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Proposed Soil Quality Guidelines
Antimony
Environmental and Human Health Effects

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1.0 INTRODUCTION

Soil quality guidelines are numerical soil concentrations intended to be protective of human and environmental health for current and potential future uses of land. They are frequently used for the assessment and remediation of contaminated sites.

The Canadian Council of Ministers of the Environment (CCME) published *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 2006) to provide a framework and methodology for developing risk-based soil quality guidelines protective of human health and the environment. The guidelines developed using this protocol have been published in the Canadian Soil Quality Guidelines (CCME 1999) and also formed the basis for some provincial guidelines, including the *Alberta Tier 1 Soil and Groundwater Remediation Guidelines (*AEP 2016). The current Alberta guidelines for antimony, date back to the earlier Interim Canadian Environmental Quality Criteria for Contaminated Sites (CCME 1991), which did not use modern risk-based approaches.

This report provides the basis for proposed soil quality guidelines for antimony based on the CCME (2006) protocol. It includes a review of sources of antimony, concentrations in the environment, fate and behaviour, and toxicological effects on soil microorganisms, plants, animals and humans. Guidelines are derived for the agricultural, residential/parkland, commercial and industrial land uses as defined by CCME (2006), as well as the natural area land use defined by AEP (2016).

2.0 BACKGROUND INFORMATION

2.1 Physical and Chemical Properties

Antimony (Sb; CAS # 7440-36-0) is a Group 5A metalloid element with an atomic number of 51 and an atomic weight of 121.75 (ATSDR, 1992). Antimony has four oxidation states, Sb(-3), Sb(0), Sb(+3), and Sb(+5), with the +3 state being the most stable and commonly found. Physical and chemical properties of antimony and some antimony compounds are presented in Table 1.

Table 1 Physical and Chemical Properties of Antimony and Select Antimony Compounds

Property	Antimony	Antimony Trioxide	Antimony Trisulfide	Stibine
Chemical formula	Sb	O_3Sb_2	S_3Sb_2	H_3Sb
CAS Registry Number	7440-36-0	1309-54-4	1345-04-6	7803-52-3
Molecular weight (g/mol)	121.75	291.5	339.69	124.77
Physical state at 25°C	Solid	Solid	Solid	Gas
Melting point (°C)	630	656	550	-88
Boiling point (°C)	1750	1550	1150	-17
Density (g/cm3)	6.7	5.2	4.64	2.204

Table 1 Physical and Chemical Properties of Antimony and Select Antimony Compounds				
Property	**Antimony**	**Antimony Trioxide**	**Antimony Trisulfide**	**Stibine**
Water solubility (g/100mL)	Slightly soluble	Slightly soluble	0.000175	0.41

a – values from (ATSDR, 1992)

2.2 Analytical Methods

Antimony in environmental media is most commonly analyzed using inductively coupled plasma-optical emission spectrometry (ICP-OES) or inductively coupled plasma-mass spectrometry (ICP-MS), although atomic absorption spectrophotometry (AAS) can also be used.

The first step in both ICP-OES and IOCP-MS is conversion of an analyte solution into ions by passing it though a plasma source. ICP-OES measures the emission of light from the heated ions, which occur at a specific wavelength and which has an intensity correlated to the concentration in the original analyte solution. ICP-MS directs these ions into a magnetic field, which deflects their path based on their mass to charge ratio onto a detector. This allows identification and quantification of chemical species in the original analyte solution (Harris 2003).

AAS involves conversion of an analyte solution into a gaseous state within a flame or furnace. A light source is then directed through the flame, and the concentration of the analyte is determined by the absorption of the light source at specific frequencies characteristic of individual elements (Harris 2003).

Typical soil sample preparation techniques using strong acid leachate are intended to indicate the environmentally available concentration of antimony (CCME 2013). Water samples are field filtered and preserved in solutions with pH values less than 2 (CCME 2013).

United States Environmental Protection Agency (USEPA) methods for analysis of antimony include EPA 200.5 *Determination of Trace Elements in Drinking Water by Axially Viewed Inductively Coupled Plasma – Atomic Emission Spectrometry* (USEPA 2003) EPA 3005A *Acid Digestion of Waters for Total Recoverable or Dissolved Metals for Analysis by FLAA or ICP Spectroscopy* (USEPA 1992), and method 3050B *Acid Digestion of Sediments, Sludges, and Soils* (USEPA 1996); the extraction method is critical for determining concentrations and needs to be consistent.

USEPA recommends analysis using either method 6010C *Inductively Coupled Plasma-Atomic Emission Spectrometry* (USEPA 2007a) or method 6020a *Inductively Coupled Plasma-Mass Spectrometry* (USEPA 2007b).

CCME (1993a) previously recommended method 3120B *Inductively Coupled Plasma (ICP) Method* (CCME 1993b) for determination of metals in water samples, and USEPA method 6010 for determination from metals in soils, sludges, sediments. Current CCME (2013) draft recommendations are for ICP-OES, ICP-MS, or AAS. No standard methodologies are recommended, but analytical standards are required to be matrix matched to samples.

2.3 Sources and Concentrations in the Canadian Environment

The assessment of soil quality for naturally occurring metals must take into consideration regional variations in background concentrations in Canada. Background concentrations and environmental fate of metals strongly depend on geological and biological characteristics; therefore, any assessment of potential risks should take into consideration regional differences in metal content in the natural environment (Chapman and Wang 2000).

Relatively high concentrations of metals can occur naturally in Canadian soils, stream sediments, and water blurring the distinction between anthropogenic pollution versus naturally occurring geological formations and natural bodies of ore. Soils and sediments reflect the composition of parent material, resulting in higher metal concentrations in mineralized areas (Wilson et al. 2010). In the determination of anthropogenic metal contamination of soils, no single guideline concentration can adequately represent the variance in background concentrations across Canada (Painter et al. 1994; Chapman and Wang 2000).

Antimony occurs naturally as sulphide ore and to a lesser extent oxide ore, with the most important source globally being Stibnite (antimony trisulphide). It is often found in trace amounts in the ores of other metals such as copper, lead, silver, gold, iron and arsenic and is a common component in coal and oil (Filella et al 2002). Within Canada, antimony deposits are most commonly found in the Cordilleran and Appalachian regions, but have also been identified in the Yukon (USGS 2007), Northwest Territories (Fawcett et al 2015), British Columbia and Quebec.

Mining districts are characterized by naturally occurring metals in soil, sediment, rock, and water at concentrations that could result in their classification as "contaminated sites" (Painter et al. 1994). Antimony ores have historically been mined in a number of locations in eastern Canada, including West Gore in Nova Scotia, York County in New Brunswick and more recently at Beaver Brook, Newfoundland until operations were suspended in 2013 (Newfoundland and Labrador Department of Natural Resources, 2015).

Production of antimony in Canada is currently limited to recovery as a by-product of lead smelting. This includes primary production from lead ores and secondary metal production from recycled lead (primarily car batteries). The majority of antimony used in Canada is imported (Environment Canada and Health Canada 2010).

Antimony is used in a wide range of manufactured products. Two thirds of total antimony production is used in manufacture of flame retardants (Krachler et al. 2005). Other uses include automobile brake linings, pigments, paints and glass. Antimony is found in products containing lead such as batteries or ammunition, where it increases the hardness and strength of the metal alloy. Antimony is also used as a catalyst in the production of polyethylene terephthalate (PET) plastic. The concentrations of antimony in two drinking water bottles made of PET were found to be 351 and 397 mg/kg (Shotyk et al. 2005). Antimony is also used as a weighting material in some drilling fluids (Adebayo et al., 2011).

Additional anthropogenic sources of antimony released to the environment aside from mining and smelting activities include disposal or incineration of manufactured products containing antimony; coal combustion; waste water treatment plants; dust from brake linings and tires; drilling fluids; refinery processes; oil releases; and spent ammunition (Filella et al. 2002, Scheinost et al. 2006, Mansson et al. 2009, Environment Canada and Health Canada, 2010, and Shotyk et al. 2005).

A summary of background antimony concentrations is included in Appendix A.

2.3.1 Atmosphere

Natural sources of atmospheric antimony emissions include windborne soil particles, volcanoes and sea salt spray (Pacyna and Pacyna 2001). The concentration of antimony in oceans is typically about 0.2 μg/L (Filella 2002).

Anthropogenic sources of atmospheric antimony include emissions from smelting, combustion of fossil fuels and waste incineration. Environment Canada's National Pollutant Release Inventory (NPRI) includes records of releases of antimony exceeding a set threshold at select facilities. In 2013, the reported mass of total antimony released to the air was 0.499 tonnes. Over 80% of the total was released from two lead smelters and a coke refinery (Environment Canada 2015).

Antimony emitted from coal combustion and refuse incineration is associated with fine particulates in stack emissions and has been decreasing as the efficiency of recovery of these particles has increased (Pacyna and Pacyna 2001).

Based on a review of data in literature, antimony concentrations in aerosols are reported to range from <0.1 ng/m^3 in the atmosphere over remote oceans to several ng/m^3 over industrialised areas (Filella 2002).

Antimony concentrations were measured in ice cores from the Canadian Arctic which provide a record of atmospheric antimony deposition extending approximately 16,000 years into the past (Krachler et al. 2005, 2008). The results showed that more than 99% of the antimony deposition was a

result of anthropogenic sources. The study confirms that long range transport of antimony is occurring in the atmosphere. Therefore, measured concentrations of atmospheric antimony at remote locations are expected to include a significant proportion of antimony from diffuse anthropogenic sources.

Concentrations of antimony in air were assessed in 161 samples collected between 1988 and 2004 from three locations in the Great Lakes basin in Ontario. The locations were identified as having low anthropogenic impacts (Environment Canada and Health Canada 2010). The air concentrations ranged between the limit of detection (not specified) and 29 ng/m^3, with a 90th percentile concentration of 10 ng/m^3. The median concentration was below the limit of detection. A separate study (Biegalski 1998) attributes the metal concentrations in air at these sampling points to anthropogenic emissions, including an area of smelting more than 200 km from the sites. No point sources were identified in close proximity to the sampling locations.

Lynch et al. (1980) studied the concentration of antimony in airborne particulate matter in Trail, British Columbia, which is located adjacent to a lead/zinc smelter. The results were compared with concentrations in the nearby community of Nelson, British Columbia. Nelson was considered unaffected by the emissions from the smelter. The mean concentration of antimony measured in airborne particulate matter in Trail was 125 ng/m^3 compared with a reference concentration of less than 75 ng/m^3 measured in Nelson. For comparison, an antimony concentration in air of 62,400 ng/m^3 was reported at a gold smelter in rural Canada (Environment Canada and Health Canada 2010).

Median atmospheric antimony concentrations in fine particulate matter (PM2.5) at seven sites in eastern and western Canada ranged from 0.05 to 0.5 ng/m^3 (Celo and Dabek-Zlotorzynska 2010). The lowest concentrations were recorded at a rural Site and the highest concentrations were recorded in a city with intensive industrial activities (Windsor, Ontario).

Atmospheric antimony concentrations in Canada are typically expected to be less than 10 ng/m^3 at locations not proximal to mineralised areas or point source emissions.

2.3.2 Soil and Dust

Antimony is widely reported to be present in the Earth's crust at an average concentration of around 0.2 mg/kg (Filella 2002). In general, geochemical data indicates that shales and coal are enriched in antimony relative to the average antimony concentration in the crust. The reported antimony concentration in sub-bituminous coal, which makes up 82% of coal produced in Canada, is 0.722 mg/kg (Environment Canada and Health Canada 2010). The estimated median concentration of antimony in shale worldwide is reported to be 1 mg/kg (Reimann et al. 2010). Natural background levels of the antimony in soil overlying rocks with enriched values are also expected to be enhanced (Rencz et al, 2006).

Globally, the average background concentration of antimony in soils is reported to be typically between 0.5 and 1 mg/kg (Telford et al. 2009), although other sources report wider ranges (e.g. Hammel et al. 2000). Review of published data indicates antimony concentrations in Canadian soils are generally similar to this range.

The median antimony concentration measured in 1,273 samples of Ap horizon soil across Canada is reported to be 0.6 mg/kg (Reimann et al. 2010).

Grunsky et al. (2012) reported a median antimony concentration of 0.4 mg/kg in 12,629 samples of glacial clay from across Canada. The range between the first and third quartiles was 0.2 to 0.9 mg/kg and the maximum concentration was 23.8 mg/kg. Elevated concentrations were found in central British Columbia, central Baffin Island, the Labrador Trough and the Bathurst Camp of New Brunswick (which include areas identified as having antimony mineralisation).

Significantly elevated concentrations of antimony can occur in the soil due to anthropogenic activities, including mining and smelting and at shooting ranges (due to the presence of antimony in lead shot).

Lynch et al. (1980) studied the concentrations of antimony in soil in Trail, British Columbia, which is a community located adjacent to a lead/ zinc smelter. The mean concentration of antimony measured in soil was 49 mg/kg. This was compared with soil antimony concentrations of 11 mg/kg in the nearby community of Nelson, which is reported not to be impacted by the smelting activities due to topography and the prevailing wind.

Wilson et al. (2010) provides a summary of recent studies on antimony concentrations at sites worldwide that were contaminated by anthropogenic activities. Concentrations in soil in the vicinity of several mining sites were well above 10,000 mg/kg, with concentrations up to 40 mg/kg seen at a location 300 km downgradient.

The maximum antimony concentration in soil at four Canadian shooting ranges was 570 mg/kg, with the highest concentration in the upper 10 cm. High leachabilities were also reported for antimony in the soils (Laport-Sauure et al 2011).

A comparison of total elemental concentrations in garden soil, house dust and street dust in the city of Ottawa reported geometric means for antimony of 0.25 mg/kg for garden soil, 5.54 mg/kg for house dust and 0.44 mg/kg for street dust (Rasmussen et al. 2000). Abrasion of antimony from brakes, tires and street surfaces, as wells as emission of aerosolised antimony in vehicle exhausts are the main sources of antimony in urban fine dust (WHO 2003).

2.4 Freshwater

2.4.1 Surface Water

Natural sources of antimony in the aquatic environment include natural weathering of rocks (especially granites, shale and coal) and runoff from soils into streams and lakes. Significant differences in antimony are likely due to the predominant lithology of the watershed. Naturally high concentrations have been identified around hot springs and boreholes in geothermal waters (e.g. up to 10 wt%) (Filella et al. 2002).

Based on a review of several studies, antimony concentrations in surface water in Canada are generally less than 1 ug/L.

The antimony concentrations measured in 1,194 samples of river water from the prairies provinces and the Northwest Territories were between 0.001 and 8 µg/L, with a 90th percentile concentration of 0.25 µg/L (Environment Canada and Health Canada 2010). The sample locations are reported to have low anthropogenic influence.

Thirty three samples of lake water were collected from Lac Blouin, Quebec in 2006 (DDEP, 2008). The range of antimony concentrations was between 0.04 and 0.17 µg/L, with a 90th percentile value of 0.14 µg/L. The results are reported to be in an area of low anthropogenic influence (Environment Canada and Health Canada 2010). However, it is noted that the samples were collected prior to restoration of upstream mine tailings ponds.

Median concentrations of total antimony in the Great Lakes measured in 1981 to 1985 were between 0.072 µg/L for Lake Superior and 0.25 µg/L for Lake Michigan (Rossmann and Barres 1988).

Surface water samples from the Fraser River at a location in Hope, British Columbia were analysed for antimony (Swain 2007). The samples were collected at approximately two-week intervals between 2003 and 2005. The maximum concentration was 0.18 µg/L. Up-gradient anthropogenic activity includes paper and pulp mills, which are not considered to be significant sources of antimony.

Antimony concentrations in streams in the Tintina Gold Province in Alaska and the Yukon ranged between <2 µg/L to 660 µg/L (USGS 2007). The highest concentrations were reported in samples close to areas of mineralization and in areas with exposed ore and tailings near surface.

A study of tap water in eight Port Colborne, Ontario homes found antimony concentrations between 0.45 and 0.97 µg/L (OMOE 2002). Antimony concentrations in soil were not generally elevated and drinking water was from a treated municipal supply from a surface water source.

2.4.2 Groundwater

Antimony concentrations in groundwater are very variable and are highly dependent on geology. Concentrations are frequently below 1 μg/L, but higher concentrations may occur naturally.

Thirty four samples of pristine groundwater from a calcareous aquifer in southern Ontario contained an arithmetic mean average antimony concentration of 0.0022 μg/L, (Shotyk et al. 2005).

Twenty-six samples of groundwater from domestic wells in a shale gas field in New Brunswick were analysed for antimony (Al et al, 2013). Concentrations ranged between the detection limit of 0.1 μg/L and 1.8 μg/L. Concentrations in 65% of samples were below the limit of detection. The higher concentrations likely reflect the chemistry of shales present in the area which are typically enriched in antimony compared to calcareous strata in the above study.

Antimony concentrations in groundwater in the Tintina Gold Province in Alaska and the Yukon were between <2 μg/L and 60 μg/L (USGS 2007). The higher antimony concentrations in this area are reported to be associated with subsurface shear and fault hosted sulphide zones.

In the United States groundwater samples from 3,009 monitoring and drinking water wells nationwide were analysed for antimony (USGS 2011). This included wells in a range of land use and physiographic settings. The 90th percentile value was below the limit of detection of 1 μg/L and the maximum concentration recorded was 6.3 μg/L. A lower limit of detection was used for analysis of antimony from a network of 954 groundwater wells in Minnesota (Minnesota Pollution Control Agency 1999). The median concentration was reported to be 0.016 μg/L.

Anthropogenic activities are also expected to result in elevated antimony concentrations in groundwater. The maximum antimony concentration measured in landfill leachates in Switzerland was 300 μg/L (Shotkyk et al. 2005). Groundwater impacted by the leachates contained in the order of 0.1 μg/L antimony.

2.5 Sediments

In general concentrations of antimony in sediments are of a similar range to those measured in soils. Average concentrations are generally below 1 mg/kg and higher concentrations are associated with mineralised areas and anthropogenic activities such as smelting and mining. Elevated antimony concentrations in sediments near the outfalls of sewers and fertiliser facilities have also been reported (Filella et al, 2002).

The median antimony concentration of 50,434 samples of stream sediments in Canada is reported to be 0.3 mg/kg, with a range between <0.1 and 170 mg/kg (Reimann et al. 2010).

Samples of lake sediments from northwest Ontario were reported to contain between <0.2 and 60 mg/kg antimony. The median concentration of the 8,661 samples was 0.2 mg/kg. The 90th percentile value was 0.3 mg/kg (Friske et al. 1998 cited in Environment Canada and Health Canada, 2010).

Lalonde et al. (2011) measured antimony concentrations in the vicinity of outfalls from two water filled coal-combustion ash-lagoons in Nova Scotia and New Brunswick. A median antimony concentration of 1.2 mg/kg was reported for samples close to the outfall into the lake compared with 0.2 mg/kg in the two background samples. The difference was reported to be statistically significant. No significant difference was identified between the antimony concentrations in sediment from the outfall area in a river (median of 0.1 mg/kg) and the two samples from the background location (0.3 mg/kg).

Concentrations of antimony in stream sediments collected downstream from gold deposits in the Yukon and Alaska varied greatly. Antimony concentrations had a median of 192 mg/kg and a maximum of 7,200 mg/kg. The concentrations downstream of known lodes were at least an order of magnitude higher than sediments in streams draining unmineralized areas (USGS 2007).

2.6 Aquatic Organisms

Antimony has no known biological function (Filella et al. 2002). Concentrations of antimony in microbiota (primarily marine and freshwater algae) are reported to be generally between 0.1 and 0.2 µg/g dry weight (Filella et al. 2007).

A study of trout from Cayuga Lake in Ontario reported antimony concentrations between 0.46 and 0.86 µg/kg in trout aged between one and twelve years old (Tong et al. 1974). There was no apparent trend in concentrations with age. The lake is not identified as being near to a point source of antimony.

Samples of aquatic horsetail in a creek downgradient of a gold mine in Northwest Territories contained 123 and 146 mg/kg of antimony (Fawcett et al. 2015). The plants were growing in sediment with measured antimony concentrations ranging between 354 and 433 mg/kg.

Antimony concentrations were analysed in macroinvertebrates from a creek downstream of an antimony and arsenic metal mine and processing site in Australia (Telford 2009). All invertebrate samples with the exception of gastropods were reported to show elevated concentrations of antimony. Concentrations in the gastropods were below 0.03 µg/g and the highest concentrations were in the worms, with a mean concentration of 316 µg/g. The results indicate that concentrations of antimony were low compared with other metals, as well as having a low enrichment factor. The study found that five samples of floating green algae contained average antimony concentrations of

96 µg/g, compared with reported background concentrations in the region of 2.3 µg/g. Algal uptake was correlated with concentrations in stream water.

2.7 Plants

Antimony is not an essential element for plants (Tschan et al. 2009). Background antimony levels in the tissue of terrestrial vascular plants are reported to range from 0.0002 to 0.05 mg/kg (Shtangeeva et al. 2011). Bowen (1979) reports higher antimony concentrations with median concentrations in moss up to 0.171 mg/kg, lichen up to 1 mg/kg and spruce bark (from Canada) of 3.6 mg/kg.

Antimony concentrations in nine samples of lichen from Ontario recorded concentrations between the limit of detection (0.04 mg/kg) and 0.21 mg/kg (Matschullat et al. 1999). Mean concentrations of antimony in lichen from the Northwest Territories were reported to be between 0.08 and 0.13 mg/kg (Puckett and Finegan 1980).

Uptake of anthropogenic antimony, leading to increased concentrations in plant material, has been confirmed in a number of studies.

The concentrations of antimony in nine samples of moss were between 0.04 and 0.09 mg/kg at a site more than 2 km from a former mercury mine in British Columbia (Plouffe et al. 2004). The concentration of a sample collected within 2 km of the mine was 0.1 mg/kg and the concentrations reported in moss at the mine site were 0.44 and 4.2 mg/kg.

Tree bark samples from urban, rural and industrial (harbour) areas and close to major roads in France and Germany were analysed for antimony to assess the level of enrichment due to anthropogenic sources (Guéguen et al. 2012). The highest antimony concentrations were reported in the samples adjacent to a major road intersection, with a maximum concentration in bark of 13.85 mg/kg. The median concentration for the high traffic areas were more than four times higher than the background sample. This is thought to be due to antimony from vehicle brake pads. The results for rural areas also show evidence of transport of antimony over longer distances.

Maximum plant concentrations of 336 mg/kg antimony were identified by Ainsworth et al. (1990) in an area downgradient of an antimony smelter in the United Kingdom. Background levels were reported to be <1 mg/kg.

Bech et al. (2012) measured antimony concentrations in native plants in an abandoned antimony mine in the Spanish Pyrenes. Three samples of plants were taken at six sites including one considered unpolluted. The mean soil concentration was 8.1 mg/kg at the unpolluted location (by aqua regia extract). Mean concentrations in the plant shoots at this location ranged between 1.1 to 1.6 mg/kg and

the concentration in roots were 2.3 and 3.0 mg/kg. At the more heavily contaminated locations mean concentrations were up to 69 mg/kg in shoots and 402 mg/kg in roots.

Feng et al. (2013) summarises the results of several studies on antimony concentrations in different plants. The report refers to three species growing in an abandoned mining area where very high concentrations up to 1,367 mg/kg were recorded in plant tissue with no obvious toxic symptoms. These species are referred to as 'hyperaccumulators'.

Hammel et al. (2000) studied antimony concentrations in nineteen plant species growing in a historical mining area with soil antimony concentrations up to 500 mg/kg. The maximum antimony concentrations in plants were 0.34 mg/kg in shoots and 2.2 mg/kg in leaves. In grain, concentrations up to 0.09 mg/kg were reported. Results are reported to be similar to literature values (typically less than 1 mg/kg) for plants grown in uncontaminated soil.

2.8 Animals

A survey of slaughtered bovine and porcine specimens in Ontario reported mean concentrations of antimony in muscle between 46 and 113 µg/kg, and between 23 and 352 µg/kg in kidneys. Mean antimony contents in 115 avian carcases were 152 and 123 µg/kg in muscles (for birds aged under and over 14 weeks respectively) and were 179 and 167 µg/kg in livers (Frank et al. 1986).

2.9 Humans

People can be exposed to antimony primarily through ingestion of food and to a lesser extent from air, soil, dust and drinking water (Environment Canada and Health Canada 2010). Exposure can occur due to dermal contact with a wide range of household products containing antimony (e.g. plastics, flame retardants, fabrics) and inhalation of tobacco smoke (Health Canada 2013). Low concentrations of antimony have been reported in bottled beverages which have been attributed to leaching of antimony from containers (Shotyk et al. 2005). Cullen et al. (1998) report that an estimated 10 to 23 µg of antimony is ingested daily from food. Dust is reported to contribute more than 1 µg in antimony intake each day.

The adsorption, distribution and excretion of antimony varies depending on oxidation state and solubility (CDC 2015, Health Canada 2013). Humans excrete antimony mainly via the kidneys so urine concentrations are valid biomarkers for exposure.

Antimony concentrations in the urine of participants aged 6 to 79 years were measured in the Canadian Health Measures Survey in two cycles, Cycle 1 in 2007 to 2009 and Cycle 2 in 2009 to 2011 (Health Canada 2013). The geometric mean antimony concentrations in the two cycles were 0.042 and 0.048 µg/L.

A study in Quebec included analysis of antimony in 471 blood and 318 urine samples (INSPQ 2004). The study found more than half of samples contained antimony concentrations below the detection limit of 0.12 μg/L. The geometric mean antimony concentrations in blood and urine were 0.267 and 0.584 μg/L respectively.

Data on exposure to antimony for the United States is collated in the Fourth National Report on Human Exposure to Environmental Chemicals (CDC 2015). The report presents data on concentrations in urine in samples collected between 1999 and 2012. Median concentrations for each two year period range between 0.047 μg/L (in 2011 to 2012) and 0.13 μg/L (in 1999 to 2002).

Jain (2013) used data from the United States National Health and Nutrition Examination Survey from 2003 to 2010 to evaluate the effect of various factors on the levels of metals (including antimony) in urine. Antimony concentrations were shown to be declining over the study period and were also higher in smokers than non-smokers. Adjusted geometric mean antimony concentrations were 0.070 μg/L for non-smokers and 0.085 μg/L for smokers.

A study of antimony concentrations in urine in infants under 2 years old was completed by Dezateux et al in 1997. The study was completed in the United Kingdom and found that concentrations in urine ranged between the limit of detection of 0.02 μg/L and 0.41 μg/L. Geometric mean concentrations for four groups studied ranged between 0.05 and 0.28 μg/L. The mean antimony concentration for healthy adults is reported to be 0.2 μg/L and a reference value for the UK based on concentrations in plasma and urine is reported to be 1 μg/L.

Cullen et al. (1998) carried out a study to determine a reference range for antimony in serum and urine of infants less than 1 year old living in Ireland. Mean antimony concentrations in the 97 samples of serum ranged between 0.16 and 0.18 μg/L. The reference range was reported to be between 0.09 and 0.25 μg/L. A concentration of less than 0.9 μg/L was reported in urine (uncorrected for creatine).

Gebel et al. (1998) studied antimony concentrations in urine and scalp hair in 218 residents living in an area of Germany with high antimony concentrations in the soil and 76 people in reference group who were not exposed to high soil concentrations. Concentrations of antimony in drinking water were generally below 0.002 mg/L. Antimony concentrations in the soil were reported to be less than 0.5 mg/kg at background locations compared with 776 mg/kg at the residences in the mining area. Concentrations of antimony in urine were a median of 0.46 μg/L in the mining area compared with 1.11 μg/L in the background location. The median concentrations in hair samples were 0.028 μg/L in the mining area compared with 0.044 μg/L in the background area. Neither urine nor scalp hair showed increasing antimony levels with increasing concentrations of antimony in the soil. The higher

antimony concentrations in the reference group were not explained by any of the factors assessed in the study.

A reference concentration range of 0.19 to 1.1 µg/L for antimony in urine was reported by Minoia et al. (1989) based on measurements in samples from 360 'unexposed' subjects living in Italy. An indicative concentration range in blood was 0.03 to 3.6 µg/L (based on 27 samples) and an indicative concentration range in serum was 0.01 to 1.7 µg/L. It is noted that the concentrations of antimony in urine appear to vary widely between the studies and are generally much higher in the older studies (pre 2000) compared with more recent data.

2.10 Existing Soil and Water Quality Criteria and Guidelines

Soil and water quality criteria and guidelines for antimony have been developed by several agencies, and are summarized in Appendix B. The current Canadian soil quality guidelines, 20 mg/kg for agricultural and residential/parkland and 40 mg/kg for commercial and industrial (CCME, 1999) were first published in 1991 and were not developed using current risk-based protocols.

3.0 ENVIRONMENTAL FATE AND BEHAVIOUR IN SOIL

Antimony is not degraded in the environment and its fate is dependent on a series of physiochemical and biological factors that influence cycling among biotic and abiotic components of the environment. Available data indicate that antimony in soil accumulates near the surface and concentrations decrease with depth (Foster 1989, Ainsworth 1988, Trnovsky et al. 1988, Van der Sloot et al. 1982). This is consistent with deposition from the atmosphere, where antimony can be transported over long distances.

In natural soils, antimony is generally present as Sb(V). Experiments indicate that after addition of Sb_2O_3 to soil, more than 70% was in the form of Sb(V) after two days (Filella et al. 2009). In soil solution Sb(V) is dominantly present as the anionic species $Sb(OH)_6^-$ at most pH levels. This is also the most common form observed in contaminated soils near smelters and at shooting ranges (Okkenhaug et al. 2011).

Antimony in soils is often considered immobile and rather non-reactive under normal environmental conditions (Filella et al. 2009, ATSDR 1992). Hammel et al. (2010) report mobilities of less than 1%. However several studies have identified substantially higher mobility in soil in active mining areas (Okkenhaug et al. 2011).

3.1 Solubility

Antimony can exist at environmentally relevant concentrations and conditions as a soluble species (Wilson et al. 2010). The solubility of antimony oxides is reported to increase under more oxidising

conditions and appears to be independent of pH (Ashley et al. 2013). Under oxidising conditions, the solubility of antimony is not expected to limit its mobility. Under reducing conditions the mobility of antimony may be limited by the solubility of antimony sulphides such as stibnite (Krupka and Serne 2002). The composition of the antimony compound has also been shown to substantially affect its solubility (Okkenhaug et al. 2011).

Dissolved antimony occurs in the form Sb(V) and Sb(III). In oxic media dissolved antimony is predominantly found as Sb(V) .Thermodynamic calculations predict that Sb(III) is more likely to be found in anoxic media. However, significant concentrations of thermodynamically unstable antimony species have been measured in oxic and anoxic systems (Filella et al. 2009). The reason for this is not clear.

3.2 Adsorption

The concentration of antimony in soils and sediments is likely controlled by adsorption reactions. The adsorption of anions is generally higher at lower pH (Krupka and Serne 2002). Therefore adsorption of $Sb(OH)_6^-$ (the dominant form of antimony in soil solution) to hydroxide and oxide minerals is predicted to be limited at above neutral values (Krupka and Serne 2002). This is supported by several laboratory studies.

Laboratory studies have shown that Sb(V) adsorbs strongly to iron and manganese oxides and hydroxides (Ritchie et al. 2013). Iron oxides and hydroxides have been identified as particularly important sorbents in the near surface for Sb(V). Over a pH range of 2.5 to 7 the maximum sorption of Sb(V) to iron hydroxide in floodplain soils occurred at a pH of 4, with a reported 95% adsorbed (Wilson et al. 2010). Antimony also adsorbs strongly to clay silicate minerals (Ritchie et al. 2013, Wilson et al. 2010). One study was identified that suggests antimony adsorption to aluminum silicate minerals can be greater when the antimony has a primary mineral origin (Wilson et al. 2010). However, in general, processed forms of antimony are thought to sorb more strongly to soils (ATSDR 1992).

Dissolved antimony in the form Sb(III) has been found to adsorb more strongly than Sb(V) and over a wider pH range (Ritchie et al. 2013, Okkenhaug et al. 2011). Sb(III) is reported to sorb most strongly to manganese hydroxides, followed by aluminium hydroxide then iron hydroxides. More than 80% of total Sb(III) was adsorbed below a pH of 6 (Wilson et al. 2010). Sb(III) adsorbed to clay, manganese and iron hydroxides was also able to oxidise to Sb(V) (Ritchie et al. 2013, Krupka and Serne 2002).

Adsorption of antimony to soil organic matter has been confirmed, with up to 30% of total Sb(III) binding to humic acids at environmentally relative conditions (Wilson et al. 2010). Less information is available relating to adsorption to organic phases than other sorbents.

Antimony does not readily volatilize except under conditions of combustion in which it subsequently condenses on particulate matter (ATSDR 1992).

3.3 Biota

Antimony has no known function in living organisms (Filella et al. 2009, Maher 2009).

Antimony does not accumulate in fish or aquatic animals (ATSDR 1992) and accumulation of antimony in terrestrial biota is very low under typical environmental conditions (Filella et al. 2002). In general uptake from the soil by biota is expected to be minimal. However, studies of antimony uptake in active mining areas with high antimony concentrations in soil found higher bioaccumulation (Okkenhaug et al. 2011 and Bech et al 2012).

4.0 BEHAVIOUR AND EFFECTS IN TERRESTRIAL BIOTA

The available information on the toxicological effects of antimony on terrestrial plants and invertebrates and soil microbial processes has been reviewed and summarized below.

Plants and invertebrates may accumulate contaminants over time if the amount of contaminant uptake is greater than the amount of contaminant elimination through excretion and metabolic processes. The process by which contaminants are directly taken up by an organism from the exposure medium (e.g. soil) is referred to as bioconcentration. The process of contaminant uptake occurring through direct uptake as well as ingestion at a rate faster than it is metabolized or excreted is referred to as bioaccumulation (CCME 2006).

4.1 Terrestrial Plants

4.1.1 Uptake, Metabolism and Elimination

Natural concentrations of antimony in the environment are low but as a result of its use in a variety of industrial products it is the ninth most mined metal worldwide (USEPA 1979). As a result, studies on the uptake of antimony into terrestrial plants are typically located in areas where mining and smelting operations have taken place (Okkenhaug et al. 2011, Tschan et al. 2009a). Numerous soil factors have been noted to influence the uptake of antimony by plants including speciation, soil chemical parameters, and variations in microbial populations (Feng 2013).

Antimony exists in a variety of oxidation states (-III, 0, III, V) and in both organic (e.g. potassium antimony tartrate) and inorganic (e.g. antimony trioxide and antimony pentoxide) forms (Okkenhaug et al. 2011, ATSDR 1992). Oxidation states Sb(III) and Sb(V) are the most common inorganic species in the environment and plant affinity for either oxidation state has been shown to vary between plant

species (Wan et al. 2013). In a study by Meharge and Jardine (2003) root uptake of Sb(III) in rice (*Oryza sativa*) competed with arsenite (As[III]) uptake in a dose-dependent manner. It has been speculated that as a result of antimony's location below arsenic on the periodic table that their similar chemistry could mean that Sb(III) behaves as an As(III) analogue and utilizes similar uptake routes (Meharge and Jardine 2003). Since the uptake of As(III) was found to be facilitated by glycerol channels (Meharge and Jardine 2003, Wysocki et al. 2003) it is reasonable to speculate that the Sb(III) uptake pathway in plants occurs passively through glycerol channels as well (Feng et al. 2013). In contrast, whilst evidence exists to suggest that As(V) is taken up into plants via the phosphate pathway (Asher and Reay 1979) it has not been proven as an effective uptake pathway for Sb(V) in plants (Tschan et al. 2009a). Two hypothesis have been proposed by Tschan et al. (2009a) to help explain the uptake process of Sb(V) in plants: i) entry into the plant occurs via an anion transport, similar to those that transport chloride (Cl^-) or nitrate (NO_3^-), or ii) entry into the plant is facilitated via the xylem by passing an incompletely sealed or damaged Casparian strip (a band of cells which regulates the flow of solutes into the plant root).

Plant uptake is critically dependent on the solubility of antimony (Chang et al. 2014), and the solubility of antimony compounds depend on a variety of soil chemical parameters and properties. Concentrations of phosphorus and calcium in soil have been shown to affect the solubility of antimony. In a study conducted on lead and antimony contaminated firing range soils, the addition of phosphorus-based fertilizers as an amendment significantly increased antimony solubility (Kilgour et al. 2007). Conversely, the oxidation and dissolution of stibnite (Sb_2S_3) as described by Okkenhaug et al. (2011) is an acidifying process neutralized by calcium carbonate ($CaCO_3$) and elevated concentrations of calcium in soil may control the concentration of antimony in soil pore-water. In the same study Okkenhaug et al. (2011) found that greater than 98% of the total water extractable antimony occurred as Sb(V) and that a relatively low percentage (<10%) of the total antimony in soil is water extractable. These findings are in close agreement with those of Johnson et al. (2005) who found leachable antimony in antimony contaminated shooting range soils were more than 99% Sb(V). The overall dominance of the pentavalent form in soils could be as a result of preferential sorption of Sb(III) to soil particulate, specifically iron-oxides (Wilson et al. 2010). Conversely, speciation analysis of antimony in plant tissues have shown a much lower percentage of the total antimony concentration in Sb(V) form with as much as 28% being Sb(III) (Okkenhaug et al. 2011). Provided the low solubility of Sb(III) into soil pore-water the increased percentage of Sb(III) in plant tissues would suggest that plant uptake of Sb(III) exceeds that of Sb(V).

Contaminant uptake may also occur via atmospheric deposition directly onto the plant (Tschan et al. 2009a). Ainsworth et al. (1990) compared antimony concentrations in vegetation both in contaminated soils and in control pots at varying set back distances from an antimony smelter. Similar concentrations of antimony were noted in both contaminated soil and control pot plants

suggesting that antimony contamination in plants was largely due to aerial deposition and not root uptake.

Plants that have accumulated high levels of antimony have been shown to have increased levels of malondialdehyde (MDA) which may indicate enhanced production of reactive oxidative species (ROS) (Feng et al. 2013). Reactive oxygen species are key signaling molecules produced in response to biotic and abiotic stresses, triggering a variety of plant defence responses. In certain plants (such as rice) efficient ROS scavenging systems have been associated with high antimony tolerance (Feng et al. 2013). Less clear is the role of phytochelatins (PCs) and the ability of plants to methylate antimony. While it has been shown that certain metabolites such as PCs have a role in arsenic detoxification (Schmoger et al. 2000), information on the responses of PCs to antimony exposure in plants is limited (Feng et al. 2013). Methylation of antimony has been shown to occur in both bacteria and fungi but whether or not Sb(V) can be reduced to Sb(III) and subsequently methylated within the plant is unknown (Feng et al. 2013).

4.1.2 Bioconcentration

Various studies have looked at the concentration of antimony in plants (Feng et al. 2013, Bech et al. 2012, Hammel et al. 2000) and despite considerably variability in antimony uptake between plant species, the relationship between plant and soil antimony concentration is relatively well described by a linear log-log regression model (Tschan et al. 2009a). The log-log regression is approximately equal to 1 indicating that on average plant antimony uptake is nearly proportional to the soluble antimony concentration in soil. Okkenhaug et al. (2011) noted a similar relationship extending several orders of magnitude across various plant and soil samples collected from the Xikuangshan mine in China. In their study it was concluded that the average plant bioconcentration value equated to 0.13 when compared to total soil antimony and 4.86 when compared to the water soluble antimony fractions. This is slightly lower than the values reported for Chinese cabbage which ranged from 0.28 (341 mg/kg in plant tissue and 1,200 mg/kg in dry soil) to 1.41 (2,263 mg/kg in plant tissue and 1,600 mg/kg in dry soil) by Baek et al. (2013).

4.1.3 Toxicity

Plants are capable of accumulating a relatively high level of antimony with varying degrees of supressed plant development (Baek et al. 2013, Shtangeeva et al. 2011). Baek et al. (2013) tested the effect on shoot, root and seedling growth for a variety of crop species (Chinese cabbage [*Brassica campestris*] wheat [*Triticum aestivum*] cucumber [*Cucumis sativus*] and Mung bean [*Phaseolus radiatus*]) and found that although plant growth was affected at increasing concentrations of antimony for each of the endpoints tested, the most sensitive endpoint was root growth. Reductions in plant biomass may be a result of the influence of antimony on the distribution of essential plant nutrients. Shtangeeva et al. (2011) found that plants grown in antimony-contaminated growth medium have a

significantly reduced tissue content of potassium and copper and that these reductions correlated well with a decrease in root and leaf biomass. Deficiencies of potassium and copper may cause disruptions in the photosynthetic process which leads to suppressed plant development (Shtangeeva et al. 2011).

The direct comparison of plant toxicity studies is difficult due to the variability between antimony compound toxicity as a result of potentially toxic counter ions and variable solubility. For example, soils spiked with an identical chloride concentration of calcium chloride ($CaCl_2$) to and antimony trichloride ($SbCl_3$) produced an identical toxicological effect on lettuce shoot yield (Oorts et al. 2008). This suggests that toxicity testing with antimony soluble salts is of little relevance for predicting risks associated with antinomy trioxide (Sb_2O_3) emission to soil (Oorts and Smolders, 2009). As a result, Sb_2O_3 is the preferred compound for toxicity testing but unlike antimony chlorides, which are more soluble, liberation of antimony oxides is far slower (EURAR 2008). Over time the slow process of transforming Sb_2O_3 into soluble and bioavailable Sb continues until equilibrium is reached, meaning that any toxicity study conducted prior to full equilibrium will underestimate the toxic potential of antimony in soil (EURAR 2008).

Oorts et al. (2008) studied the effect of Sb_2O_3 amended soil that had been aged for 31 weeks on barley (*Hordeum vulgare*) root elongation and lettuce (*Lactuca sativa* cv. Pontiac) shoot yield. Toxicity thresholds calculated in the study based on pore water antimony concentrations resulted in a no observed effects concentration (NOEC) of 9.7 mg/L, an EC_{10} of 13 mg/L and an EC_{50} of 39 mg/L for barley root elongation. Lettuce shoot yield was less affected with a calculated NOEC of 18 mg/L, an EC_{10} of 18 mg/L and an EC_{50} of 41 mg/L. These results are in close agreement with those of Tschan et al. (2009b) and Shtangeeva et al. (2011) who reported root inhibition in wheat (*Trticum aestivum*) at concentrations for 30 and 50 mg/L, respectively. Certain cultivars such as Indian mustard (*Brassica pratense*) appear to have relatively low sensitivity to antimony with no signs of distress at concentrations up to 300 mg/L while sunflower (*Helianthus annuus*) and clover (*Trifolium pratense*) are less resilient with reportable reduction in root growth at an antimony concentration of 100 mg/L (Tschan et al. 2009b).

The study results presented by Oorts et al. (2008) were used in development of the European Union predicted no effect concentration (PNEC) and in the Health Canada screening assessment of Sb_2O_3 (EURAR 2008, Environment Canada and Health Canada 2010). However, since not all of the Sb_2O_3 in amended test soils were fully transformed after 31 weeks the soil pore-water concentration was multiplied by the soil-water partition coefficient (K_d = 38 L/kg) as calculated by Smolders et al. (2007) to account for equilibrium achievement (EURAR 2008). Recalculation of the NOEC, EC_{20} and EC_{50} values for barely root elongation (NOEC = 370, EC_{20} = 508 and EC_{50} = 1,479 mg/kg) and lettuce shoot yield (NOEC = 693, EC_{20} = 1,109 and EC_{50} = 1,579 mg/kg) are similar to those reported by Baek et al. (2013).

A summary of the reviewed toxicity studies is included in Appendix C.

4.2 Terrestrial Invertebrates

4.2.1 Uptake, Metabolism, and Elimination

Dermal uptake of metals is considered the most important route for metal exposure in earthworms, and it is the soluble metal concentration that is the best descriptor of metal accumulation (Vijver et al. 2003). In the European Union assessment it was postulated that earthworms would therefore be under the same "toxic pressure" as plants as long the bioavailable antimony concentrations continue to increase as a result of the slow transformation of Sb_2O_3 in soil (EURAR 2008). In a study of increasing set-back distances downwind of an antimony smelter, Ainsworth et al. (1990) found antimony concentrations were highest in earthworms out of all the species studied and increased with increasing soil contamination. It was further noted that the concentration of antimony was greater in detritivores then either herbivorous or predatory invertebrates suggesting exposure to invertebrate consumers is relatively low.

Gal et al. (2007) compared total metal content in soil with earthworm (*Lubricus terrestris* and *Octolasion cyaneum*) tissue concentrations at an abandoned mining area in Scotland. They found that when the data was plotted on a log-log scale the regression line for antimony soil and antimony earthworm was less than 1 and indicated that antimony uptake is greatest when antimony levels in the soil were lower. It was speculated that this could mean elimination rates for antimony by earthworms increase as soil antimony levels increase (Gal et al. 2007).

4.2.2 Bioaccumulation

Gal et al. (2007) reported bioconcentration factors for the earthworms *L. terrestris* and *O. cyaneum* of 0.034 and 0.063, respectively. These values are considerably less than 1 and provide an indication that antimony does not tend to accumulate into invertebrates. This finding is in agreement with that of Ainsworth et al. (1990) who found that the mobility of antimony in food chains is low and that invertebrates contain lower concentrations of antimony then estimated by their diet.

4.2.3 Toxicity

USEPA provides an ecological soil screening level (Eco-SSL) for invertebrates of 78 mg/kg (USEPA 2005). The Eco-SSL is described by the USEPA as the concentration in soil protective of ecological receptors that commonly come into contact with and/or consume biota that live in or on the soil.

Kuperman et al. (2006) studied the effects of antimony sulphate ($Sb_2[SO_4]_3$) and antimony tartrate ($Sb_2[C_4H_4O_6]_3 \cdot 6H_2O$) on earthworm (*Eisenia fetida*), potworm (*Enchytraeus crypticus*), and collembolan (*Folsomia candida*) survival and reproduction. In their study they noted that the more soluble

antimony sulphate had a greater toxicological response in test organisms then the less soluble antimony tartrate. Bounded NOEC values for adult survival in earthworm, potworm and collembolan test species were 617, 384, and 100 mg/kg, respectively. However, juvenile production was the most sensitive endpoint when compared to adult survival with bounded NOEC values for all test species of 100 mg/kg and EC_{20} values for the earthworm, potworm and collembolan test species of 30, 194 and 81 mg/kg, respectively.

Lobella sokamensis was tested for its applicability as a new indicator species for soil quality testing of metal contamination by An et al. (2013). Adult median lethality concentrations (LC_{50}) were 10 to 20 times less than that of juveniles suggesting life stage is an important factor to consider when assessing invertebrate toxicity results. Adult and juvenile LC_{50} values were 4,702 and 447 mg/kg, respectively, which is similar to the LC_{50} reported by Moser (2007) for springtails (*Folsomia candida)* when adjusted for equilibrium achievement (>1,230 mg/kg). The study by Moser (2007) used the same soil as Smolders et al. (2007) and when adjusted for the low solubility of Sb_2O_3 the recalculated NOEC for springtail reproduction is 370 g/kg (EURAR 2008).

A summary of the reviewed toxicity studies is included in Appendix C.

4.3 Guideline Derivation

Canadian soil quality guidelines are derived for different land uses following the process outlined in CCME (2006) using different receptors and exposure scenarios for each land. Derivation of the direct soil contact guideline followed the CCME (2006) Weight of Evidence Method. This method uses a percentile of the effects data set, or combined effects and no effects data set, to estimate a concentration expected to cause no adverse biological effects. Insufficient data exists for the calculation of a guideline using IC_{25} and EC_{25} data. Instead, a combination of effects (LOEC, EC_{50}) and no effects (NOEC) data sets were selected.

Data for plants and invertebrates should, where possible, be evaluated separately, with the lower of the generated guidelines being taken as the soil quality guideline (CCME 2006). However, this requires that the data requirements for the method be met by each of the plant and invertebrate data sets which require at least ten data points be selected from at least three different studies. While the minimum data requirements were met for plants, only two studies were available from literature and considered suitable in their scope (provision of effects and no effects) and quality to derive a suitable guideline. For this reason, both the plants and invertebrate data sets were assessed separately, and combined.

Three ranked percentile graphs were generated (one plant [Figure 1.], one invertebrate [Figure 2.] and one combined [Figure 3.]) using the USEPA Causal Analysis/Diagnosis Decision Information System (CADDIS) Species Sensitivity Distribution (SSD) model (USEPA 2012).

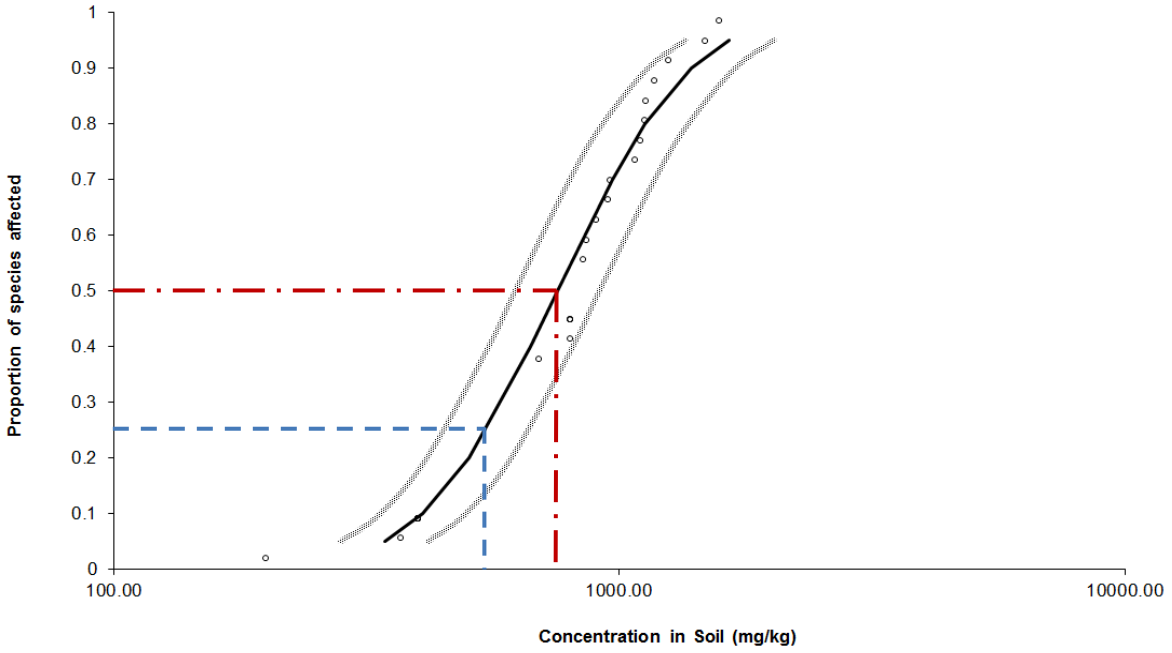

Fig 1. SSD plot showing the distribution of NOEC, LOEC and EC_{50} data for plants exposed to antimony in soil derived using the USEPA CADDIS SSD model with 95% confidence intervals (gray lines). Estimated species sensitivity distribution (ESSD) 25th percentile (TEC) and ESSD 50th percentile (ECL) are represented by the blue and red dashed lines, respectively. Estimates of $ESSD_{25}$ and $ESSD_{50}$ (intercept with x-axis) were 550 and 750 mg/kg, respectively.

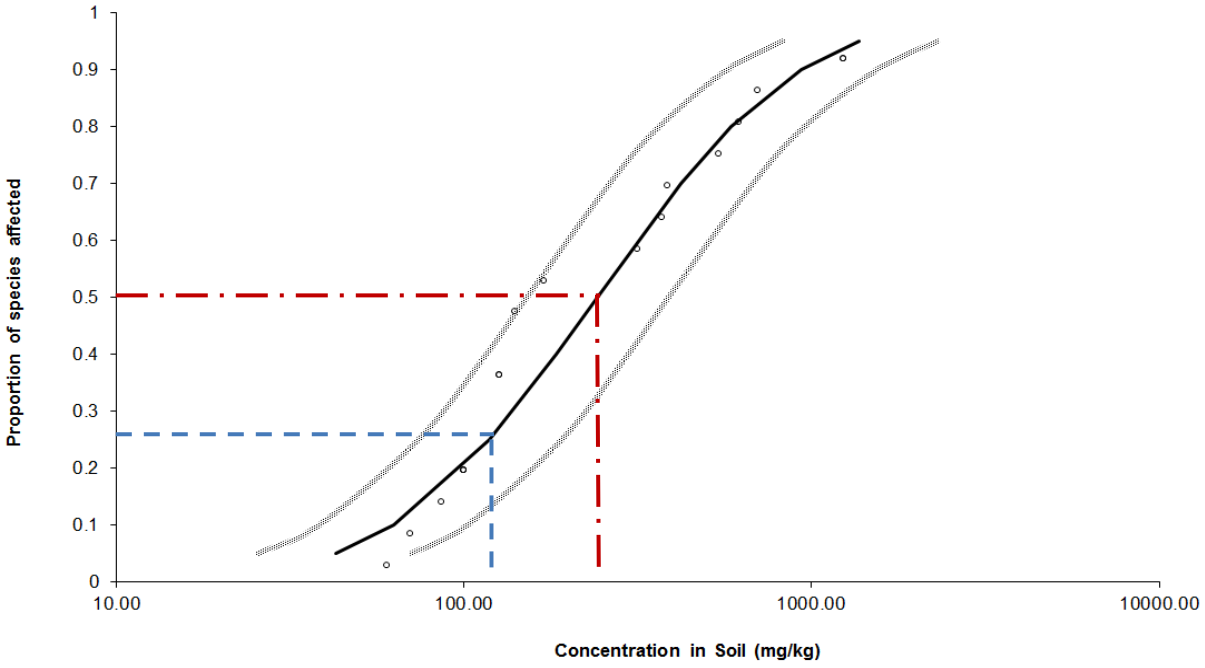

Fig 2. SSD plot showing the distribution of NOEC, LOEC and EC_{50} data for invertebrates exposed to antimony in soil derived using the USEPA CADDIS SSD model with 95% confidence intervals (gray lines). ESSD 25th percentile (TEC) and ESSD 50th percentile (ECL) are represented by the blue and red dashed lines, respectively. Estimates of $ESSD_{25}$ and $ESSD_{50}$ (intercept with x-axis) were 120 and 240 mg/kg, respectively.

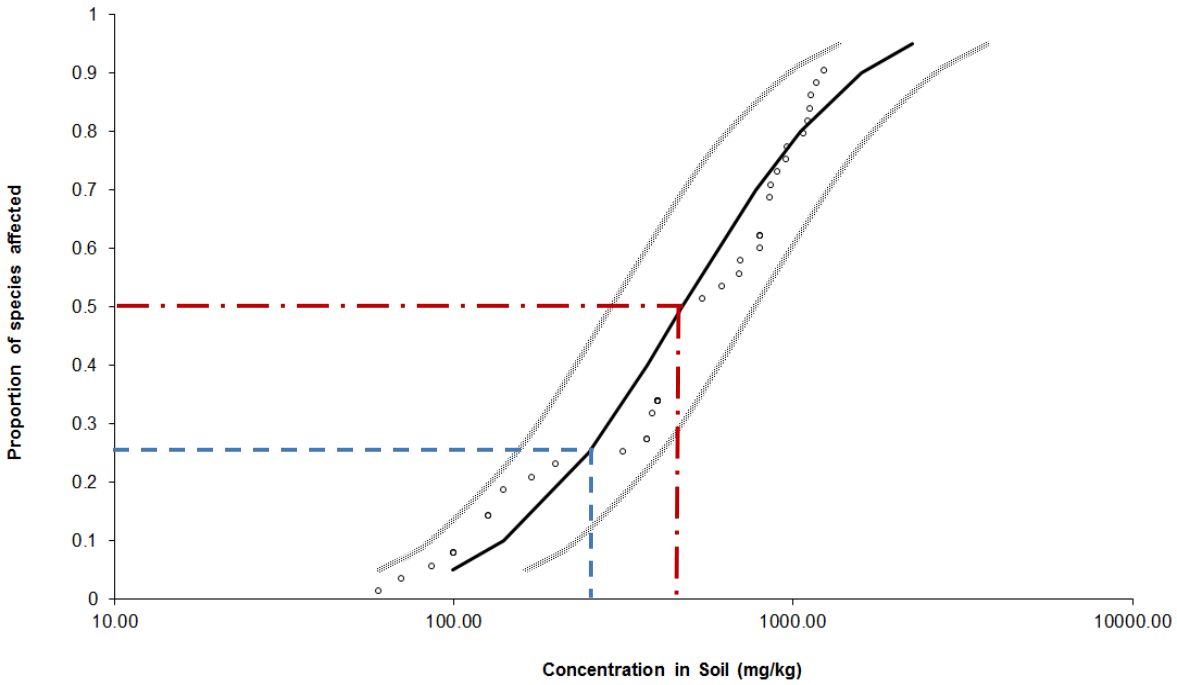

Fig 3. SSD plot showing the distribution of NOEC, LOEC and EC_{50} data for plants and invertebrates exposed to antimony in soil derived using the USEPA CADDIS SSD model with 95% confidence intervals (gray lines). ESSD 25th percentile (TEC) and ESSD 50th percentile (ECL) are represented by the blue and red dashed lines, respectively. Estimates of $ESSD_{25}$ and $ESSD_{50}$ (intercept with x-axis) were 250 and 470 mg/kg, respectively.

A review of the sensitivity distributions (Fig 1. through Fig 3.) suggests that invertebrate species are more sensitive to antimony than plants. Based on this observation a combination of both plant and invertebrate toxicity data in the calculation of an SSD graph is not suggested. The suggested guideline for direct soil contact in an agricultural, residential and parkland land use scenario is based on the TEC for invertebrates of 120 mg/kg and in a commercial and industrial land use scenario the guideline is based on the ECL for invertebrates of 240 mg/kg.

5.0 BEHAVIOUR AND EFFECTS IN HUMANS AND EXPERIMENTAL ANIMALS

5.1 Overview

The toxicology of antimony and its compounds to humans is primarily known from three sources: its medicinal use over centuries, studies of process workers, and its presence in modern city and domestic environments. Gross chronic exposures of miners and process workers to antimony compounds, most commonly antimony sulfide (SbS_3) or antimony oxide (Sb_2O_3), have occurred but the few studies available have poor details (Cooper and Harrison 2009). Exposures during antimony

processing have decreased significantly over the past 40 years due to automation of the process (Cooper and Harrison 2009). Antimony has been studied as a possible cause of sudden infant death syndrome (SIDS) but extensive investigations have not confirmed this effect (Cooper and Harrison 2009). A summary of the reviewed toxicity studies is included in Appendix C.

5.2 Pharmacokinetics

5.2.1 Absorption

A small amount of antimony may enter the blood a few hours after oral exposure (ATSDR 1992). The amount and the form of antimony ingested will affect how much antimony enters the blood. Eating or drinking very large doses of antimony results in vomiting and may prevent most of the antimony from being absorbed into the body. An oral absorption of 1% was proposed for antimony trioxide (EURAR 2008); however, high uncertainty was associated with the data used. The studies used were performed with different study protocols which did not meet current standards and indicated an average intestinal absorption of 3 to 8% (with a range of 0.15 to 40%). EURAR (2008) also proposed a 6.82% inhalation absorption rate based on data on physical particle size and density, a multiple-path particle deposition model and gastrointestinal tract absorption rate. A 0.26% dermal absorption rate was proposed by EURAR (2008) based on an *in vitro* study using human skin (Roper and Stupart 2006).

The default dermal relative absorption factor of 0.01 recommended for metals by Health Canada (2010) is generally consistent with the available data and applied for guideline derivation.

5.2.2 Distribution

A study in which rats were exposed to potassium antimony tartrate in drinking water for 13 weeks showed tissue antimony levels were dose related (Poon et al. 1998). The distribution of antimony in the organs was: red blood cells > > spleen, liver > kidney > brain, fat > serum. After a 4 week recovery period, antimony levels in the highest dose group decreased for all tissues except the spleen, which remained the same as before recovery.

The highest concentration of antimony was found in the spleen of male rats after subcutaneous injections of meglumine antimoniate (21-day treatment at 300 mg antimony/kg bw-day). The distribution of antimony in the organs was: spleen >> bone, thyroid, kidneys > liver, epididymis, lungs, adrenals > prostate > thymus, pancreas, heart, small intestines > skeletal muscle, testes, stomach > brain (Coelho et al. 2014).

A significant portion of the antimony that was excreted in the bile was deduced to have undergone enterohepatic circulation in rats (Bailly et al. 1991).

5.2.3 Metabolism

Antimony is a metal and is not metabolized, but it can interact with sulfhydryl groups and phosphate. No information is available on possible interconversion of antimony oxidation states within the body.

5.2.4 Elimination

After a single intravenous or intraperitoneal administration of antimony trichloride to rats, approximately 45 to 55% of the administered antimony was excreted in the urine or faeces within four days regardless of the dose (Bailly et al. 1991).

Male rats treated with a single intravenous injection of 75 mg/kg bw of meglumine antimoniate showed a two-compartment kinetic model with fast ($t_{1/2}$ = 0.6 h) and slow elimination phases (Coelho et al. 2014). A sharp fall in the blood concentration of antimony was seen within 6 to 12 hours after injection (to approximately 2 μg/g), followed by a slower decrease. At the terminal elimination phase (105 days after treatment), most of the antimony was found in whole blood rather than plasma (Coelho et al. 2014). This is consistent with the notion that red blood cells account for most of the antimony contained in the blood at the terminal elimination stage.

Antimony was no longer detectable in bile and gastric fluid 100 hours after ingestion of an unknown amount of antimony sulfide by an adult woman who had attempted to commit suicide (Bailly et al. 1991). However, antimony concentrations were above normal values in blood and urine one week after ingestion of the antimony sulfide.

Studies on workers employed in the production of antimony pentoxide and sodium antimoniate suggest an airborne concentration of 500 μg/m^3 of antimony led to an increase in urinary concentration of approximately 35 μg/g creatinine during an 8 hour shift (Bailly et al. 1991).

Biological elimination half-times were estimated to be 600 to 1,100 days for non-smokers and 1,700 to 3,700 days for smokers based on seven male workers who were accidentally exposed to radioactive antimony trioxide aerosols (Garg et al. 2003).

5.2.5 Physiologically-based Pharmacokinetic Models

No physiologically-based pharmacokinetic models for antimony were identified.

5.3 Essentiality

Antimony has long been used as a medicine and in cosmetics but it is not believed to play a nutritional role in humans and animals.

5.4 Acute Exposure

5.4.1 Acute Oral Toxicity

Health effects have been observed in humans and animals following oral exposure to a variety of antimony compounds, including potassium antimony tartrate (an organic form of antimony), antimony trichloride, antimony trioxide, and metallic antimony.

5.4.1.1 Human Acute Oral Toxicity

Shortly after drinking an average of 10 ounces of lemonade contaminated with potassium antimony tartrate (equivalent to 0.53 mg antimony/kg for a 70 kg man) in one day, workers began to vomit (Dunn 1928).

5.4.1.2 Animal Acute Oral Toxicity

Mortality was not observed in rats following a single oral exposure to 188 to 16,714 mg antimony/kg as inorganic antimony (Fleming 1982, Myers et al. 1978, Smyth and Carpenter 1948, Smyth and Thompson 1945) or to 7,000 mg antimony/kg of metallic antimony (Bradley and Frederick 1941). However, a lower single dose of organic antimony (300 mg antimony/kg dose as potassium antimony tartrate) resulted in death in rats (Bradley and Frederick 1941). The cause of death was reported to be myocardial failure. These animal studies suggest that organic antimony is more lethal than the inorganic compounds.

5.4.2 Acute Inhalation Toxicity

5.4.2.1 Human Acute Inhalation Toxicity

No data on acute toxicity of antimony inhalation were identified for humans.

5.4.2.2 Animal Acute Inhalation Toxicity

Many of the respiratory effects reported in animals with acute exposure to antimony are associated with the physiological response to dust accumulation in the lung (pneumoconiosis). The effects progress from pneumoconiosis and a proliferation of alveolar macrophages to fibrosis.

Parenchymatous degeneration was observed in the liver and kidneys of rabbits exposed to 19.94 mg antimony/m^3 as antimony trisulfide for 5 days (Brieger et al. 1954). Lung inflammation was also noted in these rabbits. Five days of exposure to 19.94 mg antimony/m^3 as antimony trisulfide also resulted in myocardial damage (EKG alterations) in rabbits.

Tubular dilation was observed in rats and guinea pigs exposed to stibine gas for 30 minutes at a concentration of 799 mg antimony/m^3 (Price et al. 1979). Guinea pigs and rats exposed to 1,395 mg

antimony/m^3 as stibine gas for 30 minutes died; however, none of the guinea pigs or rats exposed to 799 mg antimony/m^3 for 30 minutes died (Price et al. 1979). Pulmonary edema was a contributing factor to the death of guinea pigs and rats exposed to stibine.

5.4.3 Acute Dermal Toxicity

5.4.3.1 Human Acute Dermal Toxicity

In human volunteers, patch tests with fibre containing antimony trioxide did not induce skin reaction (Stevenson 1965, Haskell Laboratory for Toxicology and Industrial Medicine 1970, Motolese et al. 1993, Environment Canada and Health Canada 2010).

5.4.3.2 Animal Acute Dermal Toxicity

Death was observed in one out of six rabbits following a single application of 6,685 mg antimony/kg antimony oxide (Myers et al. 1978). The cause of death was not reported.

Two out of four rabbits died after 6 to 8 topical applications of antimony trioxide paste to shaven and abraded skin. The antimony trioxide was combined with a mixture formulated to resemble acidic sweat. The application area was not occluded; thus, there is a possibility of oral ingestion of the paste (Fleming 1982).

The application of 79 to 100 mg antimony as antimony oxide or antimony thioantimonate into the eyes of rabbits resulted in eye irritation (Horton et al. 1986, Wil Research Laboratories 1979). However, the application of antimony trioxide (34.5 to 83.6 mg antimony) did not result in eye irritation (Gross et al. 1955, Myers et al. 1978).

5.5 Subchronic Exposure

5.5.1 Subchronic Oral Toxicity

5.5.1.1 Human Subchronic Oral Toxicity

No data on subchronic oral toxicity of antimony to humans were identified.

5.5.1.2 Animal Subchronic Oral Toxicity

Prenatal and postnatal exposure, or postnatal exposure alone to 0.0748 mg antimony/kg/day as antimony trichloride for 30 days resulted in a decreased pressor response to 1-noradrenaline and a decreased hypotensive response to 1-isoprenaline and acetylcholine (Marmo et al. 1987).

Dogs administered 84 mg antimony/kg/day as antimony trioxide for 32 days experienced severe diarrhea. No gastrointestinal effects or gross abnormalities were noted in rats exposed to 501 mg antimony/kg/day or less as antimony trioxide for 20 days (Fleming 1982).

Small reductions in plasma alkaline phosphatase activity, increases in aspartate aminotransferase and increase in liver weight were noted in rats fed up to 1879 mg/kg/day antimony trioxide for 90 days; however, in the absence of any histological effects, these changes were considered to be incidental to treatment (Hext et al. 1999).

Severe weight loss, muscle weakness and difficulty in moving hind limbs were observed in dogs administered 6,644 mg antimony/kg/day as antimony trioxide for 32 days (Fleming 1982). Severe diarrhea was also observed in these dogs at 84 mg/kg/day.

A no observable adverse effects level (NOAEL) of 0.5 ppm (equivalent to an average intake of 0.06 mg/kg/day) antimony in drinking water was reported in a study where rats were exposed to potassium antimony tartrate in drinking water at concentrations of 0.5, 5, 50 and 500 ppm for 13 weeks (Poon et al. 1998). The NOAEL was based on histological and biochemical changes observed at antimony concentrations of 5.0 ppm and higher but not at 0.5 ppm.

5.5.2 Subchronic Inhalation Toxicity

5.5.2.1 Human Subchronic Inhalation Toxicity

No data on subchronic toxicity of antimony inhalation were identified for humans.

5.5.2.2 Animal Subchronic Inhalation Toxicity

The effective exposure levels resulting in cardiovascular effects were at least four times lower (2 to 4 mg antimony/m^3) in rats, rabbits, and dogs exposed to airborne antimony for 6 to 10 weeks compared to rabbits acutely exposed to antimony (Brieger et al. 1954).

Dogs exposed to 3.81 mg antimony/m^3 as antimony trisulfide for 7 weeks) did not exhibit changes in EKG readings but did show changes in EKG readings when exposed to 3.98 mg/m^3 of antimony trisulfide for 10 weeks (Brieger et al. 1954). The degenerative changes of the myocardium observed in rats, rabbits, and dogs exposed to antimony trisulfide consisted of hyperemia and swelling of myocardial fibers (Brieger et al. 1954).

A decreased number of rat offspring resulted after exposure to 209 mg antimony/m^3 as antimony trioxide prior to conception and throughout gestation for a duration of 63 to 78 days (Belyaeva 1967).

After rats were exposed to 209 mg antimony/m^3 as antimony trioxide for 63 days, 67% failed to conceive. Metaplasia in the uterus and disturbances in the ovum- maturing process were noted in animals that failed to conceive but not in animals that conceived (Belyaeva 1967).

Rats were exposed to antimony trioxide dust at exposure levels of 0, 0.25, 1.08, 4.92 and 23.46 mg/m^3 for 6 hours per day, 5 days per week for 13 weeks followed by a 27-week observation period (Newton et al. 1994). Corneal irregularities were seen after about 2 weeks of exposure. Reduced body weights were observed in males exposed to 23.46 mg/m^3 of antimony trioxide compared to the control group. Higher mean absolute and relative lung weights were observed in the 4.92 and 23.46 mg/m^3 groups. These observed effects persisted during the observation period.

5.6 Chronic Exposure

5.6.1 Chronic Oral Toxicity

5.6.1.1 Human Chronic Oral Toxicity

No data on chronic toxicity of antimony to humans were identified.

5.6.1.2 Animal Chronic Oral Toxicity

Chronic low dose administration of potassium antimony tartrate (5 ppm) resulted in decreased lifespan in rats (Schroeder et al. 1970). The dose was not precisely stated and has been reported by the ATSDR (1992) as 0.262 mg antimony/kg/day and by the USEPA (1987) as 0.35 mg/kg/day. In a similar study (Kanisawa and Schroeder 1969), groups of CD-1 mice were given potassium antimony tartrate in drinking water at 0 or 5 mg/L (5 ppm) for 540 days (18 months). Lifespans were significantly reduced in both males and females; however, the degree of antimony toxicity was less severe in mice than rats.

Rats exposed to 500 to 1,000 mg antimony/kg/day (metallic antimony) for 12 to 24 weeks showed decreased hematocrit and hemoglobin levels and decreased plasma protein levels (Hiraoka 1986, Sunagawa 1981).

Decreased red blood cell count was observed in rats exposed to 418 mg antimony/kg/day as antimony trioxide for 24 weeks (Sunagawa 1981).

Cloudy swelling of the hepatic cords has been observed in rats exposed to 418 mg antimony/kg/day as antimony trioxide or 500 mg antimony/kg/day as metallic antimony (Sunagawa 1981).

Rats exposed for a lifetime to low levels of potassium antimony tartrate (5 ppm) in drinking water had increased serum cholesterol and decreased non-fasting serum glucose levels (Schroeder *et al*. 1970).

5.6.2 Chronic Inhalation Toxicity

5.6.2.1 Human Chronic Inhalation Toxicity

Workers exposed to airborne antimony have experienced alterations in pulmonary function, including airway obstruction, bronchospasm, and hyperinflation (Cooper et al. 1968, Potkonjak and Pavlovich 1983). Other respiratory effects reported in workers include chronic bronchitis, chronic emphysema, inactive tuberculosis, pleural adhesions and irritation (Potkonjak and Pavlovich 1983).

Chronic occupational exposure to antimony trioxide and/or pentoxide dust (8.87 mg antimony/m^3 or greater) resulted in antimony pneumoconiosis (Cooper et al. 1968; Potkonjak and Pavlovich 1983; Renes 1953). The respiratory irritation reported in the workers diagnosed as having pneumoconiosis was characterized by chronic coughing, wheezing, and upper airway inflammation. However, the workers inhaled a variety of compounds including antimony pentoxide, arsenic oxide, iron oxide, hydrogen sulfide, and sodium hydroxide (Cooper et al. 1968, Potkonjak and Pavlovich 1983, Renes 1953).

Respiratory irritation was not noted in workers exposed to antimony trisulfide by inhalation for 8 months to 2 years; however, increased blood pressure (greater than 150/90) and altered EKG readings were observed in these workers (Brieger et al. 1954). Of the 75 workers examined, 37 showed changes in the EKG, mostly of the T-waves; these workers had also been exposed to phenol formaldehyde resin (Brieger et al. 1954).

A variety of gastrointestinal disorders have been noted in factory workers with repeated prolonged exposure to airborne antimony trichloride (Taylor 1966), antimony trisulfide (Brieger et al. 1954) or antimony oxide (Renes 1953). These disorders include abdominal pain, diarrhea, vomiting, and ulcers. A causal relationship to antimony exposure has not been definitely established because workers were exposed to a variety of other agents in addition to antimony that might cause or contribute to gastrointestinal effects (e.g., hydrogen chloride, sodium hydroxide). Additionally, both inhalation and oral exposure to antimony may have occurred in the workplace. Assuming that gastrointestinal effects are only related to antimony exposure, effective exposure levels may range from approximately 2 to 70 mg antimony/m^3.

Workers exposed to an antimony oxide concentration of 10.07 mg antimony/m^3 experienced nerve tenderness and a tingling sensation (Renes 1953). However, the factory workers were also exposed to arsenic, lead, copper, and possibly hydrogen sulfide and sodium hydroxide.

An increased incidence of spontaneous abortions, compared to a control group, were reported in women working at an antimony metallurgical plant. The women were exposed to a mixture of antimony trioxide, antimony pentasulfide, and metallic antimony (Belyaeva 1967). The level of airborne antimony and presence of other compounds was not known and details about the control group were not available.

Women exposed to airborne metallic antimony, antimony pentasulfide, and antimony trioxide in a metallurgical plant experienced disturbances in their menstrual cycle. No other details were provided (Belyaeva 1967).

The dermatitis associated with exposure to airborne antimony is characterized as epidermal cellular necrosis with associated acute inflammatory cellular reactions (Stevenson 1965). The dermatitis is seen more often during the summer months and in workers exposed to high temperatures (Potkonjak and Pavlovich 1983, Stevenson 1965). Stevenson (1965) concluded that the dermatitis resulted from the action of antimony trioxide on the dermis after dissolving in sweat and penetrating the sweat glands. Transferring the worker to a cooler environment often resulted in the rash clearing up within 3-14 days. It is unclear whether dermal and ocular effects are the result of inhalation exposure or dermal contact with airborne antimony.

5.6.2.2 Animal Chronic Inhalation Toxicity

A dose-related increase in the number of alveolar and/or intraalveolar macrophages was observed in rats exposed to 0.07 mg/m^3 antimony trioxide for 1 year or to 0.92 mg antimony/m^3 for 13 weeks. The proliferation of macrophages was still present for 12 months or 28 weeks, respectively, after exposure termination (Bio/dynamics 1985, 1990). Chronic interstitial inflammation was also observed in rats exposed to 0.07 mg antimony/m^3 for 1 year with a 1 year recovery.

Interstitial fibrosis and lipoid pneumonia were observed in rats exposed to antimony trisulfide or antimony trioxide for 1 year at exposure levels between 1.6 and 83.6 mg antimony/m^3 (Bio/dynamics 1990; Gross et al. 1952; Groth et al. 1986; Watt 1980, 1983; Wang et al. 1979). No respiratory effects were reported in pigs exposed to 4.2 mg antimony/m^3 as antimony trioxide for 1 year (Watt 1983).

Gastrointestinal symptoms were not reported in animals and no histopathological alterations were observed in rats exposed to antimony trioxide (4.2 mg antimony/m^3) for 1 year (Watt 1980).

Hematological effects were not observed in rats and pigs following long-term exposure to antimony aerosols ranging from 4 to 20 mg antimony/m^3 as antimony trioxide (Bio/dynamics 1985, 1990; Watt 1983), with the exception of small (but statistically significant) changes in the hemoglobin concentration of erythrocytes and erythrocyte volume in rats exposed to 4.01 mg antimony/m^3 as antimony trioxide (Bio/dynamics 1990).

Renal effects were not noted in rats exposed to 17.5 mg antimony/m^3 as antimony trisulfide or up to 36 mg antimony/m^3 as antimony trioxide for 1 year (Bio/dynamics 1990, Groth et al. 1986, Wang et al. 1979).

Rats and guinea pigs exposed to stibine gas (Price et al. 1979) and antimony trioxide (Bio/dynamics 1985) experienced eye irritation. Rats exposed to antimony trioxide for 1 year with a 1 year recovery period experienced cataracts and chromodacryorrhea (Bio/dynamics 1990). The chromodacryorrhea may have been secondary to dental abnormality, infectious disease, or xerosis.

Alopecia was noted in rats exposed to 0.92 mg antimony/m^3 or greater as antimony trioxide for 13 weeks (Bio/dynamics 1985). However, since high levels of antimony are measured in the skin or hair of animals following nose-only exposure to antimony aerosols, alopecia may not be due to dermal contact with airborne antimony (Felicetti et al. 1974a, 1974b).

Rats exposed to 0.07 mg antimony/m^3 antimony trioxide for 1 year with a 1 year recovery period experienced hyperplasia of the reticuloendothelial cells in the peribronchiolar lymph nodes (Bio/dynamic 1990).

A 12 month inhalation study was conducted on rats at exposure levels of 0, 0.06, 0.51 and 4.50 mg/m^3 of antimony trioxide, followed by a 12-month observation period (Newton et al. 1994). A dose related increase in cataracts was observed in the chronic study for males and females. Microscopic changes in the lungs, including interstitial inflammation, increased numbers of alveolar-intraalveolar macrophages, granulomatous inflammation/granulomas and fibrosis, were noted. Clearance of antimony trioxide from the lung was burden dependent in the chronic study. The increased lung burden in the high-exposure group decreased pulmonary clearance by 80%; however, antimony trioxide was not found to be carcinogenic under the exposure and study conditions used.

5.7 Carcinogenicity and Genotoxicity

5.7.1 Human Data

The limited human studies of antimony carcinogenicity have been inconclusive (IARC, 1989).

5.7.2 Animal Data

Lung tumors were observed in rats exposed to 4.2 or 36 mg antimony/m^3 as antimony trioxide (Groth et al. 1986; Watt 1980, 1983; Wang et al. 1979) or 17.48 mg antimony/m^3 as antimony trisulfide for 1 year (Groth et al. 1986, Wang et al. 1979). OECD (2008) concluded that "the most likely mechanism for carcinogenicity appears to be impaired lung clearance and particle overload followed by an inflammatory response, fibrosis and tumours." Based on the available evidence, antimony trioxide can be regarded as a threshold carcinogen (Environment Canada and Health Canada 2010).

No change in the incidence of cancer was observed in rats (Schroeder 1970) or mice (Kanisawa and Schroeder 1969, Schroeder 1968) fed 0.262 or 0.35 mg antimony/kg/day, respectively, as potassium antimony tartrate for a lifetime.

5.7.3 Genotoxicity/Mutagenicity

Incubation of Chinese hamster ovary cells for 2 hours with antimony trichloride (0.2 mM) or antimony potassium tartrate (0.4 mM) inhibited the repair of radiation-induced DNA double strand breaks (Takahashi et al. 2002). Antimony trichloride concentrations of 50 μM or greater induced micronuclei formation in cultured Chinese hamster ovary cells, human bronchial epithelial cells and human fibroblasts. Delayed DNA fragmentation and/or apoptosis were observed in these cells after incubation with antimony trichloride (Huang et al. 1998).

Antimony trioxide was not genotoxic when examined in a range of *in vitro* and *in vivo* genotoxicity assays (Elliot et al. 1998).

5.7.4 Carcinogenic Classification

Health Canada and USEPA have not classified antimony with respect to carcinogenicity. IARC (1989) has classified antimony trioxide as possibly carcinogenic to humans (Group 2B). The classification is based on evidence of carcinogenicity from animal inhalation studies. Antimony trisulfide was determined to be not classifiable as to its carcinogenicity to humans (Group 3) (IARC 1989). The available data do not conclusively suggest antimony is carcinogenic to humans and indicate that if it is carcinogenic to humans it likely acts as a threshold carcinogen.

5.8 Toxicological Limits

5.8.1 Chronic Oral

The Health Canada, World Health Organization (WHO) and the Dutch National Institute for Public Health and the Environment (RIVM) chronic oral toxicity reference values (TRVs) discussed below are all based on the same critical study (Poon et al. 1998) but differ based on the interpretation of the critical effect.

5.8.1.1 Health Canada

Health Canada (1999) established a tolerable daily intake (TDI) of 0.0002 mg/kg/day (0.2 μg/kg-bw/d) based on a 13-week rat water ingestion study (Poon et al. 1998), where a NOAEL of 0.5 mg/L in drinking water, equivalent to 0.06 mg/kg/day was established. An uncertainty factor of 300 (10 for intraspecies variation, 10 for interspecies variation and 3 for use of a short-term study) was applied.

5.8.1.2 WHO

WHO (2003) used the same Poon et al. (1998) study as Health Canada; however, they incorporated a review by Lynch et al. (1999) that concluded the critical effects suggested by Poon et al. (1998) were expected to be reversible/adaptive and not consistent with other studies, and that therefore a NOAEL of 6.0 mg/kg/day should be applied based on decreased body weight gain, food intake and water intake. An uncertainty factor of 1,000 (10 for intraspecies variation, 10 for interspecies variation and 10 for a subchronic study) was then applied to calculate a TDI of 0.006 mg/kg/day.

5.8.1.3 RIVM

RIVM (2009) used the same study and rationale as the WHO (2003) to derive an identical TDI of 0.006 mg/kg/day.

5.8.1.4 USEPA

USEPA (1987) published an oral RfD for antimony of 0.0004 mg/kg/day (0.4 µg/kg/day) based on a rat study (Schroeder et al 1970) where rats were exposed to 5 ppm potassium antimony tartrate in water. The experimental group had on average shorter lifespans, decreased blood glucose levels (males only) and altered cholesterol levels compared to controls. The estimated exposure dose (0.35 mg/kg/day) was adjusted by an uncertainty factor of 1,000 (10 for interspecies conversion, 10 for sensitive individuals, and 10 for use of a LOAEL). A notation in the assessment indicates that a 2002 literature review identified one or more significant new studies.

5.8.2 Acute Inhalation

5.8.2.1 Ontario Ministry of the Environment (OMOE)

OMOE (2012) has specified a ½ hour standard for antimony of 75 µg/m^3 and a 24-hour standard of 25 µg/m^3; these values are identified as being health-based but the detailed basis is not provided.

5.8.2.2 Texas Commission on Environmental Quality (TCEQ)

TCEQ (2014) specified an interim short-term effects screening level (ESL) of 5 µg/m3 (in PM_{10}) for antimony. While it is identified as being health-based, the basis is not provided.

5.8.3 Chronic Inhalation

5.8.3.1 TCEQ

TCEQ (2014) specified an interim long-term ESL of 0.5 µg/m^3 (in PM_{10}) for antimony. While it is identified as being health-based, the basis is not provided.

5.8.3.2 USEPA

USEPA (1987) did not publish inhalation toxicity limits for antimony, but references historical studies suggesting an inhalation NOEL for myocardial damage of approximately 0.5 mg/m^3 in historical occupational studies. USEPA (1995) did establish a reference concentration (RfC) for antimony trioxide of 0.0002 mg/m^3 based on respiratory effects in rats observed by Newton *et al.* (1994) in a 1-year inhalation study. A benchmark concentration (BMC_{10}) of 0.87 mg/m^3, or 0.074 mg/m^3 as a human-equivalent concentration, was calculated. An uncertainty factor of 10 for sensitive humans was applied, along with a factor of 3 for interspecies extrapolation, 3 for less-than lifetime exposure and 3 for database inadequacies (lack of reproductive and developmental studies).

5.8.3.3 ATSDR

ATSDR (1992) evaluated the toxicity of antimony but concluded that data were inadequate to derive minimal risk levels (MRLs).

5.9 Toxicity Reference Values Selected for SQG Development

The WHO TDI of 0.006 mg/kg/day is recommended, since it is newer than the Health Canada (1999) value, which was not adopted into more recent Health Canada documents (e.g. Health Canada, 2010) and reflects an updated evaluation using the same data and methodologies similar to Health Canada.

6.0 DERIVATION OF ENVIRONMENTAL AND HUMAN HEALTH SOIL QUALITY GUIDELINES

6.1 Environmental Soil Quality Guidelines

Canadian Soil Quality Guidelines are derived for four different land uses: agricultural, residential/parkland, commercial and industrial. Alberta also adds guidelines for the natural area land use.

All data for use in the following derivations have been screened for ecological relevance and are presented in the preceding sections. For the soil contact pathway, data were not selected from soils that are outside the typical conditions found in Canada (e.g. pH <4), or from studies that did not use soil or artificial soil, did not record soil texture and pH, did not use appropriate statistical analyses, did not use controls, or involved sewage sludge or mixtures of toxicants.

6.1.1 Soil Quality Guidelines for Soil Contact

Soil quality guidelines for soil contact (SQG_{SC}) are based on toxicological data for plants and soil invertebrates. The preferred approach is to use a weight of evidence method using EC_{25} or similar values; if the data do not meet the requirements for this method, then additional approaches using

other data points can be applied, such as effects/ no effects data, lowest observed effects concentrations, and median effects.

The data requirements for the preferred weight of evidence approach include:

- At least 10 discrete data points from at least 3 studies.
- A minimum of 2 soil invertebrate and 2 crop/plant data points.

The plant and invertebrate data available for antimony did not meet the minimum requirements for this approach and therefore the second approach of using an effects/no-effects concentration distribution was applied. The data set did meet the minimum requirements for this approach

In some cases it is prudent to combine data points to eliminate redundancy by calculating the geometric mean of individual data points (CCME 2006). For example, data points representing the same type of response in the same species under highly similar exposure conditions, or different responses that are known to be directly, causally connected should be combined. Consideration can also be given to combining data for different soil types – in general variations in toxicity due to exposure conditions such as soil type are considered to be a valid part of the sensitivity distribution, but in some cases it may be appropriate to combine data points to prevent a significant bias of the sensitivity distribution to a single species.

Plant and invertebrate toxicity data from Baek et al. (2013), Shtangeeva et al. (2011), Tschan et al. (2009b), Oorts et al. (2008), An et al. (2012), Mosser (2007), and Kuperman et al. (2006) meet minimum data requirements. Multiple response levels are available; EC_{50} values were used from Baek et al. (2013), Oorts et al. (2008), Mosser (2007), and Kuperman et al. (2006), LOEC values were used from Shtangeeva et al. (2011) and Tschan et al. (2009b). These studies included a total of 12 different plant species with 28 data points, and 5 invertebrate species with 18 data points.

As specified by CCME (2006), the selected data were ranked and rank percentiles determined for each data point. The protocol allows plant and invertebrate data to be either combined or treated separately; both approaches have been examined herein. The resulting species sensitivity distributions were found to be similar, and therefore the combined data set was used for greater statistical power. The resulting species sensitivity distributions along with EC_{25} and EC_{50} values were shown previously in Figures 1 through 3.

6.1.1.1 Guidelines for the Agricultural, Residential/Parkland and Natural Area Land Uses

The soil contact guidelines are calculated from the 25th percentile of the estimated species sensitivity distribution ($ESSD_{25}$). Values for invertebrates were used as they were considerably more sensitive than the tested plant species. The $ESSD_{25}$ has been calculated at 120 mg/kg.

The threshold effects concentration is then calculated as:

$$TEC = ESSD_{25}/UF$$

Where:

TEC	=	threshold effects concentration (mg/kg)
$ESSD_{25}$	=	estimated species distribution – 25th percentile (mg/kg)
UF	=	uncertainty factor (if needed)

An uncertainty factor is only applied if the data are borderline, such as if only the minimum number of studies is available, fewer than three taxonomic groups are represented, greater than 50% of either the plant or invertebrate toxicity data are in the lower 25th percentile of the combined distribution, short-term toxicity studies were used, or more than 50% of the data reflect low bioavailability conditions (CCME 2006). None of these conditions apply, and therefore an uncertainty factor is not considered to be warranted.

The SQG_{SC} for the agricultural, residential/parkland and natural area land uses is set at the TEC, or 120 mg/kg.

6.1.1.2 Guidelines for the Commercial and Industrial Land Uses

The soil contact guidelines are calculated from the 50th percentile of the estimated species sensitivity distribution ($ESSD_{50}$). Values for invertebrates were used as they were considerably more sensitive than the tested plant species. The $ESSD_{50}$ has been calculated at 240 mg/kg.

The effects concentration - low is then calculated as:

$$ECL = ESSD_{50}$$

Where:

ECL	=	threshold effects concentration (mg/kg)
$ESSD_{50}$	=	estimated species distribution – 50th percentile (mg/kg)

An uncertainty factor is not normally applied to the ECL. The SQG_{SC} for the commercial and industrial land uses is set at the ECL, or 240 mg/kg.

6.1.1.3 Confidence Ranking for the Soil Contact Guideline

CCME (2006) uses a ranking system to indicate the confidence in the guideline, based on the method used and whether there were enough data to evaluate plants and invertebrates separately.

For antimony, the effects/no-effects distribution approach was used. There was sufficient data to evaluate plants and invertebrates individually or when the data were combined; therefore, a confidence ranking of 'C' is assigned.

6.1.2 Nutrient and Energy Cycling

The nutrient and energy cycling guideline (SQG_{NEC}) is used to evaluate biological processes in the soil that are expected to affect the overall soil ecosystem performance. Professional judgment is applied as to whether this guideline should be used in the overall soil quality guideline calculation (CCME 2006). The preferred data for the calculation of guidelines for this pathway are nitrification and nitrogen-fixation data. In the absence of sufficient nitrification and nitrogen fixation values, the data set can be supplemented with decomposition, respiration and nitrogen mineralization data.

Available studies related to this exposure pathway either did not meet reliability requirements or showed no effect at high (10,000 mg/kg) doses (Smolders et al. 2007). While the data are not considered adequate for guideline calculation, the available information indicates that guidelines for this pathway would be substantially higher than for ecological soil contact and SQG_{NEC} has not been calculated as part of the overall soil quality guideline.

6.1.3 Soil Quality Guidelines for Soil and Food Ingestion

Soil and food ingestion guidelines (SQG_{I}) are calculated for the agricultural and natural area land uses to protect domestic animals and wildlife.

The CCME (2006) process normally evaluates grazing herbivores on agricultural lands, although other species can be considered if identified as being particularly sensitive to the contamination. The first step is to identify the species most at threat based on oral toxicological data for grazing/foraging species. The minimum data requirements include at least two oral mammalian studies, only one of which can be a laboratory rodent study and at least one of which should reflect a grazing herbivore, and one oral avian study.

6.1.3.1 Development of the Daily Threshold Effect Dose (DTED)

The data requirements specified by CCME (2006) for this pathway are not met with current data. In the absence of toxicological studies specific to livestock and wildlife species, the rat NOAEL of 6

mg/kg-bw/d used to derive the human TDI (Section 5.8.1) is applied as a provisional wildlife and livestock DTED.

6.1.3.2 Receptor Parameters

A cattle body weight of 550 kg and soil ingestion rate of 0.747 kg/d have previously been used for soil quality guideline derivation (CCME 2008), (ESRD 2010). A typical cattle food ingestion rate is 2.5% of body weight per day or 13.75 kg/d dry weight. This value is slightly higher than what would be calculated using the allometric equation recommended by CCME (2006). These values were used to calculate the agricultural guideline herein.

For Alberta natural area land use calculations, a vole with a body weight of 0.017 kg and soil ingestion rate of 0.000058 kg/d is used (ESRD 2010). A food ingestion rate of 0.00241 kg/d is calculated using the CCME (2006) allometric equation. These values were used to calculate the natural area guideline herein.

6.1.3.3 Bioavailability

There is no information on the relative bioavailability of antimony in natural food/soil compared to bioavailability in the critical toxicity studies. A bioavailability factor of 1 is therefore assumed.

6.1.3.4 Bioconcentration Factors

In general the bioconcentration of antimony in plants appears to be low. Many of the available studies where plant uptake do not contain sufficient information to calculate a meaningful bioconcentration factor – often available instead of total antimony concentrations are reported, or antimony is provided in solution. USEPA have previously endorsed a literature-derived bioconcentration factor of 0.03 (USEPA 2013), which has also been previously proposed by Oak Ridge National Laboratory (Baes et al. 1984); since this is the only BCF that was identified it is adopted herein.

6.1.3.5 Calculation of the Soil Quality Guideline for Ingestion

The guideline for soil and food ingestion can be calculated for a primary consumer using the following equation (CCME 2006):

$$SQG_I = \frac{0.75 \times DTED \times BW}{(SIR \times BF) + (FIR \times BCF)}$$

Where,

SQGI = soil quality guideline for food and soil ingestion (mg/kg)

DTED = daily threshold effects dose (mg/kg-bw/d)

BW	=	body weight (kg)
SIR	=	soil ingestion rate (kg/d)
FIR	=	food ingestion rate (kg/d dry weight)
BF	=	bioavailability factor
BCF	=	bioconcentration factor (mg/kg plant per mg/kg soil)

The resulting SQG_I for the agricultural land use is 2,100 mg/kg. For the Alberta natural area land use, the wildlife SQGI is 590 mg/kg. This guideline is considered provisional due to the limitations of the toxicity dataset.

6.1.4 Guidelines for the Protection of Groundwater

No guidelines for protection of groundwater (freshwater life, livestock water and irrigation water) were derived for antimony due to restrictions on the mathematical model when applied to metals (CCME 2006).

6.1.5 Off-site Migration Guidelines for Commercial and Industrial Land Uses

The guideline for offsite migration (SQG_{OM-E}) is calculated for the commercial and industrial land uses to protect against transfer of contaminated soil to a more sensitive nearby property through processes such as wind and water erosion. CCME (2006) derived the following equation to evaluate this pathway, based on the Universal Soil Loss Equation and Wind Erosion Equation:

$$SQG_{OM-E} = 14.3 \times SQG_A - 13.3 \times BSC$$

Where,

SQG_{OM-E}	=	environmental soil quality guideline for off-site migration (mg/kg)
SQG_A	=	soil quality guideline for agricultural land use (120 mg/kg)
BSC	=	background concentration of chemical in receiving soil (0.6 mg/kg)

The background antimony concentration in soil of 0.6 mg/kg was selected based on the Reimann et al. (2010) Canada-wide study. The resulting SQGOM-E for commercial and industrial land uses is 1700 mg/kg.

6.1.6 Summary and Selection of the SQG_E

The lowest ecological guideline for all land uses is the SQG_{SC} (120 mg/kg for natural areas, agricultural and residential/parkland, and 240 mg/kg for commercial and industrial land uses). All mandatory pathways have been evaluated and therefore a SQG_E can be established.

6.1.7 Data Gaps in the Derivation of Environmental Soil Quality Guidelines

The data set for the direct soil contact pathway is considered to be robust, with several different plant and invertebrate species evaluated in multiple soil types using standard test protocols.

The nutrient and energy cycling guideline could not be calculated reliably, but the information suggests that any calculated guidelines would likely be higher than the direct contact guideline.

No toxicity data specific to wildlife and livestock species were identified, and therefore the wildlife and livestock guidelines were based on laboratory rodent studies.

6.2 Human Health Soil Quality Guidelines

6.2.1 Estimated Daily Intakes

A background antimony concentration in soil of 0.6 mg/kg was selected based on the Reimann et al. (2010) Canada-wide study. Concentrations of antimony in water and the atmosphere were consistently low, and in most studies in areas unaffected by anthropogenic impacts median concentrations were below laboratory detection limits. Based on data summarized in Section 2.3, a conservative estimate of 0.01 μg/m^3 was applied for a background concentration in air, and 0.1 μg/L as a background concentration in drinking water. No Canadian data on concentrations in food were identified; antimony was included in the UK Total Diet Study (Ysart et al., 1999) and values from that study, generally below 0.001 mg/kg, were applied for EDI calculations.

Receptor characteristics used for the Estimated Daily Intake (EDI) calculations are summarized in Table 2 below. The calculated EDI for antimony is summarized in Table 3, and is based entirely on soil ingestion.

Table 2 Receptor Characteristics for EDI Calculation[a]						
Age Group / Characteristic	0-6 months	0.5 – 4 yr	5 – 11 yr	12 – 19 yr	20 – 64 yr	65+ yr
Inhalation Rate (m3/day)	2.1	9.3	14.5	15.8	16.2	14.3
Water Ingestion Rate (L/day)	0.2	0.2	0.4	0.4	0.4	0.4
Soil & Dust Ingestion Rate (mg/day)	30	100	65	30	30	30
Body Weight (kg)	7.5	15.5	31	59.4	70.9	72
Time Spent Outdoors (day/day)	0.125	0.125	0.125	0.125	0.125	0.125
Time Spent Indoors (day/day)	0.875	0.875	0.875	0.875	0.875	0.875

a – values provided by Health Canada for EDI calculation based on EHD (1998)

Table 3 Estimated Intake (µg/kg-bw/d) of Antimony						
Age Group / Route of Exposure	0-6 months	0.5 – 4 yr	5 – 11 yr	12 – 19 yr	20 – 64 yr	65+ yr
Ambient Air	3.5E-04	7.5E-04	5.9E-04	3.3E-04	2.9E-04	2.5E-04
Indoor Air	2.5E-03	5.3E-03	4.1E-03	2.3E-03	2.0E-03	1.7E-04
Drinking Water	0.003	0.001	0.001	0.0007	0.0006	0.0006
Food and Beverages	0.16	0.12	0.08	0.05	0.04	0.03
Soil	2.4E-03	3.9E-03	1.3E-03	3.0E-04	2.5E-04	2.4E-04
Total Intake	0.17	0.13	0.09	0.05	0.04	0.03

6.2.2 Soil Guideline for Direct Contact with Soil

Direct human contact with soil is calculated for the agricultural, residential/parkland, commercial and industrial land uses. Since antimony is treated as a threshold substance a toddler is considered to be the most sensitive human receptor, except for the industrial land use where only adults are assumed to spend significant amounts of time (CCME 2006).

The direct contact guideline includes three separate exposure pathways: incidental soil ingestion, dermal contact with soil, and soil particulate inhalation. While there is an inhalation TRV available, due to the small contribution of inhalation exposures separation of inhalation as a separate exposure route does not affect the guideline calculations and the direct exposure routes were combined:

$$SQG_{DH} = \frac{(TDI - EDI) \times SAF \times BW}{[(AF_G \times SIR) + (AF_S \times SR) + (AF_L \times IR_S) \times ET_2] \times ET_1} + BSC$$

Where:

SQG_{DH}	=	direct contact human health soil quality guideline (mg/kg)
TDI	=	tolerable daily intake (mg/kg-bw/d)
EDI	=	estimated daily intake (mg/kg-bw/d)
SAF	=	soil allocation factor (dimensionless)
BW	=	body weight (kg)
BSC	=	background soil concentration (mg/kg)
AF_G	=	relative absorption factor for the gut (dimensionless)
AF_L	=	relative absorption factor for the lung (dimensionless)
AF_S	=	relative absorption factor for the skin (dimensionless)
SIR	=	soil ingestion rate (kg/d)
IRS	=	soil inhalation rate (kg/d)
SR	=	soil dermal contact rate (kg/d)
	=	(hand surface area x hand soil adherence factor) + (arm/leg surface area x arm/leg adherence factor) x events/day
ET_1	=	exposure term 1 (dimensionless) – days per week/7 x weeks per year/52
ET_2	=	exposure term 2 (dimensionless) – hours per day/24

Input values are summarized in Table 4 below for each land use, along with the calculated guidelines.

Table 4 SQG_{DH} Input Values for Each Land Use

Parameter	Agricultural	Residential / Parkland	Commercial	Industrial
TDI (mg/kg-bw/d)	0.006	0.006	0.006	0.006
EDI (mg/kg-bw/d)	1.3E-04	1.3E-04	1.3E-04	4.0E-05
SAF	0.2	0.2	0.2	0.2
BW (kg)	16.5	16.5	16.5	70.7
BSC (mg/kg)	0.6	0.6	0.6	0.6
AF_G	1	1	1	1
AF_L	1	1	1	1

Table 4 SQG$_{DH}$ Input Values for Each Land Use				
Parameter	**Agricultural**	**Residential / Parkland**	**Commercial**	**Industrial**
AF_S	0.01	0.01	0.01	0.01
SIR (kg/d)	0.00008	0.00008	0.00008	0.00002
IRS (kg/d)	$7.07x10^{-9}$	$7.07x10^{-9}$	$7.07x10^{-9}$	$1.20x10^{-8}$
SR (kg/d)	0.000069	0.000069	0.000069	0.00011
ET_1	1	1	0.66	0.66
ET_2	1	1	0.42	0.42
SQG$_{DH}$	**240**	**240**	**870**	**14,000**

6.2.3 Guideline for the Protection of Potable Groundwater

No guideline for protection of potable groundwater was derived for antimony due to restrictions on the mathematical model when applied to metals (CCME 2006).

6.2.4 Guideline for the Protection of Indoor Air Quality

Antimony is not a volatile chemical and therefore a guideline for the protection of indoor air quality is not required.

6.2.5 Produce, Meat and Milk Ingestion Check

The produce, meat and milk ingestion check is not normally calculated for inorganics (CCME 2006). Bioaccumulation of inorganics in food is highly affected by soil chemistry as well as specific plant species and can vary significantly between sites.

This pathway should be evaluated on a site-specific basis if food crops are grown in antimony-contaminated soils.

6.2.6 Off-site Migration Guidelines for Commercial and Industrial Land Uses

The guideline for offsite migration (SQG$_{OM-HH}$) is calculated for the commercial and industrial land uses to protect against transfer of contaminated soil to a more sensitive nearby property through processes such as wind and water erosion. CCME (2006) derived the following equation to evaluate this pathway, based on the Universal Soil Loss Equation and Wind Erosion Equation:

$$SQG_{OM-HH} = 14.3 \times SQG_A - 13.3 \times BSC$$

Where,

SQG_{OM-HH}	=	human health soil quality guideline for off-site migration (mg/kg)
SQG_A	=	soil quality guideline for agricultural land use (240 mg/kg)
BSC	=	background concentration of chemical in receiving soil (0.6 mg/kg)

The resulting SQG_{OM-HH} for commercial and industrial land uses is 3,400 mg/kg.

6.2.7 Discussion of Uncertainties Associated with the Human Health Soil Quality Guidelines

The human health soil quality guidelines were derived using a TDI derived by WHO. This value was were based on a subchronic rat water ingestion study. There is some uncertainty associated with the extrapolation from subchronic to chronic exposures and rats to humans, although this is addressed through the use of uncertainty factors.

Further conservatism is introduced by the assumption that gastrointestinal absorption from soil is equivalent to gastrointestinal absorption in the critical toxicity study (i.e. from water). It is likely that absorption is lower from soil than water, but this has not been quantified.

Some components of the EDI are relatively uncertain due to the limited data on antimony concentrations in the environment, primarily due to measured concentrations being consistently lower than laboratory detection limits. The calculated EDI is expected to be conservative based on the use of upper-percentile values for some concentrations and detection limits where antimony was not detected.

Soil guidelines have not been calculated based on uptake by plants and subsequent ingestion by humans. This pathway is not normally quantitatively evaluated for inorganics (CCME 2006), but can be a significant source of exposure if crops are grown directly in contaminated soils. If food crops are grown directly in soils with antimony contamination in the rooting zone, then this pathway may need to be evaluated on a site-specific basis.

7.0 DERIVATION OF THE FINAL SOIL QUALITY GUIDELINE

The final soil quality guideline (SQG_F) considers both environmental and human health.

7.1 Considerations Other than Toxicity

If there is evidence that a contaminant may cause significant adverse effects other than toxicity to human and ecological receptors, this evidence may be used to derive a soil quality guideline for management considerations (SQG_M). This may include aesthetic concerns, damage to infrastructure, explosive hazards, or mobile free-phase liquid formation.

There is no indication that significant adverse effects other than toxicity are of concern for antimony, and no SQG_M is proposed.

7.2 Evaluation Against Plant Nutritional Requirement, Geochemical Background and Practical Quantitation Limits

Antimony is not an essential nutrient for plants or animal species, and the proposed guidelines are above geochemical background at locations within Canada unless influenced by anthropogenic sources. Therefore no adjustment of the guidelines for these factors is necessary.

7.3 Final Soil Quality Guidelines

The guidelines are summarized in Table 5 below. Since both human and environmental guidelines were developed, these guidelines are considered to supersede the previous (1991) guidelines.

Table 5 Soil Quality Guidelines for Antimony

Pathway	Natural Area	Agricultural	Residential/ Parkland	Commercial	Industrial
Guideline (SQG_F)	**120**	**120**	**120**	**240**	**240**
Human health guidelines					
SQG_{HH}	***NA***	***240***	***240***	***870***	***3,400***
Direct Contact (SQG_{DH})	NA	240	240	870	14,000
Protection of Indoor Air Quality (SQG_{IAQ})	NA	NA	NA	NA	NA
Protection of Potable Water (SQG_{PW})	NA	NA	NA	NA	NA
Off-site migration check (SQG_{OM-HH})	NA	NA	NA	3,400	3,400
Produce, meat & milk check (SQG_{FI})	NC	NC	NC	NC	NC
Environmental health guidelines					
SQG_E	***120***	***120***	***120***	***240***	***240***
Soil contact (SQG_{SC})	120	120	120	240	240
Soil and food ingestion (SQG_I)	590[a]	2100[a]	NA	NA	NA
Protection of freshwater life (SQG_{FL})	NA	NA	NA	NA	NA
Livestock watering (SQG_{LW})	NA	NA	NA	NA	NA
Irrigation water (SQG_{IR})	NA	NA	NA	NA	NA
Nutrient and energy cycling (SQG_{NEC})	NC	NC	NC	NC	NC
Off-site migration check (SQG_{OM-E})	NA	NA	NA	1,700	1,700
SQG_M (non-toxicity considerations)	NA	NA	NA	NA	NA
Interim soil quality criterion (CCME 1991)	20	20	20	40	40

a – provisional value

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