1000 mg/kg used in these mutagenicity studies, indicating that, despite the high doses that were administrated, the bone marrow may only have been marginally exposed. Therefore, it is not possible to conclude whether the negative *in vivo* results are due to lack of mutagenic potential or due to inability of the test to detect a mutagenic effect at the low concentrations achieved in the bone marrow following oral exposure. Still, it can be concluded that diantimony trioxide does not cause systemic mutagenicity *in vivo* after oral administration. However, it is not possible to conclude on mutagenicity in specific site of contact tissues (local mutagenicity) and thus, whether the result is relevant for the situation in the lung after inhalation exposure, which is the site where tumours have been found in the carcinogenicity studies. However, the *in vivo* data might suggest that a possible mutagenic potency of diantimony trioxide would be low and it is believed that a possible local genotoxic effect of diantimony trioxide would only be biologically relevant at concentration levels that also cause particle overload. Therefore, there is also no concern for local mutagenicity of diantimony trioxide. In humans no induction of micronuclei or sister chromatid exchanges could be seen in lymphocytes from workers occupationally exposed to antimony trioxide, but a higher proportion of the workers in the "high exposure group" showed oxidative DNA damage in their lymphocytes. However, "high exposure" is in the range of ambient air exposure, i.e. less than 0.001 mg/m^3, the workers were exposed to diverse chemicals and no monitoring was performed on the control group, therefore a correlation between the oxidative DNA damage and air concentration of antimony trioxide is uncertain.

4.1.2.8 Carcinogenicity

4.1.2.8.1 Studies in animals

In vivo studies

Inhalation

Three chronic toxicity/carcinogenicity studies are available where the carcinogenicity of inhaled antimony trioxide in rats has been evaluated (Watt, 1983; Groth et al., 1986a; Newton et al., 1994). The exposure duration in all three studies is 12 months and thus all studies deviates from the OECD guideline on chronic toxicity/carcinogenicity, which suggests an exposure period of 24 months in rats. It should be noted that two of the studies (Groth et al., 1986a; Newton et al., 1994) might reflect sub-chronic disease among the animals, and that one study (Watt, 1983) only included females.

In the first study, a chronic inhalation toxicity study, the carcinogenicity of antimony trioxide was investigated in female CDF Fisher rats (Watt, 1983). This study is also reported in chapter 4.1.2.6. Repeated dose toxicity. 148 female rats from the Charles River Laboratories, 14 weeks of age, were divided into three groups (the number per group was not specified) and exposed to 0, 1.9 ± 1.8 and 5.0 ± 3.8 mg diantimony trioxide/m^3 for 6 h/day, 5 days/week for one year in whole body exposure chambers. Since the exposure concentrations were reported as Sb, 1.6 ± 1.5 and 4.2 ± 3.2 mg Sb/m^3, respectively, the corresponding values as Sb_2O_3/m^3 have been calculated by the Rapporteur. Air samples were taken within the exposure chambers at the same level as the suspended rat cages. It should be noted that air samples were not taken in the cages. The antimony trioxide used was 99.4% pure with arsenic (0.02%) and lead (0.2%) as the major contaminants. Only particles with mean aerodynamic diameter of 15 μm or less would pass into the chamber. The particle size (Ferret's diameter) was 0.44 and 0.40 μm for the low and high concentrations (GSD 2.23 and 2.13, respectively). Control animals were moved to other chambers during each exposure session but no more information regarding the conditions of those chambers are given by the author. Surviving animals were kept up to 15 months post-exposure for observation.

Prior to and after approximately 3, 6, 9 and 12 months of exposure, and 2 to 12 and 12 to 15 months post-exposure, animals were sacrificed and evaluated for evidence of toxicity. At sacrifice, the heart, lung, liver, spleen and kidney were weighed and fixed in 10 % buffered formalin for subsequent light microscopic examination. Blood samples were taken for analyses of differential count, red and white blood cell count, hemoglobin, hematocrit, mean corpuscular volume, mean corpuscular hemoglobin and mean corpuscular hemoglobin volume, serum enzymes and chemistry. Animals were weighed periodically throughout the exposure period.

No exposure-related effects on survival, hematology, or clinical chemistry were noted. The body weights of the exposed animals were significantly but reversibly higher than the controls; in the low-dose group during most of the exposure period (range of % increase in body weight: 5.5-10.5) but not by the end of the exposure period (% increase in body weight: 3.7) or post-exposure (range of % increase in body weight: 2.5-7.3), whereas high-dose

animals had significantly higher body weights pre-exposure (range of % increase in body weight: 2.1-7.8) and during exposure (range of % increase in body weight: 1.9-6.3, but not post-exposure (range of % increase in body weight: 3.9-5.6). Since some of the weight differences occurred before the start of exposure, it cannot be determined to what extent, if any, antimony trioxide contributed to it. The lung weight was significantly but reversibly increased at nine months in the high dose group and at 12 months in both exposure groups, but no significant difference was observed at one year post exposure. A number of neoplastic effects were observed in the lungs of exposed animals (Table 4-52) and in many animals the neoplasms were multifocal. Scirrhous carcinomas (malignant, poorly-differentiated adenocarcinomas surrounded by dense fibrous tissue) were the most common type of lung neoplasm, found only in the high dose rats at the end of exposure and during the observation period (Table 4-52). The increased incidence was statistically significant in animals sacrificed between 2 and 12 months postexposure and between 12 and 15 months postexposure. The scirrhous carcinomas appeared to arise from the alveolar epithelial lining cells and the cells exhibited a variety of morphologic changes; they were hypertrophied, hyperchromatic and exhibited varying degrees of mitotic activity and anaplasia. There was evidence of local invasion but no evidence of metastasis. An abundance of fibrous connective tissue was associated with these carcinomas and considered in excess of what would be expected as a stroma for scirrhous carcinomas and alsounusual for a primary neoplasm of the rat lung.

Two squamous cell carcinomas were observed in the 18 high dose rats killed at one year post exposure, however the incidence was not statistically significant (Table 4-52). Squamous cell carcinomas also appeared to be originating from the alveolar lining epithelial cells. These cells, in addition to exhibiting varying degrees of anaplasia, had gone through a metaplastic change to a more squamous type cell with evidence of keratin production. A fibrous connective tissue stroma was a prominent part of these neoplasms, however it was less than seen in the scirrhous carcinomas.

Bronchiolar adenomas were found in both exposed and control animals and were not considered as treatment-related but similar to those that occur naturally in the rat lung. The incidence in the high dose group (4 tumours in 45 animals) was not statistically significantly higher than that of the controls (1 tumour in 41 animals). A variety of neoplastic changes were observed in other tissues, most notably mammary glands, but they were not considered treatment-related since their incidence and severity was not significantly different between the exposed and the control rats

This study shows that exposure to diantimony trioxide significantly increase the incidence of pulmonary scirrhous carcinomas in female rats, 2 to 15 months after 12 months inhalation of 5.0 ± 3.8 mg diantimony trioxide/m^3. The incidence of scirrhous carcinomas in animals sacrificed between 0 and 15 months post exposure in the highest exposure group was 44 % (15/34). The corresponding value for the control was 0/28. A LOAEC of 5.0 mg/m^3 is suggested based on the development of scirrhous carcinomas. The NOAEC is set to 1.9 mg/m^3. There is however some uncertainty regarding this LOAEC of 5.0 mg/m^3 as a re-evaluation of the histopathology tissue sections from the Watt- and the Newton-studies indicated higher lung deposition of antimony and more severe lung damage in exposed rats in the Watt-study than in the Newton-study, which allegedly were conducted at similar exposure levels (1.9-5.0 and 0.06-4.50 mg/m^3, respectively). This suggests that the exposure levels in the Watt study may have been above those reported, but the difference could also be due to different particle generation techniques or different strains of rats.

Table 4-52. Lung tumour incidence in Fisher rats after inhalation exposure to diantimony trioxide. Only females were included in this study.

		Any lung neoplasm			Scirrhous carcinoma			Squamous cell carcinoma			Bronchioalveolar adenoma		
Group	**Death or Sacrifice**	**Control**	**Low**	**High**	**Control**	**Low**	**High**	**Control**	**Low**	**High**	**Control**	**Low**	**High**
A	Pre-exposure	0/3	0/1	0/0	0/3	0/1	0/0	0/3	0/1	0/0	0/3	0/1	0/0
B	From start through 5 months of exposure	0/2	0/4	0/3	0/2	0/4	0/3	0/2	0/4	0/3	0/2	0/4	0/3
C	From 6 through 9 months of exposure	0/4	0/3	0/3	0/4	0/3	0/3	0/4	0/3	0/3	0/4	0/3	0/3
D	From 9 through 12 months of exposure	0/4	0/5	0/5	0/4	0/5	0/5	0/4	0/5	0/5	0/4	0/5	1/5
E	At the end of exposure (12 months)	0/9	0/9	2/9	0/9	0/9	1/9	0/9	0/9	0/9	0/9	0/9	0/9
F	Between 2 and 12 months post-exposure	1/6	0/5	5/7	0/6	0/5	5/7 *	0/6	0/5	0/7	1/6	0/5	0/7
G	12 to 15 months post-exposure	1/13	1/17	14/18	0/13	0/17	9/18 **	0/13	0/17	2/18	0/13	1/17	3/18
Total A-G		2/41	1/44	21/45	0/41	0/44	15/45	0/41	0/44	2/45	1/41	1/44	4/45
Total E-G		2/28	1/31	21/34	0/28	0/31	15/34	0/28	0/31	2/34	1/28	1/31	3/34

Statistically different from control: $*p < 0.05$; $**$ $p < 0.01$

The same study also included eight female Sinclair S-1 miniature swine that were exposed under similar conditions as the rats and housed in the same exposure chambers. The animals were divided into high dose (n=3), low dose (n=3) and control (n=2) groups. No exposure-related histopathological changes were observed in the swine. No real conclusions can be drawn on the carcinogenicity of antimony trioxide from this limited study.

In the second study, the carcinogenic effects of antimony trioxide and antimony ore (Sb_2S_3) were evaluated in Wistar rats, 90 males and 90 females per group (Groth et al., 1986a). This

study is also described in section 4.1.2.6. Repeated dose toxicity. The animals, 8 months of age, were exposed by inhalation to antimony trioxide [time-weighted average (TWA) 45 mg/m^3 (range = 0-191.1)], antimony ore [TWA 36-40 mg/m^3 (range = 0-91.1)] or filtered air (controls) in exposure chambers, 7 h/day, 5 days/week for up to 52 weeks. The MMADs for diantimony trioxide and antimony ore were 2.80 and 4.78 μm, respectively (GSD not reported). The antimony content in the diantimony trioxide was 80 % and in the antimony ore it was 46 %. Major contaminants in the diantimony trioxide were lead (0.23 %), tin (0.21 %) and arsenic (0.004 %) and in the antimony ore they were aluminium (0.48 %), iron (0.33 %), lead (0.25 %), tin (0.16 %) and arsenic (0.079 %). At 6, 9, and 12 months after initiating exposures 5 animals/sex/group were sacrificed and autopsied, the remainder of the animals were sacrificed 18-20 weeks post-exposure. In addition, all animals that died or were sacrificed due to ill health were autopsied. At autopsy all organs were examined grossly and tissue sections from the lungs, liver, kidneys, pancreas, spleen, adrenal, thyroid, pituitary, bladder, brain, eye, bone marrow, skin, lymph nodes (mesenteric and tracheobronchial), stomach and colon (ascending and descending) from each rat were fixed in buffered 10 % formalin, embedded, sectioned and stained with hematoxylin and eosin for examination by light microscopy. Samples from the testicle and prostate from males and mammary gland, ovary, uterus and cervix from females were also examined as well as any abnormal tissue. In addition, at the final sacrifice heart tissue was sampled and examined by light microscopy. At sacrifices, portions of liver, lungs, kidneys, brains, spleens and blood from 5 animals/sex/group were sampled for antimony concentration analysis.

The data indicate that there was no treatment related mortality. However, for each of the groups, the female survival rate was significantly greater than the male counterparts. The mean body weight of the males exposed to antimony trioxide and the females exposed to antimony ore was slightly but statistically significantly reduced (6.2 % and 6.4 %, respectively). Sporadic bleeding from eyes and hematuria occurred in all groups, but appeared to occur more frequently in the diantimony trioxide and antimony ore groups (data not shown). Although, according to the authors, no significant pathological alterations were seen in any of the control lungs, occasional foci containing lymphocytes, typical of chronic pneumonia, were seen in a few rats.

No lung tumours were seen in the control rats of either sex or in the male rats exposed to either compound. In contrast, both the diantimony trioxide and antimony ore exposed female rats developed lung tumours, including squamous-cell carcinomas, bronchoalveolar adenomas and carcinomas and scirrhous carcinomas (these neoplasms were observed.

Table 4-53 and Table 4-54). The first lung tumour was observed in a female rat that died after 41 weeks of antimony ore-exposure. The first lung tumour in the diantimony trioxide group was seen at 12 months of exposure. If only the animals at risk (those alive at the first lung tumour was found) and subsequently examined are used in the calculations, then the incidence of lung tumours for antimony trioxide-exposed rats was 27 % and that for antimony ore-exposed rats was 25 %. These incidences were significantly ($p < 0.001$) greater than in the control group, which had no lung tumours.

Different types of tumours that are typical for this strain of rats were observed in thyroid, skin (subcutis), mammary gland, pituitary and adrenal tissue in all groups; however, no exposure-related differences in the incidence of these neoplasms were observed.

Table 4-53. Lung tumour incidence in female rats after inhalation exposure to diantimony trioxide and antimony ore.

Weeks on experiment	Controls	Diantimony trioxide	Antimony ore
53 (serial sacrifice)	0/5	2/5	2/5
54-71 (died)	0/15	5/23	3/21
71-73 (serial sacrifice)	0/39	12/31***	11/33***
Total (18-73 weeks)	0/89	19/89***	17/87***
41-72 "animals at risk"	0/69	19/70***	17/68***

Statistically different from control: *** $p < 0.001$

Table 4-54 Tumour type frequency in female rats after inhalation exposure to diantimony trioxide or antimony ore

Compound	Scirrhous carcinomas	Squamous cell carcinomas	Bronchoalveolar adenomas and carcinomas	Multiple lung tumours (2-4/rat)
Sb_2O_3	5/19	9/19	11/19	6/19
Sb ore	4/17	9/17	6/17	3/19

This study shows that the incidence of various lung neoplasms significantly increased in female rats exposed to 45 mg diantimony trioxide/m^3 for 12 months via inhalation. In general the lung tumours occurred after 12 months of exposure with an incidence of 32 % (19/59) in animals examined after 12 months of exposure (including animals from weeks on experiment 53, 54-71 and 71-73; these neoplasms were observed.

Table 4-53). No lung tumours were found in males despite higher antimony concentrations found in male lungs than in female lungs, suggesting that female rats are more susceptible to the induction of lung cancer by antimony trioxide and antimony ore and also indicating that the tumour response was not only a function of lung tissue concentration of antimony. It is noted that only one dose per compound was investigated, which does not allow a dose-response assessment. Moreover, all animals including controls showed signs (further described in the Repeated dose toxicity section 4.1.2.6) which might reflect sub-chronic disease of unknown aetiology among the animals.

In conclusion, diantimony trioxide caused a statistically significant increase in lung cancer in female rats under the present test conditions. However, the underlying mechanism of the tumour formation is unclear. The authors state that interstitial fibrosis is a frequent precursor to the induction of lung tumours in rats exposed to particulates containing beryllium compounds (Groth et al., 1980) and quartz (Groth et al., 1986b). However, in the present study on antimony a high incidence of lung neoplasms in female rats and a lack of neoplasms in male rats were observed although pulmonary interstitial fibrosis was seen in both males and females. A LOAEC of 45 mg/m^3 is suggested based on observed lung cancer in females.

The oncogenicity of antimony trioxide was also evaluated in a whole-body inhalation study, performed by Bio/dynamics Inc. and published by Newton and co-workers, (Newton and Daly, 1990; Newton et al., 1994). This study, which is also described in section 4.1.2.6, Repeated dose toxicity, is based on the results from both the Watt and Groth studies. Fisher 344 rats, 65 males and 65 females per dose group, 8 weeks of age, were exposed to diantimony trioxide at 0, 0.06, 0.51 or 4.50 mg/m^3 for 6h/d, 5d/wk for 12 months followed by a 12-month observation period. Control animals were exposed to clean air only. The flow rate was 18-25 complete air changes per hour (the recommended flow rate in OECD guideline 412, 413 and 453 is 12-15 air changes per hour, Rapporteur comment). Five animals per sex per group were sacrificed at 6 and 12 months of exposure and at 6 months postexposure. All surviving animals were sacrificed at 24 months (12 months postexposure). The purity of the diantimony trioxide was 99.68 % and the particle MMAD was 3.76 ± 0.84 µm with a geometric standard deviation of 1.79±0.32 for all concentrations.

Animals were observed twice daily for viability and overt signs of toxicity. Detailed observations were conducted weekly and body weights were measured twice pretest, weekly for the first 13 weeks, monthly thereafter and at termination. Ophtalmoscopic examinations were performed on all animals pretest and on the day before their scheduled sacrifice. Hematological effects were evaluated at 12, 18 and 24 months. Complete gross postmortem examinations of all major organs were performed in all animals. Histological examinations were performed on hematoxylin-eosin-stained tissue sections of heart, nasal turbinates, larynx, trachea, lung and peribronchial lymph node. At each sacrifice, the left lung lobe was frozen for later antimony analyses and blood samples were collected. Fecal samples were collected at the 18- and 24-month sacrifices.

Survival was not affected by the exposures to antimony trioxide. At termination at 24 months, there was 56% survival of the males and 48% survival of the females.

Pulmonary carcinomas were seen in only three animals, two males (one each from the control and high dose groups) and in one female (from the medium dose group). These carcinomas were not considered to be antimony trioxide exposure related. No other primary lung neoplasms were seen. Elevated total leukocyte counts and atypical lymphocytes in some animals in all groups at the terminal euthanization indicated the presence of leukemia. However, leukemia is a common finding in aged Fischer 344 rats. Other postmortem neoplastic findings (not further specified) occurred mostly with comparable incidence and severity in the treated and control animals or they occurred sporadically. None of these findings were considered to be of either oncological or toxicological significance with respect to antimony trioxide.

In conclusion, no evidence of antimony trioxide-induced carcinogenicity was found under the exposure conditions of this study. The chronic interstitial pneumonia and the chromodacryorrhea (shedding of bloody tears) observed without dose-response in both control and antimony trioxide exposed animals suggest that the animals suffered from sub-chronic disease of unknown etiology. These non-neoplastic findings are further described in the 4.1.2.6, Repeated dose toxicity section.

Dermal

No studies on carcinogenic potential of antimony trioxide after dermal exposure have been located.

Oral

No studies on carcinogenic potential of antimony trioxide after oral exposure have been located.

4.1.2.8.2 Studies in humans

In vivo studies

Inhalation

The mortality and the underlying causes were investigated in workers employed in an antimony plant where roasting of antimony ore started in the 1920s (Jones, 1994). This study is also described in the 4.1.2.6.Repeated dose toxicity section.

Over the years, antimony metal, antimony alloys and antimony trioxide were produced in the plant. Until the early 1970s considerable quantities of lead alloys were made, containing as much as 80 % lead and 10 % arsenic. All production of antimony metal and its alloys stopped in 1973 and after that only antimony trioxide was manufactured. Since the 1960s the bulk of ore used was a sulphide ore containing about 60 % antimony and up to 0.5 % of arsenic. Arsenic metal and its trioxide were also brought into the plant to make arsenic alloys.

Since 1950 another non-antimony process, the milling of zircon sand, has also been carried out at the site. The process is purely physical, with no chemical change involved. The zircon and antimony workers have worked separately, except for a few cases where temporary transfers of workers between the processes have been necessary.

No exposure data were presented but variable occupational exposure to lead, metallic antimony, metallic arsenic, diantimony trioxide and arsenic trioxide was assumed by the author. It was also assumed that there was exposure to polycyclic aromatic hydrocarbons due to the combustion in the furnaces used.

All men employed between 1961 and 1992, and with at least three months of employment were recruited into the survey. Of the 1452 men that were recruited, 32 were not traced. Of the 1420 men that were traced, 357 had died and 29 emigrated by December 1992.

The workers were subdivided into four occupational groups: a) antimony workers, b) maintenance workers, c) zircon workers and d) others (including office workers and management staff).

Two sets of expected death rates were worked out: the first based on national rates for England and Wales and the second based on local rates (Tyneside conurbation 1961-83, Tyne

and Wear 1974-83). Man-years at risk were calculated for the population in separate decennial age periods for each year from January 1961 to December 1983. Appropriate age specific death rates were then applied to calculate the number of deaths expected for each cause of death considered. The causes of death were divided as follows; lung cancer, stomach cancer, other neoplasms, circulatory disease, ischemic heart disease, respiratory disease, genitourinary disease, accident and suicides and others. The observed numbers of deaths for each cause were compared with the expected figures calculated as above.

In the antimony workers there was a significant increase in mortality from lung cancer (37 v. 23.9, p = 0.016) but no difference in mortality from stomach cancer or other neoplasms. For maintenance men there was also a significant increase in mortality from lung cancer (15 v. 8.1, p = 0.038) but not from stomach cancer or other neoplasms. No increased death rates were observed in the groups of zircon workers or others. When the employees were divided into those employed before and after 1st of January 1961 it was shown that for the antimony workers employed before January 1961 there was a significant excess of mortality from lung cancer (32 v. 14.7, p = 0.001). Significant excess was also seen in the maintenance workers before 1961 (12 v. 5.3, p = 0.016). No evidence of an excess of mortality from lung cancer was found in the zircon and other groups, or in any of the groups employed after 1st of January 1961. When deaths from lung cancer in antimony workers were divided into calendar year of first exposure (from 1940-1990) an excess of mortality from lung cancer for all five-year periods of first exposure up to 31st of December 1960 was found. People who started work before 1955 showed an excess of three to four times the lung cancer expected, the group who started work from 1955 to 1960 showed less than a doubling of lung cancer, whereas people who started work after 1960 showed no such excess.

Analysis of deaths from lung cancer in antimony workers by years since their first exposure to antimony showed that < 20 years after the first exposure there is no excess of lung cancer but after that time a significant twofold excess emerges. However there was no trend of greater risk with increased years of exposure.

This study shows a significant excess of mortality from lung cancer among antimony smelter workers and maintenance workers. The excess of deaths from lung cancer in smelter workers was confined to those joining before 1961 and did not appear until 20 years after their first exposure to the antimony process. There was an excess of lung cancer for workers first exposed in all the five-year calendar periods before 1960. This suggests that antimony workers were occupationally exposed to some carcinogen before 1960. Because at the stage of follow up, very few of the men employed after 1960 were first exposed to the antimony process > 20 years previously and because the lung cancer excess was not obvious until 20 years after first exposure it is not possible to be certain whether the carcinogenic effect persisted after 1960. Considering the variable exposure (including in addition to antimony compounds also arsenic compounds, lead and polycyclic aromatic hydrocarbons) and the lack of exposure data it is not possible to determine what factors that have been responsible for the lung cancer.

Dermal

No studies on carcinogenic potential of antimony trioxide after dermal exposure have been located.

Oral

No studies on carcinogenic potential of antimony trioxide after oral exposure have been located.

4.1.2.8.3 Summary of carcinogenicity

Three chronic inhalation studies in rats are available for carcinogenicity assessment of diantimony trioxide (Watt, 1983; Groth et al., 1986a; Newton et al., 1994). Two animal studies indicate neoplastic properties of diantimony trioxide, whereas one animal study showed negative results. There is also one human study available (Jones, 1994). However, due to lack of exposure data the human study is regarded inconclusive. The exposure duration in all three animal studies is 12 months and thus all studies deviate from the OECD guideline on chronic toxicity/carcinogenicity, which prescribes an exposure period of 24 months for rats. In the first animal study (Watt, 1983) inhalation of 5.0 mg Sb_2O_3/m^3 for 12 months produced lung neoplasms in 44 % of the animals tested (only females were exposed). In the second study, (Groth et al., 1986a) a 9 times higher dose (45 mg Sb_2O_3/m^3) produced pulmonary neoplasms in 32 % of the female rats exposed under similar conditions, but none in male rats. It is noted that the female survival rate was significantly higher than the male counterparts in the study by Groth et al., (1986a). The differences in incidence between the studies might be explained by a longer observation period (12 months vs 20 weeks) and by the use of older animals (8 months vs 14 weeks) in the study by Watt (1983). The study by Newton et al. (1994) showed no diantimony trioxide-related lung tumours, neither in males nor females, at any dose level up to 4.5 mg/m^3. This is in contrast with the data reported by Watt and Groth and the cause of the difference is not entirely clear. However, the histopathology slides from the negative Newton study was re-evaluated by the pathologist who evaluated the slides from the Groth and of the Watt studies. The re-examination confirmed a lack of diantimony trioxide-related neoplastic changes in the Newton study. In addition, the comparison of the Watt and the Newton studies, which were conducted at similar exposure levels, showed that the exposed rats had more lung damage and appeared to have considerably more antimony deposited in the lungs in the Watt study than in the Newton study. This may suggest that the exposure levels in the Watt study may have been above those reported. Given that the dose level in the study by Groth is 10 times higher and also the dose levels in the study by Watt were likely higher than 1.9 and 5.0 mg/m^3 the dose levels in the Newton study most probably fit in the dose range where no tumous were observed. However, the difference could also be due to different particle generation techniques or different strains of rats. The particle size, which will affect lung deposition, clearance and retention and hence target organ dose, was similar among the studies although they were all measured using different techniques.

In the study by Newton and co-workers it was shown that diantimony trioxide reduced the pulmonary clearance rate in a dose dependent manner, interpreted by the authors as a toxic effect of diantimony trioxide rather than a general effect due to pulmonary overload. However, it is wellknown that reduced lung clearance rate at chronic exposure of rats to poorly soluble particles (PSPs) can result in pulmonary overload, subsequently followed by an inflammatory response, epithelial cell hypertrophy and/or hyperplasia and squamous metaplasia. The persistence of these tissue responses over chronic time periods can lead to

secondary development of lung tumours (Hext, 1994). Thus, it could be speculated that the neoplastic effects seen in the Watt and Groth studies is a result of pulmonary overload and an inflammatory response to particulate diantimony trioxide. The tumour development as a consequence of pulmonary overload is an inflammatory-driven process which usually takes over a year (15-18 months) of PSP exposure via inhalation (Driscoll et al., 1997). In the present studies on diantimony trioxide, development of lung tumours occurred earlier – already at 12 months of antimony inhalation.

PSPs have several common characteristics such as low solubility, low order of toxicity and they are generally non-genotoxic (Miller, 2000; ILSI Risk Science Institute Workshop Participants., 2000). PSPs include among others, titanium dioxide, carbon black, diesel soot, shale, talc and coal mine dust. *In vitro* studies on titanium dioxide and carbon black have not shown these materials to be genotoxic (Kanematsu et al., 1980; Kirwin et al., 1981; Tennant et al., 1987). In contrast, diantimony trioxide is generally negative for gene mutations but has the potential to induce structural chromosome aberrations *in vitro* (see section 4.1.2.7 Mutagenicity). Whether or not diantimony trioxide should be regarded as a PSP can thus be discussed.

The inflammatory response to inhalatory PSP exposure includes macrophage breakdown, neutrophil accumulation in alveolar airspaces and bronchioalveolar epithelial cell proliferation. At pulmonary overload, the particle clearance function of the macrophages is impaired resulting in persisting lung burden. Extended impairment of clearance leads to development of pulmonary tumors in rats but not in mice or hamsters (Heinrich et al., 1986; Muhle et al., 1990). Accumulation of particles starts and inflammatory cell influx increases sharply (ILSI Risk Science Institute Workshop Participants., 2000; Oberdorster, 1995). The surface dose is considered the best indicator for developing pulmonary overload and a surface dose to cause lung cancer in rats has been identified between 2 000 and 3 000 cm^2 particle surface/lung (Borm et al., 2004). The surface dose needed to cause neutrophilic inflammation by PSPs is 10-fold lower, 200-300 cm^2 (Tran et al., 1999; ILSI Risk Science Institute Workshop Participants., 2000).

Experimental evidence thus supports the hypothesis that there are links between chronic inflammation and epithelial changes leading to pulmonary cancer in rats (Donaldson, 2000), although the causal mechanisms to explain this association are still unclear (Coussens and Werb, 2002; Schottenfeld and Beebe-Dimmer, 2006). Particles generate reactive oxygen species (ROS) and reactive nitrogen species (RNS), which are thought to be involved in genotoxic as well as proliferative effects (Mossman and Churg, 1998; Schins, 2002; Fubini and Hubbard, 2003).

In the majority of studies with particle exposure it appears that females develop the higher incidences of lung tumors (Lee et al., 1985). This may be attributable to a more potent response to the particle in females or to the fact that in a number of studies the females had a greater survival rate. Even a relatively small increase in longevity in one sex may result in an apparent disproportion increase in tumour incidence as the tumours develop towards the end of the life span of the rat (Hext, 1994). In the present animal studies, diantimony trioxide produced lung cancer in female rats. A higher concentration of diantimony trioxide was however found in lungs from male rats and there appeared to be more inflammatory cells in the male rat lung than in the female rat lung (Groth et al., 1986a) indicating that the tumour response was not only a function of lung tissue concentration of diantimony trioxide. In this study, a 32 % incidence of lung neoplasms in female rats and a lack of neoplasms in male rats

exposed to diantimony trioxide were observed although pulmonary interstitial fibrosis was seen in both males and females.

The issue whether genotoxicity or particle overload may be the reason for diantimony trioxide-induced lung tumors is still not entirely clear. Despite the lack of conclusive data on local genotoxicity in the lung, the overall expert judgement by TC NES is that the most likely mechanism for carcinogenicity appears to be impaired lung clearance and particle overload followed by an inflammatory response, fibrosis and tumours. Consequently, diantimony trioxide can be regarded as a threshold carcinogen and as a starting point for a quantitative risk characterisation the NOAEC of 0.51 mg/m^3 derived for local repeated dose toxicity is also used for carcinogenicity. There may be some uncertainty regarding the accuracy of the NOAEC numerical value as the study had a high background incidence of lung inflammation in control animals. In addition, the exposure duration was 12 months and thus deviates from the OECD guideline on chronic toxicity/carcinogenicity, which prescribes an exposure period of 24 months for rats. It could be discussed whether effects caused by pulmonary overload in the rat is also relevant for humans. Positive (Hext, 1994; Oberdorster, 1995) and negative (Tran and Buchanan, 2000; Kuempel et al., 2001) findings of particle overload in human lungs are reported. Macrophage transport of particles into the alveolar interstitium is the major clearance mechanism in humans but of minor importance to the rat. These species differences are related to morphological features of the lung, i.e. to the relative short pathway length from the alveoli to the ciliated terminal bronchioles in rats (Bailey et al., 1989; Kreyling, 1990; Kreyling et al., 1991). In the absence of mechanistic data to the contrary, it must be assumed that the rat model of tumorigenicity can identify potential carcinogenic hazards to humans and the rat presently remains the appropriate model for both neoplastic and non-neoplastic responses to PSP exposure (ILSI Risk Science Institute Workshop Participants., 2000).

Antimony trioxide is currently classified in Annex 1, Directive 67/548/EEC as “Carcinogenic Category 3”.

4.1.2.9 Toxicity for reproduction

Available data on the reproductive toxicity of antimony trioxide include studies on rats and mice and one human occupational study, but none is performed according to standard guidelines.

4.1.2.9.1 Effects on fertility

Studies in animals

The effect of antimony trioxide on fertility was investigated in 24 white female rats (strain not stated) exposed to antimony trioxide dust (250 mg/m^3) by inhalation daily for 4 h/day for a period of 1.5-2 months before mating (Belyaeva, 1967). Exposure continued through mating and gestation until 3-5 days prior to delivery, when the animals were moved to clean cages. A control group of 10 rats was kept under the same conditions as the exposed rats but with no exposure to antimony trioxide. 16 of the 24 exposed rats became pregnant while in the control

group all 10 rats were pregnant. The average litter size was smaller in the exposed rats, 6.2 ± 1.0 vs. 8.3 ± 0.2 in the controls. It was not reported how the pregnancy was determined and no data were presented on the incidence of resorptions or foetal deaths. The weight, sucking, fur growth and eye-and ear-opening were observed in the pups, but only data on weight, which was not affected by exposure to diantimony trioxide, was reported.

The eight rats which did not became pregnant were pooled with 15 other female rats which had been exposed to a single i.p. injection of finely dispersed metallic antimony (50 mg/kg bw) suspended in 1 ml of sterile oil (the particle size was < 5µm), mated and found non-pregnant. The group of rats administrated i.p. were 30 at number, 15 became pregnant and 15 did not. A control group of 12 rats was administrated i.p. with sterile oil, 11 of them became pregnant. In the pooled group of antimony exposed non-pregnant females, morphological changes in sexual organs (uterus, ovaries, and oviducts) were observed. The changes observed in the ovaries was disturbed egg cell development; in some rats the follicles contained no egg cells or the egg cells did not show their normal characteristics (not further specified). Hyperaemia in the cortex (strongly affected) of the ovaries was observed in some rats and some of the ovaries had follicular cysts. Metaplasia in the uterus, single cell epithelia was transformed into multi layer epithelia, was observed in almost all rats. A similar change was also observed in the oviducts of some animals. Of the 31 female rats that became pregnant (i.p. and inhalation), 19 were checked for and found to have a normal pregnancy and a for the pregnancy normal morphological picture. Sometimes there were metaplasia in the uterus and sometimes follicles contained no egg cells. The remaining 12 rats were checked and found to have a normal placenta and normal morphological picture of the sexual organs. The females in the inhalation group had no morphological changes and the rats in the i.p. group had the same normal picture, but occasional metaplasia in the uterus and occasional ovaries with follicles containing no egg cells. The control group had normal pregnancies and deliveries, but there is no information on reproductive organs.

No statistical calculations were presented in the study report. However, calculations performed by the Rapporteur showed a significant difference between exposed and control animals in the number of rats that became pregnant (Fisher´s Exact Test, one-sided p-value = 0.0405) and in the average litter size (Unpaired t-test with Welch correction, one-sided p-value <0.0001).). The bodyweight on exposed and control female rats were 195.5 ±7.2 g and 197.8±14.0 g, respectively, and therefore the conclusion can be drawn that there was no maternal toxicity.

In conclusion, based on an effect on pregnancy rate and average litter size this study indicates that antimony trioxide may have an adverse effect on fertility on rats. The value of the morphological changes that were observed is difficult to assess since no morphological examinations of control rats were performed. However, the study report gives an unclear picture of the overall experimental conditions and the purity of the substance is not stated, therefore the results are regarded inconclusive and cannot be used for risk assessment.

Testicular toxicity of antimony trioxide was evaluated in rats and mice (Omura et al., 2002). 30 male mice (Crj: CD-1) and 24 male rats (Crj: Wistar) were randomly divided into 3 groups at 8 weeks of age, a low exposure group, a high exposure group and a control group, each comprising 10 mice and 8 rats. The antimony trioxide was suspended in distilled water and were administered by gavage; mice, 5 days per week for 4 weeks; rats, 3 days per week for 4

weeks. Four weeks are shorter than the period needed to complete spermatogenesis in rats (approximately 8 weeks) and mice (approximately 39 days). However, the International conference on harmonisation tripartite guidelines on detection of toxicity to reproduction for medical products suggest that a 4-week treatment period is appropriate for detection of drug effects an male fertility in rats, provided that adequate histology and organ weight findings are available from repeat-dose studies. Such examinations were performed in this study, therefore, a 4 weeks administration period was used. The low exposure group was exposed to 12.0 mg antimony trioxide/kg bw/day, the high exposure group was exposed to 1200 mg antimony trioxide/kg bw/day and the control group was administered distilled water. There is no information on the size of the particles, but the purity was > 99.999 %. The mice were kept in an air-conditioned conventional room with a 12 h light/12 h dark light cycle, 24-26 °C and 40-80 % air humidity. Rats were kept in a SPF room with a 12 h light/12 h dark light cycle, 22-25 °C and 50-60 % air humidity.

One day after the final administration, the animals were killed by inhalation of carbon dioxide. The testes, epididymidis, ventral prostate and seminal vesicle were removed and weighed. The seminal vesicle was weighed without fluid. In addition, the number, motility and morphology of sperm in the cauda epididymidis and histological changes in the testis were investigated. Sperm motility findings of mice were not included in the report due to "too long interval between removal of the epididymidis and sperm motility analysis". In the histopathologic examination of the testis, all round or ovoid cross-sections of the seminiferous tubule (173-449 cross sections in rats and 55-165 cross sections in mice) in one transverse section were examined. The following histopathologic changes were examined: disorganisation and exfoliation of the seminiferous epithelium, degeneration of germ cells, vacuolisation of the epithelium, sperm retention in the epithelium and delayed spermiation.

Two mice and one rat in the high dose group and one control mouse died due to accidents at administration. No other deaths occurred. No effects on body weight gain of mice or rats were observed. A small, but not statistically significant, decrease in seminal vesicle weight was observed in mice in both the low and high dose exposure groups. This effect was not observed in rats of either exposure group. In both species, the weights of the other organs in the antimony-treated groups were comparable to the controls.

No effects on the count, motility or morphology of sperm in the cauda epididymidis in rats or the count, and morphology of sperm in mice were observed in any of the two dose groups. Delayed spermiation was found in one of the 8 rats in the low dose group and one of the 7 rats in the high dose group. However, the frequencies of delayed spermiation in these two rats were less than 1 %. One of the 10 mice in the low dose group showed exfoliation of the seminiferous epithelium; the frequency in this mouse was greater than 50 %. However, this was not observed in mice in the high dose group or rats in either of the groups and therefore this was suggested to be a non-treatment related finding.

This study indicates that a repetitive gavage administration of antimony trioxide to rat (3 days/week) and mouse (5 days/week) at a dose of 1200 mg/kg bw for 4 weeks was not toxic to testes. The concentration of antimony in testes was not stated. An oral NOAEL of 1200 mg/kg bw for testicular toxicity is suggested.

In a 90-day oral feeding study of antimony trioxide (further described in the section Repeated dose toxicity) groups of 12 male and 12 female Wistar rats of the Alpk:APSD strain were fed diets containing 0, 1000, 5000 or 20000 ppm antimony trioxide corresponding to mean daily doses of 84, 421 and 1686 mg/kg in males and 97, 494 and 1879 mg/kg in females (Hext et al., 1999, also reported by CTL, 1997). According to the authors the test protocol conforms to modern guidelines. Complete necropsies were performed on all rats. The epididymis and testes were weighed. Epididymis, testes, prostate gland, seminal vesicle, ovary and uterus were examined for macroscopic lesions and fixed in 10 % neutral buffered formalin or other appropriate fixative. These tissues from the controls and the top dose group were examined under the light microscope, together with any macroscopically abnormal tissue from the intermediate groups. No changes were seen in the female or male reproductive organs.

An oral NOAEL of 1879 mg/kg, based on no histopathological changes reported for female (ovary and uterus) and male (epididymis, testes, prostate gland, seminal vesicle) reproductive organs, is suggested from this study.

Studies in humans

Gynaecological examinations were performed on women (number not specified) occupationally exposed to dust containing metallic antimony, antimony trioxide and antimony pentasulfide over a period of 2 years, 1962-1964 (Belyaeva, 1967). The report is in Russian and an English summary is available. These women were compared with a group of control women, however it was not stated how the control group was selected. No exposure data were reported but antimony was detected in the blood of the exposed workers at levels 12-16 times higher than in the controls. Antimony was also detected in breast milk (3.3 ± 2.2 mg/l), amniotic fluid, placental tissue and umbilical cord blood of exposed workers.

A higher incidence of various sexual disturbances was reported in the exposed women as compared with controls (77.5 % vs. 56.0 %); these included disturbances of the menstrual cycle in 61.2 % of the exposed women compared with 35.7 % of the controls, infections in the sex organs in 30.4 % of the exposed women compared with 55.3 % of the controls, and other ailments of the sexual organs in 8.4 % of the exposed workers (the corresponding figure for controls is not stated in the study report). The incidence of late spontaneous abortions was 12.5 % in the exposed women as compared with 4.1 % in the controls, and the incidence of premature births was 3.4 % in the exposed compared with 1.2 % in the controls. The birth weights of children born to the exposed women were comparable to those of children born to the controls, but the body weight of the children of exposed women began to lag already after 3 months and continued to do so also after six and 12 months. No statistical calculations were presented.

This study indicates that occupational exposure to antimony trioxide, metallic antimony and antimony pentasulfide may affect the fertility of female workers. Among the effects observed was disturbed menstrual cycle, increased incidence of late spontaneous abortions and premature births. There was also an effect on the body weight gain of the babies up to 12 months. However, no statistical calculations were presented and no information was given about the control group or how it was selected. Further on, it is not mentioned if other factors,

like concurrent exposure to other potentially toxic substances or the workplace environment (e.g. high temperatures, heavy work or stress), were controlled for. Therefore, due to the limitations of the study it is regarded as inconclusive and can not be used for risk assessment.

4.1.2.9.2 Developmental toxicity

Studies in animals

To determine the developmental toxicity of antimony trioxide, three treatment groups and one control group, each containing 26 female Sprague-Dawley [Crl: CD®(SD)IGS Br] rats, were exposed to diantimony trioxide and clean air respectively, through nose-only inhalation (MPI, 2003). A purity of Sb_2O_3 of 99.87% is stated. The study was conducted in accordance with Standard Operating Procedures and was based on the draft guideline published in the US EPA Health Effects Test Guidelines, Inhalation Developmental Toxicity Study, Office of Prevention, Pesticide, and Toxic Substances (OPPTS) 870.3600, issued June 1996 and the OECD Guideline Number 414, Prenatal Developmental Toxicity Study (2001 01 22). However, some alterations in the conduct of the study have been made. The dose intervals are not according to guidelines. Low and mid dose give almost the same internal dose (antimony level in RBC) and the high dose does not give any maternal toxicity.

Prior to exposure the female rats were mated to untreated male rats of the same strain and the day on which positive evidence of mating (vaginal plug and/or sperm) was observed was considered Day 0 of gestation. The females were approximately 10 weeks old and weighed between 193 and 270 g on Day 0 of gestation. The mated females were exposed from Day 0 (fertilization) to Day 19 (one day prior to scheduled euthanasia and laparohysterectomy) of gestation, six hours per day, at concentrations of 2.6 (SD ± 2.43), 4.4 (SD ± 3.88), and 6.3 (SD ± 4.18) mg diantimony trioxide/m^3. Control females received clean air by the same procedure and dosing regime as the treated females. This dosing regime was, according to the authors, proposed to identify possible effects on preimplantation loss of the fertilized ova as well as effects on the developing foetus *in utero*.

Dust aerosol atmospheres of the test article were generated into the breathing air of the treated animals using a Wright Dust feeder. A purity of Sb_2O_3 of 99.87 % was given. A chamber airflow of at least 0.6 litres per minute per animal resulted in at least 10 chamber air changes per hour. The mass median aerodynamic diameters (MMAD) and geometric standard deviations (GSD) ranged from 1.59 to 1.82 μm and 1.713 to 1.744, respectively. A sample of each water and diet were collected and analyzed for total antimony.

Observations of the dams included clinical signs, conducted daily following exposures, gestation body weights and gestation food consumption. Blood samples were collected from 10 dams/treatment group and the Sb concentration in RBC was determined. Litters were delivered by caesarean section on Day 20 of gestation and gravid uterine weights were recorded. The total number of corpora lutea on each ovary and the number of implantations, resorptions, and foetuses on each uterine horn were recorded. A complete necropsy was performed on all dams. Special emphasis was placed on structural abnormalities or pathologic changes that may have influenced the pregnancy. The lungs and brain were weighed and the lungs then infused via the trachea with 10 % neutral buffered formalin. The brain weights

were used to calculate lung/brain weight ratio. Subsequently, based on organ weight changes, the lungs of 10 females/group randomly selected were processed for histopathological examination. The nasopharyngeal tissue and gross lesions were also preserved in formalin. Foetuses were weighed individually, sexed externally and measured for crown-rump distance. All foetuses were given a gross external examination. Approximately one-half of the foetuses in each litter were placed in Bouin's solution for visceral examination and the remaining foetuses were fixed in alcohol and processed for skeletal examination of bone and cartilage. Foetal external, visceral and skeletal findings were classified as malformations or developmental variations.

All animals survived to scheduled euthanasia on Day 20 of gestation. The food consumption in the lowest dose group (2.6 mg/m^3) was comparable to the control over the GD 0-20 but, in the 4.4 and 6.3 mg/m^3 groups, food consumption was statistically higher than controls over GD 15-18, 18-20 and 0-20. These increases in food consumption corresponded with slight increase in body weight gains over these same intervals, but the differences in weight gain compared to controls were not statistically significant and according to the author not considered toxicologically meaningful. For all other intervals during gestation, food consumption in these groups was comparable to controls.

The mean antimony level in RBC were statistically higher than controls in each of the treated groups, 0.128 ± 0.0286, 3.275 ± 1.0391, 3.078 ± 0.5624 and 5.591 ± 1.3248 µg/g for the dose groups 0, 2.6, 4.4, 6.3 mg/m^3, respectively. These data do not show a clear dose-response relationship but the highest mean level of antimony was seen in the highest exposure group. This indicates a systemic exposure of the article and therefore it is likely that the fetuses were being exposed.

A dose-related increase in lung weights, absolute and relative to brain weights, was seen in the antimony trioxide-treated groups. These differences in lung weights from controls were statistically significant and considered indicative of a treatment-related response. The absolute lung weights were 24.2 %, 31.1 % and 38.6 % heavier than control in the 2.6, 4.4, and 6.3 mg/m^3 groups, respectively. Lung weights relatively to brain weights were 20.3 %, 26.3 %, and 34.8 % heavier, respectively.

Test article-related microscopic findings were observed in the lungs of all animals evaluated at all exposure levels. The primary test article-related microscopic change was a diffuse accumulation of pigmented alveolar macrophages, which likely reflected phagocytosis and accumulation of the test article particulate matter. These types of findings (pigmented alveolar macrophages) are common with exposure to particulate matter. Pulmonary alveoli contained variable numbers of macrophages with abundant eosinophilic cytoplasm with minimal to moderate quantities of brown granular pigment, as well as small to moderate quantities of extracellular eosinophilic proteinaceous material containing similar pigment. Throughout the lungs, scattered foci of acute inflammation (0/10, 7/10, 4/10 and 6/10 in control, 2.6, 4.4, and 6.3 mg/m^3 groups respectively) and type II cell hyperplasia (0/10, 5/10, 4/10 and 5/10 in control, 2.6, 4.4, and 6.3 mg/m^3 groups respectively) were observed. Accumulations of pigmented macrophages and associated inflammation were likely the cause of the increased lung weights of treated animals compared to controls. The microscopic finding of increased numbers of alveolar macrophages containing foreign material noted in the current study is similar to findings observed in previous subacute and chronic inhalation studies of antimony trioxide in Fischer rats. However, as would be expected, the inflammation and type II cell

hyperplasia noted in the current study was generally of acute to subacute duration as opposed to the granulomatous inflammation and interstitial fibrosis observed in the previous studies.

Pregnancy rates were comparable between the control and antimony trioxide-treated groups providing 25, 25, 26, and 25 GD 20 litters (26 females in each group, respectively) for evaluation in the control, 2.6, 4.4, and 6.3 mg/m^3 groups, respectively.

Uterine implantation data showed no statistical significant effect of treatment with diantimony trioxide on the mean number of corpora lutea, implantations, viable fetuses and mean preimplantation loss. An increase in the mean number of resorptions and postimplantation loss was observed in the 6.3 mg/m^3 group (Table 4-55), but these values did not differ statistically from controls (p-value 0.11) and were within the range of recent historical control data (4-8 %, mean 6 %).

Table 4-55 Mean number of resorptions and postimplantation

Exposure dose mg/m^3	Mean number of resorptions	Postimplantation loss %
0	0.5 ±0.82	3.37 ±5.45
2.6	0.5±0.77	3.59±5.36
4.4	0.5±0.76	3.48±5.55
6.3	1.0±1.19	7.11±8.6

Gravid uterine weights, adjusted GD 20 body weights, and adjusted body weight change GD 0-20 for the treated groups were comparable to controls.

No treatment related effects were evident from fetal body weights or from fetal crown-rump distance distinguished by sex and for both sexes combined. Mean fetal sex ratios (% male fetuses per litter) in the treated groups were comparable to controls. No skeletal malformations or ossification variation were seen among the control and treated fetuses.

During the external examinations, no malformations or developmental variations were seen in control or treated fetuses except for anophthalmia (absence of the eye), seen in a single fetus in the 6.3 mg/m^3 group (litter incidence 4.0 %). While this malformation has not been seen in recent historical control data for this laboratory, its low incidence in occurrence in this study was, according the author, considered spontaneous and unrelated to treatment. No other visceral malformations or developmental variation were seen among the fetuses. Likewise, no visceral malformations were seen in these fetuses from the 2.6 and 4.4 mg/m^3 groups or in control fetuses.

In conclusion, this study showed no statistically significant developmental toxicity at 2.6, 4.4 or 6.3 mg diantimony trioxide/m^3. A dose related increase in lung weight and a diffuse accumulation of pigmented alveolar macrophages, which likely reflected phagocytosis and accumulation of the test article particulate matter, was observed in the dams. Scattered foci of

acute inflammation (0/10, 7/10, 4/10 and 6/10 in control, 2.6, 4.4, and 6.3 mg/m^3 groups respectively) and type II cell hyperplasia (0/10, 5/10, 4/10 and 5/10 in control, 2.6, 4.4, and 6.3 mg/m^3 groups respectively) were also observed. It should also be noted that the exposure doses in this study were close in the three exposure groups. This is also reflected in the mean level of antimony in RBC, which did not show a dose dependent increase. The concentration of antimony in RBC was almost the same in the two lowest groups and barely twice as high in the highest dose group. This indicates that not three different dosage groups, which is the minimum number of dosage groups recommended in The OECD guideline 414, were achieved in this study. Maternal toxicity (increased lung weight, scattered foci of acute inflamation and type II cell hyperplasia) was observed at 2.6 mg/m^3. However, food intake and maternal body weight were not affected at any dose level.

This study suggests that the NOAEC for developmental toxicity is 6.3 mg/m^3, the highest exposure level evaluated.

The embryotoxic effect of inhaled antimony oxide has been investigated in rats (Grin et al., 1987). The original Russian publication is available in an English translation. Four groups of female albino rats, each group comprised 6-7 animals, were exposed to 0, 0.027 ± 0.0038, 0.082 ± 0.0047 or 0.27 ± 0.42 mg diantimony oxide/m^3 each. The purity and particle size of the test compound was not stated. The animals were exposed throughout gestation, 24 h/day for 21 days. After sacrifice, on the 21st day of pregnancy, all animals were autopsied and the following were determined: ovary, foetus and placenta weight, foetus craniocaudal distance, placenta diameter, foetal death parameters, external and internal embryo developmental abnormalities and certain biochemical parameters of the maternal rats and the foetus and its amniotic fluid.

No effects on any of the parameters studied were observed in animals exposed to 0.027 diantimony oxide/m^3. In the group exposed to 0.082 mg diantimony oxide/m^3 the embryo weight was according to the authors reliably (p value not stated) lower than that of control (2.93 ± 0.20 g versus 3.29 ± 0.10 g), however this effect was not observed in the high dose group (3.36 ± 0.23 g). Overall embryo mortality and preimplantation deaths was according to the authors reliably higher (p value not stated) in the mid dose group (19.91 ± 2.31 % and 14.63 ± 1.56 %). In this group there was no effect on postimplantation deaths (6.08 ± 2.82 %). In the high-dose group, the overall embryo mortality as well as embryo pre- and postimplantation deaths were significantly higher (p value not stated) (24.32 ± 4.29 %, 13.56 ± 2.69 % and 12.22 ± 4.90 %, respectively) compared to the control group (14.03 ± 1.92 %, 8.60 ± 1.45 % and 5.90 ± 2.10 %, respectively).

In the high-dose group a number of macroscopic embryotoxic effects were seen, such as haemorrhage into the foetal cerebral (arachnoid) membrane (15 %) and liver (10 %) and enlargement of the kidney cavity (10 %) and the cerebral ventricles (10 %). Similar observation albeit to a lesser extent was also observed in the embryos from the mid-dose group.

In the high-dose group, a statistically significant decrease ($p < 0.05$) in placenta weight (0.500 ± 0.013 g versus 0.519 ± 0.016 g) was recorded. In amniotic fluid from rats in the high dose group a statistically reliable increase in total protein and cholesterol levels was detected. It should be noted that the change in cholesterol content is especially symptomatic, since this

compound is the structural component of the cell membranes from which bile acid, female and male sex hormones, suprarenal cortex hormones are formed. In amniotic fluid from rats in the mid dose group a significant ($P < 0.05$) decrease in total protein levels and LDH activity was observed.

By the end of the pregnancy, there was a small reduction on female weight increase in the high dose group but no effect on weight increase was observed in the other dose groups. However no statistical calculations are available. In the high dose group a statistically reliable increase in total serum protein levels, blood haemoglobin and leukocyte count, erythrocyte peroxide haemolysis, and total serum lipids was observed. In the mid dose group a reduction of the lactate dehydrogenase (LDH) activity in the blood serum and reduced total lipid content was observed.

This study indicates that antimony oxide may cause embryo mortality, embryo pre- and postimplantation deaths and other embryo toxic effects. However, it is not clear whether the studied substance (only reported as antimony oxide) was antimony trioxide or antimony pentoxide. Likewise, neither purity nor particle size is reported. The study was reported in 1987 (in a journal which may or may not have a peer-review procedure; Gigiyena i Sanitariya) in a two-page publication, and it is not performed according to current guidelines. It is not reported how the diantimony oxide aerosol was generated or measured. With regard to the concentration of the diantimony trioxide aerosol, it is notable that the variations in concentration at the low exposure levels (0.027 and 0.082 mg/m^3) were 14 % and 6% of mean value (the unit of the variation is not stated), whereas the variation at the highest exposure level (0.27 mg/m^3) was 155 %. It could be doubted whether the extremely low concentrations used can actually be technically generated with such precision. Considering 24 hours exposure per day, it is assumed that it was whole-body exposure, but there is no information on how the animals were actually exposed (ventilation rate, etc). In conclusion, because of the lack of information on the substance being studied and how the study has been conducted the study is considered inconclusive.

Studies in humans

No data are available.

4.1.2.9.3 Summary of toxicity for reproduction

For fertility there are two non-standard fertility studies in animals, one on female rats and one on male rats and mice, and one human occupational report study. The inhalation study on female rats indicated that antimony trioxide might have an adverse effect on fertility after repeated exposure to 250 mg/m^3. However, the results are regarded inconclusive and cannot be used for risk assessment since the study report gives an unclear picture of the overall experimental conditions and the purity of the substance is not stated. The gavage study on male rats and mice showed no testicular toxicity after 4 weeks repeated exposure up to 1200 mg/kg bw. The human case report study on women occupationally exposed to antimony trioxide indicates that antimony trioxide might affect the fertility of female workers. However, this study is inconclusive due to the lack of information on the control group, the exposure situation and the overall workplace environment. Based on the fertility studies in

animals and humans no conclusion on female fertility can be made. However, a 90-day oral feeding study in male and female rats, of antimony trioxide reported no effects on reproductive organs up to a dose of 1686 mg/kg in males and 1879 mg/kg in females. It should be noted that in this study the fertility of the animals was not specifically assessed.

For developmental toxicity there is only one acceptable animal study available. However, it should be noted that the dose intervals are not according to guidelines and the high dose does not give relevant maternal toxicity. This rat inhalation study, with exposure 6 h/day throughout gestation, showed no statistically significant developmental toxicity at 2.6, 4.4 or 6.3 mg diantimony trioxide/m^3. Although a slight increase of resorptions and postimplantation loss was observed in the highest dose group, these values did not differ statistically from controls ($p = 0.11$) and were within the range of recent historical control data. This study suggests that the NOAEC for developmental toxicity is 6.3 mg diantimony trioxide/m^3.

4.1.3 Risk characterisation

4.1.3.1 General aspects

Physical/chemical properties

Diantimony trioxide is a solid substance at room temperature (melting point ≈ 655 °C) and is as a substance most often handled as solid powder; dry or in wetted form, pellets, or granules. The vapour pressure of solid diantimony trioxide is low (1 mm Hg at 574 °C) and it is very slightly soluble in most solvents. The content of antimony in diantimony trioxide is 83.5 % (w/w).

Particle size

The particle size has a crucial impact on the absorption via the lung, the localisation of diantimony trioxide in the respiratory tract, clearance from the respiratory tract and subsequently on the local effects. The particle size of diantimony trioxide differs between technical products and the particle size of diantimony trioxide powder is in a range of 0.85 μm to about 13 μm (0.8-1.5 μm is estimated by the manufacturers to be the most common particle size in flame retardant applications). The material with a physical particle size between 1.18 and 1.26 μm, corresponding to a MMAD (mass median aerodynamic diameter) of 4.12-4.33 μm, yielded the finest airborne particles after mechanical agitation (described in section 4.1.2.1. Toxicokinetics). Except for the two samples with the highest median physical particle size all other samples showed bimodal distributions, indicating that the diantimony trioxide particles have a tendency to agglomerate. In a workplace measurement survey of airborne dust, particle size distributions were estimated during conversion, refining and

packaging (final handling) in a diantimony trioxide producing plant using personal cascade impactors (Hughson 2005). The particle size distributions were quite variable, both within and between job titles. For example, data from the packaging workers showed that between 7 % and 35 % of the particles were equal to or smaller than 3.5 μm (aerodynamic diameter). Whether the diantimony trioxide content in the various particle size fractions was identical is not known. From that study a MMAD of 11.12 μm, with a geometric standard deviation (GSD) of 3.59, was estimated for exposure during packing (IAOIA-EBRC, 2008).

Possible differences in the particle size distribution between animal experiments and human scenarios need to be considered when a N(L)OAC from an inhalational experimental animal study is compared with a human inhalation exposure scenario. The controlled exposure used for animal exposure typically consists of a rather uniform particle size distribution in the region of respirable particle sizes, while also coarser particles are part of occupational exposure, typically measured as total or inhalable dust. Since particles of respirable size are to a larger extent deposited in the lung, and subject to pulmonary absorption, than larger particles, assuming that the human inhalable exposure is indeed respirable would tend to overestimate the human pulmonary exposure and consequently the risk with respect to pulmonary toxicity. However, it must also be kept in mind that a considerably higher lung deposition of respirable particles occur in humans compared to rats (Winter-Sorkina de and Cassee, 2002). This aspect would then cause an underestimation of the risk to humans when extrapolating from inhalational toxicity studies in rats. In addition, clearance of particles from the lung is faster in rats compared to humans (Hofmann and Asgharian, 2003) and this aspect also needs to be considered as it is the retained dose rather than the deposited dose that cause lung effects.

Altogether, the uncertainies in the different parameters (particle size distribution, particle deposition and particle clearance) influencing the retained dose are considered too large in order to be considered in a quantitative risk characterisation. Higher lung deposition and slower lung clearance in humans compared to rats counteract the possible lower ratio of respirable particles at work places compared to those used in the animal experiment. Industry has used the MPPD (multiple-path particle deposition) model to calculate the lung deposition ratios in rats (IAOIA-EBRC, 2008) using the particle size distribution from the chronic Newton study (Newton et al., 1994), compared to humans, using particle size distributions from the Production industry (Hughson, 2005), described in section 4.1.1.2.2. These calculations indicate that even though the particle size distribution in the working environment indicates larger and more diverse particle sizes (eg. MMAD=11.1 μm with GSD= 3.6 in packaging during Production) compared to the animal experiments (eg. MMAD= 3.8 μm with GSD= 1.8 in the Newton study), the percentage of the airborne diantimony trioxide that is deposited in the alveoli differs with a factor less than 2 (5.1% in the rat versus 3.7 % in humans when using the particle size distributions shown above).

For consumers, the particle size is to a large part higher than 10 μm. As a result, the percentage pulmonary deposition in consumers is lower than the pulmonary deposition in the Newton-study animals. This will be considered in a qualitative way in the risk characterisation.

Life-cycle stages

There are several industries in which antimony trioxide is produced or used, in the life cycle stages production, formulation (i.e. industrial use of antimony trioxide as an additive) and processing (i.e. industrial and professional use of semi- or end-products containing antimony trioxide). At some sites, both formulation and processing may take place. The end use of products containing antimony trioxide as well as recycling and disposal of articles containing antimony trioxide should also be taken into account. Information about exposure during all lifecycle stages except recycling and disposal is available. Exposure from recycling and disposal is however assumed to be limited compared to the exposure from production, formulation and industrial and professional use, and therefore this scenario is not considered further. In addition to production of diantimony trioxide, the following uses have been identified:

Use as catalyst in production of PET

Use as flame-retardant in production of plastics

Use as flame-retardant in treated textiles

Use in pigments, paints, coatings and ceramics

Use as flame-retardant in production of rubber

Use in production of crystal glass

Exposure

The human population may be exposed to diantimony trioxide at the workplace, from use of consumer products containing diantimony trioxide and indirectly via the environment through contact with contaminated air. Humans may also be exposed indirect via the environment through consumption of food, water and soil. However, the exposure will then be to the antimony ion. Since there is a continuous exposure to diantimony trioxide via the environment, humans may be exposed during their entire lifetime. In addition, there might be a combined exposure.

In the environment, diantimony trioxide will dissolve to the trivalent and predominantly to the pentavalent forms of antimony (as further described in the Environmental section 3.1.3). As a consequence, diantimony trioxide, originating from production and use, will be present as antimony in drinking water, food and breast milk.

Toxicokinetics

There are no quantitative data regarding the absorption, disposition and retention of diantimony trioxide in humans after inhalation, dermal or oral exposure available. However, antimony has been detected in blood and urine of workers exposed to antimony trioxide via inhalation, indicating that trivalent antimony can be absorbed from the lungs and excreted in the urine. Furthermore, the absorption of antimony through human skin has been measured in an *in vitro* percutaneous study which showed a dermal absorption value of 0.26. Based on this result dermal absorbtion of diantimony trioxide is considered negligible.

One study in rat has measured the absorption of antimony trioxide after oral exposure to diantimony trioxide suspension. The absorption was 0.3 % and 0.05 % at 100 mg/kg bw and 1000 mg/kg bw, respectively. It should however, be noted that the exposure levels in this study in rats are at least 2-3 orders of magnitude higher than the exposure levels for humans. Since the absorption of antimony trioxide particle suspension is clearly dose dependent and for "consumers" and "man via the environment" the source of antimony is mainly food and water and therefore, these groups are not likely orally exposed to particles of antimony trioxide, but to the antimony ion we are hesitant to use 0.3 % oral absorption derived from this study also for oral absorption for humans. Data published on intestinal absorption of more soluble antimony compounds than diantimony trioxide. indicate an average intestinal absorption of 3-8 % (0.15-40 %), although all these studies have been performed with different study protocols which do not meet current standards. Considering the data above and the poor solubility of diantimony trioxide 1 % oral absorption for diantimony trioxide is proposed.

Based on particle size, the deposition of inhaled antimony trioxide in the airways may be calculated by the use of the MPPD model. For diantimony trioxide deposited in the alveolar region 100 % absorption is assumed even though histopathology reveals alveolar accumulation of inhaled diantimony trioxide. This absorbtion value is chosen in the absence of relevant scientific data regarding alveolar antimony absorption. It should be noted that as no adverse systemic effect has been identified, the choice of 100 % alveolar absorption is of no significant importance to the risk assessment. Diantimony trioxide deposited in the upper airways is assumed to be transported via mucociliary transport to the gastrointestinal tract where absorption occurs. Therefore, the estimated total absorption via inhalation is 6.82 % (6.0 % deposited in the alveolar region + (81.6 % deposited in the upper airways corrected by 1 % oral absorption).

Studies in humans and laboratory animals show that antimony derived from inhalation of trivalent antimony is primarily distributed to lung, bone, liver, thyroid and pelt and is also concentrating in red blood cells. Excretion occurs both via faeces and urine but it seems like it is a relatively greater excretion in the faeces. Antimony is retained in the lungs for long periods of time (the biological elimination half times in humans has been estimated to 600-1100 days for non-smokers and 1700-3700 days for smokers) and accumulates in lung tissue after repeated exposure to diantimony trioxide in the air. The extent of deposition and subsequent clearance of antimony from the lung is primarily dependant on solubility and particle size. Excretion of antimony from the lung occurs in two phases, one rapid initial phase followed by a slower second phase. Antimony has also been detected in human foetal liver as well as in human breast milk, placenta, amniotic fluid and umbilical cord blood, indicating that antimony can be distributed to the foetus and excreted in breast milk, thus exposure may occur both in utero and during breast-feeding.

After oral exposure of diantimony trioxide to rats, antimony is distributed to most organs with the highest concentrations found in whole blood, thyroid and bone marrow (TNO, 2005). Levels of antimony can also be detected in testis and brain. Also after oral exposure the elimination occurs in two phases; a rapid decline of antimony in both faeces and urine lasting for about a week followed by a slower decrease lasting for more than 30 days.

Trivalent antimony may form a complex with GSH followed by excretion via the bile, release of antimony and enterohepatic recycling. Antimony does not seem to be methylated *in vivo*.

Effects

Acute toxicity. No information is available on the effects of single exposure via any route in humans. However, assessment of available animal data indicates that antimony trioxide has low acute toxicity by the oral, dermal and inhalation route. For acute toxicity in animals, there is one guideline study available for inhalation, which shows no signs of acute toxicity after inhalation exposure to antimony trioxide, indicating a LC_{50} > 5.20 mg/L (5200 mg/m^3). The animal studies on acute oral exposure are all old, they do not comply with today standards and in many of the studies mortality was the only parameter investigated. Still, they indicate that oral LD_{50} is in excess of 20000 mg/kg bw in rats. There is one valid study on dermal exposure, that indicates that the LD_{50} for dermal exposure is higher than 8300 mg/kg bw.

Irritation. The only animal study which can be used for risk assessment of the skin irritation potential of antimony oxide shows that antimony oxide is not irritating to rabbit skin. However, several human case report studies indicate that diantimony trioxide may cause dermatitis on skin damp with perspiration and the lesions appear to be closely associated with sweat ducts. Thus the lack of dermal irritation in rabbits may be explained by the fact that rabbits lack sweat glands. There are five human case report studies on workers, occupationally exposed to diantimony trioxide, where conjunctivitis and irritation to the eyes and/ or irritation in the respiratory tract have been described. However, there is little exposure data in these studies and therefore it is unclear wether or not diantimony trioxide was the causative agent. Two animal studies indicate that diantimony trioxide is mildly irritating when applied to the eyes of rabbits. One of the studies also shows that diantimony trioxide may cause necrosis of the lower conjunctivae and the nictitating membrane. However, neither of these effects fulfils the EU criteria for classification as irritating to eyes. There is one acute inhalation toxicity animal study available, which has also assessed the irritation potential of diantimony trioxide to the respiratory tract, indicating that diantimony trioxide is not irritating to the respiratory system. In conclusion, based on available animal data diantimony trioxide is not irritating to eyes or to the respiratory system. Based on practical experience in humans, diantimony trioxide should be classified as irritating to skin (R38).

Antimony trioxide is not considered to be a corrosive agent.

Sensitisation. There are no human studies of adequate quality that can be used for assessing the sensitising potential of antimony trioxide. However, there is one reliable animal study, performed according to TG 406 and GLP, which shows that antimony trioxide has no sensitising properties.

Repeated dose toxicity. The majority of the repeated dose studies in animals are considered inconclusive, either because they do not comply with current test guidelines or because both control and exposed animals showed signs of non-treatment related illness. Still, there are studies that indicate that diantimony trioxide is toxic to lung (Newton et al., 1994; Watt, 1983; Groth et al., 1986a; MPI, 2003). A NOAEC of 0.51 mg/m^3 is derived from the study by Newton et al. (1994) based on impaired lung clearance observed at 4.50 mg/m^3. Although there might be some uncertainty regarding the accuracy of the LOAEC and NOAEC numerical values, as the study had a high background incidence of lung inflammation in

control animals, the NOAEC of 0.51 mg/m^3 is used in the risk characterisation. Furthermore, acute pneumonia has been reported in a reproductive toxicity inhalation study performed by (MPI, 2003), supporting adverse lung effects at exposure levels ≥ 2.6 mg/m^3. In this context, it could be discussed whether effects caused by pulmonary overload in the rat is also relevant for humans. Positive (Hext, 1994; Oberdorster, 1995) and negative (Tran and Buchanan, 2000; Kuempel et al., 2001) findings of particle overload in human lungs are reported. Macrophage transport of particles into the alveolar interstitium is the major clearance mechanism in humans but of minor importance to the rat. These species differences are related to morphological features of the lung, i.e. to the relative short pathway length from the alveoli to the ciliated terminal bronchioles in rats (Bailey et al., 1989; Kreyling, 1990; Kreyling et al., 1991). In the absence of mechanistic data to the contrary, it must be assumed that the rat model can identify potential hazards to humans and the rat presently remains the appropriate model for both neoplastic and non-neoplastic responses to PSP exposure (ILSI Risk Science Institute Workshop Participants, 2000). Two repeated dose oral studies suggest that diantimony trioxide may be toxic to the liver based on a 10 % increase in liver weight, supported by significantly elevated ALP and ASAT levels (Sunagawa, 1981; Hext et al., 1999). However, in the absence of histological change or any clinical signs of antimony intoxication to support that the liver findings are adverse, the findings are regarded as adaptive or incidental to treatment with diantimony trioxide and a NOAEL of 1686 mg/kg/d for repeated dose toxicity is derived from these studies. In the absence of any measured systemic toxicity, no quantitative risk characterisation is performed for systemic repeated dose toxicity.

In humans, all the data comes from case report studies on workers employed in industries manufacturing diantimony trioxide. These studies indicate that repeated inhalation exposure to diantimony trioxide may cause pulmonary inflammation, lung emphysema and pneumoconiosis. Only isolated cases of changes in lung function were reported.

Mutagenicity. Considering the available genotoxicity data, antimony trioxide does not induce gene mutations *in vitro,* but induces structural chromosome aberrations in cultured mammalian cells *in vitro*. *In vivo* studies on the induction of chromosome aberrations and micronuclei in the bone marrow and unscheduled DNA synthesis in the liver have produced negative results. However, due to the poor bioavailability of diantimony trioxide via the oral route, it is not possible to conclude whether the negative *in vivo* results are due to lack of mutagenic potential or due to inability of the test to detect a mutagenic effect at the low concentrations achieved in the bone marrow following oral exposure. Still, it can be concluded that diantimony trioxide does not cause systemic mutagenicity *in vivo* after oral administration. Furthermore, the absorption by the dermal and inhalation route is also low and therefore it can be concluded that there is no concern for systemic mutagenicity after oral, dermal or inhalation exposure to diantimony trioxide . However, it is not possible to conclude on mutagenicity in specific site of contact tissues (local mutagenicity) and thus, whether the results are relevant also for the situation in the lung after inhalation exposure, which is the site where tumours have been found in the carcinogenicity studies. Despite the lack of conclusive data on local genotoxicity in the lung, the overall expert judgement by TC NES is that the most likely mechanism for carcinogenicity appears to be impaired clearance and particle overload followed by an inflammatory response, fibrosis and tumours. Further, *in vivo* data might suggest that a possible mutagenic potency of diantimony trioxide would be low and it is believed that a possible local genotoxic effect of diantimony trioxide would only be biologically relevant at concentration levels that also cause particle overload. Therefore, there

is also no concern for local mutagenicity of diantimony trioxide. In humans no induction of micronuclei or sister chromatid exchanges could be seen in lymphocytes from workers occupationally exposed to diantimony trioxide, but a higher proportion of the workers in the "high exposure group" showed oxidative DNA damage in their lymphocytes. However, "high exposure" is in the range of ambient air exposure, i.e. less than 0.001 mg/m^3, the workers were exposed to diverse chemicals and no monitoring was performed on the control group, therefore a correlation between the oxidative DNA damage and air concentration of diantimony trioxide is uncertain.

Carcinogenicity. Three chronic toxicity/ carcinogenicity studies in rats with inhalation exposure to antimony trioxide are available. Two of these studies indicate neoplastic properties of antimony trioxide, whereas one animal study showed negative results. There is also one human study available (Jones, 1994). However, due to lack of exposure data, the human study is regarded inconclusive. The exposure duration in all three animal studies is 12 months and thus all studies deviates from the OECD guideline on chronic toxicity/carcinogenicity, which prescribes an exposure period of 24 months for rats. In the first animal study (Watt, 1983) inhalation of 5.0 mg Sb_2O_3/m^3 for 12 months produced lung neoplasms in 44 % of the animals tested (only females were exposed). In the second study, (Groth et al., 1986a) a 9 times higher dose (45 mg Sb_2O_3/m^3) produced pulmonary neoplasms in 32 % of the female rats exposed under similar conditions, but none in male rats. It is noted that the female survival rate was significantly higher than the male counterparts in the study by Groth et al. (1986a). The differences in incidence between the studies might be explained by a longer observation period (12 months vs 20 weeks) and by the use of older animals (8 months vs 14 weeks) in the study by (Watt, 1983). The study by Newton et al. (1994) showed no diantimony trioxide-related lung tumours, neither in males nor females, at any dose level up to 4.5 mg/m^3. This is in contrast with the data reported by Watt and Groth and the cause to the difference is notentirely clear. However, a comparison of the Watt and the Newton studies, which were conducted at similar exposure levels, showed that the exposed rats had more lung damage and appeared to have considerably more antimony deposited in the lungs in the Watt study than in the Newton study. This may suggest that the exposure levels in the Watt study may have been above those reported. Given that the dose level in the study by Groth is 10 times higher and also the dose levels in the study by Watt were likely higher than 1.9 and 5.0 mg/m^3 the dose levels in the Newton study most probably fit in the dose range where no tumous were observed. The particle size, which will affect lung deposition, clearance and retention and hence target organ dose, was similar among the studies although they were all measured using different techniques. The three studies together also show a picture of impaired pulmonary clearance, pulmonary overload, macrophage infiltration, chronic interstitial pneumonia and fibrosis. Based on the data from all three studies it can be argued that the tumours found are due to pulmonary overload and subsequently to inflammation and neoplastic transformation of epithelial cells. In this context, it could be discussed whether effects caused by pulmonary overload in the rat is also relevant for humans. Positive (Hext, 1994; Oberdorster, 1995) and negative (Tran and Buchanan, 2000; Kuempel et al., 2001) findings of particle overload in human lungs are reported. However, in the absence of mechanistic data to the contrary, it must be assumed that the rat model can identify potential carcinogenic hazards to humans and the rat presently remains the appropriate model for both neoplastic and non-neoplastic responses to PSP exposure (ILSI Risk Science Institute Workshop Participants, 2000). Consequently, diantimony trioxide can be regarded as a threshold carcinogen and as a starting point for a quantitative risk characterisation the NOAEC of 0.51 mg/m^3 derived for local repeated dose toxicity is also

used for carcinogenicity (Newton et al., 1994) and is used in the risk characterisation. It is based on impaired lung clearance observed at 4.50 mg/m^3. There may be some uncertainty regarding the accuracy of the NOAEC numerical value as the study had a high background incidence of lung inflammation in control animals. In addition, the exposure duration was 12 months and thus deviates from the OECD guideline on chronic toxicity/carcinogenicity, which prescribes an exposure period of 24 months for rats. Based on the present data it is concluded that diantimony trioxide induces lung tumours in experimental animals and therefore antimony trioxide is currently classified in Annex 1, Directive 67/548/EEC as Carc. Cat. 3: R40 (Limited evidence of a carcinogenic effect).

Toxicity for reproduction. For fertility there are two non-standard fertility studies in animals, one on female rats and one on male rats and mice, and one human occupational report study. The inhalation study on female rats indicated that antimony might have an adverse effect on fertility after repeated exposure to 250 mg/m^3. However, the results are regarded inconclusive and cannot be used for risk assessment since the study report gives an unclear picture of the overall experimental conditions and the purity of the substance is not stated. The gavage study on male rats and mice showed no testicular toxicity after 4 weeks repeated exposureup to 1200 mg/kg bw. A 90-day oral feeding study of diantimony trioxide in male and female rats, reported no histopathological changes on reproductive organs up to a dose of 1686 mg/kg in males and 1879 mg/kg in females. The human case report study on women occupationally exposed to diantimony trioxide indicates that diantimony trioxide might affect the fertility of female workers. However, this study is inconclusive due to the lack of information on the control group, the exposure situation and the overall workplace environment. Based on the fertility studies in animals and humans no conclusion on female fertility can be made. However, the rat 90-day oral feeding study of antimony trioxide, reported no effects on reproductive organs up to a dose of 1879 mg/kg in females. For male fertility no testicular toxicity after four weeks repeated exposure up to 1200 mg/kg bw was observed and the rat 90-day oral feeding study showed no histopathological changes in epididymis or testes up to a dose of 1686 mg/kg/d.

For developmental toxicity there is only one acceptable animal study available (MPI, 2003). However, it should be noted that the dose intervals are not according to guidelines and the high dose does not give relevant maternal toxicity. This inhalation study, with exposure 6 h/day throughout gestation, showed no statistically significant developmental toxicity at 2.6, 4.4 or 6.3 mg diantimony trioxide/m^3.

Based on these studies there is no concern for reproductive toxicity and thus no quantitative risk characterisation will be made for the fertility or developmental toxicity.

Calculation of MOS- and reference MOS values

For risk assessment the MOS approach as outlined in the TGD (Human health Risk Characterisation, Final draft) is applied.

Table 4-56 Summary of critical effect measures. NOAEL/ NOAEC values are given for repeated dose and developmental toxicity. For acute toxicity LD_{50} values are given. The skin irritating properties and the genotoxic and carcinogenic potential are indicated by a + sign.

Endpoint	In vitro	Inhalation measure	Dermal measure	Oral measure
Acute toxicity		5.20 mg/L (5 200 mg/m³)	8 300 mg/kg bw	20 000 mg/kg bw
Irritation / corrosivity		-	+	-
Sensitization		-	-	-
Repeated dose toxicity		0.51 mg/m³	-	1 686 mg/kg/d
Mutagenicity	+	?	-	-
Carcinogenicity		0.51 mg/m³	-	-
Fertility impairment				1 686 mg/kg bw
Developmental toxicity		6.3 mg/m³		

The question mark on mutagenicity indicate that there are no available inhalation studies for this endpoint.

4.1.3.2 Workers

There are several industries in which diantimony trioxide is produced or used, and the life cycle stages where occupational exposure may occur are; production, formulation, i.e. industrial use of diantimony trioxide as an additive, and processing, i.e. industrial and professional use of semi- or end-products containing diantimony trioxide. At some sites, both formulation and processing may take place. In addition, exposure might occur during recycling and disposal of articles containing diantimony trioxide, but there is no available information about this. It is assumed that exposure during recycling and disposal is limited compared to the exposure from production, formulation and industrial and professional use, and therefore this scenario is not considered further in this exposure assessment.

In addition to production of diantimony trioxide, the following uses have been identified:

- Use as catalyst in production of PET
- Use as flame-retardant in production of plastics
- Use as flame-retardant in treated textiles
- Use in pigments, paints, coatings and ceramics
- Use as flame-retardant in production of rubber
- Use in production of crystal glass

The following data were used for the occupational exposure assessments of diantimony trioxide:

- measured workplace data from production and uses of diantimony trioxide
- physico-chemical data of diantimony trioxide
- qualitative data, such as process description and use pattern of the product, and quantitative data regarding frequency and duration of handling of diantimony trioxide
- concentrations of diantimony trioxide used in the different products

For all scenarios of occupational exposure, a respiratory volume of 10 m^3/work day is used.

The main routes of occupational exposure to diantimony trioxide are inhalation of airborne solid dust and dermal exposure to solid diantimony trioxide. Dermal exposure may occur during direct handling, either by contamination of skin surfaces or by dermal deposition of airborne dust. However, due to negligible dermal absorbtion of antimony, dermal exposure is not calculated in the quantitative risk characterisation.

4.1.3.2.1 Acute toxicity

Inhalation

For acute inhalation toxicity there is one animal study which has been performed according to OECD TG 403 and which shows no signs of acute toxicity after inhalation exposure to diantimony trioxide, indicating a LC_{50} > 5.20 mg/ L (5200 mg/m^3).

Considering the inhalation exposure levels for workers reported in the exposure assessment part, workers are not expected to be exposed to diantimony trioxide in the range of hazardous dose. Therefore, the substance is of no concern for workers in relation to inhalation acute toxicity and **conclusion (ii)** is reached.

Dermal

LD_{50} for dermal exposure is higher than 8300 mg/kg bw.

Considering the dermal exposure levels for workers reported in the exposure assessment part and negligible dermal absorbtion, workers are not expected to be exposed to diantimony trioxide in the range of hazardous dose. Therefore, the substance is of no concern for workers in relation to dermal acute toxicity and **conclusion (ii)** is reached for acute dermal toxicity.

4.1.3.2.2 Irritation and corrosivity

Diantimony trioxide is considered irritating to skin. Given the skin irritating potential of diantimony trioxide it is concluded that the substance is of concern for workers with regard to skin irritation and **conclusion (iii)** is reached to indicate the need for classification. Once classified the conclusion (iii) will be changed to conclusion (ii).

Diantimony trioxide is not irritating to eyes or respiratory tract and **conclusion (ii)** is reached for these endpoints.

Diantimony trioxide is not considered to be a corrosive agent and **conclusion (ii)** is reached.

4.1.3.2.3 Sensitisation

Diantimony trioxide is not a sensitizer and **conclusion (ii)** is reached.

4.1.3.2.4 Repeated dose toxicity

Repeated inhalation exposure to diantimony trioxide gives local toxic effects in the lung and a NOAEC of 0.51 mg/m^3 is derived from a 12 month inhalation exposure study in rat, supported by observations of acute pneumonia in a 19 days inhalation developmental toxicity study. No systemic toxicity is observed after repeated exposure, therefore no quantitative risk characterisation is performed for systemic repeated dose toxicity.

Inhalation (local)

To calculate MOS-values for local pulmonary toxicity the exposure levels (see Table 4-57) should be compared with a corrected NOAEC of 0.26 mg/m^3 calculated as follows: The experimental NOAEC of 0.51 mg/m^3 adapted by a factor of 6/8 to account for differences between the experimental inhalation duration of 6 h per day and the average working day of 8 h per day and then multiplied by a factor of 6.7/10 for activity driven differences of respiratory volumes in workers (0.51 mg/m^3 · 6/8 · 6.7/10). The achieved MOS-values are then compared with a reference MOS of 12.5 (see below).

The following assessment factors are applied in the setting of a reference MOS to a local effect in the lung;

- a factor of 2.5 for interspecies differences
- a factor of 5 for intraspecies differences; this covers the variation in sensitivity expected between workers

Table 4-57 Occupational risk assessment for repeated dose toxicity (local effects). The NOAEC value is compared with typical and reasonable worst-case (RWC) exposures to calculate MOS values.

	Inhalation (local effects)							
	Typical exposure* (mg/m³)	Corrected NOAEC (mg/m³)	MOS	Conclu-sion	RWC exposure* (mg/m³)	Corrected NOAEC (mg/m³)	MOS	Conclu-sion
Production of Diantimony Trioxide								
Conversion, with RPE	0.027	0.26	9.6	(iii)	0.15	0.26	1.7	(iii)
Refuming, with RPE	0.012	0.26	22	(ii)	0.047	0.26	5.5	(iii)
Final handling, with RPE	0.040	0.26	6.5	(iii)	0.11	0. 26	2.4	(iii)
Conversion, without RPE					2.9	0.26	0.09	(iii)
Refuming, without RPE					0.94	0.26	0.28	(iii)
Final handling, without RPE					2.1	0.26	0.12	(iii)
Use as a catalyst in production of PET								
Powder handling	0.002	0.26	130	(ii)	0.026	0. 26	10	(iii)
Use as flame-retardant in								

production of plastics								
Raw material handling	0.13	0. 26	2	(iii)	0.57	0. 26	0.46	(iii)
Use as flame-retardant in treated textiles								
Formulation	0.13	0. 26	2	(iii)	0.57	0. 26	0.46	(iii)
Processing	<0.001	0. 26	-		0.001	0. 26	260	(ii)
Further handling	negl.	0. 26	-		negl.	0. 26	-	
Use in pigments, paints, coatings and ceramics								
Loading and mixing	0.036	0. 26	7.2	(iii)	0.16	0. 26	1.6	(iii)
Use as flame-retardant in production of rubber								
Formulation	0.051	0. 26	5.1	(iii)	0.22	0. 26	1.2	(iii)
Processing	0.064	0. 26	4	(iii)	0.14	0. 26	1.9	(iii)
Use in production of crystal glass								
Cutting	0.003	0. 26	87	(ii)	0.015	0. 26	17	(ii)

*Exposure values from sections 4.1.1.2.2-8

It can be seen that there is a need for limiting the risks (conclusion (iii) for a number of occupational exposure scenarios.

4.1.3.2.5 Mutagenicity

It is concluded that diantimony trioxide does not cause systemic mutagenicity by the oral, dermal or inhalation route of exposure. It is not possible to conclude on mutagenicity in specific site of contact tissues (local mutagenicity) and thus, whether the results are relevant for the situation in the lung after inhalation exposure, which is the site where tumours have been found in the carcinogenicity studies. However, the overall expert judgement by TC NES is that the most likely mechanism for diantimony trioxide induced lung carcinogenicity appears to be impaired clearance and particle overload followed by an inflammatory response, fibrosis and tumours. Further, it is believed that a possible local genotoxic effect of diantimony trioxide would only be biologically relevant at concentration levels that also cause particle overload. Therefore, there is also no concern for local mutagenicity of diantimony trioxide. In conclusion, there is no concern for genotoxicity and **conclusion (ii)** is reached.

4.1.3.2.6 Carcinogenicity

Diantimony trioxide is considered to be a carcinogenic substance and is classified for carcinogenicity. Although the mechanism for pulmonary tumour formation is still unclear it may be assumed that particle deposition followed by macrophage infiltration, pulmonary inflammation and impaired clearance are pivotal initial steps in the process. Consequently, diantimony trioxide can be regarded as a threshold carcinogen and as a starting point for a

quantitative risk characterisation the NOAEC of 0.51 mg/m^3 derived for local repeated dose toxicity is also used for carcinogenicity.

Inhalation

To calculate MOS-values for pulmonary carcinogenicity the exposure levels (see Table 4-57) should be compared with a corrected NOAEC of 0.26 mg/m^3 calculated as follows: The experimental NOAEC of 0.51 mg/m^3 adapted by a factor of 6/8 to account for differences between the experimental inhalation duration of 6 h per day and the average working day of 8 h per day and then multiplied by a factor of 6.7/10 for activity driven differences of respiratory volumes in workers (0.51 mg/m^3 · 6/8 · 6.7/10). The achieved MOS-values are then compared with a reference MOS of 12.5 (see below).

The following assessment factors are applied in the setting of a reference MOS to a local effect in the lung;

- a factor of 2.5 for interspecies differences
- a factor of 5 for intraspecies differences; this covers the variation in sensitivity expected between workers

Table 4-58 Occupational risk assessment for pulmonary carcinogenicity. The NOAEC value is compared with typical and reasonable worst-case (RWC) exposures to calculate MOS values.

	Inhalation (pulmonary carcinigenicity)							
	Typical exposure* (mg/m^3)	Corrected NOAEC (mg/m^3)	MOS	Conclu-sion	RWC exposure* (mg/m^3)	Corrected NOAEC (mg/m^3)	MOS	Conclu-sion
Production of Diantimony Trioxide								
Conversion, with RPE	0.027	0.26	9.6	(iii)	0.15	0.26	1.7	(iii)
Refuming, with RPE	0.012	0.26	22	(ii)	0.047	0.26	5.5	(iii)
Final handling, with RPE	0.040	0.26	6.5	(iii)	0.11	0. 26	2.4	(iii)
Conversion, without RPE					2.9	0. 26	0.09	(iii)
Refuming, without RPE					0.94	0. 26	0.28	(iii)
Final handling, without RPE					2.1	0. 26	0.12	(iii)
Use as a catalyst in production of PET								
Powder handling	0.002	0.26	130	(ii)	0.026	0. 26	10	(iii)
Use as flame-retardant in production of plastics								
Raw material handling	0.13	0. 26	2	(iii)	0.57	0. 26	0.46	(iii)
Use as flame-retardant in treated textiles								

Formulation	0.13	0. 26	2	(iii)	0.57	0. 26	0.46	(iii)
Processing	<0.001	0. 26	-		0.001	0. 26	260	(ii)
Further handling	negl.	0. 26	-		negl.	0. 26	-	
Use in pigments, paints, coatings and ceramics								
Loading and mixing	0.036	0. 26	7.2	(iii)	0.16	0. 26	1.6	(iii)
Use as flame-retardant in production of rubber								
Formulation	0.051	0. 26	5.1	(iii)	0.22	0. 26	1.2	(iii)
Processing	0.064	0. 26	4	(iii)	0.14	0. 26	1.9	(iii)
Use in production of crystal glass								
Cutting	0.003	0. 26	87	(ii)	0.015	0. 26	17	(ii)

*Exposure values from sections 4.1.1.2.2-8

It can be seen that there is a need for limiting the risks (conclusion (iii) for a number of occupational exposure scenarios.

4.1.3.2.7 Toxicity for reproduction

Effects on fertility

A gavage study on male mice showed no testicular toxicity after four weeks repeated exposure up to 1200 mg/kg bw. In a rat 90-day oral feeding study no histopathological changes were observed in epididymis or testes up to a dose of 1686 mg/kg/d nor in female reproductive organs up to a dose of 1879 mg/ kg. Therefore, there is no concern for male or female fertility and **conclusion (ii)** is reached.

Developmental toxicity

For developmental toxicity there is only one acceptable animal study available. However, it should be noted that the dose intervals are not according to guidelines and the high dose does not give relevant maternal toxicity. This inhalation study, with exposure 6 h/day throughout gestation, showed no statistically significant developmental toxicity at 2.6, 4.4 or 6.3 mg diantimony trioxide/m^3. There is no concern for developmental toxicity and **conclusion (ii)** is reached.

4.1.3.2.8 Summary of risk characterisation for workers

When considering the risks to human health arising from occupational exposure to diantimony trioxide, the key areas are for local pulmonary toxicity and carcinogenicity. Therefore it is considered that risk reduction measures are required and **conclusion (iii)** applies to most exposure scenarious. Although control measures are available in these industry sectors, there is no evidence that the appropriate equipment is in place in all EU

workplaces, and that it is used in the correct manner. Furthermore, in the diantimony trioxide production industry, where the risk was assessed both with and without use of RPE, **conclusion (iii)** is reached even when RPE is taken into account. In addition, there is concern for skin irritation and **conclusion (iii)** is reached to indicate the need for classification.

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4.1.3.3 Consumers

There is no known direct private use of diantimony trioxide as such. However, diantimony trioxide is used in several products, some of which are available to consumers. Some examples of end products containing diantimony trioxide or antimony are: PET, cuddly toys, flat and pile upholstered furniture (residential and commercial furniture), upholstery seating and interior textiles in private and public transportation, textiles, wall coverings, electrical and electronic equipment e.g. distribution boxes for electrical lines, polyvinyl chloride wire and cable. Due to wear and tear of the diantimony trioxide containing materials, the material will partly be abraded into small particles. These particles will become a part of the inhouse dust. Consequently, exposure of consumers todiantimony trioxide may occur via the inhalation and dermal route. Although originating from the use of diantimony trioxide the actual oral exposure via PET-bottles and cuddly toysis not to diantimony trioxide itself but to antimony. To calculate the concentration of diantimony trioxide, the measured levels of antimony are adjusted with a factor of 1.2 (correction for molecular weight). This approach is taken in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide.

The specific hand-to-mouth behaviour of small children may play a particular role for their exposure. Therefore oral exposure to diantimony trioxide for children via sucking on cuddly toys and ingestion of dust has been estimated.

Four scenarios for consumer exposure are presented. The values are based on measured data (Consumer exposure, section 4.2.1.3).

Scenario No. 1 PET-bottle: The reasonable worst-case *oral exposure* is 0.035 µg/kg bw/day for *an adult* drinking from a PET bottle.

Scenario No. 2 fabrics: The worst-case *dermal* exposure is 1.8 µg/kg bw/day for *an adult* sitting on upholstery fabric.

Scenario No. 3 cuddly toys: For *a child* sucking on cuddly toys the reasonable worst-case *oral exposure* is 0.25 µg/kg bw/day.

Scenario No. 4 indoor air: The reasonable worst-case exposure of diantimony trioxide via indoor air is $3.15 \cdot 10^{-6}$ mg/m^3.

For *a child* the reasonable worst-case *oral exposure*, via ingestion of dust from indoor air, is 0.60 µg/kg bw/day.

4.1.3.3.1 Acute toxicity

For acute inhalation toxicity there is one animal study which has been performed according to OECD TG 403 and which shows no signs of acute toxicity after inhalation exposure to diantimony trioxide, indicating a LC_{50} > 5.20 mg/ L (5200 mg/m^3). The animal studies on acute oral exposure are all old, they do not comply with today standards and in most of them mortality was the only parameter investigated. Still, they indicate that oral LD_{50} is in excess of 20,000 mg/kg bw in rats. However, already at a dose of 160-225 mg/ kg bw a discoloration of the liver was observed. There is one valid study on dermal exposure, which indicates that the LD_{50} for dermal exposure is higher than 8300 mg/kg bw.

Considering the low exposure levels for consumers reported in the exposure assessment part, consumers are not expected to be exposed to animony trioxide in the range of hazardous doses. Therefore, the substance is of no concern for consumers in relation to inhalation, oral or dermal acute toxicity and **conclusion (ii)** is reached.

4.1.3.3.2 Irritation and corrosivity

Diantimony trioxide is considered irritating to skin. However, considering the low exposure levels expected for consumers there is no concern for skin irritation and **conclusion (ii)** is reached.

4.1.3.3.3 Sensitisation

Diantimony trioxide is not a sensitizer and **conclusion (ii)** is reached.

4.1.3.3.4 Repeated dose toxicity

Repeated inhalation exposure to diantimony trioxide gives local toxic effects in the lung and a NOAEC of 0.51 mg/m^3 is derived from a 12 month inhalation exposure study in rat, supported by observations of acute pneumonia in a 19 days inhalation developmental toxicity study. No systemic toxicity is observed after repeated exposure, therefore no quantitative risk characterisation is performed for systemic repeated dose toxicity.

Inhalation

Scenario No. 4 indoor air: Based on measured data a realistic worst-case inhalation exposure to diantimony trioxide via indoor air has been calculated. House-dust containes relatively high amounts of antimony. The 90th percentiles in one publication were close to 50 µg antimony/g dust, corresponding to 60 µg diantimony trioxide /g dust. When taking the CSOIL estimate for particulate matter (dust) in indoor air of 52.5 µg/m^3 into consideration (Otte et al., 2001) 60 µg diantimony trioxide/g dust corresponds to 3.15 ng diantimony trioxide/m^3.

The average non-occupational indoor residence time, for both men and women (25-69 years) during both summer and winter, is 15.3 h/day (German Ausschuss für Umwelthygiene, 2000).

Local effects: To calculate a MOS-value for local pulmonary toxicity for exposure via indoor air, the exposure level (see Table 4-57) should be compared with a corrected NOAEC of 0.20 mg/m^3 calculated as follows: The experimental NOAEC of 0.51 mg/m^3 adapted by a factor of 6/15.3 to account for differences between the experimental inhalation duration of 6 h per day and the average of non-occupational indoor residence time of 15.3 h/day (0.51 mg/m^3 · 6/15.3). The achieved MOS-value of 63 492 (0.20 mg/m^3 / 3.15 ng/m^3) is then compared with a reference MOS of 25 (see below) and **conclusion (ii)** is reached.

The following assessment factors are applied in the setting of a reference MOS;

- a factor of 10 for intraspecies differences; this covers the variation in sensitivity expected in the whole human population

- a factor of 2.5 for interspecies differences

4.1.3.3.5 Mutagenicity

It is concluded that diantimony trioxide does not cause systemic mutagenicity by the oral, dermal or inhalation route of exposure. It is not possible to conclude on mutagenicity in specific site of contact tissues (local mutagenicity) and thus, whether the results are relevant for the situation in the lung after inhalation exposure, which is the site where tumours have been found in the carcinogenicity studies. However, the overall expert judgement by TC NES is that the most likely mechanism for the diantimony trioxide induced lung carcinogenicity appears to be impaired clearance and particle overload followed by an inflammatory response, fibrosis and tumours. Further, it is believed that a possible local genotoxic effect of diantimony trioxide would only be biologically relevant at concentration levels that also cause particle overload. Therefore, there is also no concern for local mutagenicity of diantimony trioxide. In conclusion, there is no concern for genotoxicity and **conclusion (ii)** is reached.

4.1.3.3.6 Carcinogenicity

Antimony trioxide is considered to be a carcinogenic substance and is classified for carcinogenicity. Although the mechanism for pulmonary tumour formation is still unclear it may be assumed that particle deposition followed by macrophage infiltration, pulmonary inflammation and impaired clearance are pivotal initial steps in the process. Consequently, diantimony trioxide can be regarded as a threshold carcinogen and as a starting point for a quantitative risk characterisation the NOAEC of 0.51 mg/m^3 derived for local repeated dose toxicity is also used for carcinogenicity.

Scenario No. 4 indoor air: To calculate a MOS-value for pulmonary carcinogenicity for exposure via indoor air, the exposure level (see Table 4-57) should be compared with a corrected NOAEC of 0.20 mg/m^3 calculated as follows: The experimental NOAEC of 0.51 mg/m^3 adapted by a factor of 6/15.3 to account for differences between the experimental inhalation duration of 6 h per day and the average of non-occupational indoor residence time of 15.3 h/day (0.51 mg/m^3 · 6/15.3). The achieved MOS-value of 63 492 (0.20 mg/m^3 / 3.15 ng/m^3) is then compared with a reference MOS of 25 (see below) and **conclusion (ii)** is reached.

The following assessment factors are applied in the setting of a reference MOS;

- a factor of 10 for intraspecies differences; this covers the variation in sensitivity expected in the whole human population
- a factor of 2.5 for interspecies differences

4.1.3.3.7 Toxicity for reproduction

Effects on fertility

A gavage study on male mice showed no testicular toxicity after four weeks repeated exposure up to 1200 mg/kg bw. In a rat 90-day oral feeding study no histopathological changes were observed in epididymis or testes up to a dose of 1686 mg/kg/d nor in female reproductive organs up to a dose of 1879 mg/ kg. Therefore, there is no concern for male or female fertility and **conclusion (ii)** is reached.

For developmental toxicity there is only one acceptable animal study available. However, it should be noted that the dose intervals are not according to guidelines and the high dose does not give relevant maternal toxicity. This inhalation study, with exposure 6 h/day throughout gestation, showed no statistically significant developmental toxicity at 2.6, 4.4 or 6.3 mg antimony trioxide /m^3 There is no concern for developmental toxicity and **conclusion (ii)** is reached.

4.1.3.3.8 Summary of risk characterisation for consumers

The risk characterization for consumers results in no concern and **conclusion (ii)** is reached for all endpoints.

4.1.3.4 Humans exposed via the environment

The sources of human exposure to diantimony trioxide handled in this chapter are food, breast milk, water and air. Diantimony trioxide is released to the environment through air effluents and wastewater from manufacture, formulation, processing, use and disposal of diantimony trioxide containing products. In the environment, diantimony trioxide, originating from production/use of diantimony trioxide, will dissolve to the trivalent and predominantly pentavalent ions. As a consequence, the actual exposure from drinking water, food and breast milk will be to the antimony ions.

Antimony is also a naturally occurring element. Therefore, it's presence in the environment, and thereby also indirect in water and in food and beverages produced from agricultural goods, may also be attributed to natural sources.

To calculate the concentration of diantimony trioxide, the measured levels of antimony are adjusted with a factor of 1.2 (correction for molecular weight). This approach is taken in order to enable comparison between exposure values and effect data, which are based on diantimony trioxide.

Antimony is present at trace levels in food and beverages. According to a study by Moll and Moll (2000), the main contributors to antimony intake are cereals, sweeteners, fish and crustaceans, fruits and vegetables and alcoholic beverages.

Four regional scenarios for human exposure to diantimony trioxide via the environment are presented. The exposure values are based on measured data (see exposure via the environment).

Scenario No. 1 Food: The reasonable worst-case *oral* exposure via food is 0.096 µg/kg bw/day for *adults*.

Scenario No. 2 Breast milk: The reasonable worst-case *oral* exposure via breast milk is 0.087 µg/kg bw/day for *infants* during the first 0-3 months.

Scenario No. 3 Drinking water: A reasonable worst-case *oral* exposure via drinking water is estimated to 0.029 µg/kg bw/day.

Scenario No. 4 Outdoor air: The reasonable worst case concentration in outdoor air is 3.12 ng/m^3.

Local exposure: The maximum local concentration in air is 1 µg/m^3. The maximum local exposure via water is 2.8 µg/kg/day. It should however be noticed that this exposure is a gross overestimate as it is based on untreated surface water which is not representative for drinking water in the EU.

4.1.3.4.1 Acute toxicity

For acute inhalation toxicity there is one animal study which has been performed according to OECD TG 403 and which shows no signs of acute toxicity after inhalation exposure to diantimony trioxide, indicating a LC_{50} > 5.20 mg/ L (5200 mg/m^3). The animal studies on acute oral exposure are all old, they do not comply with today standards and in most of them mortality was the only parameter investigated. Still, they indicate that oral LD_{50} is in excess of 20,000 mg/kg bw in rats. However, already at a dose of 160-225 mg/kg bw a discoloration of the liver was observed. There is one valid study on dermal exposure, which indicates that the LD_{50} for dermal exposure is higher than 8300 mg/kg bw.

Considering the low regional and local exposure levels for man via the environment reported in the exposure assessment part, environmental exposure to diantimony trioxide is not expected to be in the range of hazardous doses. Therefore, the substance is of no concern for man via the environment in relation to inhalation and oral acute toxicity and **conclusion (ii)** is reached.

4.1.3.4.2 Irritation and corrosivity

Diantimony trioxide is considered irritating to skin. However, considering the low regional and local exposure levels expected from environmental exposure there is no concern for irritation and **conclusion (ii)** is reached.

4.1.3.4.3 Sensitisation

Diantimony trioxide is not a sensitizer and **conclusion (ii)** is reached.

4.1.3.4.4 Repeated dose toxicity

Repeated inhalation exposure to diantimony trioxide gives local toxic effects in the lung and a NOAEC of 0.51 mg/m^3 is derived from a 12 month inhalation exposure study in rat, supported by observations of acute pneumonia in a 19 days inhalation developmental toxicity study. No systemic toxicity is observed after repeated exposure, therefore no quantitative risk characterisation is performed for systemic repeated dose toxicity.

Regional exposure

Outdoor air: The reasonable worst case concentration in outdoor air is 3.12 ng/m^3. To calculate a MOS-value for local pulmonary toxicity for exposure via outdoor air, the exposure level (seeTable 4-57) should be compared with a corrected NOAEC of 0.38 mg/m^3 calculated as follows: The experimental NOAEC of 0.51 mg/m^3 adapted by a factor of 6/8 to account for differences between the experimental inhalation duration of 6 h per day and the average of outdoor residence time of 8 h/day (0.51 mg/m^3 ・ 6/8). The achieved MOS-value of 121 800 (0.38 mg/m^3 / 3.12 ng/m^3) is then compared with a reference MOS of 25 (see below) and **conclusion (ii)** is reached.

The following assessment factors are applied in the setting of a reference MOS;

- a factor of 10 for intraspecies differences; this covers the variation in sensitivity expected in the whole human population
- a factor of 2.5 for interspecies differences

Local exposure

The maximum local concentration in air is 1 µg/m^3. To calculate a MOS-value for local pulmonary toxicity for local exposure via outdoor air, the exposure level (seeTable 4-57) should be compared with a corrected NOAEC of 0.38 mg/m^3 calculated as follows: The experimental NOAEC of 0.51 mg/m^3 adapted by a factor of 6/8 to account for differences between the experimental inhalation duration of 6 h per day and the average of outdoor residence time of 8 h/day (0.51 mg/m^3 ・ 6/8). The achieved MOS-value of 380 (0.38 mg/m^3 / 1 µg/m^3) is then compared with a reference MOS of 25 (see below) and **conclusion (ii)** is reached.

The following assessment factors are applied in the setting of a reference MOS;

- a factor of 10 for intraspecies differences; this covers the variation in sensitivity expected in the whole human population
- a factor of 2.5 for interspecies differences

4.1.3.4.5 Mutagenicity

It is concluded that diantimony trioxide does not cause systemic mutagenicity by the oral, dermal or inhalation route of exposure. It is not possible to conclude on mutagenicity in specific site of contact tissues (local mutagenicity) and thus, whether the results are relevant for the situation in the lung after inhalation exposure, which is the site where tumours have

been found in the carcinogenicity studies. However, the overall expert judgement by TC NES is that the most likely mechanism for the diantimony trioxide induced lung carcinogenicity appears to be impaired clearance and particle overload followed by an inflammatory response, fibrosis and tumours. Further, it is believed that a possible local genotoxic effect of diantimony trioxide would only be biologically relevant at concentration levels that also cause particle overload. Therefore, there is also no concern for local mutagenicity of diantimony trioxide. In conclusion, there is no concern for genotoxicity and **conclusion (ii)** is reached.

4.1.3.4.6 Carcinogenicity

Diantimony trioxide is considered to be a carcinogenic substance and is classified for carcinogenicity. Although the mechanism for pulmonary tumour formation is still unclear it may be assumed that particle deposition followed by macrophage infiltration, pulmonary inflammation and impaired clearance are pivotal initial steps in the process. Consequently, diantimony trioxide can be regarded as a threshold carcinogen and as a starting point for a quantitative risk characterisation the NOAEC of 0.51 mg/m^3 derived for local repeated dose toxicity is also used for pulmonary carcinogenicity.

Regional exposure

Outdoor air: The reasonable worst case concentration in outdoor air is 3.12 ng/m^3. To calculate a MOS-value for pulmonary carcinogenicity for exposure via outdoor air, the regional exposure level (see Table 4-57) should be compared with a corrected NOAEC of 0.38 mg/m^3 calculated as follows: The experimental NOAEC of 0.51 mg/m^3 adapted by a factor of 6/8 to account for differences between the experimental inhalation duration of 6 h per day and the average of outdoor residence time of 8 h/day (0.51 mg/m^3 · 6/8). The achieved MOS-value of 121 800 (0.38 mg/m^3 / 3.12 ng/m^3) is then compared with a reference MOS of 25 (see below) and **conclusion (ii)** is reached.

The following assessment factors are applied in the setting of a reference MOS;

- a factor of 10 for intraspecies differences; this covers the variation in sensitivity expected in the whole human population
- a factor of 2.5 for interspecies differences

Local exposure

The maximum local concentration in air is 1 µg/m^3. To calculate a MOS-value for pulmonary carcinogenicity for local exposure via outdoor air, the exposure level (see Table 4-57 should be compared with a corrected NOAEC of 0.38 mg/m^3 calculated as follows: The experimental NOAEC of 0.51 mg/m^3 adapted by a factor of 6/8 to account for differences between the experimental inhalation duration of 6 h per day and the average of outdoor residence time of 8 h/day (0.51 mg/m^3 · 6/8). The achieved MOS-value of 380 (0.38 mg/m^3 / 1 µg/m^3) is then compared with a reference MOS of 25 (see below) and **conclusion (ii)** is reached.

The following assessment factors are applied in the setting of a reference MOS;

- a factor of 10 for intraspecies differences; this covers the variation in sensitivity expected in the whole human population

- a factor of 2.5 for interspecies differences

4.1.3.4.7 Toxicity for reproduction

Effects on fertility

A gavage study on male mice showed no testicular toxicity after four weeks repeated exposure up to 1200 mg/kg bw. In a rat 90-day oral feeding study no histopathological changes were observed in epididymis or testes up to a dose of 1686 mg/kg/d nor in female reproductive organs up to a dose of 1879 mg/ kg. Therefore, there is no concern for male or female fertility and **conclusion (ii)** is reached .

Developmental toxicity

For developmental toxicity there is only one acceptable animal study available. However, it should be noted that the dose intervals are not according to guidelines and the high dose does not give relevant maternal toxicity. This inhalation study, with exposure 6 h/day throughout gestation, showed no statistically significant developmental toxicity at 2.6, 4.4 or 6.3 mg diantimony trioxide/m^3. There is no concern for developmental toxicity and **conclusion (ii)** is reached.

4.1.3.4.8 Summary of risk characterisation for exposure via the environment

The risk characterization for humans exposed via the environment results in no concern and **conclusion (ii)** is reached for all endpoints.

4.1.3.5 Combined exposure

Due to the use of diantimony trioxide in the society and the diffuse emissions from products, humans may be exposed from different sources. The total exposure (body burden) is the summary of all the specific exposures. The most important sources of human exposure to diantimony trioxide are probably identified. Additions of individual scenarios are not considered to change any of the conclusions, and no calculation on combined exposure has therefore been performed.

4.2 HUMAN HEALTH (PHYSICO-CHEMICAL PROPERTIES)

Flammability, explosive and oxidising properties are not considered to form a hazard hence further characterisation is not undertaken in this report and **conclusion (ii)** is reached. In addition, there is no need for further information and/or testing with regard to physico-chemical properties.

5 RESULTS [11]

5.1 INTRODUCTION

Diantimony trioxide is a solid substance at room temperature and is mostly handled as solid powder; dry or in wetted form, pellets, paste, or granules. The particle size of diantimony trioxide differs between different technical products. The vapour pressure of solid diantimony trioxide is low and it has a low solubility in most solvents.

The major use of diantimony trioxide is as a flame-retardant. However, it does not itself have flame-retarding properties; instead it is a synergist for halogenated flame-retardants in plastics, paints, adhesives, sealants, rubber, and textile back coatings. Other uses of diantimony trioxide include: as polymerisation catalyst used in PET resin manufacture and as a clarifying aid in certain glasses, and in pigments. Approximately 25000 tonnes per year are used in EU, mainly (>70%) in the production of flame-retarded plastics (PVC and non-PVC). Diantimony trioxide is presently produced in four plants in EU.

Diantimony trioxide is released to the environment via emissions to air, waste water, surface water and soil from manufacture, formulation, processing, use and disposal of diantimony trioxide, but also via coal combustion and refuse incineration, non-ferrous metal production (e.g. Cu), and road traffic.

The human population may be exposed to diantimony trioxide at the workplace, from use of consumer products containing diantimony trioxide and indirectly via the environment through contact with contaminated air. In the environment,t diantimony trioxide will dissolve to the trivalent and pentavalent forms of antimony. Consequently, humans may also be exposed indirectly via the environment to the antimony ion through consumption of food, water and soil.

5.2 ENVIRONMENT

The results summarised here are presented in more details in section 3.3 “Risk characterization”.

The compartments of concern are: fresh water sediment (generic scenarios for formulation and application of flameretardant textile back-coating and one production site), marine water (generic scenario only) and marine sediment (generic scenario only).

[11] Conclusion (i) There is a need for further information and/or testing.
Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.
Conclusion (iii) There is a need for limiting the risks; risk reduction measures which are already being applied shall be taken into account.

Aquatic compartment

Surface water

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Sediment

Conclusion (iii) There is a need for limiting the risks; risk reduction measures which are already being applied shall be taken into account.

This conclusion applies to the generic scenarios for formulation and application of flameretardant textile back-coating and to one production site (site P1).

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all other scenarios, including nineteen sites using diantimony trioxide in textile applications and three production sites, that all report releases.

Waste water treatment plants

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Terrestrial compartment

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Atmosphere

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

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Secondary poisoning

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Marine compartment

PBT-assessment

There is currently no agreed approach to perform a PBT-assessment of a metal, therefore a PBT-assessment will not be performed.

Marine water

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Despite having RCR >1 conclusion iii) is not drawn for application of flame-retardant back-coating. The reason for this is that, according to information from IAOIA, none of the sites covered by the survey IAOIA performed to collect exposure data from all their customers is located by the sea. However, it has to be pointed out that the coverage of this survey regarding textile backcoating sites was rather low. Therefore, it cannot be ruled out that textile backcoating sites located at the sea having emissions to the marine environment may exist.

Marine sediment

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

Despite having RCR >1 conclusion iii) is not drawn for formulation and application of flame-retardant back-coating. The reason for this is that, according to information from IAOIA, none of the sites covered by the survey IAOIA performed to collect exposure data from all their customers is located by the sea. For the formulation of flame-retardant in textiles, the coverage of this survey is high and there is a high probability that for this use area the marine scenario may not be relevant. For application of textile back-coating on the other hand the coverage of the survey is lower and it cannot be ruled out that sites located at the sea having emissions to the marine environment may exist.

Secondary poisoning in the marine environment

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

This conclusion applies to all scenarios.

5.3 HUMAN HEALTH

5.3.1 Human health (toxicity)

The results summarised here are presented in detail in section 4.1.3 "Risk characterization for Human health".

Human populations exposed to diantimony trioxide include workers, consumers and humans exposed via the environment. Indirect exposure via the environment to the antimony ion may also occur as diantimony trioxide is readily dissolved to the trivalent and pentavalent antimony ions in the environment. For exposure assessment, both measured data, analogues data, calculations and modelling have been used.

The endpoints of concern are: skin irritation, local pulmonary toxicity and carcinogenicity.

Repeated inhalation exposure to diantimony trioxide gives local toxic effects in the lung and a NOAEC of 0.51 mg/m^3 is derived from a 12 month inhalation exposure study in rat, supported by observations of acute pneumonia in a 19 days inhalation developmental toxicity study. No systemic toxicity was observed after repeated exposure.

Diantimony trioxide is considered to be a carcinogenic substance and is classified for carcinogenicity. Although the mechanism for pulmonary tumour formation is still unclear it may be assumed that particle deposition followed by macrophage infiltration, pulmonary inflammation and impaired clearance are pivotal initial steps in the process. Consequently, diantimony trioxide can be regarded as a threshold carcinogen and as a starting point for a quantitative risk characterisation the NOAEC of 0.51 mg/m^3 derived for local repeated dose toxicity is also used for carcinogenicity.

5.3.1.1 Workers

Conclusion (iii) There is a need for limiting the risks; risk reduction measures which are already being applied shall be taken into account.

Conclusion (iii) applies to skin irritation for all scenarios to indicate the need for classification. Once classified, **conclusion (iii)** will be changed to **conclusion (ii)**.

Conclusion (iii) also applies to repeated dose toxicity (local pulmonary toxicity after inhalation) and carcinogenicity (pulmonary carcinogenicity) for the following scenarios: **Production of diantimony trioxide:** Conversion, Refuming and Final handling with and without RPE, **Use as a catalyst in production of PET**: Powder handling, **Use as flame-retardant in production of plastics**: Raw material handling, **Use as flame-retardant in treated textiles**: Formulation, **Use in pigments, paints, coatings and ceramics:** Loading and mixing, **Use as flame-retardant in production of rubber:** Formulation and Processing.

5.3.1.2 Consumers

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

Conclusion (ii) applies to all end points and all exposure scenarios.

5.3.1.3 Humans exposed via the environment

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

Conclusion (ii) applies to all end points and exposure sources.

5.3.1.4 Combined exposure

The most important sources of human exposure to diantimony trioxide are probably identified. Additions of individual scenarios are not considered to change any of the conclusions, and no calculation on combined exposure has therefore been performed.

Conclusion (iii) There is a need for limiting the risks; risk reduction measures which are already being applied shall be taken into account.

Conclusion (iii) applies to workers (see workers above).

5.3.2 Human health (risks from physico-chemical properties)

Conclusion (ii) There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already.

Conclusion (ii) applies to flammability and explosive and oxidising properties. These properties are not considered to form a hazard, hence further risk characterisation has not been undertaken. In addition, there is no need for further information and/or testing with regard to physico-chemical properties.

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6 REFERENCES

Anonymous. Table of data from contract 68-01-4646. U.S. EPA 1978. In-depth studies on health and environmental impacts of selected water pollutants. (Table of data available from Charles E. Stephan, U.S. EPA, Duluth, MN.). 1978;.

Anonymous. Nationalencyclopedin. 1994; Bokförlaget Bra Böcker, Höganà4s.

Anonymous. Inside pages, Polyethylene Terephtalate. European Chemical News 2000; 3-9 July: 20-.

Anonymous. Update of the risk assessment of bis(pentabromodiphenyl)ether, final environmental draft. 2004; May:

Anonymous. Comments from PET industry January 2006. 2006;.

2003/33/EC. Council Decision of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 of and Annex II to directive 1999/31/EC. Official Journal of the European Communities 2003; L 11/27-LL 11/49.

Abu-Hilal AH and Riley JP. The spectrophotometric determination of antimony in water, effluents, marine plants and silicates. Anal Chim Acta 1981; 131: 175-186.

Ainsworth N. Distribution and biological effects of antimony in contaminated grasslands. Thesis. 1988;.

Ainsworth N, Cooke JA and Johnson MS. Distribution of antimony in contaminated grassland: 1-Vegetation and soils. Environ Pollut 1990a; 65: 65-77.

Ainsworth N, Cooke JA and Johnson MS. Distribution of antimony in contaminated grassland: 2-small mammals and invertebrates. Environ Pollut 1990b; 65: 79-87.

Ainsworth N, Cooke JA and Johnson MS. Biological significance of antimony in contaminated grassland. Water Air Soil Pollut 1991; 57-58: 193-199.

Al-Sibaai AA and Fogg AG. Stability of dilute standard solutions of antimony, arsenic, iron and rhenium used in colorimetry. Analyst 1973; 98: 732-738.

Alagna G and D´Aquino S. Disturbance of the eye from cobalt chloride. Archivio di Ottalmologia 1956; 60: 5-29.

Albéric P, Viollier E, Jézéquel D, Grosbois C and Michard G. Interactions between trace elements and dissolved organic matter in the stagnant anoxic deep layer of a meromictic lake. Limnol Oceanogr 2000; 45: 1088-1096.

Allen RO and Steinnes E. Proc. Int. Conf. Heavy Metal in the Environment, London. Contribution from long-range atmospheric transport to the heavy metal pollution of surface soil. 1979; pp 271-274.

Amiard-Triquet C, Lassus P, Amiard JC, Devineau J and Denuit C. Influence des procedures experimentales sur la determination de la toxicite aigue de quelques polluants a Légard de divers organismes marins et euryhalins. Colloq INSERM 1982; 106: 485-496.

Anderson BG. The apparent thresholds of toxicity to *Daphnia magna* for chlorides of various metals when added to Lake Erie water. Trans Am Fish Soc 1948; 78: 96-113.

Andreae MO.The determination of the chemical species of some of the "hydride elements" (arsenic, antimony, tin and germanium) in seawater: methodology and results. In Trace metals in sea water. Edited by Wong CS, Boyle E, Bruland KW, Burton JD and Goldberg ED 1983; pp 1-19. Plenum, New York.

Andreae MO, Asmodé J-F, Foster P and Van´t dack L. Determination of antimony(III), antimony(V), and methylantimony species in natural waters by atomic absorption spectrometry with hydride generation. Anal Chem 1981; 53: 1766-1771.

Andreae MO and Froelich PN. Arsenic, antimony, and germanium biogeochemistry in the Baltic Sea. Tellus Ser B Chem Phys Meteorol 1984; 36B: 101-117.

Andrewes P, Cullen WR and Polishchuk E. Antimony biomethylation by Scopulariopsis brevicaulis: characterization of intermediates and the methyl donor. Chemosphere 2000; 41: 1717-1725.

APME, Association of Plastic Manufacturers in Europe. An analysis of plastics consumption and recovery in Western Europe 1999. APME - Association of Plastics Manufacture in Europe 2001a;

APME, Association of Plastic Manufacturers in Europe. Plastics - a material of innovation for the electrical and electronic industry. 2001b;.

APME, Association of Plastic Manufacturers in Europe. Recycling of bromine from plastics containing brominated flame retardant in state-of-the-art combustion facilities, a technical report. APME, Association of Plastic Manufacturers in Europe 2002;

Appelgate VC, Howell JH and Hall Jr AE. Toxicity of 4,346 chemicals to larval lampreys and fishes. Spec Sci Rep Fish, No 207 1957; 1-157.

Apte SC and Howard AG. Determination of dissolved inorganic antiomony(V) and antimony(lll) species in natural waters by hydride generation atomic absorption spectrometry. J Anal At Spectromet 1986; 1: 221-225.

Apte SC, Howard AG, Morris RJ and McCartney MJ. Arsenic, antimony and selenium speciation during a spring phytoplankton bloom in a closed experimental ecosystem. Marine Chem 1986; 20: 119-130.

Aras NK and Kumulainen JT. Trace elements in Turkish wheat and human diets. Proceedings of the technical Workshop on trace elements, natural antioxidants and contaminants (Finland: Helsinki-Espoo 25-26 August). 1995; www.mtt.fi/etl/etk/fao/fao.htm.

Arimoto R, Duce RA, Ray BJ, Ellis Jr WG, Cullen JD and Merrill JT. Trace elements in the atmosphere over the North Atlantic. J Geophys Res 1995; 100: (D1): 1199-1213.

Arnot JA and Gobas FAPC. A review of bioconcentration factor (BCF) and bioaccumulation factor (BAF) assessments for organic chemicals in aquatic orgaisms. Environ Rev 2006; 14: 257-297.

Ashley PM, Craw D, Graham BP and Chappell DA. Environmental mobility of antimony around mesothermal stibnite deposits, New South Wales, Australia and southern New Zealand. J Geochem Explor 2003; 77: 1-14.

DRAFT

ATSDR. Toxicological profile for antimony. 1992; Department of Health & Human Services, USA.

Austin LS and Millward GE. Atmosphere-coastal ocean fluxes of particulate arsenic and antimony. Continental Shelf Research 1985; 6: 459-474.

Baccini P, Henseler G, Figi R and Belevi H. Water and element balances of municipal solid waste landfills. Waste Manag Res 1987; 5: 483-499.

Bailey MR, Kreyling WG, Andre S, Batchelor A, Collier CG, Drosselmeyer E, Ferron GA, Foster P, Haider B, Hodgson A, Masse R, Metivier H, Morgan A, Müller H-L, Patrick G, Pearman I, Pickering S, Ramsden D, Stirling C and Talbot RJ. An interspecies comparison of the lung clearence of inhaled monodisperse cobalt oxide particles - Part 1: objectives and summary of results. Journal of Aerosol Science 1989; 20: (2): 169-188.

Bailly R, Lauwerys R, Buchet JP, Mahieu P and Konings J. Experimental and human studies on antimony metabolism:their relevance for the biological monitoring of workers exposed to inorganic antimony. Br J Ind Med 1991; 48: 93-97.

Barbante C, Schwikowski M, Döring T, Gäggeler HW, Schotterer U, Tobler L, van de Velde K, Ferrari C, Cozzi G, Turetta A, Rosman K, Bolshov M, Capodaglio G, Cescon P and Boutron C. Historical record of European emissions of heavy metals to the atmosphere since the 1650s from alpine snow/ice cores drilled near Monte Rosa. Environ Sci Technol 2004; 38: 4085-4090.

Barghigiani C, D`Ulivo A, Lampugnani L, Pellegrini G, Romboli L and Zamboni R.Distribution of As, Hg, Sb and Se in some sediments of the Venice Lagoon. In Mediterraneanchem. Taranto, 23-27 Maggio 1995 Atti del Congresso p. 139-140 ND ICAS:A2-03/95. 1995;.

Beijer K and Jernelöv A.Sources, transport and transformation of metals in the environment. In Handbook on the Toxicology of Metals, 2nd edition. Edited by Friberg L, Nordberg GF and Vouk V 1986; pp 68-84. Elsevier Science Publishers B.V., Amsterdam.

Belyaeva AP. The effect produced by antimony on the generative function. Labor Hygiene and Occupational Dieses 1967; 11: 32-37.

Belzile N, Chen Y-W and Wang Z. Oxidation of antimony (III) by amorphous iron and manganese oxyhydroxides. Chem Geol 2001; 174: 379-387.

Benson AA.Antimony metabolites in marine algae. In The biological alkylation of heavy elements. Edited by ??? 1998; pp 135-137. ??, ??

Berg JE and Skyberg K.123. Antimony. In The Nordic Expert Group for Criteria Documentation of Health Risks from Chemicals. Edited by Kjellberg A 1998; 11, pp 1-37. Arbete och Hälsa, Arbetslivsinstitutet, Solna.

Bertine KK and Lee DS.Antimony content and speciation in the water column and interstitial waters of Saanich inlet. In Trace metals in sea water. 1983; pp 21-38.

BfR Expert Opinion. Migration of toxicologically relevant substances from toys. Risike erkennen - Gesundheit schützen 2003; 31/1: 1-6.

BGAA. Diantimontrioxide: Exposition am areitsplatz, berufsgenossenschaftlicher arbeitskreis altstoffe. Expositionsbeschreibung N4. 2001; Sept., 48, Bundesreplublik Deutschland.

Bird ML and Challenger F. J Chem Soc 1939; 163-169.

Birge WJ.Aquatic toxicology of trace elements of coal and fly ash. In Dep Energy Symp Ser, Energy and Environmental Stress in Aqatic Systems, GA. Edited by Thorp JH and Gibbons JW 1978; pp 219-240.

Birge WJ, Black JA, Westerman AG and Hudson JE.Aquatic toxicity tests on inorganic elements occurring in oil shale. In Oil shale symosium: sampling, analysis and quality assurance, March 1979. Edited by Gale C 1980; pp 519-534. EPA-600/9-80-022, US EPA, Cincinatti, OH:519-534, US NTIS PB80-221435.

Biros SM, Bridgwater BM, Villeges-Estrada A, Tanski JT and Parkin G. Antimony Ethylene Glycolate and Catecholate Compounds: Structural Characterization of Polyesterification Catalysts. Inorganic Chemistry 2002; 41: (15): 4051-4057.

BLIC and van der Velde K. BLIC feedback in context of Antimony trioxide risk assessment, personal communication. 2006;.

Bock W, Kleine H, Pfeiffer W, Pflaumbaum W, Stamm R, Stückrath M and Blom H. Übersicht über die Expositionssituation bei krebserzeugenden Gefahrstoffen. Die BG Heft 1995; 1:

Bockting GJM, van de Plassche EJ, Struijs J and Canton JH. RIVM-reoprt no.: 679101003. Soil-water partitioning coefficients for some trace metals. 1992;.

Bognar A, Schelenz R, Grünewald T, Frahm H, Heine K and Wiechen A. Mineralstoff- und Spurenelementgehalt von Speisen. Berichte der Bundesfoschungsanstalt f_FCr Ernährung. 1981;, Karlsruhe.

Bondavalli C, Croce E, Meloni S, Oddone M and Triulzi C. A chemical characterization of a lagoon ecosystem: The Sacca Di Goro (Po River Delta, Italy). Chem Ecol 1996; 12: 279-286.

Bonner NA. The exchange reaction between antimony(III) and antimony(V) in hydrochloric acid solution. J Am Chem Soc 1949; 71: 3909-3914.

Bonner NA and Goishi W. Antimony in HCl solutions. Kinetics of complex exchange and hydrolysis reactions. J Am Chem Soc 1961; 83: 85-99.

Bonotto S, Bossus A, Nuyts G, Kirchmann R, Mathot P, Colard J and Cinelli F.Experimental uptake study of ^{60}Co, ^{137}Cs, ^{125}Sb and ^{65}Zn in four marine algae. In Radioactive wastes and the ocean. Edited by Killho P 1983; pp 287-302. John Wiley & Sons, Inc., USA.

Borg H. Trace metals in Swedish natural fresh waters. Hydrobiol 1983; 101: 27-34.

Borgmann U, Norwood WP and Dixon DG. Re-evaluation of metal bioaccumulation and chronic toxicity in Hyalella azteca using saturation curves and the biotic ligand model. Environ Pollut 2004; 131: 469-484.

Borgmann U, Couillard Y and Grapentine LC. Relative contribution of food and water to 27 metals and metalloids accumulated by caged Hyalella azteca in two rivers affected by mining. Environ Pollut 2007; 145: 753-765.

Borm PJA, Schins RPF and Albrecht C. Inhaled particles and lung cancer. Part B: Paradigms and risk assessment. International Journal of Cancer 2004; 110: 3-14.

Bowen HJM. Trace elements in Biochemistry. 1966; Academic Press, London and New York.

Bowen HJM. Environmental Chemistry of the Elements. 1979; Academic Press, London.

Brannon JM and Patrick WH. Fixation and mobilization of antimony in sediments. Environ Pollut [B] 1985; 9: 107-126.

Brewer NR and Cruise LJ.Chapter 4: Physiology. In The Biology of the Laboratory Rabbit, 2:a ed. Edited by Manning PJ, Ringler DH and Newcomer CE 1994; pp 67. Academic Press.

Brihaye C, Gillain G and Duyckaerts G. Determination fo traces of metals by anodic stripping voltammetry at a rotating glassy carbon ring-disc electrode. Anal Chim Acta 1983; 148: 51-57.

Bringmann G. Bestimmung der biologishen schadwirkung wassergefährdender stoffe gegen protozoen. Z Wasser- und Abwasser-Forschung 1978; 11: (6): 210-215.

Bringmann G and Kuhn R. Grenzwerte der schadwirkung wassergefährdender stoffe gegen blaualgen (Microcystis aeruginosa) und grunalgen (Scenedesmus quadricauda) im zellvermehrugshemmtest. Vom Wasser 1978; 50: 45-60.

British standard. Guide to implementing an effective respiratory protective device programme. 1997; BS4275:

Brooke LT, Call DJ, Poirier SH, Lindberg CA and Markee TP. Acute toxicity of antimony III to several species of freshwater organisms. Report to Battelle Memorial Research Institute, Columbus, OH 43201 In partial fulfillment of work assignment no 45, task order no 4, subcontract no F-4114(8834)-411 1986; 1-12.

Brookins DG. Eh-pH Diagrams for Geochemistry. 1988; Springer-Verlag, New York.

Brooks RR and Rumsby MG. The biogeochemistry of trace element uptake by some New Zealand bivalves. Limnol Oceanogr 1965; 10: 521-527.

Brown RA and Swift EH. The formal potential of the antimonous-antimonic half cell in hydrochloric acid solution. J Am Chem Soc 1949; 71: 2719-2732.

BSEF and IAOIA. Final report on flame retardant release from textiles. Polymer research centre report. 2006; 01, 16, Surrey.

Buccafusco RJ, Ells SJ and LeBlanc GA. Acute toxicity of priority pollutants to Bluegill (Lepomis macrochirus). Bull Environ Contam Toxicol 1981; 26: 446-452.

Buchter B, Davidoff B, Amacher MC, Hinz C, Iskandar IK and Selim HM. Correlation of Freundlich Kd and n retention parameters with soils and elements. Soil Sci 1989; 148: 370-379.

Budavari, S. (ed.). The Merck Index – An Encyclopedia of Chemicals, Drugs, and Biologicals. Whitehouse Station, NJ: Merck and Co., Inc., p. 120

Buschmann J and Sigg L. Antimony(III) binding to humic substances: Influence of pH and type of humic acid. Environ Sci Technol 2004; 38: 4535-4541.

Butte W and Heinzow B. Pollutants in house dust as indicators in indoor contamination. Reviews of Environmental Contamination and Toxicology 2002; 175: 1-46.

Butterman W and Carlin JJF. Mineral commodity profiles - Antimony Open file report 03-019. US Geological Survey 2004;

Cabrera F, Clemente L, Díaz Barrientos E, López R and Murillo JM. Heavy metal pollution of soils affected by the Guadimar toxic flood. Sci Total Environ 1999; 242: 117-129.

Carbol P, Skålberg M and Skarnemark G. Cesium and antimony behaviour in water and forest soil after the Chernobyl accident. Radiochim Acta 1995; 69: 259-270.

Carlsberg. http://www.carlsberg.se.

Carlsson E. Användning, utsläpp och ackumulering av arsenik och antimon vidd Orrefors glasbruk. Degreeproject work 2002:M18. Kalmar University (in Swedish). 2002;.

Carnegie-Mellon Institute of Industrial Research UoP. The single dose and subacute toxicity of antimony oxide (Sb_2O_3). EPA/OTS; Doc # 878210812. 1945; pp 1-3.

Carnegie-Mellon Institute of Research UoP. Antimony trioxide range-finding toxicity studies. EPA/OTS; Doc # 878210813. 1978; rev.1983, pp 1-4.

Cato I.Proceedings of the Fourth Marine Geological Conference-the Baltic. In Contaminants in the Skagerrak and Kattegat sediments. Edited by Klingberg F 1997; pp 21-35. Geological Survey of Sweden, Uppsala, Sweden.

Cavallo D, Iavicoli I, Setini A, Marinaccio A, Perniconi B, Carelli G and Iavicoli S. Genotoxic risk and oxidative DNA damage in workers exposed to antimony trioxide. Environ Mol Mutagen 2002; 40: 184-189.

Challenger F.Biosynthesis of organometallic and organometalloidal compounds. In American Chemical Society, ACS Symp. Ser. No. 82. Edited by Brinckman FE and Bellama JM 1978; pp 1-22., Washington DC.

Challenger F, Higginbottom C and Ellis L. The formation of organo-metalloidal compounds by microorganisms. J Chem Soc 1933; 5: 95-101.

Challenger F and North HE. J Chem Soc 1934; 68-71.

Chan D and Stachowiak GW.Review of automotive brake friction materials. In Proc Instn Mech Engrs, Part D: J Automobile Engineering. 2004; 218, pp 953-966. IMechE.

Chapman WH, Fisher HL and Pratt MW. UCRL-50564. Concentration factors of chemical elements in edible aquatic organisms. 1968;.

Chattopadhyay A and Jervis RE. Multielement determination in market-garden soils by instrumental photon activation analysis. Anal Chem 1974; 46: 1630-1639.

Chau YK, Wong PTS, Kramar O and Bengert GA. in: *Proc. 3rd Int. Conf. Heavy Metals in the Environment*, Amsterdam, The Netherlands, 14 Sept. 1981. 1981; CEP Consult. Ltd, Edingburgh.

Cheek CH, Bonner NA and Wahl AC. Antimony(III)-antimony(V) exchange reaction in HCl solutions. J Am Chem Soc 1961; 83: 80-84.

Chen Y-W, Deng T-L, Filella M and Belzile N. Distribution and early diagenesis of antimony species in sediments and porewaters of freshwater lakes. Environ Sci Technol 2003; 37: 1163-1168.

Cherrie J W. Personal and static sample measurements are related. Occup Environ Med 2004; 61: 374-375.

Chertok RJ and Lake S. Availability in the dog of radionuclides in nuclear debris from the plowshare excavation cabriolet. Health Phys 1970; 19: (sept.): 405-409.

CIIT. Multiple-path Particle Deposition Model. Chemical Industry Institute of Toxicology (CIIT), National Institute of Public Health and the Environment (RIVM), the Netherlands, and The Ministry of Housing, Spatial Plannaing and the Environment, the Netherlands 2002-2006;

Clemente GF, Ingrao G and Santaroni GP. The concentration of some trace elements in human milk from Italy. Sci Total Environ 1982; 24: 255-265.

Cole J, Fox M, Garner RC, McGregor DB and Thacker J.Gene mutation assays in cultured mammalien cells. In Basic Mutagenicity Tests. UKEMS recommended procedures. 1990; pp 87-114. Cambridge University Press, Cambridge.

Cooper DA, Pendergrass EP, Vorwald AJ, Mayock RL and Brieger H. Pneumoconiosis among workers in an antimony industri. The American Journal of Roentgenology Radium therapy and nuclear medicine 1968; 103: 495-508.

Cornfield AH. Effects of addition of 12 metals on carbon dioxide release during incubation of an acid sandy soil. Geoderma 1977; 19: 199-203.

Cotton FA and Wilkinson G.13. The Group V Elements: P, As, Sb, Bi. In Advanced Inorganic Chemistry. 1972; pp 367-402. Interscience Publishers, New York.

Couillard Y, Grapentine LC, Borgmann U, Doyle P and Masson S. The amphipod Hyalella azteca as a biomonitor in field deployment studies for metal mining. Environ Poll 2008, doi:10.1016/j.envpol.2008.03.001.

Coughtrey PJ, Jackson D and Thorne MC.Chapter 26. Antimony. In Radionuclide distriibution and transport in terrestrial and aquatic ecosystems. A critical review of data. 1983; 3, pp 263-304. A.A.BALKEMA, Rotterdam.

Coussens LM and Werb Z. Inflammation and cancer. Nature 2002; 420: 860-867.

Covance Laboratories Ltd. Evaluation of micronuclei and chromosome aberrations in the bone marrow of Sprague Dawley rats following a 21 day repeated exposure to antimony trioxide. Covance Laboratory Ltd 2005; 2515/2-d6172: 66-.

Craig PJ, Jenkins RO, Dewick R and Miller DP. Trimethylantimony generation by Scopulariopsis brevicaulis during aerobic growth. Sci Total Environ 1999; 229: 83-88.

Crecelius EA. The geochemical cycle of arsenic in Lake Washington and its relation to other elements. Limnol Oceanogr 1975; 20: (3): 441-451.

Crecelius EA. The solubility of coal fly ash and marine aerosols in seawater. Marine Chem 1980; 8: 245-250.

CTL, Central Toxicology Laboratory. Antimony trioxide: 90 day feeding study in rats. CTL report 1997; 5145: 1-86.

Cunningham WC. Radiochemical determination of As, Cr, Mo, Sb and Se in foods. J Radioanal Nucl Chem 1987; 113: 423-430.

Curtis MW and Ward CH. Aquatic toxicity of forty industrial chemicals: testing in support of hazardous substance spill prevention regulation. J Hydrol 1981; 51: 359-367.

Cutter GA. Dissolved arsenic and antimony in the Black Sea. Deep-Sea Research 1991; 38: 825-843.

Cutter GA. Kinetic controls on metalloid speciation in seawater. Marine Chem 1992; 40: 64-80.

Cutter GA. Metalloids in wet deposition on Bermuda: concentrations, sources, and fluxes. J Geophys Res 1993; 98: 16777-16786.

Cutter GA and Cutter LS. Metalloids in the high latitude North Atlantic Ocean: Sources and internal cycling. Marine Chem 1998; 61: 25-36.

Cutter GA, Cutter LS, Featherstone AM and Lohrenz SE. Antimony and arsenic biogeochemistry in the western Atlantic Ocean. Deep-Sea Res II 2001; 48: 2895-2915.

D´Souza TJ and Mistry KB. Behaviour of gamma-emitting fission products 106Ru, 125Sb. 134Cs and 144Ce Deposited on established pastures in tropical environs. J Nuclear Agric Biol 1980; 9: 50-53.

Danish Environmental Protection Agency. Survey of chemical compounds in textile fabrics, survey of chemicals in consumer products no.23, 2003a. 2003;.

Danish Environmental Protection Agency. Undersögelse af naturlegetöj af plantefibre, uldfibre og massivt trae, Kortlaeggning af Kemiske stoffer i Forbrugerprodukter no.33, 2003b. 2003;.

Darnerud P-O, Eriksen GS, Johannesson T, Larsen PB and Viluksela M. Polybrominated Diphenyl ethers: Food contamination and Potential Risks. TemaNord, Nordic council of ministers. 1998; 503, Copenhagen.

Davidsson A. Miljörapport: För Kosta Glasbruk (0761-108) år: 2006 version 1. 2007;.

De la Calle Guntinas MB, Madrid Y and Cámara C. Flow-injection and continuous-flow systems to determine antimony(III) and antimony(V) by hydride generation atomic absorption spectrometry. Anal Chim Acta 1991; 252: 161-166.

de la Calle Guntinas MB, Madrid Y and Cámara C. Stability study of total antimony, Sb(III) and Sb(V) at the trace level. Fresenius J Anal Chem 1992; 344: 27-29.

de Miguel E, Llamas JF, Chacón E, Berg T, Larssen S, Röyset O and Vadset M. Origin and patterns of distribution of trace elements in street dust: unleaded petrol and urban lead. Atmosph Environ 1997; 31: (17): 2733-2740.

De Temmerman LO, Hoenig M and Scokart PO. Determination of "normal" levels and upper limit values of trace elements in soil. Z Pflanzenernahr Bodenknd 1984; 147: 687-694.

Deng T-L, Chen Y-W and Belzile N. Antimony speciation at ultra trace levels using hydride generation atomic fluorescence spectrometry and 8-hydroxyquinoline as an efficient masking agent. Anal Chim Acta 2001; 432: 293-302.

Dernehl CU, Nau CA and Sweets HH. Animal studies on the toxicity of inhaled antimony trioxide. The Journal of Industrial Hygiene and Toxicology 1945; 27: 256-262.

DHV. Basisdocument cadmium, draft chapters 8 and 9. 1985; DHV Consulting Engineers, Amersfoort.

Dietl C, Reifenhäuser W and Peichl L. Association of antimony with traffic - occurence in airborne dust, deposition and accumulation in standardized grass cultures. Sci Total Environ 1997; 205: 235-244.

Dissanayake CB, Tobschall HJ, Palme H, Rast U and Spettel B. The abundance of some major and trace elements in highly polluted sediments from the Rhine River near Mainz, West Germany. Sci Total Environ 1983; 29: 243-260.

DMT. Bericht über die bestimmung des Staubungsverhaltens, ausgedrückt in den gesundsheitsrelevanten Staubfraktionen nach EN 481, an 1 Probe,. GS-Nr.00 042 05,24.11.2005. 2005;.

Docherty A. written communication. 2001;.

Dodd M, Grundy SL, Reimer KJ and Cullen WR. Methylated antimony(V) compounds: synthesis, hydride generation properties and implications for aquatic speciation. Appl Organometall Chem 1992; 6: 207-211.

Dodd M, Pergantis SA, Cullen WR, Li H, Eigendorf GK and Reimer KJ. Antimony speciation in freshwater plant extracts by using hydride generation-gas chromatography-mass spectromtry. Analyst 1996; 121: 223-228.

Doe KG, Parker WR, Ponsford SJ and Vaughan JDA.The acute and chronic toxicity of antimony to Daphnia magna and rainbow trout. Extended abstract. In Proceedings of the Thirteenth Annual Aquatic Toxicity Workshop: November 12-14, 1986 Moncton, New Brunswick. Edited by Lakshminarayana JSS 1987; pp 40-43. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1575.

Donaldson K. Nonneoplastic lung responses induced in experimental animals by exposure to poorly soluble nonfibrous particles. Inhalation Toxicology 2000; 12: 121-139.

Dooren De Jong LE. Tolerance of chlorella vulgaris for metallic and non-metallic ions. Antonie van Leeuwenhoek J Microbiol Serol 1965; 31: 301-313.

Dorfman D. Tolerance of Fundulus heteroclitus to different metals in salt water. Bull New Jersey Acad Sci 1977; 22: (2): 21-23.

Dost AA, Redman D and Cox G. Exposure to rubber fume and rubber process dust in the general rubber goods, tyre manufacturing and retread industries. Ann Occup Hyg 2000; 44: (5): 329-342.

Driscoll KE, Deyo LC, Carter JM, Howard BW, Hassenbein DG and Bertram TA. Effects of particle exposure and particle-elicited inflammatory cells on mutation in rat alveolar epithelial cells. Carcinogenesis 1997; 18: 423-430.

DTI DoTaI and Research commisioned by the consumer and Competition Policy Directorate. Executive Summary; Research into the mouthing behavior of children up to 5 years old. 2002; july, pp 1-19.

DTU. Waste related emission scenario´s for risk assessment of chemicals. A background document for the revision of the EU Technical Guidance Document on risk assessment of new and existing substances. 2001;.

Dunn JT. A curious case of antimony poisoning. Analyst 1928; 53: 532-533.

Dutkiewicz VA, Parekh PP and Husain L. An evaluation of regional elemental signatures relevant to the Northeastern United States. Atmos Environ 1987; 21: 1033-1044.

EBRC. 2007. Antimony in drinking water. Evaluation of public domain data. Draft report 13th October 2007, EBRC Consulting GmbH, Hannover, Germany.

EBRC. 2005a. Occupational inhalation exposure data, Antimony trioxide production sector, Summary and statistical evaluation. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, November 15th, 2005.

EBRC. 2005c. Dustiness and particle size testing, of eight diantimony trioxide samples. Final Report. EBRC Consulting GmbH, Hannover, Germany , 01 November 2005.

EBRC. 2005b. Occupational inhalation exposure data, for the use of antimony trioxide in the PET production sector, Summary and statistical evaluation. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, November 15th, 2005.

EBRC. 2008. Occupational dermal and inhalation exposure to diantimony trioxide during the production of flame-retardant treated textiles. Revised Final report. February 4th, 2008.

EBRC. 2006b. Occupational dermal and inhalation exposure to diantimony trioxide during the production of crystal glass. Extrapolation from data on exposure to lead oxides. Final report. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, October 8th, 2006.

EBRC. 2006e. Occupational dermal and inhalation exposure to diantimony trioxide during the production of flame-retardant treated rubber. Final report. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, October 15th, 2006.

EBRC. 2006h. Occupational dermal and inhalation exposure to diantimony trioxide during the production of PET resin. Final report. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, November 2nd, 2006.

EBRC. 2006d. Occupational dermal and inhalation exposure to diantimony trioxide during the production of flame-retardant treated PVC. Final report. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, October 31th, 2006.

EBRC. 2006i. Occupational dermal and inhalation exposure to diantimony trioxide in the Pigments/Paints/Coatings/Ceramic sector. Final report. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, October 15th, 2006.

EBRC. 2006c. Occupational dermal and inhalation exposure to diantimony trioxide during the production of flame-retardant treated non-PVC plastics. Final report. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, October 26th, 2006.

EBRC. 2006f. Occupational dermal and inhalation exposure to diantimony trioxide during the production of flame-retardant treated textiles. Final report. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, October 8th, 2006.

EBRC. 2006g. Occupational dermal and inhalation exposure to diantimony trioxide during the production of PET fibres and films. Final report. Prepared on behalf of IAOIA by EBRC Consulting GmbH, Hannover, Germany, November 3rd, 2006.

EBRC. 2006k. Summary statement; Impurity profile of commercial grades of Diantimony trioxide placed on the market in the EU. Prepared on behalf of IAOIA. 2006; 03 31, pp 1-3.

EC. Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption. Offical Journal of the European Communities 1998; L330/32: (5.12.98): 1-23.

EC. Commission Directive 2002/2002/72/EC of 6 August 2002 relating to plastic materials and articles intended to come into contact with foodstuffs. Official Journal of the European Communities 2002; L 220/18: (15.8.2002): 1-41.

ECB. European Union Risk Assessment report on zinc oxide. Existing chemicals, 2nd Priority List. 2004; 43, EBC.

ECB 2005. Document ECB4/05/05 Rev. 1 CD ROM. Document ECB4/05/05 2005; 0805:

Elliot BM, Mackay JM, Clay P and Ashby J. An assessment of the genetic toxicology of antimony trioxide. Mutat Res 1998; 415: 109-117.

Enders R, Vater C and Jekel M. Antimon im Abwasser von Abfallverbrennungsanlagen (Antimony in waste water from MSW incinerators). Mull und Abfall 1990; 22: 784-784.

EPAS. Determination of nitrification inhibition by antimony. PR05.021 RP001v3 22/09/05. 2005;.

Eriksson J. Halter av 61 spårelement i avlopsslam, stallgödsel, handelsg_F6dsel, nederbörd samt jord och gröda. Swedish environmental protection agency. Report 5148. 2001;.

EURAS. Diantimony Trioxide (DAT) exposure assessment compilation and review of local exposure data DAT downstream users. Final report. 2006a;.

EURAS. Diantimony Trioxide (DAT) exposure assessment compilation and review of local exposure data DAT producers. Final report. 2006b;.

EURAS. Diantimony trioxide (DAT) exposure assessment compilation and review of local exposure data DAT downstream users update for FR textile industry - PPCC sector; Study commissioned by IAOIA; Final report, 14 August 2007-10-27. 2007;.

EURAS. Summary of TAUW report L001-9551808NEP-V01-BE: Research sediment "Diepteloop". 2008;.

EURAS bvba. Diantimony trioxide (DAT) Exposure assessment compilation and review of local exposure data, revised final report, 10 March 2003; study commissioned by Diantimony trioxide (DAT) Exposure assessment compilation and review of local exposure data, revised final report, 10 March 2003; study commissioned by International Antimony Oxide Association (IAOIA). 2003;.

EURAS comments on section 3.1.2.7 releases from disposal *Draft RAR, June, 2004*. 2004;.

EUROMETAUX Copper Industry. Prepared for the EIPPC Bureau, ITPS-European Commission. Draft: Section 7, Information Copper Indusrty, Basic Information Emissions. 1999;.

EUROPEAN COMMISSION, Health and consumer protection directorate-general. Opinion of the scientific committee on toxicity, ecotoxicity and the environment (CSTEE) on "Assessment of the bioavailability of certain elements in toys.". EUROPEAN COMMISSION 2004; 0622: 1-5.

European IPPC bureau. Reference document for Best available Techniques for the textile industry, November 2002. 2002;.

EUROSTAT. The European Union coal industry in 2002. Statistics in focus: Environment and Energy: Theme 8 - 16/2003. 2003;.

EUSES 2 0. 2004;.

Feldmann J and Hirner AV. Occurrence of volatile metal and metalloid species in landfill and sewage gas. Int J Environ Anal Chem 1995; 60: 339-359.

Felicetti SA, Thomas RG and McClellan RO. Metabolism of two valence states of inhaled antimony in hamsters. American Industrial Hygiene Association Journal 1974a; 35: 292-300.

Fergusson JE. The heavy elements: chemistry, environmental impact and health effects. 1990; Pergamon Press, Oxford.

Filella M, Belzile N and Chen Y-W. Antimony in the environment: a review focused on natural waters - I. Occurence. Earth-Sci Rev 2002a; 57: 125-176.

Filella M, Belzile N and Chen Y-W. Antimony in the environment: a review focused on natural waters II. Relevant solution chemistry. Earth-Science Reviews 2002b; 59: 265-285.

Filella M and May PM. Computer simulation of the low-molecular-weight inorganic species disrtibution of antimony(III) and antimony(V) in natural waters. Geochim Cosmochim Acta 2003; 67: 4013-4031.

Fleming AJ. Case no 84003A Doc I.D 878220297. The Toxicity of Antimony Trioxide. 1938; pp 1-15. EI Dupont Demours & CO, Wilmington.

Flyhammar P. Analysis of the cadmium flux in Sweden with special emphasis on landfill leachate. J Environ Qual 1995; 24: 612-621.

Flynn HC, Meharg AA, Bowyer PK and Paton GI. Antimony bioavailability in mine soils. Environ Pollut 2003; 124: 93-100.

Fordham PJ, Gramshaw JW, Crews HM and Castle L. Element residues in food contact plastics and their migration into food simulants, measured by inductively-coupled plasma - mass spectrometry. Food Addit Contam 1995; 12: (5):

Franke, J. Final Report, 1st Original of 2, Antimony Trioxide – Particle Size Distribution OECD 110. Report no. 20050309/310/312/313/314. 2005

Fritsche J. Anthropogener Eintrag von Platingruppenelementen in straßennahe Böden, Institut für Allgemeine and Analytische Chemie Montanuniversität Leoben in Zusammenarbeit mit dem Umweltbundesamt Wien. 2003;.

Fubini B and Hubbard A. Reactive oxygen species (ROS) and reactive nitrogen species (RSN) generation by silica in inflammation and fibrosis. Free Radical Biology and Medicine 2003; 34: 1507-1516.

Furr AK, Kelly WC, Bache CA, Gutenmann WH and Lisk DJ. Multi-element uptake by vegetables and millet grown in pots on fly ash amended soil. J Agric Food Chem 1976; 24: 885-888.

Furuta N, Iljima A, Kambe A, Sakai K and Sato K. Concentrations, enrichment and predominant sources of Sb and other trace elements in size classified airborne particulate matter collected in Tokyo from 1995 to 2004. J Environ Monit 2005; 7: 1155-1161.

Galloway JN, Thornton JD, Norton SA, Volchok HL and McLean RAN. Trace metals in atmospheric deposition: A review and assessment. Atmosph Environ 1982; 16: 1677-1700.

Gangolli S. (Editor)The dictionary of substances and their effects. 1999; 2:nd edition,.

Garg B, Cadle SH, Mulawa PA, Groblicki PJ, Laroo C and Parr GA. Brake wear particulate matter emissions. Environ Sci Technol 2000; 21: 4463-4469.

Garg SP, Singh IS and Sharma RC. Long term lung retention studies of ^{125}Sb aerosols in humans. Health Phys 2003; 84: 457-468.

Gebel T. Arsenic and antimony: comparative approach on mechanistic toxicology. Chem Biol Interact 1997; 107: 131-144.

Gebel T, Christensen S and Dunkelberg H. Comparative and Environmental Genotoxicity of antimony and arsenic. Anticancer Res 1997; 17: 2603-2608.

Gerber GB, Maes J and Eykens B. Transfer of antimony and arsenic to the developing organism. Archives of Toxicology 1982; 49: 159-168.

Gerhardsson L, Brune D and Nord. Antimony in lung, liver and kidney tissue from deceased smelter workers. Scand J Work Environ Health 1982; 8: 201-208.

German Ausschuss für Umwelthygiene. Standards zur expositionsabschätzung. Areitsgemeinschaft der leitenden medizinalbeamtinnen und beamten der länder. 2000; Behörde f¨r Arbeit, Gesundheit und Soziales, Hamburg.

Gerritse RG, Vriesema R, Dalenberg JW and de Roos HP. Effect of sewage sludge on trace element mobility in soils. J Environ Qual 1982; 11: (3): 359-364.

GHS A10. Annex 10. Globally Harmonized System of Classification and Labelling of Chemicals (GHS). 2005; United Nations.

GHS, A9. Annex 9. Globally Harmonized System of Classification and Labelling of Chemicals (GHS), ST/SG/AC.10/30. 2003; United Nations.

Gillain G and Brihaye C. A routine speciation method for a pollution survey of coastal sea water. Oceanol Acta 1985; 8: 231-235.

Gillain G, Duyckaerts G and Disteche A. Direct and simultaneous determinations of Zn, Cd, Pb, Cu, Sb and Bi dissolved in sea water by differential pulse anodic stripping voltammetry with a hanging mercury drop electrode. Anal Chim Acta 1979; 106: 23-37.

Gilmour CC, Tuttle JH and Means JC.In Estaurine Geochemistry. Edited by Sigleo AC and Mattori A 1985; pp 239.

Ginsburg S and Buschke A. Eye changes in rats after thallium feeding. Augenheilkunde 1923; 71: 385-399.

Goldberg ED.Minor elements in sea water. In Chemical Oceanography. Edited by Riley JP and Skirrow G 1965; pp 163-196. Academic Press, London and New York.

Grimanis AP, Vassilaki-Grimani M and Griggs GB. Pollution studies of trace elements in sediments from the upper Saronikos Gulf, Greece. J Radioanal Chem 1977; 37: 761-773.

Grin NV, Govorunova NN, Bessemrnyy AN and Pavlovich LV. Study of the embryotoxic effects of antimony oxide under experimental conditions. Gig Sanit 1987; 10: 85-86.

Gross P, Brown JHU, Westrick ML, Srsic RP, Butler NL and Hatch TF.Toxicologic study of calcium halophosphate phosphors and antimony trioxide. !. Acute and chronic toxicity and some pharmacologic aspects. In A. M. A. Archives of Industrial Health. Edited by Drinker P 1955a; 11, pp 473-478. American Medical Association, Chicago.

Gross P, Westrick ML, Brown JHU, Srsic RP, Schrenk HH and Hatch TF.Toxicologic study of calcium halophosphate phosphors and antimony trioxide II. Pulmonary studies. In A. M. A. Archives of Industrial Health. Edited by Drinker P 1955b; 11, pp 479-486. American Medical Association, Chicago.

Groth DH, Kommineni C and Mackay GR. Carcinogenicity of beryllium hydroxide and alloys. J Toxicol Environ Health 1980; 21: 63-84.

Groth DH, Stettler LE, Burg JR, Busey WM, Grant GC and Wong L. Carcinogenic effects of antimony trioxide and antimony ore concentrate in rats. J Toxicol Environ Health 1986a; 18: 607-626.

Groth DH, Stettler LE, Lal JB, Platek SF and Burg JR.Lung tumors in rats treated with quartz by intratracheal instillation. In In Silica, Silicosis and Cancer. Edited by Goldsmith DF, Winn DM and Shy CM 1986b; pp 243-253. Praeger, New York.

Grund SC and Hanusch K.Antimony and Antimony Compounds. In Ullmann's Encyclopedia of Industrial Chemistry. 2000; Wiley-VCH Verlag.

Guard HE, Cobet AB and Coleman WM. Methylation of trimethyltin compounds by estuarine sediments. Science 1981; 213: 770-771.

Guieu C, Zhang J, Thomas AJ, Martin JM and Brun-Cottan JC. Significance of atmospheric fallout on the upper layer water chemistry of the north-western mediterranean. J Atmosph Chem 1993; 17: 45-60.

Gurleyuk H, Van Fleet-Stalder V and Chasteen TG. Confirmation of the biomethylation of antimony compounds. Appl Organometal Chem 1997; 11: 471-483.

Gurnani N, Sharma A and Talukder G. Comparison of the clastogenic effects of antimony trioxide on mice *in vivo* following acute and chronic exposure. Biometals 1992; 5: 47-50.

Gurnani N, Sharma A and Talukder G. Comparison of clastogenic effects of antimony and bismuth as trioxides on mice in vivo. Biol Trace Elem Res 1993; 37: (2-3): 281-292.

Habib S and Minski MJ. Incidence and variability of some elements in the non tidal region of the river Thames, and river Kennet, U.K. Sci Total Environ 1982; 22: 253-273.

Hall RA, Zook EG and Meaburn GM. National marine fisheries service survey of trace elements in the fishery resource. NOAA Technical Report NMFS SSRF-721. 1978; U.S. Department of Commerce.

Hallas LE, Means JC and Cooney JJ. Methylation of tin by estuarine microorganisms. Science 1982; 215: 1505-1507.

Hamdy MK and Noyes OR. Formation of methyl mercury by bacteria. Appl Microbiol 1975; 30: 424-432.

Hammel W, Debus R and Steubing L. Mobility of antimony in soil and its availability to plants. Chemosphere 2000; 41: 1791-1798.

Hammel W, Steubing L and Debus R. Assessment of the ecotoxic potential of soil contaminants by using a soil-algae test. Ecotoxicol Environ Safety 1998; 40: 173-176.

Hares RJ and Ward NI. Comparison of the heavy metal content of motorway stormwater following discharge into wet biofiltration and dry deposition ponds along the London Orbital (M25) motorway. Sci Total Environ 1999; 235: 169-178.

Haring BJA, Van Delft W and Bom CM. Determination of arsenic and antimony in water and soil by hydride generation and atomic absorption spectroscopy. Fresenius Z Anal Chem 1982; 310: 217-223.

Harkness JE and Wagner JE. The biology and medicine of rabbits and rodents. 1989; 3, pp 1-230. Lea & Febiger, Philadelphia.

Hartley J, Cairney JWG, Freestone P, Woods C and Meharg AA. The effects of multiple contamination on ecomycorrhizal Scots pine (*Pinus sylvestris*) seedlings. Environ Pollut 1999; 106: 413-424.

Hartwig A. Current aspects in metal genotoxicity. Biometals 1995; 8: 3-11.

Haskell Laboratory. Doc I.D 878220298, Case no OTS 84003A. Effects of Intratracheal injection of Titanium Dioxide pigments medical research project No MR-583, Report No 5-63. 1963; pp 1-10. DuPont, Wilmington.

Haskell Laboratory for toxicity and Industrial Medicin. Skin tests on human subjects. EPA/OTS; Doc # 878220306. 1970; pp 1.

Haskell Laboratory for Toxicology and Industrial Medicin. Toxicity of Flame Resistant Polythene Composition JRP-10083-B. EPA/OTS; Doc # 878220304. 1950; pp 2.

Haskell Laboratory for Toxicology and Industrial Medicin. Inhalation Toxicity of pyrolysis products. EPA/OTS; Doc # 878220301. pp 2.

Haskell Laboratory for Toxicology and Industrial Medicin. Acute Inhalation Toxicity of Thermal Degradation products. EPA/OTS; Doc # 878220300. 1970a; pp 2.

Haskell Laboratory for Toxicology and Industrial Medicin. Primary skin irritation and sensitization tests. EPA/OTS; Doc # 878220307. 1970b; pp 2.

Haskell Laboratory for Toxicology and Industrial Medicin. Inhalation Toxicity of Pyrolysis Products of Samples of selfextinguishing nylon and GR -PET. EPA/OTS; Doc # 878220302. 1972; pp 4.

Haskell Laboratory for Toxicology and Industrial Medicin. Single Inhalation Exposure Evaluation of Thermal Degradation Products. EPA/OTS; Doc # 878220303. 1974; pp 3.

Haskell Laboratory for Toxicology and Industrial Medicine. Acute Inhalation toxicity test. EPA/OTS; Doc # 878220299. 1962; pp 2.

Haskell Laboratory for Toxicology and Industrial Medicine. Skin tests on human subjects. EPA/OTS; Doc # 878220276. 1970; pp 1. DuPont, Wilmington.

Haskell Laboratory for Toxicology and Industrial Medicine. Approximate Lethal Temperature. EPA/OTS; Doc# 878220295. pp 1-4. DuPont, Wilmington.

Haskell Laboratory for Toxicology and Industrial Medicine. Primary skin irritation and sensitization tests in Guinea Pigs with cover, letter. EPA/OTS; Doc# 878220294. 1970;.

He M and Yang J. Effects of different forms of antimony on rice during the period of germination and growth and antimony concentration in rice tissue. Sci Total Environ 1999; 243/244: 149-155.

Hedenstedt. Personal communication (e-mail) Hedenstedt, RVF. 2005;.

Heijerick D and Vangheluwe M. Analysis of the results of a 42-day soil toxicity test with Enchytraeus albidus, using SbCl3 as test substance. EURAS, pp 14 2003a;

Heijerick D and Vangheluwe M. Analysis of the results of a 21-day chronic toxicity test with the freshwater flea Daphnia magna using SbCl3 as test substance. EURAS, pp 22 2003b;

Heijerick D and Vangheluwe M. Analysis of the results of a 42-day chronic sediment test with Hyalella azteca using SbCl3 as test substance. EURAS, pp 38 2003c;

Heijerick D and Vangheluwe M. Analysis of the results of a 72h-toxicity test with the freshwater green alga *Pseudokirchneriella subcapitata* using SbCl3 as test substance. Final Report September, 2004. 2004; EURAS.

Heijerick D and Vangheluwe M. Analysis of the results of a 28-day chronic sediment test with the midge Chironomus riparius using SbCl3 as test substance. Final Report June, 2005a. 2005; EURAS.

Heijerick D and Vangheluwe M. Analysis of the results of a 28-day chronic sediment test with the oligochaete Lumbriculus variegatus using SbCl3 as test substance. Final Report June, 2005b. 2005; EURAS.

Heinrich U, Muhle H, Takenaka S, Ernst H, Fuhst R, Mohr U, Pott F and Stuber W. Chronic effects on the respiratory tract of hamsters, mice and rats after long-term inhalation of high concentrations of filtered and unfiltered diesel engine emissions. Journal of Applied Toxicology 1986; 6: 383-395.

Heitmanm RA and Staveley JP. Analysis of Algal Toxicity Tests. Report to EPA. 2003; pp 94pp. Arcadis G&M Inc.

Heitmuller PT, Hollister TA and Parrish PR. Acute toxicity of 54 industrial chemicals to sheepshead minnows (Cyprinodon variegatus). Bull Environ Contam Toxicol 1981; 27: 596-604.

Helz GR, Valerio MS and Capps NE. Antimony speciation in alkaline sulfide solutions: Role of zero-valent sulfur. Environ Sci Technol 2002; 36: 943-948.

Heubach H. Normungsvorhaben "Staubungsmessgerät. GmbH & CoKG 1991; 13-.

Hext PM. Current perspectives on particulate induced pulmonary tumours. Human and Experimental Toxicology 1994; 13: 700-715.

Hext PM, Pinto PJ and Rimmel BA. Subchronic feeding study of antimony trioxide in rats. J Appl Toxicol 1999; 19: 205-209.

Hillamo R, Pacyna JM and Bartonova A. Characterization of aerosols during long-range transport episodes of air pollution to Norway. J Aerosol Sci 1988; 19: (7): 1257-1261.

Hiraoka N. The toxicity and organ-distribution of antimony after chronic administration to rats. Kyoto fenitsu ika daigaku gasshi 1986; 95: 997-1017.

Hirner AV, Feldmann J, Goguel R, Rapsomanikis S, Fisher R and Andreae MO. Volatile metal and metalloid species in gases from municipal waste deposits. Appl Organometal Chem 1994; 8: 65-69.

Hirner AV, Kritsotakis K and Tobschall HJ. Metal organic assosiation in sediments-I. Comparison of unpolluted recent and ancient sediments and sediments affected by anthropogenic pollution. Applied Geochemistry 1990; 5: 491-505.

Hjelmar O, Mikkel Johanssen L, Knox K, Ehrig HJ, Flyvberg J, Winther P and Christensen TH. Management and composition of leachate from landfills. Report for DGXI Waste 92, B4-3040/013665/92. Prepared by Water Quality Institute and Carl Bro Environment. Final report september 1994. 1994;.

Hoede D, Wijkstra J and Van der Sloot HA. Dissolved trace oxyanions and trace elements in Indonesian Waters. Data report of the ECN participation in the Snellius II expedition. 1987; Netherlands Energy Research Foundation, Petten.

Hofmann W and Asgharian B. The Effect of Lung Structure on Mucociliary Clearance and Particle Retention in Human and Rat Lungs. Toxicological Sciences 2003; 73: 448-456.

Hollibaugh JT, Seibert DLR and Thomas WH. A comparison of the acute toxicities of ten heavy metals to phytoplankton from Saanich Inlet, B.C., Canada. Estuarine Coast Marine Sci 1980; 10: 93-105.

HSDB®. Hazardous Substances Data Bank, <http://toxnet.nlm.nih.gov/>.

Huerta-Diaz MA and Morse JW. A Quantitative method for determination of trace metal concentrations in sedimentary pyrite. Marine Chem 1990; 29: 119-144.

Huey C, Brinckman FE, Iverson WP and Grim SO.In Progress in Water Technology. Edited by Krenkel PA 1975; 7, pp p 1. Pergamon, New York.

Hughson G. Assessment of dermal exposure and classification of workplace aerosols for antimony trioxide production. Report No 602-00292 prepared by Institute of Occupational Medicine, Edinburgh, UK, for International Antimony Oxide Industry Association (IAOIA), November, 2005, 2005;

Hughson GW. An occupational hygiene assessment of dermal nickel expåosures in primary production industries. IOM Research Report TM/04/05 Institute of Environmental Medicine, Edinburgh http://wwwiom-worldorg/pubs/IOM_TM0405pdf 2004a;

Hughson GW. An occupational hygiene assessment of dermal inorganic lead exposures in primary and intermediate user industries. IOM Research report TM/04/06 Institute of Occupational Medicine, Edinburgh http://iom-worldorg/pubs/IOM_TM0406pdf 2004b;

Hughson GW and Cherrie JW. Validation of the EASE model in relation to dermal zinc exposures. Edinburgh, Institute of Occupational medicine (IOM report TM/01/01). 2001;.

Hughson GW and Cherrie JW. Comparison of measured dermal dust exposures with predicted exposures given by the EASE expert system. Ann Occup Hyg 2005; 49: 111-123.

Hvatum OÖ, Bölviken B and Steinnes E. Heavy metals in Norwegian ombrotrophic bogs. Environ Biogeochem 1983; 35: 351-356.

IAOIA. Comments to the ECB on the draft risk assessment report of antimony trioxide. IAOIA 2004;

IAOIA. IAOIA written communication; update on use in PET received as changes in relevant parts of RAR. 2006a;.

IAOIA. Occupational inhalation and dermal exposure to DAT: Downstream User Survey - Summary report. 2006b;.

IAOIA-EBRC consulting. Consideration of particle size in RC for occ.inhalation exposure. 2008; pp 7.

IAOIA, written communication. Antimony Trioxide catalyst for the polycondensation of polyethyleneterephtalate (PET). IAOIA, written communication. 2000;.

Iavicoli I, Caroli S, Alimonti A, Petrucci F and Carelli G. Biomonitoring of a worker population exposed to low antimony trioxide levels. J Trace Elem Med Biol 2002; 16: 33-39.

IAWG. An international perspective on characterization and management of residues from municipal solid waste incineration. Final Document - International Ash Working Group. 1995;.

ICRP. Limits for Intakes of Radionuclides by Workers. ICRP Publication 30, Part 3. Annals of the ICRP. Sowby FD. 1981; Pergamon press, Oxford.

ICRP. Human Respiratory Tract Model for Radiological Protection. Pergamon Press, Elsevier Science Ltd UK, ISBN 0 08 041154 1 1994; 66:

IEA. Trace elements from coal combustion: emissions. IEA Coal Research 01. 1987;.

IFP Research AB. Antimon, en litteratursammanställning. 1999;.

ILSI Risk Science Institute Workshop Participants. The Relevance of the Rat Lung Response to Particle Overload for Human Risk Assessment: A Workshop Consensus Report. Inhalation Toxicology 2000; 12: 1-17.

Inaba J, Nishimura Y and Ichikawa R. Metabolism of antimony-125 in the rat. JRR 1983; 24: 31-.

INRA. reported by industry who has been asked to submit this report. Etude de l'alimentation totale française Mycotoxines, minéraux et éléments traces 2004;

International Antimony Oxide Industry Association. Questions raised at TC NES IV 2007. 2007; pp 1-8.

International Organization for Standardization. Inhibition of pollutants on earthworms (Eisenia fetida)-Part 2: Determination of effects on reproduction. ISO 11268-2. Soil quality. 1998a;, Geneva, Switzerland.

International Organization for Standardization. Inhibition of reproduction of collembola (Folsomia candida) by soil. ISO 11267. Soil quality. 1998b;, Geneva, Switzerland.

International Organization for Standardization. Effects of pollutants on Enchytraeidae (Enchytraeus sp)-Determination of effects on reproduction and survival. ISO 16387. Soil quality. 2003;, Geneva, Switzerland.

IUCLID. International Uniform Chemical Information Database, available on ecb.jrc.it.

Iyengar GV, Kasperek K, Feinendegen LE, Wang YX and Weese H. Determination of Co, Cu, Fe, Hg, Mn, Sb, Se and Zn in milk samples. Sci Total Environ 1982; 24: 267-274.

Iyengar GV, Tanner JT, Wolf WR and Zeisler R. Preparation of a mixed human diet material for the determination of nutrient elements, selected toxic elements and organic nutrients: A preliminary report. Sci Total Environ 1987; 61: 235-252.

Jang H and Kim SJ. The effect of antimony trisulfide (Sb2S3) and zirconium silicate (ZrSiO4) in the automotive brake friction material on friction characteristics. Wear 2000; 239: 229-229.

Janssen Biotech. The acute toxicity of antimony trioxide in the water-flea (Daphnia magna). Environmental assessment report ADK6/0009, pp 13 1990a;

Janssen Biotech. The acute toxicity of antimony trioxide in the zebra fish (Brachydanio rerio). Environmental assessment report AFBr/0007, pp 14 1990b;

Jarvie HP, Neal C, Burton JD and Tappin AD. Patterns in trace element chemistry in the freshwater tidal reaches of the River Trent. Sci Total Environ 2000; 251/252: 317-333.

Jekel M, Ritz J and Vater C.Abwässer und deren Behandlung (Waste waters and their treatment). In Mullerverbrennung und Umwelt. Edited by Thomé-Kozmiensky KJ 1990; 31, pp p 375. EF Verlag, Berlin, Germany.

Jenkins RO, Craig PJ, Goessler W, Miller D, Ostah N and Irgolic KJ. Biomethylation of inorganic antimony compounds by an aerobic fungus: Scopulariopsis brevicaulis. Environ Sci Technol 1998; 32: 882-885.

Johnson CA, Moench H, Wersin P, Kugler P and Wenger C. Solubility of antimony and other elements in samples taken from shooting ranges. J Environ Qual 2005; 34: 248-254.

Jones RD. Survey of antimony workers: mortality 1961-1992. Occup Environ Med 1994; 51: 772-776.

Juhnke I and Ludemann D. Ergebnisse der untersuchung von 200 chemischen verbindungen auf akute fischtoxizität mit dem Goldorfentest. Z Wasser- und Abwasser-Forschung 1978; 11: (5): 161-164.

Kabata-Pendias A and Pendias H. 10. Elements of group V. Trace elements in soil and plants. 1992; pp 209-210. CRC Press, BOca Raton, Ann Arbor, London.

Kallischnigg G, Legemann P, Müller J and Käferstein FK. Scwermetallgehalt in Bier. Bericht über ein Pilotprojekt. ZEBS Berichte (Zentrale Erfassungs und BewertungsStelle für Umweltchemikalien des Bundesgesundheitsamtes). 1982; Dietrich Reimer Verlag.

Kanematsu N, Hara M and Kada T. Rec assay and mutagenicity studies on metal compounds. Mutat Res 1980; 77: 109-116.

Kantin R. Chemical speciation of antimony in marine algae. Limnol Oceanogr 1983; 28: 165-168.

Karajovic D. Pneumoconiosis amongst workers in Antimony Smelting Plant. Proceedings of the International Congress on Occupational Health 1957; 2: 116-117.

Kawamoto Y and Morisawa S. The distribution and speciation of antimony in river water, sediment and biota in Yodo River, Japan. Environ Technol 2003; 24: 1349-1356.

Kelepertsis A, Alexakis D and Skordas K. Arsenic, antimony and other toxic elements in the drinking water of Eastern Thessaly in Greece and its possible effects on human health. Environmental Geology 2006; 50: 67-84.

KEMI, National Chemical Inspectorate. Projekt Varor och Barn. 2002;.

Kentner M, Leinemann M, Schaller K-H, Weltle D and Lehnert G. External and internal antimony exposure in starter battery production. Int Arch Occup Environ Health 1995; 67: 119-123.

Kersten M, Kriews M and Förstner U. Partitioning of trace metals released from polluted marine aerosols in coastal seawater. Marine Chem 1991; 36: 165-182.

Khangarot BS. Toxicity of metals to a freshwater tubificid worm, Tubifex tubifex (Muller). Bull Environ Contam Toxicol 1991; 46: 906-912.

Khangarot BS and Ray PK. Investigation of correlation between physicochemical properties of metals and their toxicity to the water flea Daphnia magna Straus. Ecotoxicol Environ Safety 1989; 18: 109-120.

Kharkar DP, Turekian KK and Bertine KK. Stream supply of dissolved silver, molybdenum, antimony, selenium, chromium, cobalt, rubidium and cesium to the oceans. Geochimica et Cosmochimica Acta 1968; 32: 285-298.

Kimball G. The effects of lesser known metals and one organic to Fathead Minnows (Pimephales promelas) and Daphnia magna. Manuscript, Department of Entomology, Fisheries and Wildlife, University of Minnesota, Minneapolis, M N 1978;

King LD. Retention of metals by several soils of the southeastern United States. J Environ Qual 1988; 17: (2): 239-246.

Kirk-Othmer. (Editor)Encyclopedia of Chemical Technology, 4:th ed. 1992a; 3, pp 368. John Wiley & Sons.

DRAFT

Kirk-Othmer. (Editor)Encyclopedia of Chemical Technology, 4:th ed. 1992b; 6, pp 439. John Wiley & Sons.

Kirk-Othmer. (Editor)Encyclopedia of Chemical Technology, 4:th ed. 1993a; 10, pp 934. John Wiley & Sons.

Kirk-Othmer. (Editor)Encyclopedia of Chemical Technology, 4:th ed. 1993b; 10, pp 988. John Wiley & Sons.

Kirk-Othmer. (Editor)Encyclopedia of Chemical Technology, 4:th ed. 1996; 19, pp 615. John Wiley & Sons.

Kirk-Othmer. (Editor)Fractionation blood, cell separation, section 3.5 filtration. Kirk-Othmer Encyclopedia of Chemical Technology http://.mrv.interscience.wiley.com. 2007;.

Kirkland D, Whitwell J, Deyo J and Serex T. Failure of antimony trioxide to induce micronuclei or chromosomal aberrations in rat bone-marrow after sub-chronic oral dosing. Mutation Research 2007; 627: 119-128.

Kirwin CJ, LeBlanc JW, Thomas WC, Haworth SR, Kirby PE, Thilagar A, Bowman J and Brusick DJ. Evaluation of the genetic activity of industrially produced carbon black. Journal of Toxicology and Environmental Health 1981; 7: 973-989.

Klein DH, Andren AW, Carter JA, Emery JR, Feldman C, Fulkerson W, Lyon WS, Ogle JC, Talmi Y, van Hook RI and Bolton N. Pathways of thirty-seven trace elements through a coal-fired power plant. Environ Sci Technol 1975; 9: 973-979.

Klucik I, Juck A and Gruberová J. Lesions of the respiratory tract and the lungs caused by pulverulent Antimony trioxide. Prac Lek 1962; 14: 363-368.

Knie J, Hälke A, Juhnke I and Schiller W. Ergebnisse der untersuchungen von chemischen stoffen mit vier biotests. Deutsche Gewässerkundliche Mitteilungen 1983; 3: 77-79.

Krachler M, Burow M and Emons H. Biomonitoring of antimony in environmental matrices from terrestrial and limnic ecosystems. J Environ Monit 1999; 1: 477-481.

Krachler M and Emons H. Speciation analysis of antimony by high-performance liquid chromatography inductively coupled plasma mass spectrometry using ultrasonic nebulization. Anal Chim Acta 2001; 429: 125-133.

Krachler M, Emons H and Zheng J. Speciation of antimony for the 21st century: promises and pitfalls. Trends Analyt Chem 2001; 20: (2): 79-90.

Krachler M, Shi LF, Rossipal E and Irgolic J. Changes in the concentration of trace elements in human milk during lactation. Journal of Trace Elements in Medicine and Biology 1998; 12: 159-176.

Kreyling WG. Interspeices Comparison of Lung Clearence of "Insoluble" Particles. Mary Ann Liebert Inc, Publishers 1990; 3: (1): 93-110.

Kreyling WG, André S, Collier CG, Ferrow GA, Métivier H and Schumann G. Interspecies comparison of the clearence after inhalation of monodisperse, solid cobalt oxide aerosol particles. Journal of Aerosol Science 1991; 22: (4): 509-535.

Krupp EM, Grumping R, Furchtbar URR and Hirner AV. Speciation of metals and metalloids in sediments with LTGC/ICP-MS. Fresenius J Anal Chem 1996; 354: 546-549.

Kuempel ED, Tran C-L, Smith RJ and Bailer AJ. A Biomathematical Model of Particle Clearance and Retention in the Lungs of Coal Miners. II. Evaluation of Variability and Uncertainty. Regulatory Toxicology and Pharmacology 2001; 34: 88-101.

Kuperman RG, Checkai RT, Phillips CT, Simini M and Speicher JA. Toxicity assessments of antimony, barium, beryllium, and manganese for development of ecological soil screening levels (ECO-SSL) using enchytraeid reproduction benchmark values. 2002; Storming Media.

Kuperman RG, Checkai RT, Simini M and Phillips CT. Manganese toxicity in soil for E*isenia fetida,* E*nchytraeus crypticus (*Oligochaeta), and F*olsomia candida (*Collembola). Ecotoxicol Environ Safety 2004; 57: 48-53.

Kuroda K, Endo G, Okamoto A, Yoo YS and Horiguchi SI. Genotoxicity of beryllium, gallium and antimony in short-term assays. Mutat Res 1991; 264: 163-170.

Landsberger S, Vermette VG, Stuenkel D, Hopke PK, Cheng MD and Barrie LA. Elemental source signatures of aerosols from Canadian high Arctic. Environ Pollut 1992; 75: 181-187.

Landström O and Wenner C. Neutron-Activation Analysis of Natural Water Applied to Hydrogeology. 1965; AE-204, pp 1-15. Aktiebolaget Atomenergi, Stockholm.

Lansink CJM, Marquart I and van Hemmen JJ. Standard scenario for the handling of powdered agents. TNO report V 96.065. 1996;.

Lantzy RJ and Mackenzie FT. Atmospheric trace metals: global cycles and assessment of man´s impact. Geochimica et Cosmochimica Acta 1979; 43: 511-525.

Latimer WM. The oxidation states of the elements and their potentials in aqueous solutions, 2nd edn. 1952; pp 116-121. Prentice-Hall, Englewood Cliffs, NJ.

LDAI. Volontary risk assessment on lead and lead compounds. VRAL (2006). 2006;, London.

Leatherland TM and Burton JD. The occurrence of some trace metals in coastal organisms with particular reference to the solent region. J Marine Biol Ass Unit Kingdom 1974; 54: 457-468.

LeBlanc GA. Acute toxicity of priority pollutants to water flea (Daphnia magna). Bull Environ Contam Toxicol 1980; 24: 684-691.

LeBlanc GA and Dean JW. Antimony and thallium toxicity to embryos and larvae of fathead minnows (Pimephales promelas). Bull Environ Contam Toxicol 1984; 32: 565-569.

Lee B-G, Griscom SB, Lee J-S, Choi HJ, Koh C-H, Luoma SN and Fisher NS. Influences of dietary uptake and reactive sulfides on metal bioavailability from aquatic sediments. Science 2000; 287: 282-284.

Lee KP, Trochimowicz HJ and Reinhartd CF. Pulmonary response of rats exposed to titanium dioxide (tiO2) by inhalation for two years. Toxicology and Applied Pharmacology 1985; 79: 179-192.

Leffler P, Gerhardsson L, Brune D and Nordberg GF. Lung retention of antimony and arsenic in hamsters after the intratracheal instillation of industrial dust. Scand J Work Environ Health 1984; 10: 245-251.

Li X and Thornton I. Arsenic, antimony and bismuth in soil and pasture herbage in some old metalliferous mining areas in England. Environ Geochem Health 1993; 15: 135-144.

Li Y, Burkhardt L and Teraoka H. Desorption and coagulation of trace elements during estaurine mixing. Geochim Cosmichim Acta 1984; 48: 1879-1884.

Lin HC and Hwang PP. Acute and chronic effects of antimony chloride (SbCl3) on tilapia (Oreochromis mossambicus) larvae. Bull Environ Contam Toxicol 1998; 61: 129-134.

Lindahl L-Å. E-mail 2003-11-14. 2003;.

Lintschinger J, Koch I, Server S, Feldmann J and Cullen WR. Determination of antimony species with high-performance liquid chromatography using element specific detection. Fresenius J Anal Chem 1997; 359: 484-491.

LISEC. Alga, growth inhibition test effect of Sb2O3 on the growth of Selenastrum capricornutum. Final report WE-06-088, pp 17 1994;

LISEC. Screening transformation/dissolution testing for antimony trioxide powder in ecotox media. Draft report WE-14-012e, pp 13 2000;

LISEC. Alga, growth inhibition test effect of Sb2O3 on the growth of Raphidocelis subcapita. Final report WE-06-276, pp 33 2001;

LISEC. The effect of Sb2O3 on the nitrogen transformation activity of soil micro-organisms. LISEC study n WE-12-004, pp 39 2002a;

LISEC. Screening transformation/dissolution tests with Sb2O3 in ecotox medium at 3 different pH levels. LISEC study n WE-14-030, pp 25 2002b;

LISEC. Acute toxicity test of Sb2O3 on survival and reproduction of Enchytraeus crypticus. LISEC study no WE-23-001 2002c; 1-8.

LISEC. Screening transformation/dissolution tests with Sb203 in ecotox medium at different pH. LISEC study nr: WE-14-021 2002d;

LISEC. Screening and acute transformation/dissolution test with Sb2O3 in ecotox media. LISEC study nr WE-14-018 2002e;

LISEC. Chronic transformation/diiolution test with Sb203 in ecotox medium. LISEC study nr WE-14-020 2002f;

Lisk DJ. Trace metals in soils, plants and animals. Adv Agron 1972; 24: 267-325.

Looser MO, Parriaux A and Bensimon M. Landfill underground pollution detection and characterization using inorganic traces. Water Res 1999; 33: 3609-3616.

LPT and IAOIA. Acute eye irritation/corrosion test of antimony trioxide in rabbits. LPT report No. 19227/05. 2005a; pp 1-26.

LPT and IAOIA. Examination of antimony trioxide in a skin sensitisation test in Guinea Pigs according to Magnusson and Kligman (maximisation test). LPT Report No. 19228/05. 2005b; pp 1-38.

LPT and IAOIA. Acute inhalation toxicity study of antimony trioxide in rats. LPT Report No. 19226/05. 2006; pp 1-58.

Lucian E and Renes MS. Antimony poisoning in industry. AMA archives of industrial hygiene and occuptional medicine 1953; 7: 99-108.

Lüdersdorf R, Fuchs A, Mayer P, Skulsuksai G and Schäcke G. Biological assessment of exposure to antimony and lead in the glass-producing industry. Arch Occup Environ Health 1987a; 59: 469-474.

Lüdersdorf R, Fuchs A, Mayer P, Skulsuksai G and Schäcke G. Biological assessment of exposure to antimony and lead in the glass-producing industry. Int Arch Occup Environ Health 1987b; 59: 469-474.

Lyalikova NN. *Stibiobacter senarmontii* - a new antimony-oxidizing microoganism (English translation). Mikrobiologiya 1974; 43: 941-948.

MacPhee C and Ruelle R. Lethal effects of 1888 chemicals upon four species of fish from Western North America. Univ of Idaho Forest, Wildl Range Exp Station Bull No 3, Moscow, ID 1969; 1-17.

Maeda S, Fukuyama H, Yokoyama E, Kuroiwa T, Ohki A and Naka K. Bioaccumulation of antimony by Chlorella vulgaris and the association mode of antimony in the cell. Appl Organometal Chem 1997; 11: 393-396.

Maher W. Measurement of total antimony in marine organisms and waters by stibine generation and atomic absorption spectrometry. Anal Lett 1986; 19: (3&4): 295-305.

Maihara VA, Vasconcellos MBA, Cordeiro MB and Cozzolino SMF. Estimate of toxic element intake in diets of pre-school children and elderly collected by duplicate portion sampling. Food Addit Contam 1998; 15: 782-788.

Mann H and Fyfe WS. Biogeochemical cycling of the elements in some fresh water algae from gold and uranium mining districts. Biorecovery 1988; 1: 3-26.

Mann H, Fyfe WS and Kerrich R. The chemical content of algae and waters: Bioconcentration. Tox Assess 1988; 3: 1-16.

McCallum RI. The work of an occupational hygiene service in environmental control. Ann Occup Hyg 1963; 6: 55-64.

McCallum RI. Detection of Antimony in process workers´lungs by X-radiation. Trans Soc Occup Med 1967; 17: 134-138.

McCallum RI, Day MJ, Underhill J and Aird EGA. Measurement of antimony oxide dust in human lungs *in vivo* by X-ray spectrophotometry. Inhaled Part 1971; 611-619.

Meharg AA. Inputs of pollutants into the environment from large-scale plastics fires. TEN 1994; 1: (4): 117-122.

Meima JA and Comans RNJ. Reducing Sb-leaching from municapal solid waste incinerator bottom ash by addition of sorbent minerals. Journal of Geochemical Exploration 1998; 62: 299-304.

Melcher B and Faullant P. A comprehensive study of chemical and physical properties of metal sulfides. Soc Automotive Engineers, Annu Brake Colloquium, 2000-01-2757:39 2000;

Metzger M and Braun H. Inversvoltammetrische spurenbestimmung von antimon(III) und antimon(V) in aquatischen umweltproben nach selektiver extraktion. Anal Chim Acta 1986; 189: 263-275.

Middelburg JJ, Hoede D, Van der Slott HA, Van der Weijden CH and Wijkstra J. Arsenic, antimony and vanadium in the North Atlantic Ocean. Geochimica et Cosmochimica Acta 1988; 52: 2871-2878.

Miller FJ. Dosimetry of particles in laboratory animals and humans in relationship to issues surrounding lung overload and human health risk assessment: a critical review. Inhalation Toxicology 2000; 12: 19-58.

Mitchell RL and Burridge JC. Trace elements in soils and crops. Philos Trans R Soc Lond Biol 1979; B 288: 15-24.

MMAF, Ministry of Agriculture Fishery and Food, UK. 1994 Total Diet Study (Part 2): Dietary intakes of metals and other elements. No 149. 1998;.

Mohammad B, Ure AM, Reglinski J and Littlejohn D. Speciation of antimony in natural waters: the determination of Sb(III) and Sb(V) by continous flow hydride generation-atomic absorption spectrometry. Chemical Speciation and Bioavailability 1990; 3: 117-122.

Mok W-M and Wai CM. Distribution and mobilization of arsenic and antimony species in the Coeur DÁlene River, Idaho. Environ Sci Technol 1990; 24: 102-108.

Moll M and Moll N.Lexique Précis des risques alimentaires. In Lexique Précis des risques alimentaires. Edited by Tec & Doc 2000; 2, pp 193-383., Paris.

Mori C, Orsini A and Migon C. Impact of arsenic and antimony contamination on benthic invertebrates in a minor Corsican river. Hydrobiol 1999; 392: 73-80.

Moser T. Acute and reproduction toxicity of Antimony Trioxide aged residues in contaminated soil according to the International Standard ISO 11267 (1999) "Soil Quality - Inhibition of reproduction of Collembola (Folsomia candida) by soil pollutants" for International Antimony Oxide Industry Association (IAOIA). 2007;.

Moskalev YI. Experiments dealing with distribution of antimony 124 and tellurium 127. AEC TR 1964; 62-70.

Mosselmans JF, Helz GR, Pattrick RAD, Charnock JM and Vaughan DJ. A study of speciation of Sb in bisulfide solutions by X-ray absorption spectroscopy. Appl Geochem 2000; 15: 879-889.

Mossman BT and Churg A. Mechanisms in the pathogenesis of asbestos and silicosis. American Journal of Respiratory Critical Care Medicine 1998; 157: 1666-1680.

Motolese A, Truzzi M, Giannini A and Seidenari S. Contact dermatitis and contact sensitization among enamellers and decorators in the ceramics industry. Contact Dermatitis 1993; 28: 59-62.

MPA SMPA. Ref MPA dnr 481: 2006/43536. 2006;.

MPI RI. An inhalation developmental toxicity study in rats with antimony trioxide, Study number 952-002. 2003; 1, pp 1-351. IAOA, Washington D.C.

Muhle H, Bellmann B, Creutzenberg O, Heinrich U, Ketkar M and Mermelstein R. Dust overloading of lungs after exposure of rats to particles of low solubility: Comparative studies. J Aerosol Sci 1990; 21: 374-377.

Mulder DE, Cardinaals JM, Mak JK and van Knippenberg JAJ. Review of litterature data on antimony and some anorganic antimony compounds. NOTOX Toxicological Research & Consultancy; DHV Consultancy Engineers, 97 pp. 1986;.

Müller J. Atmospheric pathways of heavy metals. Heavy metals in the environment, Int Conf 5th 1985; 1: 214-216.

Murthy GK, Rhea U and Peeler JT. Levels of Antimony, Cadmium, Chromium, Cobalt, Manganese and Zinc in institutional total diets. Environ Sci Technol 1971; 5: 436-442.

Nadkarni RA and Ehman WD. Radiochem Radioanal Lett 1970; 4: 325-335.

Nakamaru Y, Tagami K and Uchida S. Antimony mobility in Japanese agricultural soils and the factors affecting antimony sorption behavior. Environ Pollut 2006; 141: 321-326.

Nakamura K, Kinoshita S and Takatsuki H. The origin and behavior of lead, cadmium and antimony in MSW incinerator. Waste Manag 1996; 16: 509-517.

Nash MJ, Maskall JE and Hill SJ. Methodologies for determination of antimony in terrestrial environmental samples. J Environ Monit 2000; 2: 97-109.

National Research council.Antimony Trioxide. In Toxicological risks of selected flame-retardant chemicals, ISBN 0-309-07047-3. 2000; pp 229-261. National Academy Press, Washington DC.

Naturvårdsverket. Bedömningsgrunder för miljökvalitet: Kust och hav. Rapport 4914. 1999a;.

Naturvårdsverket. Bedömningsgrunder för miljökvalitet: Sjöar och vattendrag. Rapport 4920. 1999b; Naturvårdsverket Förlag.

Naturvårdsverket. Ökande eller minskande miljögiftshalter i den svenska miljön? Rapport 5016. 1999c;.

DRAFT

NEA/OECD. Committee on radiation protection and public health. Report of an expert group on gut transfer factors. 1988; NEA/OECD, Paris.

Neal C, Williams RJ, Neal M, Bhardwaj LC, Wickham H, Harrow M and Hill LK. The water quality of the River Tames at a rural site downstream of Oxford. Sci Total Environ 2000; 251/252: 441-457.

Neumann HM and Brown H. Rate of radioactive exchange between antimony(III) and antimony(V) in hydrochloric acid solution. J Am Chem Soc 1956; 78: 1843-1847.

Newton PE, Bolte HF, Daly IW, Pillsbury BD, Terrill JB, Drew RT, Ben-Dyke R, Sheldon AW and Rubin LF. Subchronic and chronic inhalation toxicity of antimony trioxide in the rat. Fundam Appl Toxicol 1994; 22: 561-576.

Newton PE and Daly IW. A one year inhalation toxicity study of antimony trioxide in the rat (with a one year recovery period), project no 83-7647. Developmental Inhalation Study with Antimony trioxide. 1990; 1, pp 1-32. Bio Dynamics Inc., New Jersey.

Niven KJM. The development of a method to assess the contamination of skin by antimony in the furniture upholstery industry. A thesis submitted for the degree of Master of Science at the University of Aberdeen. 1993;, Aberdeen.

Noddack I and Noddack W. Die Häufigkeiten der Schwermetalle in Meerestieren. Ark Zool 1939a; 32A: 1-35.

Noddack I and Noddack W. Die Häufigkeiten der Schwermetalle in Meerestieren. Arkiv för zoologi 1939b; 32: 1-35.

Norwood WP, Borgmann U and Dixon DG. Chronic toxicity of arsenic, cobalt, chromium and manganese to Hyalella azteca in relation to exposure and bioaccumulation. Environ Pollut 2007; 147: 262-272.

Nriagu JO. A global assessment of natural sources of atmospheric trace metals. Nature 1989; 338: 47-49.

Nriagu JO and Pacyna JM. Quantitative assessment of worldwide contamination of air, water and soils by trace metals. Nature 1988; 333: 134-139.

Oberdorster G. Lung particle overload: implications for occupational exposures to particles. Regulatory Toxicology and Pharmacology 1995; 27: 123-135.

OECD. Bioaccumulation: Sequential Static Fish Test. OECD Guidelines for the Testing of Chemicals. No 305A. 1993, Paris.

OECD. Draft Guidance Document on the Statistical Analysis of Ecotoxicity Data. 2004;, Paris.

OECD ESD Plastics additives. OECD Emission Scenario Document on Plastics Additives, Revised version, final draft July 2003. 2003;.

Oekotest. Test Pluschtiere, Da brummt der Baer, da bloekt das Schaf. Oekotest 2002; 04: 40-43.

DRAFT

Offenthaler I, Reisenberger J, Schröder P, Trimbacher C, Wimmer J and Weiss P. Biomonitoring von Nährstoffen und Schwermetallen, organischen Schadstoffen und enzymatischen Reaktionen an Bäumen im Raum Linz. 2004; Umweltbundesamt Monographie, in Vorbereitung.

Öman C, Malmberg M and Wolf-Watz C. Utveckling av metoder för karakterisering av lakvatten från avfallsupplag - Slutrapport. IVL Rapport/report B 1353 2000;

Omura M, Tanaka A, Hirata M and Inoue N. Testicular toxicity evaluation of two antimony compounds, antimony trioxide and antimony potassium tartrate, in rats and mice. Environ Health Prev Med 2002; 7: 15-18.

Oorts K, Gasco G and Smolders E. Toxicity of antimony in the terrestrial environment. Research project for the International Antimony Oxide Industry Association (IAOIA). Final report. 2005;.

Otte PF, Lijzen JPA, Otte JG, Swartjes FA and Versluijs CW. Evaluation and revision of the CSOIL parameter set. Proposed parameter set for human exposure modelling and deriving intervention values for the first series of compound. RIVM Report 2001; 711701021:

Otto B, Berger U and Zimmer AG. Enhanced PET production through new heavy metal-free catalyst systems. Chemical Fibres International 2001; 51: 188-189.

Pacyna JM. Estimation of the atmospheric emissions of trace elements from anthropogenic sources in Europe. Atmos Environ 1984; 18: (1): 41-50.

Pacyna JM and Pacyna EG. An assessment of global and regional emissions of trace metals to the atmosphere from anthropogenic sources worldwide. Environ Rev 2001; 9: 269-298.

Pakkanen TA, Loukkola K, Korhonen CH, Aurela M, Mäkelä T, Hillamo RE, Aarnio P, Koskentalo T, Kousa A and Maenhaut W. Sources and chemical composition of atmospheric fine and coarse particles in the Helsinki area. Atmosph Environ 2001; 35: 5381-5391.

Pan-Hou HS and Imura N. Involvement of mercury methylation in microbial mercury detoxication. Arch Microbiol 1982; 131: 176-177.

Paoletti F, Sirini P, Seifert H and Vehlow J. Fate of antimony in municipal solid waste incineration. Chemosphere 2001; 42: 533-543.

Papakostidis G, Grimanis AP, Zafiropoulos D, Griggs GB and Hopkins TS. Heavy metals in sediments from the Athens sewage outfall area. Mar Pollut Bull 1975; 6: 136-139.

Parkmann H. Personal communication. 2004;

Parr RM, DeMaeyer EM, Iyengar VG, Byrne AR, Kirkbright GF, Schöch G, Niinist
'F6 L, Pineda O, Vis HL, Hofvander Y and Omololu A. Minor and trace elements in human milk from Guatemala, Hungary, Nigeria, Phillippines, Sweden and Zaire. Biological Trace Elements Research 1991; 29: 51-75.

Parris GE and Brinckman FE. Reactions which relate to environmental mobility of arsenic and antimony. II. Oxidation of trimethylarsine and trimethylstibine. Environ Sci Technol 1976; 10: (12): 1128-1133.

Pattenden NJ, Branson JR and Fisher EMR.Trace element measurements in wet and dry deposition and airborne particulate at an urban site. In Deposition of atmospheric pollutants. Edited by Georgii H-W and Pankrath J 1982; pp 173-184. UAEA.

Pauwels J. *Gedrag van antimoon in de bodem en zijn opname door de planten.* PhD thesis. University of Ghent, Belgium. 1985;.

Pearce JA. The effect of antimony on plants. 1978; A thesis submitted for the degree of doctor of philosophy, The University of Newcastle upon Tyne.

Personal communication with industry. Ronny Sabert, Hensoterm. 2003;.

PET container recycling Europe. What is PET, PET and the environment. www.petcore.org. 2003;.

Peterson ML and Carpenter R. Biogeochemical processes affecting total arsenic and arsenics species distributions in an intermittently anoxic fjord. Marine Chem 1983; 12: 295-321.

Pettersson R, Rogbäck J and Suér P. Bottom ash from fluidishing bed boilers as filler material in district heating pipe culverts. Chemical and geochemical characterisation. Manuscript for Q4-220, Värmeforsk. 2003;.

Phillips CT, Checkal RT, Kuperman RG, Simini M and Speicher JA. Toxicity assessments of antimony, barium, beryllium, and manganese for development of ecological soil screening levels (ECO-SSL) using Folsomia reproduction benchmark values. 2002; Storming Media.

Pierson WR and Brachaczek. Particulate matter associated with vehicles on the road. II. Aerosol Sci Technol 1983; 2: 1-40.

Portmann JE and Riley JP. The determination of antimony in natural waters with particular reference to sea water. Anal Chim Acta 1966; 35: 35-41.

Postupolski A and Golimowski J. Trace determination of antimony and bismuth in snow and water samples by stripping voltammetry. Electroanal 1991; 3: 793-797.

Potkonjak V and Pavlovich M. Antimoniosis: A prticular form of Pneumoconiosis. Int Arch Occup Environ Health 1983; 51: 199-207.

Prüeß A.Einstufung mobiler Spurenelemente in Böden. In Ergänzbares Handbuch. Edited by Rosenkranz D, Einsele G and Harress HM 1994; pp 1-59. E. Schmidt Verl., Berlin.

Qian X, Koerner RM and Gray DH. Geotechnical aspects of landfill design and construction. 2002; pp 710 p. Prentice Hall.

Ragaini RC, Ralston HR and Roberts N. Environmental trace metal contamination in Kellog, Idaho, near a lead smeting complex. Environ Sci Technol 1977; 11: 773-781.

RAR. Draft risk assessment report Tetrabromobisphenol A, Rapporteur country UK, Dec 2003 version. 2003;.

RAR. Draft Environmental Risk Assessment Report for bis(pentabromophenyl) ether. Rapporteur Country: United Kingdom. March 2004 version. 2004;.

Rehwoldt R, Karimian-Teherani D and Altmann H. Measurement and distribution of various heavy metals in the Danube River and Danube Canal aquatic communities in the vicinity of Vienna, Austria. Sci Total Environ 1975; 3: 341-348.

Reiman C and Caritat P. Chemical elements in the environment. 1998; Springer, Berlin, Heidelberg.

Renes LE. Antimony poisoning in industry. AMA archives of industrial hygiene and occupational medicine 1953; 7: 99-108.

Risk and policy analysts Limited. Risk reduction strategy and analysis of advantages and drawbacks of Pentabromodiphenyl Ether; Stage 4 report - March 2000; project J285/PeBDPE; prepared for Departmetn of the environment, transport and the regions. 2000;.

Robertson DE. Trace elements in marine organisms. 1967; pp 56-59. BNWL-481-2.

Robinson H. A review of the composition of leachates from domestic wastes in landfill sites. Report prepared for the UK Department of the Environment. 1995;.

Rognerud S and Fjeld E. Trace element contamination of Norwegian lake sediments. Ambio 2001; 30: 11-19.

Rooney CP, Zhao F-E and McGrath SP. Soil factors controlling the expression of copper toxicity to plants in a wide range of European soils. Environ Toxicol Chem 2006; 25: 726-732.

Roper SC and Stupart L. The in vitro percutaneous absorption of antimony trioxide through human skin. Charles River Laboratories Report Number 25985. 2006; pp 1-112. Charles River Laboratories, Edinburgh.

Roskill Information Services. The Economics of Antimony. The Economics of Antimony (9th ed.). 2001; pp p 122. Roskill Information Services Ltd., London.

Routh J and Ikramuddin M. Trace-element geochemistry of Onion Creek near Van Stone lead-zinc mine (Washington, USA) - Chemical analysis and geochemical modeling. Chem Geol 1996; 133: 211-224.

Salbu B, Pappas AC and Steinnes E. Elemental composition of Norwegian rivers. Nordic Hydrology 1979; 115-140.

Sauvant MP, Pepin D, Bohatier J and Groliere. Microplate technique for screening and assessing cytotoxicity of xenobiotics with Tetrahymena pyriformis. Ecotoxicol Environ Safety 1995a; 32: 159-165.

Sauvant MP, Pépin D, Grolière CA and Bohatier J. Effects of organic and inorganic substances on the cell proliferation of L-929 fibroblasts and Tetrahymena pyriformis

GL protozoa used for toxicological bioassays. Bull Environ Contam Toxicol 1995b; 55: 171-178.

Schaumberg DA, Mendes F, Balaram M, Dana M, Sparrow D and Hu H. Accumulated lead exposure and risk of age-related cataract in men. JAMA 2004; 8: (292): 2750-2754.

Schaumlöffel N and Gebel T. Heterogeneity of the DNA damage provoked by antimony and arsenic. Mutagenesis 1998; 13: 281-286.

Schelenz R. J Radioanal Chem 1977; 37: 539-548.

Schins RPF. Mechanisms of genotoxicity of particles and fibres. Inhalation Toxicology 2002; 14: 57-78.

Schmidt U and Huber F. Nature 1976; 259: 157-157.

Schottenfeld D and Beebe-Dimmer J. Chronic inflammation: a common and important factor in the pathogenesis of neoplasia. CA: a Cancer Journal for Clinicians 2006; 56: 69-83.

Schramel P, Samsahl K and Pavlu J. Some determinations of Hg, As, Se, Sb, Sn and Br in waer plants, sediments and fishes in bavarian rivers. Int J Environ Stud 1973; 5: 37-40.

Schulze D, Kruger A, Kupsch H, Segebade C and Gawlik D. Enrichment and distribution of elements in flood-plain soils of the Bitterfeld indusrtial area - studied by neutron activation analysis. Sci Total Environ 1997; 206: 227-248.

Schutz DF and Turekian KK. The investigation of the geographical and vertical distribution of several trace elements in sea water using neutron activation analysis. Geochimica et Cosmochimica Acta 1965; 29: 259-313.

Sereno D, Holzmuller P, Mangot I, Cuny G, Ouaissi A and Lemesre J-L. Antimonial-mediated DNA fragmentation in *Leishmania infantum* amastigotes. Antimicrob Agents Chemother 2001; 45: 2064-2069.

Shand CA, Aggett PJ and Ure AM. The Spark-Source Mass Spectrometric Determination of the Trace Element Composition of Human Foetal Livers. Trace elements in man and animals 5th international symposium Papers described as proceedings tema 5 1985; 5: 642-645.

Sherman DM, Ragnarsdottir KV and Oelkers EH. Antimony transport in hydrothermal solutions: and EXAFS study of antimony(V) complexation i nalkaline sulfide and sulfide-chloride brines at temperatures from 25 ºC to 300 ºC at Psat. Chem Geol 2000; 167: 161-167.

Shotyk W, Cheburkin AK, Appleby PG, Fankhauser A and Kramers JD. Two thousand years of atmospheric arsenic, antimony, and lead deposition recorded in an ombrotrophic peat bog profile, Jura Mountains, Switzerland. Earth Planet Sci Lett 1996; 145: E1-E7.

Shotyk W, Chen B and Krachler M. Lithogenic, oceanic and anthropogenic sources of atmospheric Sb to a maritime blanket bog, Myrarnar, Faroe Islands. J Environ Monit 2005; 7: 1148-1154.

Shotyk W and Krachler M. Contamination of bottled waters with antimony leaching from polyethylene terephtalate (PET) increases upon storage. Environmental Science & Technology 2007; 41: (5): 1560-1563.

Shotyk W, Krachler M and Chen B. Contamination of Canadian and European bottled waters with antimony from PET containers. J Environ Monitor 2006; 8: 288-292.

Simini M, Checkai RT, Kuperman RG, Philips CT and Speicher JA. Toxicity assessments of antimony, barium, beryllium, and manganese for development of ecological soil screening levels (ECO-SSL) using earthworm (*Eisenia Fetida*) benchmark values. 2002; Storming Media.

Sinemus H-W, Kleiner J and Stabel H-H. Combination of flow injection hydride generation and sequestration on a graphite tube for the automated determination of antimony in potable and surface waters. J Anal At Spectromet 1992; 7: 433-437.

Skeaff JM and Dubreuil AA. Calculated 1993 emission factors of trace metals for Canadian non-ferrous smelters. Atmosph Environ 1997; 31: (10): 1449-1457.

Skeaff JM and Hardy DJ. Dissolution Kinetics of Antimony rioxide in the Aqueous Medium ISO 6341. Final Report. Work performed for: The International Antimony Oxide Industry Association (IAOIA). 2004; Oct, CANMET-MMSL 04-046, CR.

Sloof W, Bont PFH, Hesse JM and Loos B. Exploratory report Antimony and Antimony compounds. 1992; National institute of of public health and environmental protection, Bilthoven, NL.

Slooff W, Bont P, Hesse J and Loos B. PB93-224004; RIVM-710401020. Exploratory report; antimony and antimony compounds. 1992;.

Smeykal, H. Relative density. Report no. 20041133.01. 2005.

Smichowski P, Madrid Y and Cámara C. Analytical methods for antimony speciation in waters at trace and ultratrace levels. A review. Fresenius J Anal Chem 1998; 360: 623-629.

Smichowski P, Madrid Y, de la Calle Guntinas B and Cámara C. Separation and determination of antimony (III) and antimony (V) species by high-performance liquid chromatography with hydride generation atomic absorption spectrometric and inductively coupled plasma mass spectrometric detection. J Anal At Spectromet 1995; 10: 815-821.

Smihily. Personal communication (e-mail) Smihily, EUROSTAT. 2003;.

Smock LA. The influence of feeding habitats on whole-body metal concentrations in aquatic insects. Freshw Biol 1983; 13: 301-311.

Smolders E, Brans K, Coppens F and Merckx R. Potential nitrification rate as a tool for screening toxicity in metal-contaminated soils. Environ Toxicol Chem 2001; 20: 2469-2474.

Smolders E and Mertens J. Kinetics of Sb2O3 solubilization in soil: Final report to the International Antimony Oxide Industry Association (IAOIA). COM415_ENV_IND4 annex 1. 2007;.

Smolders E, Mertens J and Buekers J. Toxicity and Bioavailability of Sb2O3 after ageing in Terrestrial Environments. Final report to the International Antimony Oxide Indusrty Association (IAOIA). 2007;.

Smyth HF and Carpenter CPJ. Further experience with the range finding test in the industrial toxicology laboratory. J Ind Hyg Toxicol 1948; 30: 63-68.

Springborn Life Sciences. Biotranformation of antimony oxide in natural sediments under aerobic and anaerobic conditions. SLS Report 8-5-2728 1988;

Stabel HH, Kleiner J, Merkel P and Sinemus HW. Stoffkreisläufe ausgewählter Spurenelemente im Bodense. Vom Wasser 1991; 76: 73-91.

Statens Naturvårdsverk. Om metaller, en litteratursammanställning. 1976;.

Stechmann H and Dannecker W. Characterization and source analysis of vehicle-generated aerosols. J Aerosol Sci 1990; 21: (Suppl. 1): S287-S290.

Steinnes E.Regional distribution of Arsenic, Selenium and Antimony in human layers of Norwegian soils. In Geomedical aspects in present and future research: symposium held in Oslo, on May 22-23, 1978. Edited by Låg J 1980; pp 217-222.

Steinnes E, Allen RO, Petersen HM, Rambæk JP and Varskog P. Evidence of large scale heavy-metal contamination of natural surface soils in Norway from long-range atmospheric transport. Sci Total Environ 1997; 205: 255-266.

Steinnes E, Berg T, Eidhammer Sjoback T, Uggerud H and Vadset M. Atmosfaerisk nedfall av tungmetaller i Norge; Rapport 838/01. 2001;.

Sternbeck J. IVL, Stockholm, pers comm 2003-09-03. 2003;.

Sternbeck J and Munthe J. Metaller i avfallsaskor och brännbart avfall - delrapport. IVL A 21150, 2001-06-18 2001;

Sternbeck J and Östlund P. Nya metaller och metalloider i samhället. IVL Rapport, B 1332. 1999;.

Sternbeck J, Palm A and Kaj L. Antimon i Sverige - användning, spridning och miljöpåverkan. IVL Rapport, B 1473. 2002a;.

Sternbeck J, Sjödin Å and Andréasson K. Spridning av metaller från vägtrafik. IVL Rapport, B 1431. 2001;.

Sternbeck J, Sjödin Å and Andréasson K. Metal emissions from road traffic and the influence of resuspension - results from two tunnel studies. Atmos Environ 2002b; 36: 4735-4744.

Stevenson CJ. Antimony spots. Trans St Johns Hosp Dermatol Soc 1965; 51: 40-45.

Stössel RP and Michaelis W. Wet and dry deposition of heavy metals. Environmental contamination, 2nd Int Conf 1986; 85-88.

Strohal P, Huljev D, Lulic´ S and Picer M. Antimony in the coastal marine environment, North Adriatic. Estuarine Coast Marine Sci 1975; 3: 119-123.

Stuyfzand PJ. De samenstelling van regenwater langs Hollands kust. KIWA-rapport SWE-91.010, 70p. 1991;.

Stuyfzand PJ. Sporenelementen in grondwater in Nederland (deel 2). H2O 1992; 25: 20-25.

Suér P and Lyth M. Antimony in ashes: Sources, leaching and possible remedial actions. SGI report; Projectnr: 11826/11515. 2003;.

Sun H-W, Shan X-Q and Ni Z-M. Selective separation and differential determination of antimony(III) and antimony(V) by solvent extraction with N-benzoyl-N-phenyl-hydroxylamine and graphite-furnace atomic-absorption spectrometry using a matrix-modification technique. Talanta 1982; 29: 589-593.

Sunagawa S. Experimental studies on antimony poisoning. Igaku Kenkyu 1981; 51: (3): 1-14.

Swedish National Testing and Research Institute, SP. Fire-LCA model, TV case study. SP Report 2000:13. 2000; 13,.

Swedish Work Environment Authority. Occupational exposure limit values. Occupational exposure limit values. 2000; www.av.se/regler/afs/2000-3.pdf.

TAI Environmental Sciences I. Results of acute toxicity testing of antimony trichloride using freshwater species Chironomus tentans, Physa heterostropha, Ictalurus punctatus, Hyalella azteca, Hydra oligactis and Chlorohydra viridissimus. 1990;

Takaoka M, Fukutani S, Horiuchi M, Satta N, Takeda N, Oshita K, Yoneda M, Morisawa S and Tanaka T. Determination of chemical form of antimony in contaminated soil around a smelter using X-ray absorption fine srtucture. Anal Sci 2005; 21: 769-773.

Takayanagi K. Acute toxicity of waterborne Se(IV), Se(VI), Sb(III), and Sb(V) on Red Seabream (*Pargus major*). Bull Environ Contam Toxicol 2001; 66: 808-813.

Takayanagi K and Cossa D. Vertical distribution of Sb(III) and Sb(V) in Pavin Lake, France. Water Res 1997; 31: 671-674.

Takayanagi K, Cossa D and Martin J-M. Antimony cycling in the western Mediterranean. Marine Chem 1996; 54: 303-312.

Takayanagi K and Michel P. Semi-automated determination of dissolved antimony in seawater and sediment pore water. Bunseki Kagaku 1996; 45: 1115-1120.

Tamulinas SH. A dissertation. The effects of antimony-trioxide on channel catfish Icatalurus punctatus. 1979; Texas A&M University.

Tange L and Drohmann D. Waste electrical and electronic equipment plastics with brominated flame retardants - from legislation to separate treatment - therminal processes. Polymer degradation and stability 2005; 88: 35-40.

Tarzwell CM and Henderson C. Toxicity of less common metals to fishes. Indust Wastes 1960; 5: (1): 12-.

Tennant RW, Margolin BH, Shelby MD, Zeiger E, Haseman JK, Spalding J, Caspary W, Resnick M, Stasiewicz S, Anderson B and et a. Prediction of chemical carcinogenicity in rodents from in vitro genetic toxicity assays. Science 1987; 236: 933-941.

Thanabalasingam P and Pickering WF. Specific sorption of antimony (III) by the hydrous oxides of Mn, Fe, and Al. Water Air Soil Pollut 1990; 49: 175-185.

Thayer JS. Environmental Chemistry of the Heavy Elements: Hydrido and Organo Compounds. 1995; VCH, New York.

Thomas J and Stevens G. Final report on flame retardants release from textiles; Polymer reasearch centre report. 2006; University of Surrey.

Thompson M and Thornton I. Antimony in the domestic environment and SIDS. Environ Technol 1997; 18: 117-119.

Tighe M, Lockwood P and Wilson S. Adsorption of antimony (V) by floodplain soils, amorphous iron (III) hydroxide and humic acid. J Environ Monit 2005; 7: 1177-1185.

TNO Quality of Life. A study on the Biodistribution of Antimony trioxide (Sb_2O_3) in rats. TNO report 2005; 6502: 1-63.

Tran CL and Buchanan DE. Development of a biomathematical lung model to describe the exposure - dose response realtionship for inhaled dust among UK coal miners. Institute for Occupational Medicine Edinburgh, IOM Research Report TM/00/02 2000;

Tran CL, Jones AD, Cullen RT and Donaldson K. Exploration of the mechanisms of retention and clearance of low-toxicity particles in the rat lung using a mathematical model. Inhalation Toxicology 1999; 11: 1077-1108.

TRAR302+303_env.doc. Risk Assessment: Targeted Report Cadmium (oxide) as used in batteries, draft of May 2003, Belgium. 2003;.

Trnovsky M, Oxer JP, Rudy RJ, Hanchak MJ and Hartsfield B.Site remediation of heavy metals contaminated soils and groundwater at a former reclamation site in Florida. In Hazardous Waste: Detection, Control, Treatment. Edited by Abbou R 1988; pp 1581-1590. Elsevier Science Publishers B.V., Amsterdam.

Uexküll O, Skerfving S, Doyle R and Braungart M. Antimony in brake pads - a carcinogenic component? Journal of Cleaner Production 2005; 13: 19-31.

UK Department of Health. Expert group to investigate cot death theories, Toxic gas hypothesis, Final report May 1988. 1988;.

UK Food Standards Agency. Food survey information sheet number 28/02; Antimony, Arsenic, Bromate and Nickel Contents of Bottled Water. 2002;.

UK Food Standards Agency. Food survey information sheet number 42/03; Multi-element survey of infant foods. 2003;.

UK FSA. Concentrations (mean value) of antimony (Sb) in foods and beverages. 1997;.

UK Ministry of Agriculture, Fisheries and Food. Annual Report on Food Expenditure, Consumption and Nutrient Intakes. 1998;.

Ulrich N. Speciation of antimony(III), antimony(V) and trimethylstiboxide by ion chromatography with inductively coupled plasma atomic emission spectrometric and mass spectrometric detection. Anal Chim Acta 1998; 359: 245-253.

Ulrich N. Determination of antimony species with fluoride as modifier and flow injection hydride generation inductively-coupled plasma emission spectrometry. Anal Chim Acta 2000; 417: 201-209.

UMWELTANALYTIK GMBH. 1. Determination of the antimony content in a test substance, 2. Determination of the solubility of a test substance at pH 5.0, 7.0 and 9.0. 1993;

Umweltbundesamt. Statuserhebung betreffend Überschreitungen des IG-L-Grenzwertes für PM10 an den Messstellen Illmitz, Kittsee und Eisenstadt im Jahr 2002, in Druck. 2004;.

University of Surrey, polymer research centre. Risk and benefits of the use of flame retardants in consumer products. A report for the department of trade and industry - consumer safety unit, UK. 1999;.

Ure AM, Bacon JR, Berrow ML and Watt JJ. The total trace element content of some scottish soils by spark source mass spectrometry. Geoderma 1979; 22: 1-23.

US Consumer Safety Commission. Upholstery furniture. US Consumer Safety Commission. 1994; 59, pp 30735-30738.

US E. Water-related environmental fate of 129 priority pollutants. Volume 1: Introduction and technical background, metals and inorganics, pesticides and PCB°4s. EPA-440/4-79-029a. U.S. Environmental Protection Agency, Washington. 1979;.

US EPA. Office of solid waste, Washington, DC 20460. Partitioning coefficients for metals in surface water, soil, and waste. 1999;.

USDA. "Supplement to Soil Classification System (7th Approximation)", SCS, Second Printing, March, 1967. 1967;.

USGS. US Coal Quality Database. Open file report 97-134. 1997; U.S. Geological Survey, Washington D.C.

Uthe JF and Bligh EG. J Fish Res Board Can 1971; 28: 786-788.

Van Bruwaene R, Greber GB, Kirchmann R and Colard K. Metabolism of antimony-124 in lactating dairy cows. Health Phys 1982; Nov.: (43): 733-738.

Van der Sloot HA, Hoede D and Wijkstra J. Trace oxyanions and their behaviour in the rivers porong and solo, the Java Sea and the adjacent Indian Ocean. Netherlands Journal of Sea Research 1989; 23: 379-386.

Van der Sloot HA, Hoede D, Wijkstra J, Duinker JC and Nolting RF. Anionic species of V, As, Se, Mo, Sb, Te and W in the Scheldt and Rhine Estuaries and the Southern Bight (North Sea). Estuarine, Coastal and Shelf Science 1985; 21: 633-651.

Van der Weijden CH, Middelburg JJ, De Lange GJ, Van der Sloot HA, Hoede D and Woittiez JRW. Profiles of the redox-sensitive trace elements As, Sb, Mo and U in the Tyro and Bannock Basins (eastern Mediterranean). Marine Chem 1990; 31: 171-186.

van Velzen D, Langenkamp H and Herb G. Antimony, its sources, applications and flow paths into urban and industrial waste: a review. Waste Manag Res 1998; 16: (1): 32-40.

Vandenbroele M, Van Sprang P and Vangheluwe M. Diantimony trioxide (DAT) exposure assessment: compilation and review of local exposure data. Revised Final Report 10 March 2003 by EURAS, Comissioned by International Antimony Oxide Industry Association (IAOIA). 2003; pp 1-130.

Vangheluwe M, Vandenbroele M, Van Sprang P, Smolders E, Degryse F, Ruttens A and Vangronsveld J. ONDERBOUWING NORMSTELLING BODEM- EN SECUNDAIRE GRUNDSTOFFEN VOOR BIJKOMENDE METALEN ONTWERP EINDVERSLAG - FEBRUARI 2001; Euras (internal report to OVAM, opebare Vlaamse Afvalstoffenmaatschappij. 2001;.

Vehlow J and Mark FE. Electrical and electonic plastics waste co-combustion. Technical paper from APME, Association of Plastic manufacturers in Europe 1997;

Verdonck F, Heijerick D and Vangheluwe M. Collection and evaluation of partition coefficients for antimony in the environment, Final draft report, August 2004. Prepared by Euras. 2004;.

Vermeulen R, de Hartog J, Swuste P and Kromhout H. Trends in exposure to inhalable particulate and dermal contamination in the rubber manufacturing industry: effectiveness of control measures implemented over a nine-year period Trends in exposure to inhalable particulate and dermal contamination in the rubber manufacturing industry: effectiveness of control measures implemented over a nine-year period. The Annals of Occupational Hygiene 2000; 44: (5): 343-354.

Veselý J, Majer V, Kucera J and Havránek V. Solid-water partitioning of elements in Czech freshwaters. Appl Geochem 2001; 16: 437-450.

Vien SH and Fry RC. Ultrasensitive, simultaneous determination of arsenic, selenium, tin, and antimony in aqueos solution by hydride generation gas chromatography with photoionization detection. Anal Chem 1988; 60: 465-465.

Vijver MG, Vink JPM, Miermans CJH and van Gestel CAM. Oral sealing using glue: a new method to distinguish between intestinal and dermal uptake of metals in earthworms. Soil Biol Biochem 2001; 35: 125-132.

Vinogradov AP. Elementary chemical composition of marine organisms. 1953; Transl. by Efron and Setlow. New Haven, Sears Foundation for Marine Research, Yale University.

Wagner JP, El-Ayyuobi MA, Konzen RB and Krohn JL. Quantitative identification of Sb, Ba, Cd, and Sn during controlled combustion of plastics. Polym-plast Technol Eng 1992; 31: 73-101.

Waitz JA, Ober RE, Meisenhelder JE and Thompson PE. Physiological diposition of antimony after administration of 124Sb-labelled tartar emetic to rats, mice and monkeys and the effects of tris(p-aminophenyl) carbonium pamoate on this distribution. Bull Org mond Santé, Bull Wld Hlth Org 1965; 33: 5437-5546.

Wappelhorst O, Kühn I, Heidenreich H and Markert B. Transfer of selected elements from food into human milk. Nutrition 2002; 18: 316-322.

Watt WD. Dissertation. Chronic inhalation toxicity of antimony trioxide: Validation of the threshold limit value. 1983; 1, pp 1-133. Wayne State University, Detroit, Michigan.

Wedepohl KH. The composition of the continental crust. Geochim Cosmochim Acta 1995; 59: (7): 1217-1232.

Weidenfeller. Final report of the distiness testing of powdery substances (Antimony trioxide) on behalf of EBRC Consulting Gmbh. GF-no. 00 009 05. 2005.

Weiss, Trimbacher and Moche. in Press;

Wester PO. Trace element balances in relation to variations in calcium intake. Atherosclerosis 1974; 20: 207-215.

Wheeler J and Sams C. Lead exposure in the crystal industry.HSL internal report IR/A/00/01. Health and Safety laboatories 1999;

White GP, Mathias CGT and Davin JS. Dermatitis in workers exposed to antimony in a melting process. J Occup Med 1993; 35: (4): 392-395.

WHO.Guidelines for drinking-water quality [electronic resource]: incorporating firsst addendum. Vol. 1, Recommendations. - 3rd ed. 2006; 1, World Health Organization, http://www.who.int/water_sanitation_health/dwq/gdwq3rev/en/print.html, Geneve.

Wil Research Laboratories I. Acute Eye Irritation Study in Rabbits with Antimony Oxide. Project Number: WIL-1277-79. 1979; pp 13.

Williams PL and Dusenbery DB. Aquatic toxicity testing using the nematode, Caenorhabditis elegans. Environ Toxicol Chem 1990; 9: 1285-1290.

Winter-Sorkina de R and Cassee FR. From concentration to dose: factors influencing airborne particulate matter deposition in humans and rats. RIVM report 650010031/2002 2002;

Wong PTS, Chau YK and Luxon PK. Methylation of lead in the environment. Nature 1975; 253: 263-264.

Wyttenbach A, Bajo S and Haekkinen A. Beitrg Tabakforsch Int 1976; 8: 247-249.

Yamada M and Tonomura K. Formation of methylmercury compounds from inorganic mercury by Clostridium cochlearium. J Ferment Technol 1972; 50: 159-166.

Zechmeister HG, Hochenwaller D, Riss A and Hanus-Illnar A. Estimation of element deposition deriving from road traffic sources by mosses. Environ Pollut 2005; Submitted:

abbreviations

[update the list to correspond to the substance RAR]

ADI	Acceptable Daily Intake
AF	Assessment Factor
ASTM	American Society for Testing and Materials
ATP	Adaptation to Technical Progress
AUC	Area Under The Curve
B	Bioaccumulation
BBA	Biologische Bundesanstalt für Land- und Forstwirtschaft
BCF	Bioconcentration Factor
BMC	Benchmark Concentration
BMD	Benchmark Dose
BMF	Biomagnification Factor
bw	body weight / *Bw*, *b.w.*
C	Corrosive (Symbols and indications of danger for dangerous substances and preparations according to Annex III of Directive 67/548/EEC)
CA	Chromosome Aberration
CA	Competent Authority
CAS	Chemical Abstract Services
CEC	Commission of the European Communities
CEN	European Standards Organisation / European Committee for Normalisation
CMR	Carcinogenic, Mutagenic and toxic to Reproduction
CNS	Central Nervous System
COD	Chemical Oxygen Demand
CSTEE	Scientific Committee for Toxicity, Ecotoxicity and the Environment (DG SANCO)
CT_{50}	Clearance Time, elimination or depuration expressed as half-life
d.wt	dry weight / dw
dfi	daily food intake
DG	Directorate General

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DIN	Deutsche Industrie Norm (German norm)
DNA	DeoxyriboNucleic Acid
DOC	Dissolved Organic Carbon
DT50	Degradation half-life or period required for 50 percent dissipation / degradation
DT90	Period required for 50 percent dissipation / degradation
E	Explosive (Symbols and indications of danger for dangerous substances and preparations according to Annex III of Directive 67/548/EEC)
EASE	Estimation and Assessment of Substance Exposure Physico-chemical properties [Model]
EbC50	Effect Concentration measured as 50% reduction in biomass growth in algae tests
EC	European Communities
EC10	Effect Concentration measured as 10% effect
EC50	median Effect Concentration
ECB	European Chemicals Bureau
ECETOC	European Centre for Ecotoxicology and Toxicology of Chemicals
ECVAM	European Centre for the Validation of Alternative Methods
EDC	Endocrine Disrupting Chemical
EEC	European Economic Communities
EINECS	European Inventory of Existing Commercial Chemical Substances
ELINCS	European List of New Chemical Substances
EN	European Norm
EPA	Environmental Protection Agency (USA)
ErC50	Effect Concentration measured as 50% reduction in growth rate in algae tests
ESD	Emission Scenario Document
EU	European Union
EUSES	European Union System for the Evaluation of Substances [software tool in support of the Technical Guidance Document on risk assessment]
F(+)	(Highly) flammable (Symbols and indications of danger for dangerous substances and preparations according to Annex III of Directive 67/548/EEC)
FAO	Food and Agriculture Organisation of the United Nations
FELS	Fish Early Life Stage
GLP	Good Laboratory Practice
HEDSET	EC/OECD Harmonised Electronic Data Set (for data collection of existing substances)
HELCOM	Helsinki Commission -Baltic Marine Environment Protection Commission
HPLC	High Pressure Liquid Chromatography
HPVC	High Production Volume Chemical (> 1000 t/a)
IARC	International Agency for Research on Cancer
IC	Industrial Category

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IC50	median Immobilisation Concentration or median Inhibitory Concentration
ILO	International Labour Organisation
IPCS	International Programme on Chemical Safety
ISO	International Organisation for Standardisation
IUCLID	International Uniform Chemical Information Database (existing substances)
IUPAC	International Union for Pure and Applied Chemistry
JEFCA	Joint FAO/WHO Expert Committee on Food Additives
JMPR	Joint FAO/WHO Meeting on Pesticide Residues
Koc	organic carbon normalised distribution coefficient
Kow	octanol/water partition coefficient
Kp	solids-water partition coefficient
L(E)C50	median Lethal (Effect) Concentration
LAEL	Lowest Adverse Effect Level
LC50	median Lethal Concentration
LD50	median Lethal Dose
LEV	Local Exhaust Ventilation
LLNA	Local Lymph Node Assay
LOAEL	Lowest Observed Adverse Effect Level
LOEC	Lowest Observed Effect Concentration
LOED	Lowest Observed Effect Dose
LOEL	Lowest Observed Effect Level
MAC	Maximum Allowable Concentration
MATC	Maximum Acceptable Toxic Concentration
MC	Main Category
MITI	Ministry of International Trade and Industry, Japan
MOE	Margin of Exposure
MOS	Margin of Safety
MW	Molecular Weight
N	Dangerous for the environment (Symbols and indications of danger for dangerous substances and preparations according to Annex III of Directive 67/548/EEC
NAEL	No Adverse Effect Level
NOAEL	No Observed Adverse Effect Level
NOEL	No Observed Effect Level
NOEC	No Observed Effect Concentration
NTP	National Toxicology Program (USA)
O	Oxidizing (Symbols and indications of danger for dangerous substances and preparations according to Annex III of Directive 67/548/EEC)

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OECD	Organisation for Economic Cooperation and Development
OEL	Occupational Exposure Limit
OJ	Official Journal
OSPAR	Oslo and Paris Convention for the protection of the marine environment of the Northeast Atlantic
P	Persistent
pKa	negative log of the acid dissociation constant
PBT	Persistent, Bioaccumulative and Toxic
PBPK	Physiologically Based PharmacoKinetic modelling
PBTK	Physiologically Based ToxicoKinetic modelling
PEC	Predicted Environmental Concentration
pH	logarithm (to the base 10) (of the hydrogen ion concentration $\{H^+\}$
pKa	logarithm (to the base 10) of the acid dissociation constant
pKb	logarithm (to the base 10) of the base dissociation constant
PNEC	Predicted No Effect Concentration
POP	Persistent Organic Pollutant
PPE	Personal Protective Equipment
QSAR	(Quantitative) Structure-Activity Relationship
R phrases	Risk phrases according to Annex III of Directive 67/548/EEC
RAR	Risk Assessment Report
RC	Risk Characterisation
RfC	Reference Concentration
RfD	Reference Dose
RNA	RiboNucleic Acid
RPE	Respiratory Protective Equipment
RWC	Reasonable Worst Case
S phrases	Safety phrases according to Annex III of Directive 67/548/EEC
SAR	Structure-Activity Relationships
SBR	Standardised birth ratio
SCE	Sister Chromatic Exchange
SDS	Safety Data Sheet
SETAC	Society of Environmental Toxicology And Chemistry
SNIF	Summary Notification Interchange Format (new substances)
SSD	Species Sensitivity Distribution
STP	Sewage Treatment Plant
T(+)	(Very) Toxic (Symbols and indications of danger for dangerous substances and preparations according to Annex III of Directive 67/548/EEC)

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TDI	Tolerable Daily Intake
TG	Test Guideline
TGD	Technical Guidance Document [1]
TNsG	Technical Notes for Guidance (for Biocides)
TNO	The Netherlands Organisation for Applied Scientific Research
UC	Use Category
UDS	Unscheduled DNA Synthesis
UN	United Nations
UNEP	United Nations Environment Programme
US EPA	Environmental Protection Agency, USA
UV	Ultraviolet Region of Spectrum
UVCB	Unknown or Variable composition, Complex reaction products of Biological material
vB	very Bioaccumulative
vP	very Persistent
vPvB	very Persistent and very Bioaccumulative
v/v	volume per volume ratio
w/w	weight per weight ratio
WHO	World Health Organization
WWTP	Waste Water Treatment Plant
Xn	Harmful (Symbols and indications of danger for dangerous substances and preparations according to Annex III of Directive 67/548/EEC)
Xi	Irritant (Symbols and indications of danger for dangerous substances and preparations according to Annex III of Directive 67/548/EEC)

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The report provides the comprehensive risk assessment of the substance Diantimony trioxide (DAT). It has been prepared by Sweden in the frame of Council Regulation (EEC) No. 793/93 on the evaluation and control of the risks of existing substances, following the principles for assessment of the risks to man and the environment, laid down in Commission Regulation (EC) No. 1488/94.

The evaluation considers the emissions and the resulting exposure to the environment and the human populations in all life cycle steps. Following the exposure assessment, the environmental risk characterisation for each protection goal in the aquatic, terrestrial and atmospheric compartment has been determined.

The environmental risk assessment concludes that there is concern for the sediment compartment due to formulation and application of flame-retardant textile back-coating and from one production site. There is no concern identified for the other compartments.

For human health the scenarios for occupational exposure, consumer exposure and humans exposed via the environment have been examined and the possible risks have been identified. The human health risk assessment concludes that there is concern for workers only, not for the other human categories.

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