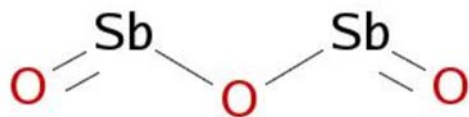United States
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EPA

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TSCA Work Plan Chemical Risk Assessment

Antimony Trioxide

CASRN: 1309-64-4

August 2014

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This report was developed by the United States Environmental Protection Agency (US EPA), Office of Chemical Safety and Pollution Prevention (OCSPP), Office of Pollution Prevention and Toxics (OPPT). The Work Plan Chemical Risk Assessment for Antimony Trioxide (ATO) was prepared based on currently available data and any additional information received during the public comment period and peer review process. Mention of trade names does not constitute endorsement by the US EPA.

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External Peer Review

EPA/OPPT released a peer review plan in August of 2012 and draft risk assessment and charge questions for peer review and public comment in January 2013. EPA/OPPT contracted with the Scientific Consulting Group, Inc. (SCG) to convene a panel of *ad hoc* reviewers to conduct an independent external peer review for the EPA's draft work plan risk assessment for ATO. As an influential scientific product, the draft risk assessment was peer reviewed in accordance with US EPA's peer review guidance. The peer review panel performed its functions by web conference and teleconference between November 13, 2013 and January 6, 2014. The panel consisted of the following individuals:

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Please visit the EPA/OPPT's Work Plan Chemicals web page for additional information on the peer review process (http://www.epa.gov/oppt/existingchemicals/pubs/riskassess.html) and the public docket (Docket: EPA-HQ-OPPT-2012-0722) for the independent external peer review report and the response to comments document.

GLOSSARY OF TERMS AND ABBREVIATIONS

ATO	Antimony Trioxide
BAF	Bioaccumulation Factor
BCF	Bioconcentration Factor
°C	Degree Celsius
CASRN	Chemical Abstract Service Registry Number
CBI	Confidential Business Information
cm^3	Centimeters Cubed
COC	Concentration of Concern
d	Day
DMR	Discharge Monitoring Report
dw	Dry Weight
EC_{50}	Half Maximal Effective Concentration
EC	European Commission
E-FAST2	Exposure and Fate Assessment Screening Tool, version 2
EPA	Environmental Protection Agency
EU RAR	European Union Risk Assessment Report
MATC	Maximum Allowable Toxicant Concentration
IARC	International Agency for Research on Cancer
IUR	Inventory Update Rule
kg	Kilogram
kg/yr	Kilogram(s) per year
lb	Pound
LC_{50}	Half Maximal Lethal Concentration
LOEC	Lowest-Observed-Effect Concentration
mg	Milligram
mg/kg	Milligram(s) per kilogram
mg/kg-bw	Milligram(s) per kilogram body weight
mg/L	Milligram(s) per liter
mg/m^3	Milligram(s) per cubic meter
µg/kg	Microgram(s) per kilogram
µg/L	Microgram(s) per liter
MMAD	Mass Median Aerodynamic Diameter
NAICS	North American Industry Classification System
NHANES	National Health and Nutrition Examination Survey
NOAEC	No-Observed-Adverse-Effect Concentration
NOEC	No-Observed-Effect Concentration
NPDES	National Pollution Discharge Elimination System
NWIS	National Water Information System
OCSPP	Office of Chemical Safety and Pollution Prevention
OECD	Organization for Economic Co-operation and Development
OPPT	Office of Pollution Prevention and Toxics
OSHA	Occupational Safety and Health Administration

PDM	Probabilistic Dilution Model
PET	Polyethylene Terephthalate
PNEC	Predicted No-Effect Concentration
PM	Particulate Matter
ppb	Parts Per Billion
PVC	Polyvinyl Chloride
REACH	Registration, Evaluation, Authorization and Restriction of Chemicals
Sb	Antimony
Sb^{3+}	Trivalent antimony
Sb^{5+}	Pentavalent antimony
$Sb(OH)_3$	Antimonite
$Sb(OH)_6^-$	Antimonate
$SbCl_3$	Antimony trichloride
Sb_2O_3	Antimony trioxide
Sb_2O_5	Antimony pentoxide
STORET	STOrage and RETrieval Data Warehouse
TRI	Toxics Release Inventory
TSCA	Toxic Substances Control Act
US	United States
USGS	United States Geological Survey
WWTP	Waste Water Treatment Plant
WHO	World Health Organization
ww	Wet Weight
7Q10	Refers to seven consecutive days of lowest streamflow over a 10-year period based on USGS.

EXECUTIVE SUMMARY

As a part of a comprehensive approach to enhance the Agency's existing chemicals management, in March 2012 EPA identified a work plan of chemicals for further assessment under the Toxic Substances Control Act (TSCA)[1]. The Agency is performing risk assessments on chemicals in the work plan. If an assessment identifies unacceptable risks to humans or the environment, EPA will pursue risk management. The category, "Antimony and Related Compounds" was assessed as part of the work plan.

ATO is the most widely used antimony compound, accounting for roughly 80 percent of primary (*i.e.*, mined) antimony consumption in the US (USGS, 2011). ATO is mainly used as a flame retardant synergist for halogenated flame retardants used in textiles and plastics (Freedman et al., 2003); approximately 25 million pounds (lbs) of ATO were consumed for this end-use application in 2011 (USGS, 2011). Available US Geological Survey (USGS) data indicated no operational antimony mines or primary antimony production in the US in 2011, except for one company in Montana that produced antimony metal and oxide from a foreign feedstock (USGS, 2012a).

Data from EPA's Toxics Release Inventory (TRI) for the 2010 reporting cycle indicated that approximately 970,000 lbs of antimony compounds were released to land and roughly 15,700 lbs were released to water by industries that use or produce ATO-containing flame retardants (TRI, 2010). On-site air releases totaled approximately 12,000 lbs, of which roughly two-thirds were from stack releases. A significant portion of the remaining fugitive air releases are likely related to the combustion of fossil fuels; however, the precise quantity cannot be determined from TRI data.

This assessment focused on ecological risks from the use of ATO as a synergist in halogenated flame retardants. Based on a review of the available data regarding antimony concentrations in food and environmental media, and biomonitoring data obtained from the latest National Health and Nutrition Examination Survey (NHANES), general population exposure to antimony is expected to be low (CDC, 2009; 2012). Because food and water are the primary sources of general population exposure, and the less toxic (*i.e.*, pentavalent) form of antimony predominates in these media, significant human health risks are not anticipated. This conclusion is supported by recent risk assessments completed for ATO in Canada and Europe. The US National Toxicology Program is currently conducting a two-year carcinogenicity study on ATO; EPA will review these results when they become available.

The remaining area of potential concern, for effects on ecological organisms, was selected as the focus of the current assessment. This decision was based on the Agency's review of available ecotoxicity information and reported industrial releases of Antimony and Related Compounds in association with the manufacture and use of ATO-containing flame retardants.

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[1] http://www.epa.gov/oppt/existingchemicals/pubs/workplans.html

To assess ecological risks, EPA/OPPT derived hazard benchmarks (*i.e.*, concentrations of concern; COC) for water- and sediment-dwelling organisms, which were compared to measured or predicted antimony concentrations in these media. Media concentrations were obtained from two independent environmental monitoring datasets, the US Geological Survey-National Water Information System (water samples from 1972 to 2012; sediment samples from 1987 to 2011; and soil samples from 2005 to 2010; USGS, 2012b) and EPA's Storage and Retrieval (STORET) Data Warehouse (water and sediment samples from 2002 to 2010; EPA, 2012c). In addition, site-specific modeling was conducted for selected TRI facilities (*i.e.*, those with identified use or production of ATO-containing flame retardants) using EPA's Exposure and Fate Assessment Screening Tool, version 2 (E-FAST2; EPA, 2010a) to estimate surface water concentrations. The measured and modeled concentrations used in this assessment are assumed to represent reasonable estimates of environmental exposures.

To evaluate the potential for adverse ecological effects, antimony concentrations in the ambient environment were compared to the hazard benchmarks (i.e., concentrations of concern; COC) identified to be protective of aquatic and benthic organisms. Under this paradigm, ecological risks were not found in association with ATO use as a flame retardant synergist. Using the E-FAST2 model, only one exceedance of the COC was predicted following acute exposure to aquatic organisms, and the available monitoring data showed no exceedances of the COC for aquatic, benthic or soil-dwelling organisms within the last three years (2009 to 2012). The available hazard and exposure information was inadequate to support a quantitative evaluation of risks to terrestrial organisms.

This assessment provides insights regarding ecological risks that may be expected in association with ATO use as a synergist in halogenated flame retardants. EPA/OPPT identified key sources of uncertainty related to limitations in the available hazard and exposure information, and conservative assumptions incorporated in the E-FAST2 model estimates of antimony surface water concentrations. Although these uncertainties may limit data interpretation, EPA/OPPT has high confidence in its minimal risk finding based on the following considerations:

- The use of release data for antimony compounds and conservative assumptions incorporated in the EFAST-2 model estimates likely overestimate ATO exposure potential.

- Environmental monitoring data obtained during the last three years show no exceedances of the hazard benchmarks identified to be protective of sensitive ecological species.

1 BACKGROUND AND SCOPE

1.1 INTRODUCTION

As a part of EPA's comprehensive approach to enhance the Agency's existing chemicals management, in March 2012 EPA identified a work plan of chemicals for further assessment under the Toxic Substances Control Act (TSCA). After gathering input from stakeholders, EPA developed criteria used for identifying chemicals for further assessment. The criteria focused on chemicals that meet one or more of the following factors: (1) potentially of concern to children's health (for example, because of reproductive or developmental effects); (2) neurotoxic effects; (3) persistent, bioaccumulative, and toxic (PBT); (3) probable or known carcinogens; (4) used in children's products; or (5) detected in biomonitoring programs. Using this methodology, EPA identified a TSCA Work Plan of chemicals as candidates for risk assessment over the next several years. In the prioritization process, ATO was identified for assessment based on: (1) high potential for exposure due to high production volume and widespread use in consumer products; (2) expected releases to aquatic, benthic, and terrestrial environments; (3) high persistence and low-to-moderate bioaccumulation potential; (4) biomonitoring data indicating its occurrence in water, air and soil; and (5) adverse effects reported in humans and laboratory animals.

The Agency is performing risk assessments on chemicals in the work plan. If an assessment identifies unacceptable risks to humans or the environment, EPA will pursue risk management. The target audience for this risk assessment is primarily EPA risk managers; however, it may also be of interest to the broader risk assessment community as well as US stakeholders that are interested in ATO. The information presented in the risk assessment may be of assistance to other Federal, State, and local agencies as well as members of the general public who are interested in the risks of ATO. EPA/OPPT relied on Agency guidance to develop this the risk assessment (US EPA, 1998 and US EPA 2014a). EPA followed the risk assessment paradigm beginning with planning and scoping and problem formulation.

The initial step in EPA's risk assessment development process, which is distinct from the initial prioritization exercise, is scoping and problem formulation. During this step EPA reviews currently available data and information, including but not limited to, assessments conducted by others (*e.g.*, authorities in other countries), published or readily available reports, and published scientific literature. During scoping and problem formulation, a more robust review of the factors that influence initial prioritization may result in refinement – either addition/expansion or removal/contraction – of specific hazard or exposure concerns previously identified using the prioritization methodology.

ATO is the most widely used antimony compound, accounting for roughly 80 percent of primary (*i.e.*, mined) antimony consumption in the US (USGS, 2011). ATO is mainly used as a flame

retardant synergist for halogenated flame retardants used in textiles and plastics (Freedman et al., 2003); approximately 25 million pounds (lbs) of ATO were consumed for this end-use application in 2011 (USGS, 2011). Available US Geological Survey (USGS) data indicated no operational antimony mines or primary antimony production in the US in 2011, except for one company in Montana that produced antimony metal and oxide from a foreign feedstock (USGS, 2012a).

Antimony compounds are subject to a number of regulations for various environmental media. Examples of programs where EPA has identified priority actions for this chemical category include:

- Listed as a Hazardous Air Pollutant (HAP) under the Clean Air Act 112(b)
- National Primary Drinking Water Regulation Maximum Contaminant Level established under the Safe Drinking Water Act (SDWA)
- Designated a priority pollutant under the Clean Water Act; effluent limitation guidelines established under CWA Section 304
- Annual reporting of chemical releases to the Toxics Release Inventory (TRI) under Section 313 of The Emergency Planning and Community Right-to-Know Act (EPCRA).
- Hazardous Waste designation and land disposal restrictions established under the Resource Conservation and Recovery Act (RCRA) and The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA)
- Listing on the TSCA Inventory and reporting requirements established under the Toxic Substances Control Act (TSCA)

1.2   PROBLEM FORMULATION

The objective of OPPT's TSCA Work Plan Chemicals assessments is to use readily available data to evaluate the human health and/or environmental risks associated with chemical exposures incurred as a result of TSCA-related uses. EPA/OPPT approached problem formulation by reviewing other recently performed assessments and searching published literature from 1978 to 2012 using EPA's ECOTOX database to identify additional information which was subsequently retrieved and evaluated for data adequacy. The literature review included ATO chemistry, uses, sources including industrial releases, fate, exposure, and hazard to humans and ecological receptors. Other sources of information reviewed were unpublished reports provided by the International Antimony Association, EPA's publicly available Inventory Update Reporting (IUR), Chemical Data Reporting (CDR) and Storage and Retrieval (STORET) databases, and the US Geological Survey National Water Information Service (NWIS). Data was acceptable if it met standard adequacy criteria which varies according to the type of information (data adequacy criteria are described in later sections of this report).

1.2.1 ATO Uses

Global antimony demand is largely driven by the production of antimony compounds used in various consumer products such as medicines, polymers, resins, and textiles. ATO is considered to be the most economically relevant and environmentally prevalent antimony compound, accounting for approximately 80 percent of global antimony consumption; a smaller portion (roughly 20 percent) of global antimony consumption is associated with metallic uses (Roskill, 2011). Based on this market information, ATO was selected as the focus of the current assessment.

ATO is primarily used as a flame retardant synergist. ATO is also used as a condensation catalyst for the manufacture of polyethylene terephthalate (PET), a large-volume commodity plastic used in bottles, films, and various food contact applications. This use scenario has been evaluated for safety and is regulated by the US Food and Drug Administration and other international authorities (EC/HC, 2010; EC, 2008; European Food Safety Authority, 2004); therefore, ATO use in PET plastics will not be further evaluated here.

1.2.2 Regulation of ATO

Antimony compounds are subject to a number of regulations for various environmental media. Examples of programs where EPA has identified priority actions for this category include:

- Listed as a Hazardous Air Pollutant (HAP) under the Clean Air Act 112(b)
- National Primary Drinking Water Regulation Maximum Contaminant Level established under the Safe Drinking Water Act (SDWA)
- Designated a priority pollutant under the Clean Water Act; effluent limitation guidelines established under CWA Section 304
- Annual reporting of chemical releases to the Toxics Release Inventory (TRI) under Section 313 of The Emergency Planning and Community Right-to-Know Act (EPCRA).
- Hazardous Waste designation and land disposal restrictions established under the Resource Conservation and Recovery Act (RCRA) and The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA)
- Listing on the TSCA Inventory and reporting requirements established under the Toxic Substances Control Act (TSCA)

1.2.3 Human Exposure Potential

Based on a review of the available data regarding antimony concentrations in food and environmental media, and biomonitoring data obtained from the latest National Health and Nutrition Examination Survey (NHANES), general population exposure to antimony is expected to be low (CDC, 2009, 2012). Because food and water are considered to be the primary sources of general population exposure, and the less toxic (*i.e.*, pentavalent) form of antimony generally predominates in these media, significant human health risks are not expected at ambient levels

to which individuals may be exposed in the environment. This conclusion is supported by previous ATO risk assessments conducted in Canada and Europe (EC/HC, 2010; EC, 2008). Additional information regarding the human health effects that may be associated with ATO exposure can be found in Appendix A. The US National Toxicology Program is currently conducting a two-year carcinogenicity study on ATO; EPA will review these results when they become available.

1.2.4 Ecological Exposure Potential

Industry reported water releases associated with the production or use of ATO as a synergist in halogenated flame retardants are possible. EPA's 2010 TRI data can be sorted by industry codes, using the North American Industry Classification System (NAICS) codes, to identify a subset of facilities that indicate production, processing or use of ATO-containing flame retardants. Because ATO is not specifically listed on the TRI, releases reported under the broader category of antimony compounds were used as a surrogate to evaluate the potential for aquatic exposures. Many different types of antimony compounds, and sources for their environmental release are known, so use of this data is therefore expected to provide a conservative estimate of ATO exposure potential.

Queries of the 2010 TRI were used to obtain information on antimony water releases for the 36 TRI facilities identified under the scope of this assessment (i.e., those indicating production, processing or use of ATO-containing flame retardants). Water releases reported by the identified TRI facilities could be modeled using EPA's Exposure and Fate Assessment Screening Tool version 2 (E-FAST2) modeling program to estimate antimony surface water concentrations.

Environmental monitoring information obtained from the USGS National Water Information System (NWIS; USGS, 2012b) and EPA STORET (EPA, 2012c) databases include antimony concentrations in the ambient environment, which can also be used to evaluate environmental exposures to antimony. Data collection specifically centered on those states in which the selected TRI facilities are located may provide an approach to linking environmental antimony concentrations to ATO use. This USGS dataset is considered robust based on the presumption that consistent internal methodologies were utilized to derive monitoring information of sufficient quality for exposure characterization, as it was collected under the guidance of the USGS National Field Manual for the Collection of Water-Quality Data, a publication which documents the methods, protocols, procedures and recommended practices for the collection of water-quality data (USGS, 1999).

1.2.5 Ecological Hazard Assessment

For hazard identification, key ecological hazard reports were obtained for critical review and evaluation of data adequacy. Details regarding the data adequacy criteria used for the ecological hazard assessment are included in Chapter 3.

Due to ATO data limitations, antimony trichloride and antimony sulfate were considered as supporting chemicals to characterize hazards to aquatic and sediment-dwelling organisms. Antimony trichloride tends to dissolve more readily than ATO and is therefore expected to provide a conservative estimate of ecological hazard potential.

Risks to soil/terrestrial organisms were not evaluated due to deficiencies in the available hazard and exposure data; however, ATO is not expected to present a major concern for soil-dwelling organisms, as it exhibits limited bioaccessibility in this environmental compartment. During problem formulation, EPA/OPPT summarized the following information, which may be used in a weight-of-evidence approach to support this assumption:

- There is considerable evidence for the natural attenuation of antimony in soils over time. Field studies of weathered or 'aged' soils show significant decreases in bioaccessibility when compared to 'antimony spiked' soils typically used for laboratory studies. This phenomenon, and the use of more soluble antimony compounds for laboratory testing, may overestimate potential toxicity to soil-dwelling organisms.

- The limited bioaccessibility of antimony in soil may be explained by its association with organic or inorganic soil constituents. Soil conditions that promote precipitation or adsorption are expected to decrease bioaccessibility to soil-dwelling organisms.

- Sequential extraction procedures have shown that antimony often occurs in the 'residual' or 'refractory' soil fraction. Largely on the basis of such results, it has been established, and is generally accepted, that antimony is essentially immobile in soils.

- Because antimony exhibits limited mobility, and low solubility in soil (*i.e.*, limited bioaccessibility), the amounts that may be released to this environmental compartment are not expected to present significant concerns for soil-dwelling organisms.

For the above reasons EPA/OPPT did not conduct a quantitative evaluation of risks to the terrestrial compartment.

1.3   CONCEPTUAL MODEL FOR ENVIRONMENTAL ASSESSMENT OF ATO

The Conceptual Model depicted in Figure 1-1 identifies environmental releases and critical exposure pathways associated with ATO use as a flame retardant synergist. Based on the information currently available, EPA/OPPT identified releases to water as a primary exposure pathway for aquatic and benthic organisms. Aqueous exposures that occur throughout the full extent of the water column, including surface sediments, were considered to be the critical exposure pathway for ATO environmental releases.

During the wastewater treatment processes, influent is treated, producing sludge and effluent that is released to surface water at outfall sites, which may be located in freshwater, estuarine, or marine environments. Under this scenario, ATO enters the environment via effluents and is diluted to varying degrees in streams, lakes, estuaries, and oceans where aquatic and benthic organisms may be exposed.

Because of its physical chemical properties, ATO partitions to solid phases (sludge) during wastewater treatment processing and with further treatment, can become concentrated in biosolids. This organic carbon-rich material may be disposed of via landfill, incinerated, or utilized for land amendments. Wastewater Treatment Plants (WWTPs) need to dispose of waste sludge continuously, and a portion of this material is applied to agricultural land each year as biosolids. ATO released to land has a long half-life in soil, and may be available for uptake by plants and soil invertebrates; however, ATO is expected to exhibit limited bioaccessibility in soil due to chemical interactions with soil matrix constituents that limit exposure potential.

Surface waters and sediments were identified as primary exposure pathways because ecological organisms are most likely to encounter ATO in these media. Soils and subsurface sediments are not expected to provide a significant route of exposure for ecological organisms; therefore, these environmental compartments were not evaluated. The ATO assessment did not assess future risks that could occur if environmental conditions change.

Assessment Questions:

1. Do ATO concentrations measured in US surface waters pose risks for adverse effects in aquatic organisms?

2. Do ATO concentrations measured in US sediments pose risks for adverse effects in benthic organisms?
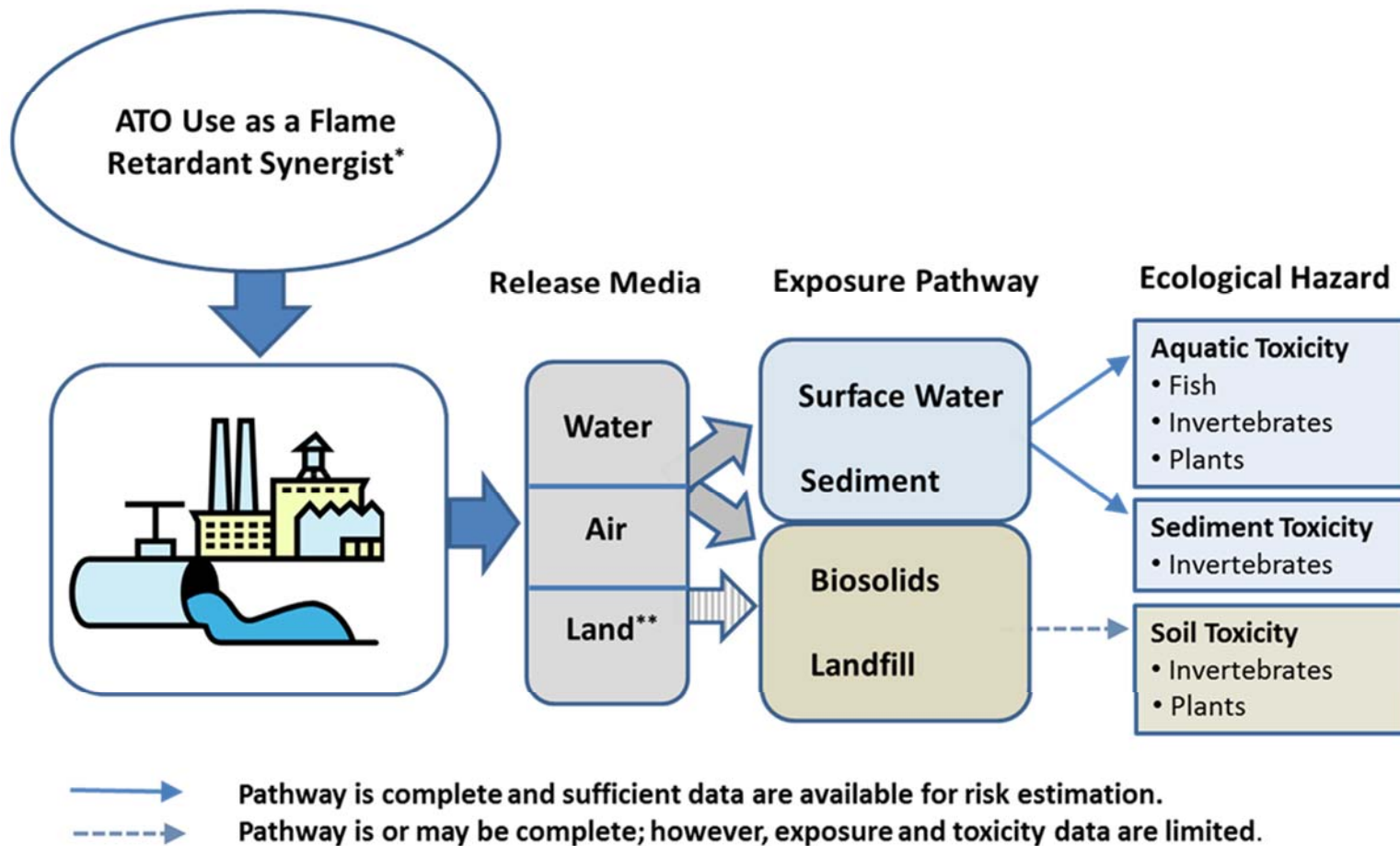
Figure 1-1. Conceptual Model

[*] There are many potential sources of antimony releases that may occur directly to environmental media, or indirectly through WWTPs.

[**] Risks are expected to be low for this compartment due to limited bioaccessibility to soil-dwelling organisms.

1.4   ANALYSIS PLAN FOR ENVIRONMENTAL ASSESSMENT

The objective of this assessment is to determine the media and pathways through which ATO may adversely impact ecological organisms.

Although information specifically related to ATO was limited, the data identified for antimony and related compounds was considered sufficient for an ecological risk evaluation. Risks to aquatic and sediment-dwelling (*i.e.*, benthic) organisms were selected as the focus of this assessment, based on the available hazard and exposure information. The ecological assessment endpoints evaluated for ATO include acute and chronic toxicity to aquatic and benthic organisms. Risks to terrestrial/soil-dwelling organisms were not evaluated due to deficiencies in the available hazard and exposure data.

A characterization of environmental hazards, as indicated by reported ecotoxicity benchmarks (concentrations of concern; COC), and environmental exposures, as indicated by reported industrial releases and environmental monitoring data, were used to evaluate risks to water- and sediment-dwelling organisms.

EPA/OPPT identified a subset of TRI facilities (*i.e.*, those reporting production or use of ATO as a flame retardant synergist) for exposure characterization. Due to the availability of a sufficient quantity of environmental monitoring data for the US, EPA/OPPT also collected information on ambient antimony concentrations for those states in which the selected TRI facilities are located.

Hazard data obtained from relevant and reliable ecotoxicological studies were used to calculate conservative (acute and chronic) COCs for aquatic and benthic organisms.

To assess ecological risks, COCs for water- and sediment-dwelling organisms are compared to the measured or modelled antimony concentrations in these media using a risk quotient (RQ) approach. For risk assessments that are based on an RQ approach, the comparison of exposure concentrations to COCs has a binary outcome: either the environmental concentrations exceed the COC, or they do not. A risk quotient greater than one (RQ > 1) suggests a potential for adverse ecological effects; however, it is generally presumed that an RQ at or below one indicates a low potential for adverse effects to ecological organisms.

2 SOURCES AND ENVIRONMENTAL FATE

2.1 INTRODUCTION

ATO enters the environment from natural (*e.g.*, weathering of rocks, aerial transport of volatile compounds) and anthropogenic sources (*e.g.*, combustion, mining, smelting activities). In order to assess ecological risks, a firm understanding of the primary uses, physicochemical properties, and ultimate fate in the environment is necessary.

2.1.1 Physical and Chemical Properties

Antimony (Chemical Abstracts Service Registry Number [CASRN] 7440-36-0) is a naturally occurring metalloid that belongs to Group 15 of the periodic table; it can exist in +5, +3, 0, or -3 oxidation states, with +5 (*i.e.*, pentavalent) and +3 (*i.e.*, trivalent) being most common.

ATO is a dimorphic crystalline solid that exists in cubic (*i.e.*, senarmontite; CASRN 12412-52-1) and orthorhombic configurations (*i.e.*, valentinite; CASRN 1317-98-2). Cubic senarmontite is thought to be stable below 570 °C, whereas valentinite (*i.e.*, infinite double chains) is formed above this temperature (Freedman et al., 2003); the cubic form is shown in Figure 2-1; the orthorhombic form is shown in Figure 2-2. Because ATO may exist in either molecular arrangement and there are no exposure or hazard data that specifically differentiate between the two, both were addressed as ATO (CASRN 1309-64-4) in this assessment.
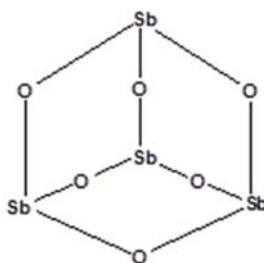
Figure 2-1. Representative Chemical Structure of Senarmontite (CASRN 12412-52-1)
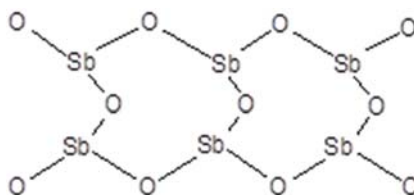
Figure 2-2. Representative Chemical Structure of Valentinite (CASRN 1317-98-2)

ATO exhibits negligible vapor pressure at 25°C and tends to be soluble in hydrochloric acid, tartaric acid, and caustic alkalies; it is insoluble in organic solvents and only slightly soluble in water (Patnaik, 2003). ATO solubility (when determined in reconstituted standard water; ISO 6341 medium) was 2.76 mg/L after a seven-day equilibration period at 22.2 °C and slightly alkaline (pH 8) conditions (EC, 2008); typical physicochemical properties of ATO are shown in Table 2-1.

Table 2-1. Physical-Chemical Properties

Property	Value	Comment
Physical state at 25 °C	White solid	
Particle size	0.2-13.89 μm[a]	
Estimated vapor pressure at 25 °C	$<1 \times 10^{-6}$ torr	
Water solubility at 22.2 °C	2.76 mg/L at pH 8[b]	Value determined after 7 days In reconstituted standard water, loading = 100 mg/L .
Water solubility at 20 °C	19.7 mg/L at pH 5[c] 25.6 mg/L at pH7[c] 28.7 mg/L at pH 9[c]	Value determined after 24 hours in distilled water.
Melting point	656 °C[d]	
Boiling point	1425 °C[d]	
[a]Weidenfeller (2005) as cited in EC (2008). [b]LISEC (2002) as cited in EC (2008). [c]Umweltanalytik GmbH (1993) as cited in EC (2008). [d]Lide (2000) as cited in EC (2008).		

2.2   PRODUCTION AND USES

Several types of antimony compounds are known to exist in nature; however, the mineral stibnite is considered to be the primary source of processed antimony in commerce. In terms of the total production volume and potential end-use applications, ATO is the most widely used antimony compound.

No primary (*i.e.*, mined) antimony production occurred in the US in 2011, with the exception of one company that produced antimony metal and oxide from a foreign feedstock (USGS, 2012a). Approximately 87 percent of the ATO consumed in the US between 2007 and 2011 was imported from other countries (USGS, 2012a). Approximately 240 million pounds (lbs) of ATO was produced globally in 2005, with the majority of production taking place in China (The Chemical Company, 2012). Total combined production of ATO in the US and Mexico was estimated at approximately 52.8 million lbs in 2005; however, the total amount of ATO consumed in the US during this time period was estimated at roughly 70 million lbs (International Antimony Association, 2010).

Currently, Chemtura Corporation and US Antimony Corporation are the two main ATO producers in the US (Roskill, 2011; USGS, 2006). While neither is listed as a manufacturer on the 2006 TSCA Inventory Update Reporting (IUR) rule, their respective websites indicated that they manufacture and sell ATO products (Chemtura, 2012; USAC, 2012). The ATO national production volume for 2012 was 5.4 million pounds (EPA, 2012). Table 2-2 provides a list of companies that manufacture or import ATO in the US.

Table 2-2. US Manufacturers and Importers

Company	City	State	Manufacture	Import
Arkema Inc.	Philadelphia	PA	CBI	CBI
Chemson Inc.	Philadelphia	PA	No	Yes
E. I. du Pont de Nemours and Company	Wilmington	DE	No	Yes
Eastman Kodak Company	Rochester	NY	No	Yes
Ferro Corporation	Washington	PA	No	Yes
General Electric Company	Mt. Vernon	IN	No	Yes
	Bridgeville	PA	No	Yes
	Washington	WV	No	Yes
ICC Chemical Corporation	New York	NY	No	Yes
LG Chem America, Inc.	Englewood Cliffs	NJ	No	Yes
PolyOne Corporation	Avon Lake	OH	No	Yes
Sojitz Corporation of America	New York	NY	No	Yes
United Mineral & Chemical Corp.	Lyndhurst	NJ	No	Yes
CBI = confidential business information. Source: EPA (2010b)				

ATO is primarily used as a synergist with halogenated flame retardants applied to textiles and plastics; other applications include plastic stabilizer, paint pigment, catalyst precursor, fining agent or antisolarant (Freedman et al., 2003). Additional uses for ATO are shown in Table 2-3.

Table 2-3. Major End-Uses of Antimony in the US in 2011

End-Use	Market Share (%)
Flame retardant synergist	36
Transportation (including batteries)	23
Chemicals	16
Ceramics and glass	12
Other	13
Source: USGS (2012a)	

ATO has no inherent flame retarding capacity (EPA, 2012a); however, when used as a flame retardant synergist, it decreases the amount of halogen needed to impart acceptable flame resistance (USGS, 2004; Weil and Levchik, 2009). ATO interacts with chloride or bromide to form flame quenching antimony trichloride or antimony tribromide (Weil and Levchik, 2009). In many instances, ATO is added to (rather than incorporated in) the actual polymer matrix (AkzoNobel, 2012; Albemarle, 2000; ChemEurope, 2004), occurring as a separate, non-reactive (*i.e.*, non-covalently bonded) phase. ATO may be found in a number of flame-retarded products, including electrical equipment, wires, automotive parts, building materials, furniture, carpets, mattress covers, plastics, and textiles (EC/HC, 2010). The average ATO content in flame retardant formulations used to treat plastics generally ranges from 25 to 33 percent by weight, whereas the relative abundance of ATO in the final (flame-retarded) product can range from 1.5 to 12 percent (USGS, 2004). The demand for ATO in halogenated flame retardant applications appears to be decreasing; approximately 25 million lbs of ATO were consumed by this market sector in 2011, as compared to approximately 50 million lbs in 1999.

2.3 ENVIRONMENTAL FATE

ATO may be released to air, water, sediment and soil where it is slowly transformed into $Sb(OH)_3$ (antimonite), which may be further oxidized to $Sb(OH)_6^-$ (antimonate) under aerobic conditions in the environment. Atmospheric releases may be dissolved in rain water, or adsorbed to particulate matter and deposited to soil and surface waters via wet and dry deposition (EC, 2008). Environmental fate processes can increase or decrease antimony mobility in these environmental compartments, thereby impacting its availability to ecological organisms. Processes that influence fate in water, sediment and soil systems are briefly described below.

2.3.1 Environmental Persistence

Though not particularly abundant, antimony compounds are naturally occurring in the earth's crust. ATO is an inorganic metal oxide; therefore, a fate analysis based on the octanol-water partition coefficient or the soil organic carbon-water partition coefficient is not applicable. Typical mass balance fugacity modeling is also inapplicable to ATO (or other metal ions) released upon dissolution, because these substances exert zero partial pressure and fugacity in air (Diamond et al., 1992). ATO is persistent in all media, because the antimony ions issued upon dissolution cannot be degraded; however, antimony can be transformed between different chemical species via biotic and abiotic processes which can impact bioaccessibility and potential toxicity to ecological organisms (EC, 2008).

2.3.1.1 Fate in Water

Industrial effluents released to the environment may contain antimony as a result of losses from various operating procedures, such as material handling (*e.g.*, textile washing, etc.) and/or washing of equipment. Most of this material will end up in industrial or municipal wastewater

treatment plants, where significant amounts may be deposited in biosolids as a result of alkalinization processes. Biosolids containing antimony may be incinerated, landfilled or applied to land, with potential impacts on terrestrial and aquatic compartments due to leaching and runoff (EC/HC, 2010).

The fate and transport of dissolved and particulate forms of antimony are influenced by various physical, chemical, and biological processes (Filella et al., 2002a, 2002b; Filella et al., 2007). Fluctuations in flow parameters such as advection, or dispersion, and colloidal interactions (*e.g.*, settling, resuspension) can directly impact bulk movement of materials throughout the water column. In addition, chemical interactions between aqueous constituents can impact transport via shifts in thermodynamic equilibria that drive phase associations (*e.g.*, adsorption, dissolution), or biological transformations (*e.g.*, methylation, mineralization) which can affect environmental mobility and potential toxicity in aquatic systems.

Antimony concentrations in unpolluted natural waters are low, usually on the order of one part per billion (ppb) or less; levels detected in ocean surface waters tend to be even lower (Filella et al., 2002a). In a review of the published literature on antimony speciation in natural waters, Filella et al. (2009) reported that relatively small amounts of Sb^{3+} were detected in oxygenated waters. This finding has been corroborated by other authors, such as Garbos et al. (2000), who determined that the majority of antimony detected in natural waters (*i.e.*, 96 to 99 percent) occurs as Sb^{5+}. Cutter (1992) evaluated the rate of oxidation of Sb^{3+} to Sb^{5+} by measuring the depth profiles for antimony species in the upper 100 meters of the Black Sea. In this study, no Sb^{3+} was detected at surface levels; however, a gradual increase in the Sb^{3+} concentration (in association with a decline in Sb^{5+} levels) was observed at depths beyond 60 meters. An estimated pseudo-first-order rate constant of 0.008 day^{-1} was calculated for the oxidation of Sb^{3+} to Sb^{5+}, corresponding to a residence time of approximately 125 days for Sb^{3+} in natural waters.

Despite prevailing thermodynamic theories, Sb^{5+} has been detected in anaerobic environments in association with the following potential transport processes: movement associated with detritus sinking from aerobic waters, advection of surface waters containing Sb^{5+} and biotransformation of Sb^{3+} by marine organisms. These transport mechanisms assume a slow rate of conversion (*i.e.*, reduction) to Sb^{3+} in the absence of oxygen; the rate constant for this reaction was estimated at approximately 1.1×10^{-6} days^{-1} (Cutter, 1992). Atmospheric deposition and biotic or abiotic reduction have been proposed as possible explanations for the occurrence of Sb^{3+} in aerobic surface waters (ATSDR, 1992).

2.3.1.2   Fate in Sediment

Antimony is mobile upon release to aquatic compartments (Filella et al., 2002a, 2002b); however, it exhibits a strong tendency for adsorption to suspended particles and sediment, such that a significant portion of dissolved antimony will ultimately end up in soil and sediment. This may occur through adsorption to suspended particles as they settle out of solution

(Hamilton-Taylor et al., 1984), or precipitation reactions; a portion of the undissolved parent material may also occur in sediment (EC, 2008).

The extent to which antimony partitions between solid and solution phases is influenced by its chemical speciation, which may be impacted by a variety of environmental processes (*e.g.*, acid neutralization, secondary mineral precipitation). Dissolved antimony concentrations are mainly controlled by adsorption to (or precipitation of) metal oxide containing particulates, and colloids of mineral secondary phases (*e.g.*, amorphous ferric-hydroxide) or other high surface area materials (*e.g.*, bacteria, fungi, organic matter). For example, acid neutralization via dissolution of carbonate or dilution with surface waters may reduce the solubility of metal ions, such as Fe^{3+}, resulting in precipitation of secondary minerals that may serve as a reservoir for antimony in sediment (Nordstrom and Alpers, 1999). Colloidal and fine-grained sorbent materials may transport antimony for long distances before settling in the streambed (Ritchie et al., 2013).

The log sediment-water partition coefficient for antimony ranges from 2.5 to 4.8 (EC/HC, 2010); suggesting a tendency for adsorption; however, under certain circumstances antimony adsorption to sedimentary organic matter may be reversed by changes in pH or redox conditions. For example, Sb^{5+} adsorption to humic acids (as the soluble antimonate anion $Sb(OH)_6^-$), which is maximal under acidic conditions (Filella et al., 2002a; Tighe et al., 2005), may be reversed under alkaline conditions, resulting in release of Sb^{5+} back into the water column. In contrast, the neutral antimonite species ($Sb(OH)_3$, which is the most stable form of Sb^{3+} under anaerobic (reducing) conditions, will adsorb to humic acids and metal oxides across a wide range of pH values, and is therefore expected to be largely immobile in the environment (Leuz et al., 2006; Scheinsost et al., 2006).

2.3.1.3   Fate in Soil

Given its negligible vapor pressure and limited water solubility, ATO released to the terrestrial compartment is expected to remain in soil; however, leaching and subsequent releases to local surface waters (via run off) may occur as a result of seasonal rainfall or snow (EC/HC, 2010). Subsequent exposures (*i.e.*, those incurred as a result of leaching from ATO containing materials) are unlikely to reach levels that would present a major concern for soil-dwelling organisms due to complex interactions with soil constituents that effectively limit antimony mobility (bioaccessibility).

Ambient antimony concentrations in US soils are low, ranging from 0.15 to 1.5 mg kg^{-1} (USGS, 2014); however, elevated concentrations have been reported in association with anthropogenic activities such as mining, smelting and the combustion of fossil fuels (ATSDR, 1992; Filella et al., 2002a, 2002b). Although high levels (up to 700 mg kg^{-1}) have been reported in association with mining activities, antimony released to soil was determined to be biologically unavailable over a wide range of environmentally relevant pH values (Flynn et al., 2003). Antimony bioaccessibility to soil-dwelling organisms may be characterized in part, by its distribution between solid and

solution phases (*i.e.*, soil-water partition coefficient k_d for soil); however, this parameter can vary widely due to the site-specific nature of phase associations. An evaluation of the published literature showed significant variation in the available partition coefficients for antimony in soil (EC, 2008); reported values ranged from 60 to 1316 (log k_d = 1.8 to 3.2) in sandy soils (pH 5.7) and 94 to 962 (log k_d = 2.0 to 2.98) in heavy clay soils (pH 7.8). The observed variability in values reported for different sites may result from numerous of factors including differences in media type and loading rate. Key limitations associated with the use of k_d values to assess bioaccessibility to soil-dwelling organisms include the following:

- The nature and stability (*i.e.*, permanence) of binding interactions are poorly defined
- A lack of site-specific information precludes quantitative evaluations of bioavailability
- The environmental factors that influence adsorption/desorption reactions are dynamic

Antimony mobility in soils is primarily limited by adsorption to hydrous oxides of iron, manganese, and aluminum (HSDB, 2012), which are fairly ubiquitous in the environment. Because antimony exhibits a strong tendency for adsorption to metal oxides at environmentally relevant pH levels, antimony is expected to exhibit limited mobility in this environmental compartment (Leuz et al., 2006). Studies suggest that over time, antimony released to soil may become sequestered (*i.e.*, removed from the dissolved phase) by weathering or 'aging' phenomena. This natural attenuation mechanism likely reflects biotic and/or abiotic surface charge interactions which induce formation of mineral aggregates of varying stability depending on the strength of the chemical association (*e.g.*, electrostatic, ionic, or covalent bond energy). Site-specific parameters (*e.g.*, pH, matrix composition), and mineralogical properties (*i.e.*, soil type, predominate mineral phase) will likely influence the relative stability of the controlling secondary mineral phase (Leuz et al., 2006).

2.3.1.4   Fate in Air

Antimony is often released into the atmosphere as a result of high-temperature anthropogenic processes, such as metal smelting and fuel combustion, which represent major sources of antimony air releases. As a non-gaseous element with negligible vapor pressure, ATO is emitted primarily as fine particulate matter (PM) which may be removed from the atmosphere by gravitational settling or wet and dry deposition, depending on particle size. Larger particles are generally removed from the atmosphere within hours, whereas smaller particles, such as those generated via combustion processes, require a longer time period (on the order of weeks). In a model comparing particle size to volatility, antimony exhibited an average atmospheric half-life of approximately 3.2 days (ATSDR, 1992).

Regardless of the original form, combustion conditions (*e.g.*, high temperature and excess oxygen) are expected to release antimony oxides to the atmosphere; it is assumed that combustion produces the trivalent form which may be further oxidized to the pentavalent form upon contact with moisture and oxygen in the environment (EC, 2008). Ashes generated from combustion processes are generally landfilled; however, small amounts may be released to air

despite use of pollution control devices (*e.g.*, bag filters, wet scrubbers) and subsequently deposited on soil, surface water and sediments.

A study of wet and dry deposition over an eight-week period in an area that was presumably far from release sources found that 87 percent of antimony deposited from air was dissolved in rain; 11 percent occurred as fine PM in rain, and approximately two percent was removed from the air via dry deposition of fine PM (Stössel and Michaelis, 1986; as cited in EC, 2008). Airborne sources of antimony releases to aquatic and terrestrial systems seem to be important even for environmental systems that are located far from the original point of release; however, the available data do not allow firm conclusions to be drawn (Filella et al., 2009).

2.3.2   Environmental Bioavailability, Transformation and Bioaccumulation

Antimony occurs in water, soil and sediment, where it can transition between various organic and inorganic forms. The potential for ecological exposure varies according to the inherent capacity and changing ability of various environmental compartments to adsorb, transform, and release antimony in forms that can impact ecological organisms.

2.3.2.1   Bioavailability

Changes in antimony bioavailability are largely a function of environmental processes that effectively increase or decrease its mobility, thereby making it more or less accessible to receptor organisms. For environmental risk assessments involving soils and sediments, the ability of matrix constituents to sequester or transform antimony will ultimately determine how much of the ambient exposure concentration is capable of affecting biological activity. Site-specific chemical and physical conditions can greatly influence the chemical form (*i.e.*, speciation) of antimony and thus the degree to which it adsorbs to matrix constituents and is thereby removed from solution.

The main purpose for bioavailability considerations is to provide a more accurate or 'realistic' characterization of exposure potential. Qualitative evidence can assist in determining how much of the total antimony concentration is actually available for uptake, and this is likely to provide a better indication of the potential for adverse ecological effects. Biochemical processes (*e.g.*, ion exchange, methylation) can change the physicochemical form, or speciation of antimony, thereby impacting its accessibility to ecological organisms. For the purposes of this discussion, bioaccessibility is defined as the amount of antimony that an organism can extract from exposure media, whereas bioavailability reflects the amount that is taken up from the gut contents into the general circulation.

Adsorption, dissolution and precipitation reactions are primary determinants of antimony availability in soils. Antimony can occur in the solid or aqueous (*i.e.*, soil solution) phase; in solution, it can exist either as the hydrated free ion or be complexed with various organic or inorganic ligands. Conversely it may be sequestered by various adsorption or precipitation

reactions that effectively remove it from solution via formation of relatively inert secondary mineral phases (Leuz et al., 2006). Dissolved antimony is generally more available for a variety of processes, including uptake and transport; therefore, soil pH is a primary determinant of availability to soil-dwelling organisms, as it directly influences solubility. Acidic conditions and high humic acid content (> 5 to 10 percent) generally decrease antimony mobility in soil (Alloway, 2005).

Dissolution kinetics represent a major controlling factor for antimony availability in soil, due to the formation of stable secondary mineral phases. Both the extent to which these mineral species occur in a particular soil and their solubility in various biological fluids will influence bioavailability to ecological organisms. Previous studies have demonstrated slow dissolution kinetics for antimony in soil and limited bioaccessibility to soil-dwelling organisms (Denys et al., 2012; Oorts et al., 2008).

Studies conducted using soluble antimony compounds freshly mixed with soils or sediments generally do not show significant reductions in bioavailability, and will not provide an accurate representation of the influence exerted by natural modifying factors that can dramatically affect exposure potential. In general, soil conditions that promote precipitation or adsorption tend to reduce mobility in the environment and bioaccessibility to ecological organisms. Because this information is largely site-specific, extrapolation across different geographic regions is not feasible; however, as a first approximation, assumptions regarding expected impacts of various physicochemical parameters (*e.g.*, soil pH, matrix constituents), can be used in a weight-of-evidence approach to support the assumption of limited availability in soil. The following information is intended as a guideline; soils at specific sites may not conform to all of the general trends discussed here.

- Both the form of antimony (speciation) and the amount of time it has been present in soil or sediment can influence its bioaccessibility to ecological organisms; antimony generally exhibits limited solubility and slow dissolution kinetics in soil.

- Site-specific parameters can influence soil weathering ('aging') reactions which further decrease antimony solubility. Limited solubility and bioaccessibility to ecological organisms is expected in soils that contain calcium, sulfide or metal oxides (which are ubiquitous in the environment) due to increased adsorption potential.

- Antimony exhibits increased mobility under alkaline conditions, and in soils containing elevated (> than 5 to 10 percent) fulvic acid content; however, these soils often contain relatively insoluble sulfides (due to the actions of sulfate-reducing bacteria in seasonally flooded soils) that can effectively remove antimony from the soil solution phase.

- Rainfall, bacterial activity and degradation of organic matter can decrease soil pH, which generally increases antimony adsorption to soil particles. Since most US soils tend to be acidic to slightly alkaline, and antimony generally exhibits limited solubility and slow

dissolution kinetics in soil, it is expected to exhibit limited bioaccessibility to soil-dwelling organisms.

2.3.2.2   Transformation

Transformations of antimony have been reported in the presence of organic matter and microorganisms. For example, humic acids have been shown to oxidize Sb^{3+} to Sb^{5+} (Buschmann et al., 2005; Steely et al., 2007) and various microorganisms (*e.g.*, bacteria, fungi), have been shown to methylate relatively small amounts (< 0.1 percent) of antimony (Duester et al., 2008; HSDB, 2012). Information on antimony methylation is limited, but measured concentrations are usually low, and the extent of antimony methylation in environmental media appears to be significantly less than that of other elements, such as arsenic (Filella et al., 2009). Certain metal hydroxides, soil bacteria and marine organisms have also been shown to transform antimony (Filella et al., 2007; HSDB, 2012); thereby impacting its bioaccessibility and potential toxicity to ecological organisms.

There is considerable evidence for the natural attenuation of antimony in soils due to formation of low-solubility species during equilibration with the soil constituents. For example, fungal involvement has been implicated in mineral formation (Fomina et al., 2007); fungal organisms can acidify their microenvironment and excrete a variety of metabolites (*e.g.* siderophores, carboxylic acids) that effectively sequester metals within a matrix of secondary minerals. All major microbial groups have demonstrated roles in metal immobilization and mineral formation. Reported mechanisms include microbial oxidation or reduction of a metal species released via mineral dissolution, and metabolite excretion or other metabolism-dependent transformations, with microbial surfaces providing chemically reactive sites for adsorption, nucleation and subsequent formation of mineral precipitates (Gadd, 2010).

Studies of sequential extraction techniques have shown that a low percentage of antimony is extracted from soils using mild or aqueous (*e.g.*, water soluble) solutions; most occurs in the "residual" or "recalcitrant" fraction. Largely on the basis of such results, it has been established, and is generally accepted, that antimony is essentially immobile in soils (Filella et al., 2009). Similar results were demonstrated in field tests used to confirm the chemical distribution of antimony in creek sediments contaminated by a local smelter operation. These results showed that antimony was predominantly associated with the residual sediment fraction, with small amounts (~ 3 to 7 percent) occurring in the water soluble or exchangeable fractions (Baeza et al., 2010). As mentioned previously, a major difficulty encountered in applying information on phase associations is their site-specific nature; uncertainties regarding the primary factors that control antimony mobility (and bioaccessibility) in aquatic environments remain.

Antimony availability to plants is often low, even in contaminated sites, due to limited water solubility (Gal et al., 2007; Hammel et al., 2000); however, accumulation has been reported in some plant species (Tschan, 2009). Antimony uptake and bioavailability to plants was investigated from field sampling of grass near a smelter emitting antimony, and in experiments

in which grass was grown under laboratory conditions using the same antimony-contaminated soils (Ainsworth et al., 1990). It was concluded that the high levels of antimony observed in grass grown near the smelter occurred primarily from atmospheric deposition; antimony uptake from soil, and translocation to shoots accounted for only a small percentage of the total antimony concentration, suggesting low bioavailability in soils. Biota obtained from the site also showed minimal uptake of antimony from soil.

2.3.2.3   Bioaccumulation

Bioaccumulation is of potential concern due to direct impacts to organisms that accumulate metals and indirect impacts to those that consume them (*i.e.*, trophic transfer). Previous studies have indicated that antimony may bioaccumulate in aquatic organisms; however, there is considerable uncertainty regarding the derivation of these values (EC, 2008).

Antimony is not expected to exhibit significant bioaccumulation potential in aquatic organisms; metal bioaccumulation in this environmental compartment is largely governed by trophic feeding groups, with detritivores accumulating higher levels than herbivores or predatory groups. Although the potential for secondary impacts as a result of trophic transfer exists, it is not well characterized. For example, *Corbicula fluminea*, a filter-feeding clam that consumes algae, bacteria, and fine detritus, has been reported to bioaccumulate antimony to a greater extent than other predators or omnivores (Duran et al., 2007); however, Baeza et al. (2010) found no significant correlation between antimony bioaccumulation in *C. fluminea* and elevated antimony levels in sediments or lower trophic organisms.

Reliable bioaccumulation studies for antimony are limited (Gal et al., 2007; Heier et al., 2009). In one study, Heier et al. (2009) studied time-dependent accumulation in brown trout residing in stream waters that exhibited elevated antimony concentrations (2 to 3 µg/L) due to runoff from a shooting range. After 23 days of exposure, antimony accumulation was identified on the gills and in liver tissue taken from exposed fish. Measured antimony concentrations ranged from 0.01 to 0.08 µg/L in gill, and 0.0 to 0.07 µg/L in liver, suggesting antimony accumulation in brown trout is neither significant nor tissue specific. In another study, Gal et al. (2007) studied antimony bioavailability in earthworms near a former antimony mining and smelting site where total antimony concentrations in sediment ranged from 10 to 1,200 mg/kg. In the two species of earthworm studied, measured antimony concentrations ranged from 0.72 to 26 mg/kg; the bioconcentration factors (*i.e.*, the ratio of antimony concentrations in biota to that in soil) were less than one, suggesting minimal concern for antimony bioconcentration in these organisms. Additional information on antimony bioaccumulation and bioconcentration can be found in Table 2-4 and Table 2-5.

Table 2-4. Antimony Bioconcentration in Terrestrial Plants

Terrestrial Plants Species[a]	
Maize Zea mays	BCF: 0.186
Sunflower Helianthus annuus	BCF: 0.266
[a]EC/HC (2010). BCF = bioconcentration factor	

Table 2-5. Bioconcentration and Bioaccumulation of Antimony in Aquatic Species

Salt Water Species[a]		Fresh Water Species[b]	
Fish	BCF[c]: 0.15 (Two spot goby); BCF: 0.40 (Shanny)	BAF[c]: 0.43 (*Labeobarbus kimberleyensis*)	
Aquatic invertebrate	No data	BAF[c]: 5.65 (*Hyalella azteca*)	
Algae	BCF[c]: 17.00 (Coralline or red algae)	BCF[c]: 24.00 (*Chlorella vulgaris*)	
[a]EPA (2012b). [b]EC/HC (2010). [c]BAF = bioaccumulation factor; BCF = bioconcentration factor			

Considerable uncertainty may be associated with the application of metal bioaccumulation (or bioconcentration) factors, as these values can be highly variable (EPA, 2007). This may reflect differences in metal speciation and the related effects on bioavailability and/or complex toxicokinetic mechanisms that influence body burden (*e.g.*, uptake, metabolism, distribution, elimination). Saturable uptake and storage mechanisms, and the innate ability to regulate bioaccumulated metal concentrations within a certain physiological range may partially explain the inverse relationship that is often reported between bioaccumulation (or bioconcentration) factors and metal exposure concentrations (McGeer et al., 2003).

2.3.3 Conclusion of Environmental Fate

ATO is an environmentally persistent metalloid that exhibits low bioaccumulation potential. Generally speaking, pentavalent antimony is the thermodynamically favored under aerobic conditions in the environment; however, trivalent antimony predominates under anaerobic conditions. Antimony is primarily removed from the atmosphere via wet and dry deposition. Antimony generally exhibits limited solubility and slow dissolution kinetics in soil, therefore it is expected to exhibit limited bioaccessibility. However, mobility in soil and sediment may be influenced by site-specific environmental parameters (*e.g.*, matrix composition, pH; redox potential) which can significantly impact antimony speciation and bioavailability to ecological organisms.

3 ENVIRONMENTAL ASSESSMENT

3.1 INTRODUCTION

The environmental assessment evaluated ecological exposures that may be associated with ATO use as a flame retardant synergist.

3.1.1 Environmental Exposure Assessment

Exposure characterization focused specifically on industrial sectors reporting ATO production or use as a flame retardant synergist. The strategy used for the assessment included: 1) a refinement of the release data reported in the 2010 TRI to identify a subset of ATO production or use facilities, 2) collection of monitoring data for water, sediment, and soil from the states in which these facilities are located, and 3) modeling of TRI releases to provide site specific estimates of environmental exposure concentrations.

3.1.2 Toxics Release Inventory

To identify the selected TRI facilities that may use or produce ATO-containing flame retardants, EPA/OPPT examined the primary North American Industry Classification System (NAICS) codes reported in the 2010 TRI for antimony compounds.[2] The NAICS codes descriptions include a general heading and several subcategories that were used to screen TRI facilities for inclusion in the exposure assessment. ATO is not specifically listed on the TRI, but is reported under the broader category of antimony compounds, which covers all chemicals that contain antimony. EPA/OPPT could not distinguish ATO releases from those of other antimony-containing chemicals reported on the TRI; therefore, exposure characterizations were based on the conservative assumption that all releases of antimony compounds reported in the 2010 TRI reflect ATO releases associated with its use as a synergist in halogenated flame retardants.

A total of 270 facilities reported releases of antimony compounds to the TRI in 2010 Table 3-1. The majority of reported environmental releases were to land and other disposal media (which included on- and off-site transfers, landfill, underground injection, land treatment, surface impoundments, and other transfers for off-site disposal). The on- and off-site land and other disposal of antimony compounds totaled approximately 970,000 lbs, of which, approximately one-third was associated with on-site land releases from nine facilities (NAICS code 325188 - All Other Basic Inorganic Chemical Manufacturing), which are subject to Subtitle C regulation of hazardous materials under the Resource Conservation and Recovery Act.

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[2] A full description of the 6-digit NAICS codes is available from the Department of Census (http://www.census.gov/eos/www/naics/).

The on-site air releases of antimony compounds totaled approximately 12,000 lbs, of which approximately two-thirds were from stack releases, and one-third were from fugitive air emissions. A significant portion of these releases likely resulted from the combustion of fossil fuels; however, the precise quantity cannot be determined from TRI data.

The 2010 TRI water releases reported for antimony-containing compounds totaled approximately 15,700 lbs. Of this total, approximately 3,100 lbs were associated with direct surface water discharges, and approximately 12,600 lbs were transferred to Waste Water Treatment Plants (WWTPs) or other off-site water treatment facilities. Table 3-1 summarizes TRI information on environmental releases reported by industry sectors identified under the scope of this assessment (i.e., those using NAICS reporting codes that indicate production or use of ATO-containing flame retardants). No other environmental release data were identified for this use scenario.

Table 3-1. Summary of 2010 TRI Releases of Antimony Compounds

Life Cycle Stage	NAICS		Total Number of Facilities Reporting	On-site and Off-site Releases and Disposal (lbs/yr)[a]		
	Primary NAICS Code	Description		Total Air Releases	Total Water Releases [b]	Total Land and Other Disposal
Manufacture of antimony oxides	325188	All other basic inorganic chemical manufacturing	9	2,106	573	364,113
Plastics material and resin, synthetic rubber, and fiber manufacturing	325211	Plastics material and resin manufacturing	27	884	4,939	47,394
	325212	Synthetic rubber manufacturing	2	85	26	2,497
	325222	Non-cellulosic organic fiber manufacturing	5	408	186	700
Plastics compounding and conversion	325991	Custom compounding of purchased resins	55	1,231	540	81,014
	326113	Unlaminated plastics film and sheet (except packaging) manufacturing	18	339	110	7,078
	326121	Unlaminated plastics profile shape manufacturing	2	3	0	4,648
	326130	Laminated plastics plate, sheet (except packaging), and shape manufacturing	2	10	1	3,229
	326150	Urethane and other foam product (except polystyrene) manufacturing	3	0	0	0
	326199	All other plastics product manufacturing	25	330	10	68,625
Rubber compounding and conversion	326220	Rubber and plastics hoses and belting manufacturing	8	14	44	20,876
	326291	Rubber product manufacturing for mechanical use	3	30	0	9,106

Life Cycle Stage	NAICS		Total Number of Facilities Reporting	On-site and Off-site Releases and Disposal (lbs/yr)[a]		
	Primary NAICS Code	Description		Total Air Releases	Total Water Releases [b]	Total Land and Other Disposal
	326299	All other rubber product manufacturing	16	1,163	33	24,298
Fabric Finishing	313210	Broadwoven fabric mills	2	60	0	2,058
	313230	Nonwoven fabric mills	1	0	2,167	11,328
	313311	Broadwoven fabric finishing mills	4	53	1,139	2,707
	313312	Textile and fabric finishing (except broad woven fabric) mills	1	669	669	6,025
	313320	Fabric coating mills	15	2,893	4,927	88,465
Fiberglass insulation manufacturing	327993	Mineral wool manufacturing	3	0	0	4,765
Drawing and insulating of wires and fiber optic cables	331422	Copper wire (except mechanical) drawing	23	844	90	60,626
	335921	Fiber optic cable manufacturing	2	0	0	4,815
	335929	Other communication and energy wire manufacturing	38	3	25	101,160
Plastic-containing electronic/electrical product manufacturing	335931	Current-carrying wiring device manufacturing	5	255	5	17,733
Textile product mills and apparel manufacturing	314110	Carpet and rug mills	3	255	207	1,942
	314999	All other miscellaneous textile product mills	1	0	0	31,415

[1] See: supplemental data file titled "Supplemental 2010 TRI Data for the TSCA Work Plan Chemical Risk Assessment of ATO (External Review Draft).xlsx"
[a] Mass in lbs/yr can be converted to kg/yr by multiplying lbs/yr by 0.4536 kg/lb (numbers are rounded to the nearest integer).
[b] Includes transfers to WWTPs and other off-site wastewater treatment facilities.

3.1.3 Measured Concentrations in the Environment

EPA/OPPT developed subsets of environmental monitoring data from targeted searches of the USGS-NWIS and EPA's STORET databases. Approximately 8,500 water, sediment and soil samples were collected from these sources; data collection focused on the ten states in which selected TRI facilities (*i.e.*, those reporting use or production of ATO-containing flame retardants) were located.

3.1.3.1 Compilation of Environmental Monitoring Data from USGS-NWIS Database

Environmental monitoring data were downloaded from the USGS-NWIS website (http://nwis.waterdata.usgs.gov/usa/nwis/qwdata, accessed in July 2012). A subset[3] of the entire dataset, which spanned more than 30 years (*i.e.*, 1987 to 2011), was used to evaluate environmental exposures.

The subset used for exposure characterization was limited to monitoring samples obtained from states that included at least one of the TRI facilities identified under the scope of this assessment (i.e., AL, IN, KY, NC, NJ, NY, SC, TN, TX, and VA). These monitoring sites are shown in Figure 3-1. The USGS dataset was further culled to focus specifically on water, sediment, and soil samples obtained from these regions. The available monitoring data were split into two groups: (1) measured antimony concentrations from all years, and (2) measured antimony concentrations from 2009 to 2012.

3.1.3.2 Compilation of Environmental Monitoring Data from the STORET Database

Another large source of environmental monitoring data was obtained from EPA's STORET database (http://www.epa.gov/storet/dw_home.html; accessed July 2012). The STORET dataset included measured antimony concentrations in water and sediment only and was limited to samples taken from states that included at least one of the TRI facilities identified in Table_Apx C-1. The STORET data were divided into two categories: (1) concentrations measured in water (µg/L), and (2) concentrations measured in sediment (mg/kg dw). An attached file has a supplemental dataset[4].

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[3]See attached file titled "Supplemental subset of USGS-NWIS Dataset for the TSCA Work Plan Chemical Risk Assessment of ATO (External Review Draft).xlsx"

[4] See attached file titled "Supplemental subset of STORET Dataset for the TSCA Work Plan Chemical Risk Assessment of ATO (External Review Draft).xlsx"

Legend:
- ▲ TRI Facilities with EFAST Conc.
- • All USGS Locations
- ▪ All STORET Locations
- ▨ States not included in this evaluation

Antimony Water and Sediment Sample Location Map
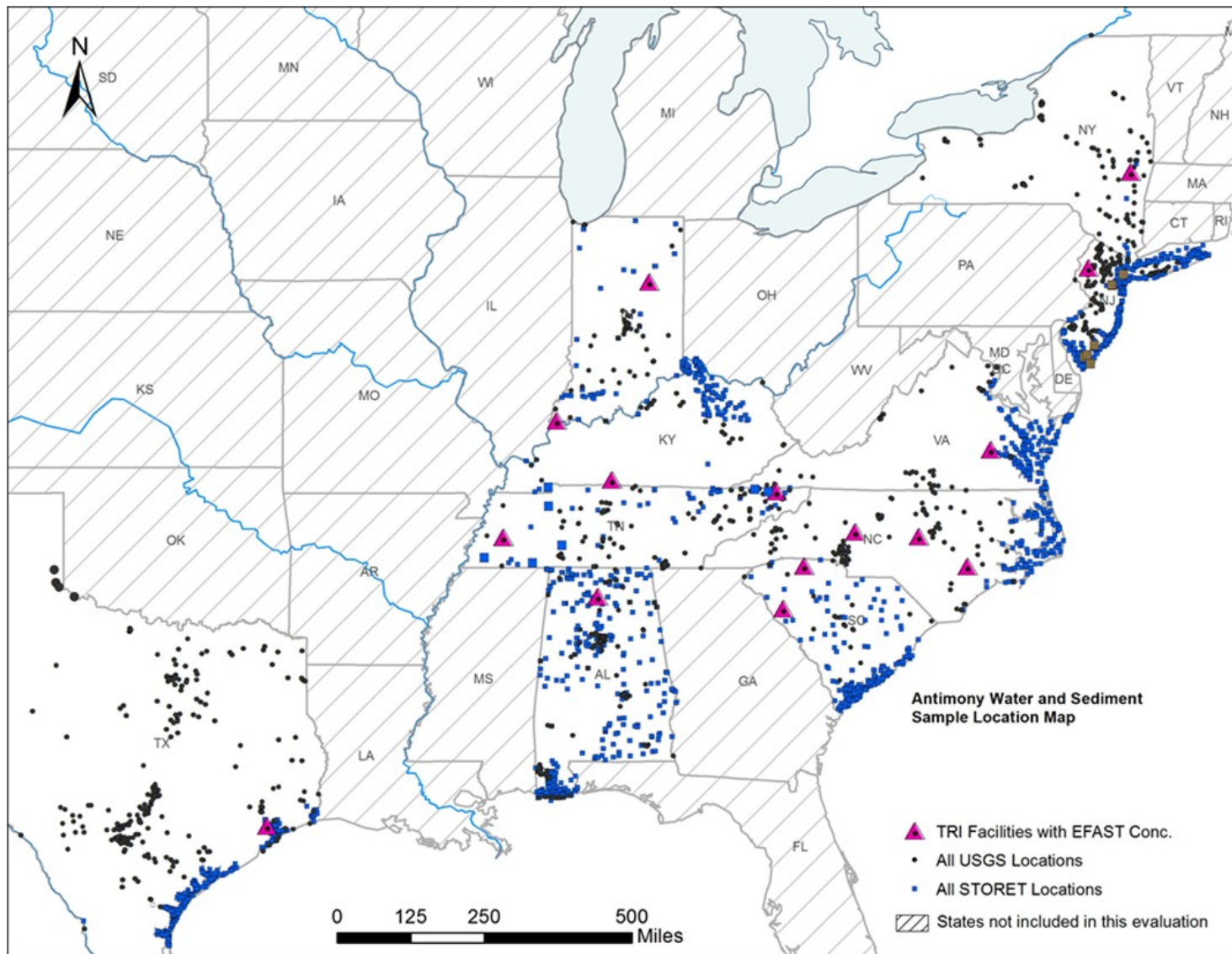
Scale: 0 — 125 — 250 — 500 Miles

Figure 3-1 Geographic Locations of TRI Facilities and Environmental Monitoring Data

3.1.3.3   Location of Monitoring Sites Relative to TRI Facilities

Upon retrieval and analysis of the monitoring data, it became clear that none of the actual monitoring sites were located within the primary watershed of the identified TRI facilities. Hence, concentrations of antimony found in water, sediment or soils from the USGS or STORET monitoring programs could not be clearly associated with ATO production or processing. Therefore, EPA/OPPT estimated environmental exposure concentrations from TRI facilities using the modeling approach described in Section 3.2.3.

3.1.4   Estimated Environmental Releases

Having found that available environmental monitoring data were not associated with industry sectors identified under the scope of this assessment (*i.e.*, those using NAICS reporting codes that indicate production or use of ATO-containing flame retardants), EPA/OPPT used domestic release data reported in the 2010 TRI to estimate ATO surface water concentrations.

3.1.4.1   EFAST Modeling

EPA's Exposure and Fate Assessment Screening Tool, Version 2.0, (E-FAST2) was developed to support EPA assessments of potential environmental exposures. For exposure characterization, the E-FAST2 model was used to estimate ATO surface water concentrations based on water releases reported by selected TRI facilities. The exposure scenarios included in the E-FAST2 model contain default parameter values that allow for exposure estimation with minimal data entry. Because E-FAST2 incorporates defaults that encompass either a combination of upper percentile and mean exposure parametric values, or all upper percentile parametric values, the resulting model predictions represent high-end exposures estimates. EPA/OPPT acknowledges the conservative nature of this approach.

E-FAST2 includes a Probabilistic Dilution Model (PDM) which predicts the number of days per year in which a designated exposure, or effect level (*i.e.*, COC) will be exceeded in ambient waters as a result of chemical discharges (effluents) released from a given facility. PDM analyses can be performed on stream reaches with measured flow data or stream reaches that incorporate estimated streamflow values. The PDM model provides chronic risk estimates that are derived from a simple mass balance approach of chemical dilution/emulsion into stream water; however, the input parameters are not single point estimates.

In reality, streams exhibit highly variable seasonal flow patterns. In addition, manufacturing processes include various operating procedures that can change intermittently, thereby affecting effluent flow rates and the total amount of chemical released to the environment over a given time interval. The PDM incorporates probability distributions from Monte Carlo simulations as analysis inputs for calculating the resulting probability distribution for the chemical concentration that may be seen in stream waters. Ultimately it predicts the number of days per year in which the modelled stream concentrations are expected to exceed the

designated effect levels (*i.e.*, COCs) identified for aquatic organisms based on the total amounts of chemical released per day. (For more information, see the user guide/background document for EFAST 2 at this web address: http://www.epa.gov/oppt/exposure/pubs/efast2man.pdf).

A summary of the input data used for the E-FAST2 model and a summary table which includes site specific input parameters is provided in Appendix B.

3.1.4.2   Criteria for Selection of TRI Facilities

Three selection criteria were used to identify the TRI facilities included in this exposure characterization: (1) reported water releases of 'antimony compounds' above 1 lb/yr; (2) reported NAICS codes that indicate production or use of ATO as a flame retardant synergist; and (3) stream releases for which the lowest streamflow data, or "7Q10" value (i.e., the average consecutive seven-day streamflow data calculated over a 10-year period) was available. The 7Q10 streamflow data are linked to the National Pollution Discharge Elimination System (NPDES) codes for each facility, which were found using EPA's Discharge Monitoring Report (DMR) Pollutant Loading Tool (available at http://cfpub.epa.gov/dmr/error_correction.cfm).

Using this approach, EPA/OPPT identified 36 TRI facilities TRI, 2010) that reported water releases of antimony compounds and listed primary NAICS codes associated with ATO use as a synergist in halogenated flame retardants; 33 of these indicated releases >1 lbs/yr. The NPDES numbers matching the reported latitudes and longitudes were found for 14 TRI facilities providing site-specific streamflow rates for use in the E-FAST2 model. The geographic location of these TRI facilities are shown in Figure 3-1 (red triangles).

3.1.4.3   Release Scenarios used for Exposure Characterization

Because TRI facilities do not indicate the total number of release days per year (i.e., a required input for E-FAST2 modeling), two hypothetical release scenarios were modeled (assuming total TRI releases of 'antimony compounds' reported during the 2010 reporting cycle occurred over a time period spanning 24 or 250 days per year). The first scenario (i.e., assuming total releases occurred over a period of 250 days) was developed based on a typical operating schedule which includes 260 work days and ten vacation days/holidays during the calendar year The more conservative scenario (*i.e.*, assuming total releases occurred over a period of 24 days), was developed based on a hypothetical situation wherein a facility might incur intermittent releases as a result of cleaning practices that span a 24-day time period. Both hypothetical release scenarios were used to estimate surface water concentrations based on E-FAST2 model results for water releases reported by the 14 TRI facilities evaluated in this assessment. These hypothetical release scenarios were developed to provide an upper and lower bound for the range of estimated surface water concentrations predicted by the E-FAST2 model. The results of this exposure assessment, which were used to assess risks to aquatic organisms, are provided in Table 3-2.

Table 3-2 Summary of E-FAST2 Model Results for TRI Facilities with Assumptions for Daily Release Amounts[1]

Facility Name	Latitude	Longitude	NPDES Code	Total Surface Water Discharge (lbs)	Scaled Annual Release (kg)	Results 250 Days (kg)	Results 24 Days (kg)
Eastman Kodak Co., Kingsport, TN 37662	36.519823	-82.540429	TN0002640	1,042	557.22	2.23	23.22
Guilford East Plant, Kenansville, NC, 28349	35.021587	-77.859994	NC0002305	669	357.75	1.43	14.91
Auriga Polymers Inc., Spartanburg, SC, 29307	35.030528	-81.861366	SC0002798	320	171.12	0.68	7.13
Sabic Innovative Plastics US LLC, Selkirk, NY, 12158-0145	42.575416	-73.853563	NY0007072	250	133.69	0.53	5.57
Sabic Innovative Plastics LLC, Mount Vernon, IN, 47620-9364	37.906389	-87.927778	IN0002101	250	133.69	0.53	5.57
Sage Automotive Interiors, Abbeville, SC, 29260	34.167222	-82.371389	SC0000353	248	132.62	0.53	5.53
Performance Fibers Operations Inc., Salisbury, NC, 28147	35.720234	-80.610867	NC0004944	95	50.80	0.20	2.12
Dupont Teijin Films, Chester, VA, 23836	37.346259	-77.285811	VA0003077	51	27.27	0.11	1.14
Haywood Co Brownsville, TN, 38012-1797	35.605911	-89.240068	TN0041939	28	14.97	0.06	0.62
Bayshore Industrial, La Porte, TX, 77571	29.63038	-95.030311	TX0022799	13.505	7.22	0.03	0.30
Covalence Adhesives, Franklin, KY, 42134	36.753889	-86.575278	KY0074659	5	2.67	0.01	0.11
General Cable Industries Inc., Marion, IN, 46953-2406	40.55425	-85.654944	IN0002372	5	2.67	0.01	0.11
Borealis Compounds Inc., Port Murray, NJ, 07865	40.813361	-74.888111	NJ0028657	5	2.67	0.01	0.11
Cerro Wire LLC, Hartselle, AL, 35640	34.412913	-86.917395	AL0054674	5	2.67	0.01	0.11

[1]Screening-level estimates are based on a range of days of release, which may not represent actual releases or environmental concentrations at these facilities.
[2] Mass in lbs/yr can be converted to kg/yr by multiplying 'lbs/yr' by 0.4536 kg/lb (numbers are rounded to the nearest integer).
NPDES = National Pollution Discharge Elimination System

3.2 ECOLOGICAL HAZARD ASSESSMENT

The ATO hazard assessment was based on readily available ecotoxicity information, including published studies and studies evaluated in previous risk assessments. Much of the information summarized in this section was obtained from the European Union Risk Assessment Report for Diantimony Trioxide (EC, 2008), and the Environment Canada Screening Assessment for the Challenge - Antimony Trioxide (EC/HC, 2010). This information was supplemented by subsequent searches of the peer-reviewed literature (from 1978 to 2012) using EPA's ECOTOX database to identify additional information from journal articles which were subsequently retrieved and evaluated for data adequacy. Data adequacy criteria include, but are not limited to: test substance identification, number of organism in each dose/concentration group, dose/concentration level, route/type of exposure, duration of exposure, species, controls (for additional details, see EPA, 1999. Studies conducted according to established EPA or OECD Guidelines as well as studies using other protocols were included if they met data adequacy criteria.

Due to the limited experimental data available for ATO [Sb_2O_3], antimony trichloride [$SbCl_3$] and antimony sulfate [$Sb_2(SO_4)_3$] were used as supporting chemicals to assess hazards to aquatic and sediment-dwelling organisms. The amount of time necessary to dissolve and liberate antimony ions capable of exerting adverse effects on ecological organisms is an important consideration for toxicity testing, as this parameter will vary for different antimony compounds. Antimony trichloride dissolves more readily (*i.e.*, the liberation of antimony ions will be slower for ATO); however, this is not expected to significantly impact testing results as the total amounts of antimony issued upon dissolution are expected to be comparable for both compounds (EC, 2008).

The majority of the available ecotoxicity studies utilized trivalent antimony compounds for testing, but did not include specific information on antimony speciation measurements. Since antimony is oxidized under aerobic conditions, the results of toxicity testing with trivalent antimony likely reflect exposure to a mixture of trivalent and pentavalent antimony. Because there is no conclusive scientific evidence to support a valence-specific differential for antimony-related ecological effects, no further distinction between antimony valence forms was made when evaluating hazard potential.

Acceptable studies were identified for aquatic and benthic organisms; however, a systematic review of the published literature revealed a relative paucity of antimony toxicity and exposure data for soil. Because the available environmental exposure information was inadequate to support a quantitative assessment of risks to soil-dwelling organisms, this environmental compartment was not evaluated in the current assessment. This conclusion is supported by the Ecological Soil Screening Levels (Eco-SSL) document for Antimony prepared by EPA's Office of Solid Waste and Emergency Response (EPA, 2005).

A summary of the available ecotoxicity data for ATO that were deemed adequate for consideration in this assessment are provided in Table 3-3, Table 3-4, Table 3-5 and the studies described in more detail in the appropriate section below.

Application of uncertainty factors based on established EPA/OPPT methods were used to calculate lower bound effect levels (referred to as the concentration of concern; COC) that would likely

encompass more sensitive species not specifically represented by the available experimental data. Uncertainty factors are included in the COC calculation to account for differences in inter- and intraspecies variability, as well as laboratory-to-field variability. These uncertainty factors are dependent upon the availability of datasets that can be used to characterize relative sensitivities across multiple species within a given taxa or species group, but are often standardized in risk assessments conducted under TSCA, since the data available for most industrial chemicals is limited (Ahlers et al., 2006; EPA, 2012d; EPA, 2013).

3.2.1 Acute Toxicity to Aquatic Organisms

Acute aquatic toxicity studies considered for this assessment are summarized in Table 3-3.

EPA/OPPT evaluated the available acute toxicity studies for fish, invertebrates, and plants; antimony trichloride ($SbCl_3$) was used as the test substance for most laboratory toxicity tests, whereas ATO (Sb_2O_3) was used in one plant study. The environmental effect levels provided below are based on measured antimony concentrations reported in these laboratory studies.

Acute toxicity to fish species was evaluated using $SbCl_3$ with red sea bream (*Pagrus major*) (Takayanagi, 2001). Juvenile sea bream (approximately 2.0 g in weight) were exposed to varying concentrations of either selenium and $SbCl_3$ or selenium and $SbCl_5$ for 96 hours under static exposure conditions. A measured 96-hr LC_{50} of 6.9 mg Sb/L was reported by the study authors. Though this study is considered to be reliable, uncertainties regarding antimony speciation and the specific test concentrations used for toxicity evaluations (*i.e.*, only a range of test concentrations was provided in the report), preclude its use for COC derivation.

For aquatic invertebrates, a 96-hr static test with $SbCl_3$ in hydra (*Chlorohydra viridissima*) was conducted in duplicate for each test concentration evaluated using five organisms per replicate. Measured antimony concentrations were 0.0, 1.11, 1.50, 2.54, 4.59, and 5.39 mg Sb/L, resulting in an LC_{50} value of 1.77 mg Sb/L (95% C.I. = ± 0.7 mg Sb/L; TAI, 1990).

Two studies using the algal species (*Pseudokirchneriella subcapitata*) were conducted using either $SbCl_3$ or Sb_2O_3. Heijerick and Vangheluwe (2004) exposed this green algal species to the following measured concentrations of $SbCl_3$: < 0.001 (control), 1.22, 2.11, 4.00, 6.73, 12.2, 21.9, 36.6, and 17.3 mg Sb/L for 72 hours under static exposure conditions according to the OECD 201 test guidelines. The starting cell density was 1 X 10^6 cells/mL, with three replicates per test concentration and six replicates per control. The LISEC (2001), which also adhered to OECD 201 test guidelines, exposed *P. subcapitata* to Sb_2O_3 (purity 98 %) at the following measured concentrations: 0, 0.074, 0.156, 0.323, 0.396, 1.32, 2.4 mg Sb/L. The starting cell density was 1 X 10^4 cells/mL, with three replicates per test concentration and six replicates per control. In both studies, biomass, growth, and growth rate were determined for control and treated groups at 24, 48, and 72 hours after test initiation.

Growth rate was determined to be the most relevant endpoint (*i.e.*, ErC_{50}; the effective concentration wherein 50 percent of tested organisms experienced a decline in growth rate), with the geometric mean of the maximum allowable toxicant concentration (MATC) determined to be 2.91 and 0.723 mg Sb/L for $SbCl_3$ and Sb_2O_3, respectively (Heijerick and Vangheluwe,2004; LISEC (2001). The LISEC (2001)

study is considered unreliable because the ErC_{50} value could not be determined due to uncertainties regarding the dose response relationship (*e.g.*, the limit dose induced minimal inhibition of growth rate, which is the most responsive endpoint); this value only decreased by three percent at the highest concentration tested (EC, 2008). Though the data reported by Heijerick and Vangheluwe (2004) are considered to be reliable (*i.e.*, based on the OECD guidelines for the testing of chemicals - Freshwater Algae and Cyanobacteria, Growth Inhibition Test), the reported ErC_{50} value (> 36.6 mg Sb/L) indicates that this algal species is not particularly sensitive to the effects of $SbCl_3$ exposure.

The acute COC for aquatic organisms was derived from the 96-hr LC_{50} of 1.77 mg Sb/L obtained using $SbCl_3$ in the aquatic invertebrate, *Chlorohydra viridissima*. This value was divided by an assessment factor (uncertainty factor) of five for invertebrate species to yield an acute COC of 0.354 mg Sb/L (354 µg Sb/L) in accordance with established EPA/OPPT methods (EPA, 2012d; 2013). Uncertainty factors are included to account for differences in inter- and intraspecies variability, as well as laboratory-to-field variability.

Table 3-3. Aquatic Toxicity Data for Sb_2O_3 and $SbCl_3$ – Acute Toxicity

Test Organism	Fresh/ Salt Water	Test Guideline/ Study Type	Duration (hr-hour)	End-point	Concentration (mg/L)	Antimony Compound	Effect	Reference
Fish - Saltwater								
Red Sea Bream *Pagrus major*	Salt	[a] Static	96-hr	LC_{50}	6.9	$SbCl_3$	Mortality	Takayanagi, 2001
Aquatic Invertebrates - Freshwater								
Hydra *Chlorohydra viridissima*	Fresh	[a] Static	96-hr	LC_{50}	1.77	$SbCl_3$	Mortality	TAI (1990)
Water flea *Daphnia magna*	Fresh	OECD TG 211 Semi-static	21-day	NOEC	1.74	$SbCl_3$	Reproduction	Heijerick and Vangheluwe (2003b)
				LOEC	3.13			
				MATC	2.33			
Aquatic Plants - Freshwater								
Algae, *Pseudo-kirchneriella subcapitata*	Fresh	OECD 201 Static	72-hr	EC_{50}	> 36.6	$SbCl_3$	Growth	Heijerick and Vangheluwe (2004)
				NOEC	2.11			
				LOEC	4.00			
				MATC	2.91			
Algae, *Pseudo-kirchneriella subcapitata*	Fresh	OECD 201 Static	72-hr	NOEC	0.396	Sb_2O_3	Growth	LISEC (2001)
				LOEC	1.32			
				MATC	0.723			

Note: The shaded row indicates the principal study used to assess acute risks to aquatic organisms.
[a] Guideline not reported

3.2.2 Chronic Toxicity to Aquatic Organisms

Chronic aquatic toxicity studies considered for this assessment are summarized in Table 3-4.

EPA/OPPT evaluated chronic toxicity studies with Sb_2O_3 and $SbCl_3$ in fish and aquatic invertebrates. No reliable chronic toxicity studies were identified for aquatic plants; therefore, acute toxicity studies were considered for this endpoint.

In a 30-day chronic toxicity study with $SbCl_3$ in the fathead minnow (*Pimephales promelas*), larvae were exposed under continuous flow-through conditions from pre-hatch (*i.e*., 16 to 40 hours post-fertilization) until 28 days post-hatch. Growth was determined to be the most responsive endpoint (Kimball, 1978). The study authors reported a measured MATC value of 1.62 mg Sb/L, which was used to calculate a chronic COC for aquatic organisms in the current assessment. In a separate 30-day chronic toxicity study, *P. promelas* were exposed to Sb_2O_3 at measured concentrations of 0 (control) 0 (positive control), 0.62, 0.74, 1.4, 3.2, or 7.5 µg/L of Sb_2O_3 under continuous flow-through conditions. The study authors reported a no-observed effect concentration (NOEC) > 0.0075 mg Sb/L, based on growth (LeBlanc and Dean, 1984). This study was not selected for the chronic COC calculation due to uncertainties regarding the dose response relationship (*e.g*., the lack of observed effects at the highest concentration tested).

Heijerick and Vangheluwe (2003b) conducted a 21-day static-renewal test using $SbCl_3$ in an aquatic invertebrate (*Daphnia magna*; conducted according to OECD 211 test guidelines) with the following measured concentrations at day 0: < 2 (control), 55.6, 98.0, 180, 312, 577, 984, 1736, 3130, 5860, and 9964 µg Sb/L and at day 21: < 2 (control), 46.4, 90.5, 166, 300, 561, 948, 1708, 2990, and 5580 µg Sb/L. Test concentrations were renewed three times per week with one organism per vessel and ten replicates per test concentration. The 21-day *D. magna* NOEC, LOEC and MATC for $SbCl_3$ are 1.74 mg/L, 3.13 mg/L and 2.33 mg/L, respectively. Reproduction was the hazard endpoint identified in this study, yielding a MATC value of 2.33 mg Sb/L. Though this study was considered reliable, the data were not selected for COC derivation because the MATC for *P. promelas* indicates that of the species evaluated, fish are more sensitive.

The chronic COC for aquatic organisms was derived from the 30-day MATC for growth (1.62 mg Sb/L) determined using $SbCl_3$ in fish (*Pimephales promelas*; Kimball, 1978). This value was divided by an uncertainty factor of ten for fish species in accordance with established EPA/OPPT methods (EPA, 2012; 2013) to yield a chronic COC of 0.162 mg Sb/L (162 µg Sb/L).

Kimball (1978) was also used to determine a predicted no-effect concentration (PNEC) of 0.113 mg Sb/L in the EU RAR for ATO (EC, 2008) and the Environment Canada/Health Canada (2010) risk assessment for ATO; the NOEC value (1.13 µg Sb/L) was used to calculate the PNEC for ATO.

Table 3-4. Aquatic Toxicity Data for Sb_2O_3 and $SbCl_3$ – Chronic Toxicity

Test Organism	Fresh/ Salt Water	Test Guideline/ Study Type	Duration	End-point	Concentration (mg/L)	Antimony Compound	Effect	Reference
Fish-Freshwater								
Pimephales promelas	Fresh	Flow-through[a]	30-day	NOEC	1.13	$SbCl_3$	Growth (length)	Kimball (1978)
				LOEC	2.31			
				MATC	1.62			
Pimephales promelas	Fresh	Flow-through[a]	30-day	NOEC	> 0.0075	Sb_2O_3	Growth	LeBlanc and Dean (1984)
Aquatic Invertebrates - Freshwater								
Water flea *Daphnia magna*	Fresh	OECD TG 211 Semi-static	21-day	NOEC	1.74	$SbCl_3$	Reproduction	Heijerick and Vangheluwe (2003b)
Aquatic Plants - Freshwater								
Pseudo-kirchneriella subcapitata	Fresh	OECD 201 Static	72-hr	EC_{50}	> 36.6	$SbCl_3$	Growth	Heijerick and Vangheluwe (2004)
				NOEC	2.11			
				LOEC	4.00			
				MATC	2.91			
Pseudo-kirchneriella subcapitata	Fresh	OECD 201 Static	72-hr	NOEC	0.396	Sb_2O_3	Growth	LISEC (2001)
Note: The shaded row indicates the principal study used to assess chronic risks to aquatic organisms. [a] Guideline not reported								

3.2.3   Toxicity to Sediment-Dwelling Organisms

Toxicity studies for sediment-dwelling organisms considered for this assessment are summarized in Table 3-5.

Toxicity to sediment-dwelling organisms in freshwater systems has been investigated in three different species (EC, 2008). These test species included: (1) the bottom-dwelling amphipod (*Hyalella azteca*), which is a surface deposit and filter feeder, (2) the midge larvae (*Chironomus riparius*), which burrows and feeds in sediment and on detritus, and (3) the oligochaete (*Lumbriculus variegatus*), which feeds in sediment and the interstitial spaces located below the water-sediment interface (Heijerick and Vangheluwe, 2003a, 2005 a, 2005b).

Heijerick and Vangheluwe (2003a) exposed the amphipod, *H. azteca* to $SbCl_3$ at five test concentrations (ranging from 44 to 355 mg Sb/kg dw) in a sediment-water exposure system for a total of 28 days. Each test concentration included 13 replicates with ten amphipods each. Four replicates were used to evaluate the 28-day growth and survival endpoints, and eight replicates were used to evaluate treatment-related effects on survival and reproduction on day 35 and 42 (including growth). The remaining replicate was used to measure the nominal antimony concentration in sediment. After the initial exposure period, the amphipods were removed from the treated sediment and placed in sediment-free chambers for another 14 days (depuration). Effects on survival (day 28, 35, 42), growth (day 28, 42), and reproduction (*i.e.*, the number of offspring per female from day 28 to 42) were subsequently evaluated. Of the monitored effects, growth (*i.e.*, weight and length) was identified as the most responsive endpoint after 28 days of $SbCl_3$ exposure, resulting in a NOEC of 124 mg Sb/kg dw.

For the toxicity study with the oligochaete (*L. variegatus*), Heijerick and Vangheluwe (2005a) exposed adult organisms to $SbCl_3$ in a sediment-water exposure system for 28 days. Effects on survival, reproduction, and growth were subsequently evaluated at the end of this exposure period. The study was performed using six exposure concentrations (23 to 445 mg Sb/kg ww; 33 to 636 mg Sb/kg dw), a control group (< 1.4 mg Sb/kg ww), and seven replicates per test concentration. Six replicates were used to evaluate effects on survival, reproduction, and growth after 28 days of exposure; the remaining replicate was used to measure the nominal antimony concentration in sediment. Of the endpoints that were evaluated, growth (*i.e.*, weight) provided the lowest effect level, resulting in a NOEC of 112 mg Sb/kg dw.

Two-day-old midge larvae (*C. riparius*) were exposed to a range of $SbCl_3$ concentrations in a sediment-water exposure system until the midge larvae emerged into the adult phase (Heijerick and Vangheluwe, 2005b). Effects on larval mortality and growth, and subsequent emergence to midges were determined after 14 and 28 days of exposure. The study was performed with six concentrations (ranging from 33 to 636 mg Sb/kg dw) and a control group (< 2.0 mg Sb/kg dw). Eleven replicates were included for each test concentration, five of which were used to evaluate effects on survival and growth after 14 days of exposure. Another five replicates were used to determine the extent of midge emergence after 28 days of exposure, and the remaining replicate was used to measure the nominal antimony concentration in sediment. Of the

endpoints evaluated, growth (*i.e.*, weight), provided the lowest effect level, resulting in a NOEC of 112 mg Sb/kg dw.

To determine the chronic COC for toxicity to invertebrates in freshwater sediment, the NOEC for the growth endpoint (78 mg Sb/kg ww) was used as 112 mg Sb/kg dw (based on a 70/100 calculated ww/dw ratio). The effect levels reported in the Heijerick and Vangheluwe (2005a, b) studies with SbCl$_3$ exposure in the oligochaete (*L. variegatus*) and midge larvae (*C. Riparius*), were divided by an uncertainty factor of 10 to yield 11.2 mg Sb/kg dw as the chronic COC for sediment-dwelling organisms. The dry weight (dw) value was used for comparison with the environmental monitoring data for suspended sediment obtained from the USGS-NWIS database (reported as mg Sb/kg dw).

The Heijerick and Vangheluwe (2005a, b) studies were also used in the EU risk assessment to derive a PNEC value of 11.2 mg/kg dw; the EU derived a PNEC by utilizing the lowest NOEC value and then applying an uncertainty factor of 10 (EC, 2008).

Table 3-5. Chronic Toxicity Data for Sediment-Dwelling Invertebrates

Test Organism	Test Guideline/ Study Type	Duration	End-point	Concentration (mg Sb/kg dw)	Antimony Compound	Effect	Reference
Amphipod *Hyalella azteca*	EPA 600/R-99/064 Section 14: Test Method 100.4 Measured	42-day	NOEC	124	SbCl$_3$	Growth (Length)	Heijerick and Vangheluwe (2003a)
Oligochaete *Lumbriculus variegatus*	OECD "Bioaccumulation; Sediment test using benthic oligochaetes" and EPA 600/R-99/064 Section 13: Test Method 100.3 Measured	28-day	NOEC	112	SbCl$_3$	Growth (Weight)	Heijerick and Vangheluwe (2005a)
Midge larvae *Chironomus riparius*	OECD TG 218 Measured	28-day	NOEC	112	SbCl$_3$	Growth (Weight)	Heijerick and Vangheluwe (2005b)
Note: The shaded row indicates the principal study used for assessing chronic risks to sediment-dwelling organisms.							

3.2.4 Chronic Toxicity to Terrestrial Organisms

The chronic toxicity studies considered for soil-dwelling organisms are summarized in Table 3-6. EPA/OPPT identified chronic toxicity studies for soil-dwelling invertebrates. Several studies were available from secondary sources; however, only one study, by Moser (2007) was considered to be reliable. This study was described in the following excerpt taken from the EU RAR (EC, 2008):

"Moser (2007) studied the toxicity of antimony to the springtail, *Folsomia candida* (reproduction and mortality) using aged (approximately 32 w) Sb_2O_3-amended soil. Ten (10-12 day old) synchronized springtails were put onto 30 g moist soil in a glass vessel during a four-week exposure period. Five replicates were used for each test concentration and the measured test concentrations were 90, 322, 999, 2930, and 10119 mg Sb/kg soil dw. Additionally, the springtails were also tested using uncontaminated field soil and an untreated artificial soil as controls, based on OECD 207 guidelines. The positive control used on springtail reproduction was the herbicide Betosip (a.i. 159 g/L phenmedipham). At the end of the exposure period, the number of juveniles in each test vessel was counted after floating. The mortality of the springtails was also recorded. At day 28, the pH-value and the moisture of the artificial soil for each concentration in additional vessels without springtails were determined for both mortality and reproduction, the highest effect observed was below 50% in any of the treatments. Consequently, LC_{50} and EC_{50} values could not be calculated, but were estimated as >10,119 mg Sb/kg soil dw for each one of the endpoints. The NOEC for reproduction was approximately 999 mg Sb/kg dw; the LOEC for reproduction was approximately 2,930 mg Sb/kg dw. All reported values refer to measured concentrations."

The remaining studies were reviewed, but had deficiencies (or discrepancies) in study design and/or uncertainties due to inadequate or missing information.

Because the available exposure data are insufficient to assess ATO risks to soil-dwelling organisms, EPA/OPPT did not calculate a COC from the springtail study.

Table 3-6. Chronic Toxicity Data for Soil-dwelling Invertebrates

Test Organism	Test Guideline/ Study Type	Duration	End-point	Concentration (mg Sb/kg dw)	Antimony Compound	Effect	Reference
Springtail *Folsomia candida*	ISO 11267 Measured	28-day	NOEC	999	Sb_2O_3	Reproduction	Moser (2007)
			LOEC	2930			
Earthworm *Eisenia fetida*	ISO 11268-2 Nominal	21-day	NOEC	617	$Sb_2(SO_4)_3$	Adult survival	Simini *et al.* (2002)
			LOEC	697			
			NOEC	60		Juvenile production	
			LOEC	86			
Enchytraeid *Enchytraeus crypticus*	ISO 16387 Nominal	14-day	NOEC	384	$Sb_2(SO_4)_3$	Adult survival	Kuperman *et al.* (2002)
			LOEC	538			
		28-day	NOEC	100		Juvenile production	
			LOEC	140			
Springtail *Folsomia candida*	ISO 11267 Nominal	28-day	NOEC	100	$Sb_2(SO_4)_3$	Adult survival	Phillips *et al.* (2002)
			LOEC	126			
			NOEC	100		Juvenile production	
			LOEC	126			

3.2.5 Conclusions of Environmental Hazard Assessment

This assessment evaluated environmental risks to aquatic and benthic organisms in association with ATO use as a flame retardant synergist. The hazard characterization for aquatic organisms was based on an evaluation of acute and chronic toxicity endpoints. The COCs derived from the studies for aquatic organisms are presented in Table 3-7. Many of the reported aquatic toxicity values for ATO were considered unreliable either because they failed to meet EPA's adequacy criteria or exceeded the solubility limits of antimony at environmentally relevant pH levels. For this reason, all of the hazard benchmarks identified for aquatic organisms were derived from toxicity studies with antimony trichloride ($SbCl_3$).

Table 3-7. Concentrations of Concern for Environmental Toxicity

Toxicity	Concentration of Concern (COC)
Acute Toxicity, aquatic organisms	354 µg Sb/L
Chronic Toxicity, aquatic organisms	162 µg Sb/L
Acute Toxicity, sediment-dwelling organisms	112 mg Sb/kg dw
Chronic Toxicity, sediment-dwelling organisms	11.2 mg Sb/kg dw

The lowest effect level identified for the acute toxicity (mortality) endpoint, an LC_{50} value of 1.77 mg Sb/L reported in green hydra, *Chlorohydra viridissimus*, was used to derive the acute COC (354 µg Sb/L) for aquatic organisms. Reliable chronic toxicity studies were also available for antimony in fish, daphnia and algae. Of the aquatic species evaluated, fish appear to be more sensitive to chronic antimony exposure; the lowest effect level identified for chronic toxicity (1.62 mg Sb/L), derived from a study on *Pimephales promelas*, was used to derive chronic COC (162 µg Sb/L) for aquatic organisms. Acceptable chronic toxicity values are also available for three benthic species which exhibit similar hazard profiles; the lowest effect level (112 mg Sb/kg dw) was identified for the midge, *Chironomus riparius* and the oligochaete, *Lumbriculus variegatus*. A slightly higher hazard was identified for the amphipod *Hyalella azteca* (124 mg Sb/kg dw); impaired growth was determined to be the most sensitive chronic toxicity endpoint for the aquatic and benthic organisms included in this hazard characterization. The lowest effect level (112 mg Sb/kg dw) was used to derive the chronic COC (11.2 mg Sb/kg dw) for benthic organisms.

3.3   ENVIRONMENTAL RISK CHARACTERIZATION

The goal of the current assessment was to evaluate the environmental risks that may be associated with ATO use as a flame retardant synergist. Point estimates used to characterize environmental exposures were based on reported TRI releases and US monitoring data for water and sediment. These values were subsequently compared to the COCs identified for these environmental compartments to determine whether this use scenario may present unacceptable risks to water- and sediment-dwelling organisms.

The following assessment questions were addressed:

1. Do ATO concentrations measured in US surface waters pose risks for adverse effects in aquatic organisms?

2. Do ATO concentrations measured in US sediments pose risks for adverse effects in in benthic organisms?

3.3.1   Summary of TRI Modeling Results

ATO surface water concentrations were predicted using E-FAST2 to model water releases reported by selected TRI facilities in 2010. Because TRI does not indicate the number of days associated with reported releases, two generic release scenarios were developed (*i.e.*, assuming either 24 or 250 days of release) to provide a range of predicted water concentrations for comparison with hazard benchmarks (COCs) identified for aquatic organisms.

The Probabilistic Dilution Model (PDM), a module of E-FAST2 that predicts the number of days a stream concentration will exceed the designated COC value, was used for both release scenarios. The estimated 7Q10 concentrations (µg/L) and the predicted days of exceedance (as predicted by the PDM model), are shown in Table 3-8. The COC exceedance days refers to the total number of release days in a given year that the chronic COC would be exceeded using the surface water concentrations predicted using the E-FAST2 model.

Table 3-8. Summary of E-FAST2 Results for Facilities Reporting TRI Water Releases

Facility name	NPDES Code	250 days of release 7Q10 Concentration (ppb)	RQ	24 days of release 7Q10 Concentration (ppb)	RQ	PDM Days of Exceedance
Eastman Kodak Co TN Eastman Co Div, Kingsport, TN, 37662	TN0002640	1.15	0.003	11.99	0.03	0
Guilford East Plant, Kenansville, NC, 28349	NC0002305	62.07	0.18	674.14	1.9	3
Auriga Polymers Inc. Spartanburg Facility, Spartanburg, 29307	SC0002798	2.29	0.006	23.97	0.07	0
Sabic Innovative Plastics US LLC, Selkirk, NY, 12158-0145	NY0007072	0.08	0	0.81	0.002	0
Sabic Innovative Plastics Mt. Vernon, LLC, Mount Vernon, IN, 47620	IN0002101	0.02	0	0.25	0.001	0
Sage Automotive Interiors Abbeville Plant, Abbeville, SC, 29620	SC0000353	29.86	0.08	311.55	0.9	6
Performance Fibers Operations Inc. Salisbury Plant, Salisbury, NC, 28147	NC0004944	2.83	0.01	30.04	0.09	0

Facility name	NPDES Code	250 days of release		24 days of release		PDM
		7Q10 Concentration (ppb)	RQ	7Q10 Concentration (ppb)	RQ	Days of Exceedance
Dupont Teijin Films, Chester, VA, 23836	VA0003077	0.07	0	0.78	0.002	0
Haywood Co, Brownsville, TN, 380121797	TN0041939	0.03	0	0.33	0.001	0
Bayshore Industrial La Porte, TX, 77571[b]	TX0022799	2.54[b]	0.01	25.36[b]	0.07	NA
Covalence Adhesives, Franklin, KY, 42134	KY0074659	0.69	0.002	7.61	0.02	0
General Cable Industries Inc., Marion, IN, 46953	IN0002372	0.19	0	2.14	0.01	0
Borealis Compounds Inc., Port Murray, NJ, 07865	NJ0028657	0.09	0	0.96	0.003	0
Cerro Wire LLC, Hartselle, AL 35640	AL0054674	0.08	0	0.92	0.003	0

[a] These screening- level estimates are based on a range of days of release, which may not represent actual releases and environmental concentrations at these facilities

[b] Dilution factor of 1 used as standard practice for sites that discharge into lakes, bays, estuaries and oceans. This facility releases to Galveston Bay.

NPDES = National Pollution Discharge Elimination System; NA = Not Applicable

3.3.2 Summary of Environmental Monitoring Data

The environmental exposure assessment included a total of 8,500 environmental monitoring samples for water, sediment, and soil. Although this data could not be associated directly with ATO production or processing facilities, EPA/OPPT compared this ambient monitoring data to the ecotoxicity benchmarks derived in this assessment to gain a broader perspective on the potential risks of environmental antimony concentrations.

A total of 5,493 water samples were evaluated in the exposure assessment. Five of the 3,643 water samples (~ 0.4 percent) obtained from the USGS-NWIS database (collected from 1972 to 2012), exceeded the chronic COC for water (162 μg/L); seven of the 1,850 water samples (~ 0.4 percent) obtained from EPA's STORET database (collected from 2002 to 2010), exceeded the chronic COC for water (See Appendix C). The average antimony concentration reported for the 3,643 water samples obtained from the USGS-NWIS dataset was 4 μg/L, which is roughly 40 times lower than the COC for aquatic organisms.

A total of 2,985 sediment samples were evaluated in the exposure assessment. Fourteen of the 1,753 sediment samples (~ 0.9 percent) obtained from the USGS-NWIS database (collected from 1987 to 2011), exceeded the chronic COC for sediment (11.2 mg/kg dw) and six of the 1,222 sediment samples (~ 0.5 percent) obtained from the STORET database (collected from 2002 to 2010), exceeded the chronic COC for sediment (11.2 mg/kg dw; See Appendix C). There were 1,753 sediment samples in the USGS-NWIS dataset with an average antimony concentration of 1.3 mg/kg dw, which is approximately ten times lower than the chronic COC for sediment-dwelling organisms.

Only 32 soil samples were retrieved from the USGS-NWIS database between 2005 and 2010; all reported antimony concentrations were less than the minimum reporting levels for soil (*i.e.*, 1.2 to 1.7 mg/kg); no soil samples were available from the STORET database.

3.3.3 Conclusions of Risk Characterization

In summary, modeling results obtained for the TRI facilities evaluated under the scope of this assessment do not indicate risks to aquatic or benthic organisms. No exceedances were predicted using the 250 days/yr release scenario; antimony stream concentrations predicted by the EFAST-2 model using this release scenario were from two to 100,000 times lower than the COCs selected to be protective of organisms that dwell in the aquatic compartment. Stream concentrations predicted using the more conservative (*i.e.*, 24 days/yr) release scenario were also well below the COCs identified for aquatic organisms for most sites; only one exceedance of the acute and chronic COC was predicted. Because the number of exceedance days predicted by the model (*i.e.*, 3 and 6 days, respectively) is less than the time required to trigger a chronic risk concern (*i.e.*, 20 days), significant risks are not anticipated (EPA, 2012d).

EPA collected environmental monitoring data from two national datasets (USGS-NWIS and STORET) to assess environmental risks that may be associated with ATO use as a synergist in halogenated flame retardants. Although this exposure data cannot be directly attributed to ATO production or processing facilities, the analysis showed very few instances where antimony concentrations measured in the ambient environment exceeded (or came near) the conservative hazard benchmarks identified to be protective of aquatic and benthic organisms. Less than 0.01 percent of the environmental monitoring samples collected from the ten states covered under the scope of this assessment showed antimony concentrations that exceeded the COC for water or sediment. No exceedances were identified within the last three years (*i.e.*, from 2009 to 2012) for any environmental compartment evaluated in this assessment.

These results must be considered within the context of the conservative assumptions, uncertainties and data limitations that underpin this assessment, described briefly below.

3.3.4   Uncertainty and Data Limitations

The primary sources of uncertainty identified in this assessment are related to limitations in the available exposure and hazard data, the E-FAST2 model, and the parameters that support them, as discussed below.

3.3.4.1   Limitations of the Exposure and Release Data

The environmental fate and transport of antimony is complex, and can be influenced by site-specific environmental conditions. Surface environments may be enriched with antimony as a result of natural or anthropogenic events and over time, this material may be transformed in ways that can significantly impact the availability and potential toxicity to living organisms. Because environmental systems are dynamic, and may be influenced by site-specific parameters that exhibit significant spatial and temporal variation, it is difficult to extrapolate the current findings to other sites or geographic regions within the US.

The USGS-NWIS database is one of the largest environmental monitoring databases in the US; however, comprehensive information needed for data interpretation is not always readily available. In some instances, proprietary information may be withheld, or specific details regarding analytical techniques may be unclear, or not reported at all. As a result of all of these shortcomings, there are uncertainties in the reported data that are difficult to quantify with regard to impacts on exposure estimates.

The quality of the data provided in the USGS-NWIS and STORET datasets varies, and some of the information provided is non-quantitative. While a large number of individual sampling results were obtained from these datasets, the monitoring studies used to collect the data were not specifically designed to evaluate ATO distribution across the US. The available data represent a variety of discrete locations and time periods; therefore it is unclear whether the data are representative of other locations in the US; however, this limitation does not diminish the overall findings reported in this assessment, as the exposure data showed very few

instances (*i.e.*, less than 0.01 percent) where measured or predicted antimony levels in the ambient environment exceeded the identified hazard benchmarks for water or sediment-dwelling organisms. It is also important to note that the USGS-NWIS and STORET data did not align with the primary watersheds of the TRI facilities identified under the scope of this assessment, therefore no direct correlation can be made between them.

The TRI database represents a comprehensive source of environmental release data for the US; however, there are a few caveats to consider when using this information. TRI data are self-reported by facilities subject to minimum reporting thresholds; therefore, it may not capture releases from smaller facilities (*i.e.*, environmental releases may be underestimated). Some of the reported information may be inaccurate because it reflects approximations based on mass balances and emission factors, rather than actual emissions or release data. Use of TRI data to estimate environmental exposures is constrained by a number of uncertainties including: the heterogeneity of processes and releases among facilities grouped within a given sector; assumptions made regarding sector definitions used to select TRI facilities covered under the scope; lack of information regarding the distribution of specific antimony compounds reported in TRI; and fluctuations in the level of production and associated environmental releases incurred as a result of changes in standard operating procedures. Uncertainty may also arise from omissions in the TRI reporting data, such as sectors that are not required to report, facilities that fall below the reporting threshold, or facilities for which forms simply are not filed. While these limitations are important, their impact on estimating exposure potential may be less than that associated with the conservative assumptions made regarding environmental releases. ATO is reported in TRI under the broader category of 'antimony compounds', which covers all chemicals that contain antimony. As such, EPA/OPPT expects the actual releases of ATO to be significantly lower than those reported in TRI.

3.3.4.2   E-FAST2 Model Uncertainties

The major limitations associated with use of the E-FAST2 model relate to the assumptions made regarding missing information that was required for model input, including the pattern of TRI releases and the types of antimony compounds that are reflected in reported TRI releases. It is important to note however, that these limitations are unlikely to change the stated conclusions of this assessment, because they are based on a series of conservative assumptions that likely overestimate exposure potential.

3.3.4.3   Limitations of the Hazard Data

The laboratory studies used to identify COCs for ecological organisms varied in terms of the analytical methods used to measure antimony concentrations and the form of antimony used to assess toxicity. For example, aquatic COCs were derived from studies using $SbCl_3$, whereas the data used to assess ecological exposures reflect not only those associated with ATO, but all antimony-containing compounds, which likely overestimates exposure potential. Risks to soil-dwelling organisms were not evaluated; however, antimony is expected to present minimal concerns due to limited bioavailability in this environmental compartment.

3.4 CONCLUSIONS

This assessment used TRI data US monitoring data to characterize environmental exposures. There are very few reliable studies of antimony toxicity to terrestrial organisms. EPA (2005) reviewed the available hazard and exposure information and determined that it was inadequate for the development of an Ecological Soil Screening Level (Eco-SSL) for antimony. Due to the relative paucity of reliable hazard and exposure data for risk estimation, a weight of evidence approach was used to support the assumption of minimal risks to soil-dwelling organisms.

Despite the data limitations and uncertainties described above, significant risks to water- or sediment-dwelling organisms are not expected based on the following considerations:

- Use of antimony compounds as a surrogate for ATO and modeled exposure estimates based on the conservative assumption of total releases (during the 2010 TRI reporting cycle) occurring over a 24-day period likely overestimate exposure potential.

- Measured antimony concentrations from environmental monitoring samples collected within the last three years showed no exceedance of any of the calculated hazard benchmarks.

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APPENDICES

Appendix A   HUMAN HEALTH HAZARD SUMMARY

A-1   TOXICOKINETICS

Precise quantitative information regarding antimony absorption, distribution, and retention in humans is lacking; however, increased levels of antimony have been reported in blood and urine obtained from workers following ATO inhalation exposure, indicating that trivalent antimony may be absorbed from the lungs and excreted in the urine (Bailly et al., 1991). A renal elimination half-life of four days was estimated in 21 employees of a starter battery manufacturing plant who were exposed to ATO via inhalation (Kentner et al., 1995).

Systemic absorption of antimony compounds following inhalation exposure is mainly influenced by chemical form, particle size, and solubility. In general, large particles (≥ 2.5 µm) tend to deposit in the upper airway and are cleared more rapidly (via mucociliary transport) than smaller particles that can deposit in the lower lung, where clearance times depend more on solubility (i.e., less soluble particles will be cleared more slowly) (Brain et al., 1994).

Antimony may be retained in the lungs for long periods following inhalation exposure. Lung retention times measured in seven male workers accidentally exposed to radiolabeled aerosols of ATO (^{125}Sb) showed that most of the radiolabeled material was confined to the lungs, with negligible amounts detected in other parts of the body. Monitoring results based on personal and stationary air sampling methods from glass-making factories showed measured antimony concentrations in air between < 5 and 840 µg/m^3 (Lüdersdorf et al., 1987). The estimated elimination half-life for antimony varied significantly, ranging from 600 to 1,100 days in non-smokers and 1,700 to 3,700 days in smokers, suggesting that ATO lung clearance may be negatively impacted by smoking (Garg et al., 2003). Deceased smelter and refinery workers exposed to ATO primarily via inhalation, exhibited elevated lung concentrations when compared to non-occupationally exposed individuals (Gerhardsson et al., 1982); however, antimony concentrations measured in other organs (e.g., liver and kidney) were comparable, suggesting either a short biological half-life or limited systemic distribution for antimony in these tissues.

Laboratory animal data indicate low systemic absorption of ATO when administered via the oral route (De Bie and Salmon-te Rietstap, 2005; Felicetti et al., 1974). Therapeutic use of antimonial compounds for the treatment of parasitic diseases has also demonstrated poor systemic bioavailability following oral exposure in humans (WHO, 2003). Based on results reported by the International Commission on Radiological Protection, limited systemic absorption (~1%) may be assumed when estimating ATO uptake from the gastrointestinal tract (as cited in EC, 2008); dermal absorption (0.26 %) is negligible (Roper and Stupart, 2006). It has been suggested that a portion of systemic absorption observed following inhalation exposure is due to the clearance of large particles (via mucociliary transport) and subsequent uptake into the gastrointestinal tract (Brain et al., 1994). Using the Multiple Path Particle Deposition model

and predicted deposition patterns for inhaled ATO particles (4 μm), the total systemic absorption (reflecting combined uptake in the lungs and gastrointestinal tract) was estimated at approximately 7% (as cited in EC, 2008).

Once taken up into the systemic circulation, trivalent antimony accumulates in red blood cells and is distributed primarily to liver, spleen, and bone (Casals, 1972). It has been suggested that this pattern of distribution may reflect clearance from circulation via phagocytic macrophages located in these organs which effectively sequester non-endogenous molecules (Sarin, 2010).

A-2   ACUTE TOXICITY

The available laboratory animal data indicate low acute toxicity via oral, dermal, and inhalation routes (as cited in EC, 2008):

- oral LD_{50} >20,000 mg/kg-bw (rats)
- dermal LD_{50} >8,300 mg/kg-bw (rabbits)
- inhalation LC_{50} (4-hour) >5,200 mg/m^3 (rats)

A-2-1   Irritation

No adverse skin reactions have been reported in laboratory animals; however, workers have reported skin irritation following dermal exposure to ATO under conditions of intense heat and heavy sweating (NIOSH, 1978; Potkonjak and Pavlovich, 1983). It has been suggested that ATO may induce skin rashes (i.e., antimony spots) as a result of physical obstruction of the sweat ducts. Lung and eye irritation have also been noted with occupational exposure, but no causal relationship has been established. Although ATO is not classified as an irritant under the Registration, Evaluation, Authorization and Restriction of Chemicals, workers are encouraged to avoid excessive contact via use of personal protective equipment.

A-3   SUBCHRONIC TOXICITY

Repeated inhalation exposure has been shown to cause respiratory irritation in rodents (Groth et al., 1986; Newton et al., 1994; Potkonjak and Vishnjich, 1983). Although detailed occupational exposure data are lacking in the US, early case reports of adverse pulmonary effects (inflammation, pneumoconiosis, emphysema) and decreased lung function (Potkonjak and Vishnjich, 1983) indicate a potential for adverse respiratory effects in humans following repeated inhalation exposure. In addition, gastrointestinal disorders (vomiting, diarrhea), fatigue, dizziness, and headaches have been reported in workers heavily exposed to smelter fumes (NIOSH, 1978). It is important to note however, that working conditions have improved significantly over the past 30 years; a permissible exposure limit of 0.5 mg/m³ has been set to protect this population.

A-4   REPRODUCTIVE TOXICITY

A relative paucity of data exists regarding the potential for ATO-induced reproductive or developmental effects; however, a weight-of-evidence approach may be used to address these endpoints. Toxicokinetic considerations (low systemic absorption), together with the absence of effects on reproductive organs in repeated-dose studies suggest low concern for reproductive effects.

Limited information is available from a Russian report on ATO (Belyaeva, 1967), which indicated adverse reproductive effects in rats (e.g., ovarian cysts, uterine metaplasia, decreased fertility) following inhalation exposure at 250 mg/m^3 for 4 hours/day during a two-month period. These study authors also reported adverse reproductive outcomes (spontaneous abortions, premature births, gynecologic problems) in women following exposure to metallic dusts (of antimony, ATO, and antimony pentasulfide) at unspecified concentrations while working in a metallurgical plant. The reported results are largely inconclusive and inadequate for risk assessment due to inadequate or missing information regarding study design (e.g., lack of information on experimental conditions, controls, and purity of test substance) and exposure conditions.

No testicular effects were observed in male rats or mice following repeated gavage exposure to ATO at doses ≥1,000 mg/kg-bw/day (Gurnani et al., 1992; Omura et al., 2002). In addition, oral exposure to ATO during a 90-day feeding in rats study showed no effects on male reproductive organs up to the highest dose tested (1,686 mg/kg-bw/day; Hext et al., 1999). Gerber et al. (1982) used a readily soluble, radiolabeled compound, antimony trichloride (^{125}SbCl3), to determine the extent of uptake in pregnant mice following dietary exposure to a mixture of antimony chloride and arsenic throughout pregnancy. Minimal absorption (~7 %) was observed in treated dams, with limited distribution to reproductive organs (~0.1 to 0.2 % in ovaries and uteri). This finding is supported by another prenatal developmental toxicity study, in pregnant ewes dosed orally throughout gestation with antimony tartrate (a highly soluble form of antimony expected to exhibit higher absorption as compared to ATO) at 2 mg/kg-bw/day showed no adverse maternal or developmental effects (James et al., 1966).

A-5   DEVELOPMENTAL TOXICITY

In a prenatal developmental toxicity study, female Sprague-Dawley rats (26/dose) were exposed to ATO via (nose-only) inhalation at 0, 2.6, 4.4, and 6.3 mg/m^3 (mass median aerodynamic diameter [MMAD] = 1.59 to 1.82 μm) for 6 hours/day from the day of fertilization through day 19 of gestation. Significant increases in lung weights and an accumulation of ATO in erythrocytes were observed in all treated dams as compared to controls. In addition, histological examination revealed an accumulation of pigmented alveolar macrophages, and discrete areas of acute inflammation and hyperplasia in the lungs. An increase in post-implantation loss (3.4, 3.6, 3.5, and 7.1 % in the 0, 2.6, 4.4, and 6.3 mg/m^3 groups; p = 0.11) was observed at the highest concentration tested; however, these changes were within the range of

historical controls and not statistically significant. No developmental effects were reported; the no-observed-adverse-effect concentration (NOAEC) for developmental toxicity was determined to be > 6.3 mg/m^3. A NOAEC for maternal toxicity could not be established, as increased lung weight was observed at all levels of exposure (Schroeder, 2003)

A-6 MUTAGENICITY

The available in vitro data for ATO-induced mutagenicity is mixed. ATO was not mutagenic when tested in Salmonella typhimurium strains TA98, TA100, TA1535, and TA1537 or Escherichia coli WP2P and WP2P uvrA with or without metabolic activation; however, ATO induced chromosomal aberrations in cultured human leukocytes and lymphocytes (Elliott et al., 1998). No mutagenic potential was observed in mouse lymphoma cells (Paton and Allison, 1972), but weak induction levels (~ two-fold above background) were reported in a sister chromatid exchange assay using V79 cells in vitro (Kuroda et al., 1991). Conflicting results have also been obtained for ATO-induced mutagenicity in vivo. Mice gavaged with aqueous ATO at 400, 668, and 1,000 mg/kg-bw/day for up to 21 days were positive for chromosomal aberrations, but not clastogenic effects (Gurnani et al., 1992). A mouse micronucleus test conducted with ATO showed no clastogenic effects following one high dose (5,000 mg/kg-bw) or repeated exposure at 400, 668, or 1,000 mg/kg-bw/day for 7, 14, or 21 days (Elliott et al., 1998). Negative results were also reported for micronuclei induction in rats following oral exposure at 250, 500, or 1,000 mg/kg-bw/day for 21 days (Kirkland et al., 2007).

Limited evidence of mutagenic potential has been demonstrated for ATO; however, a number of technical experts have independently concluded that ATO is unlikely to be mutagenic in vivo (EC/HC, 2010; European Food Safety Authority, 2004; Tweats et al., 2007a, 2007b). Since a possible local genotoxic effect of ATO would only be relevant under exposure conditions that also produce lung overload, minimal concerns are indicated for humans (EC, 2008).

A-7 CARCINOGENICITY

The International Agency for Research on Cancer (IARC) concluded that "there is sufficient evidence for the carcinogenicity of ATO in experimental animals" (IARC, 1989) based on one published, and one unpublished report of increased lung tumors in female rats following ATO inhalation exposure (Groth et al., 1986; Watt, 1983). Another chronic inhalation study has since been published that found no increased cancer incidence following ATO inhalation exposure in rats (Newton et al., 1994).

The proliferation of alveolar macrophages is generally regarded as a normal physiological response to the deposition of poorly soluble particulates in the lung; however, in rats repeatedly exposed at high levels, macrophage clearance is impaired, resulting in "lung overload". This condition is characterized by macrophage infiltration, chronic interstitial inflammation, metaplasia, and the subsequent development of lung tumors. Although the cascade of effects caused by pulmonary overload are unlikely to be relevant for humans, the rat

is assumed to be an appropriate model for both neoplastic and non-neoplastic responses to poorly soluble particulates (Mauderly, 1997). Because the responsiveness of the rat model to non-genotoxic agents (under conditions of lung overload) is dependent upon chronic active inflammation and cell proliferation, no lung cancer hazard is anticipated for humans exposed to ATO at lower, environmentally relevant concentrations, where chronic active inflammation and cell proliferation are not present (International Life Sciences Institute Workshop Participants, 2000).

IARC concluded that "there is inadequate evidence for the carcinogenicity of ATO in humans" (IARC, 1989). Since this evaluation, two epidemiological studies of increased cancer incidence in smelter workers have been published.

A mortality study conducted in 1,014 male workers employed in an antimony smelter plant between 1937 and 1971 showed a significant excess in mortality from lung cancer, ischemic heart disease and non-malignant respiratory disease; however, the authors acknowledged that the observed increase in cancer mortality may be influenced by co-exposure to arsenic (Schnorr et al., 1995).

A survey of antimony smelter workers employed before 1961 showed a significant increase in lung cancer mortality (12 expected vs. 5.3 observed, p = 0.016), with a minimum latency period of approximately 20 years between the first exposure and death from lung cancer (Jones, 1994). No correlation between the length of employment at the smelter and cancer mortality was found. In addition, no excess cancer mortality was found in the cohort recruited after 1960. This study is limited by the lack of an appropriate control population, and confounding factors such as cigarette smoking and co-exposure to polycyclic aromatic hydrocarbons, arsenic, and sulfur. As such, the increased cancer mortality observed in the earlier cohort could not be attributed to a specific agent.

While it is clear that ATO inhalation exposure can impair pulmonary clearance in rats (Newton et al., 1994), it is important to note that animal studies tend to employ significantly higher concentrations and longer durations of exposure than would generally be expected for humans. Although the available information is limited, there is no evidence of carcinogenicity via the oral route (Schroeder et al., 1968; Schroeder et al., 1970).

A-8 SYNOPSIS

Inhalation of airborne dusts is the primary concern for ATO exposure in occupational settings, whereas the general population is primarily exposed to low levels of ATO in food and water.

Chemical speciation and route of exposure are major determinants of antimony toxicity in humans. Generally speaking, pentavalent antimony is less toxic than the trivalent species; however, most antimony compounds are oxidized to ATO upon release into the environment, and the available evidence suggests that over time, further oxidation to the pentavalent form

may occur under aerobic conditions (EC, 2008). In addition, quantitative studies indicate that minimal amounts of ATO are released from flame-retarded consumer products (CPSC, 2006, 2012; Thomas and Stevens, 2005).

The general population is exposed mainly via ingestion of food and water, whereas inhalation and dermal contact are more common in occupational settings. Irritation at the point of contact (e.g., dermatitis, lung inflammation, pneumoconiosis) has been reported in workers following chronic inhalation exposure (NIOSH, 1978); however, a permissible exposure limit of 0.5 mg/m³ has been set to protect this population. ATO produces minimal systemic toxicity when administered via oral, dermal, or inhalation routes due to poor systemic absorption, which effectively limits bioavailability. Because food and water are the primary sources of general population exposure, and the pentavalent (i.e., less toxic) form of antimony predominates in these media, significant health risks are not expected at concentrations to which individuals may be exposed in the environment (ATSDR, 1992). This conclusion is supported by previous risk assessments for ATO in Europe and Canada (EC/HC, 2010; EC, 2008).

A-9   PREVIOUS ASSESSMENTS

A-9-1   Environment Canada/Health Canada: Screening Assessment for the Challenge – ATO (EC/HC, 2010)

(http://www.ec.gc.ca/ese-ees/9889ABB5-3396-435B-8428-F270074EA2A7/batch9_1309-64-4_en.pdf)

In Canada, ATO is listed on the Domestic Substances List and is characterized as "inherently toxic to aquatics," but did not meet the full categorization criteria and was not recommended for further assessment. In this assessment, it was concluded:

"The margins between upper-bounding estimates of exposure to antimony trioxide from environmental media (based on antimony) and from use of household items and levels associated with effects in experimental animals are considered to be adequately protective to account for uncertainties in the health effects and exposure databases. On the basis of the adequacy of the margins between conservative estimates of exposure to antimony trioxide and critical effect levels in experimental animals, it is concluded that antimony trioxide is not entering the environment in a quantity or concentration or under conditions that constitute or may constitute a danger in Canada to human life or health."

A-9-2   EU RAR – Diantimony Trioxide (EC, 2008)

(http://esis.jrc.ec.europa.eu/doc/risk_assessment/REPORT/datreport415.pdf)

In Europe, ATO is registered under REACH regulations; the European Commission classifies and labels ATO as "H351: suspected of causing cancer by inhalation;" therefore, "a need for limiting the risks" was identified for workers.

The assessment of human health and environmental risks concluded:

"Overall results of the risk assessment for surface water, waste water treatment plants, terrestrial compartment, atmosphere, secondary poisoning, marine water, marine sediment, secondary poisoning in the marine environment: There is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already."

A-9-3   OECD Assessment for Antimony Trioxide (OECD, 2008)

(http://webnet.oecd.org/Hpv/UI/handler.axd?id=13e93c97-6605-4eac-961f-8af23cc6ad32)

A-9-3-1   Environment

"This chemical is currently of low priority for further work."

A-9-3-2   Human Health

"Despite the poor quality of the available acute oral and dermal toxicity studies with diantimony trioxide in laboratory animals, diantimony trioxide is considered to be of low acute inhalation, oral and dermal toxicity. The chemical possesses properties indicating a hazard for human health (skin irritation, lung toxicity and lung carcinogenicity after repeated exposure). Member countries are invited to perform an exposure assessment for workers and if then indicated, a risk assessment. As all studies on chronic toxicity/carcinogenicity deviate from the OECD guideline and because of other critical shortcomings in the data set on chronic toxicity/carcinogenicity, the US NTP (National Toxicology Program) has initiated 2-year inhalation toxicology and carcinogenicity studies on diantimony trioxide in rats and mice." (http://ntp.niehs.nih.gov/?objectid=BCD52E6A-123F-7908-7BD5298A5D912AF9)

Appendix B EFAST2 MODEL INPUTS USED WITH TRI DATA

TRI does not require facilities to report the number of days associated with reported releases, therefore two release scenarios (assuming 24 or 250 release days per year) were developed to provide an upper and lower bound for the range of surface water concentrations predicted by the EFAST2 model. Typically one would enter a removal rate associated with waste water treatment, but in regard to the TRI data, the amount listed is the self-reported amount released into water and a removal rate is not applicable, therefore 0% is used. One of the cases below was a facility that released into a bay (Galveston). Additional input is needed in these types of cases, therefore the table for Galveston Bay is different from the others.

FACILITY NAME: TN EASTMAN CO KINGSPORT		

FACILITY LOCATION: KINGSPORT TN 37660

RECEIVING STREAM NAME: HOLSTON R S FK

REACH NUMBER: 06010102004	FACILITY ON REACH: Yes	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: TN0002640	GAGING STATION ID: N/A	
GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 6343.16	7q10 FLOW (MLD): 1937.37	EFFLUENT FLOW (MLD):1937.37

FACILITY NAME: GUILFORD MILLS INC/GUIL. EAST		

FACILITY LOCATION: KENANSVILLE NC 28349

RECEIVING STREAM NAME: NE CAPE FEAR R

REACH NUMBER: 03030007009	FACILITY ON REACH: Yes	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: NC0002305	GAGING STATION ID: N/A	

FACILITY NAME: GUILFORD MILLS INC/GUIL. EAST

GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 1041.00	7q10 FLOW (MLD): 23.04	EFFLUENT FLOW (MLD):2.70

FACILITY NAME: ARTEVA SPECIALTIES D/B/A KOSA

FACILITY LOCATION: SPARTANBURG SC 29307

RECEIVING STREAM NAME: PACOLET R

REACH NUMBER: 03050105066	FACILITY ON REACH: No	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: SC0002798	GAGING STATION ID: N/A	
GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 1797.77	7q10 FLOW (MLD): 297.48	EFFLUENT FLOW (MLD):3.50

FACILITY NAME: G E PLASTICS - SELKIRK OPER

FACILITY LOCATION: SELKIRK NY 12158

RECEIVING STREAM NAME: HUDSON R

REACH NUMBER: 02020006041	FACILITY ON REACH: Yes	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: NY0007072	GAGING STATION ID: N/A	
GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 3.65E+04	7q10 FLOW (MLD): 6866.81	EFFLUENT FLOW (MLD):7.38

FACILITY NAME: MILLIKEN/ABBEVILLE PLANT

FACILITY LOCATION: ABBEVILLE SC 29620

RECEIVING STREAM NAME: LONG CANE CR

REACH NUMBER: 03060103021	FACILITY ON REACH: Yes	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: SC0000353	GAGING STATION ID: N/A	
GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 88.42	7q10 FLOW (MLD): 17.75	EFFLUENT FLOW (MLD):2.79

FACILITY NAME: ARTEVA SPECIALTIES – KOSA

FACILITY LOCATION: SALISBURY NC 28144

RECEIVING STREAM NAME: SECOND CR

REACH NUMBER: 03040102028	FACILITY ON REACH: Yes	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: NC0004944	GAGING STATION ID: N/A	
GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 727.70	7q10 FLOW (MLD): 70.58	EFFLUENT FLOW (MLD):6.29

FACILITY NAME: DU PONT TEIJIN FILMS

FACILITY LOCATION: HOPEWELL VA 23860

RECEIVING STREAM NAME: JAMES R

REACH NUMBER: 02080206045	FACILITY ON REACH: Yes	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: VA0003077	GAGING STATION ID: N/A	

FACILITY NAME: MILLIKEN/ABBEVILLE PLANT

GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 1.85E+04	7q10 FLOW (MLD): 1468.34	EFFLUENT FLOW (MLD):0.68

FACILITY NAME: HAYWOOD CO-BROWNSVILLE

FACILITY LOCATION: BROWNSVILLE TN 38012

RECEIVING STREAM NAME: HATCHIE R

REACH NUMBER: 08010208012	FACILITY ON REACH: No	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: TN0041939	GAGING STATION ID: N/A	
GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 5709.76	7q10 FLOW (MLD): 1902.97	EFFLUENT FLOW (MLD):15.07

FACILITY NAME: CITY OF LA PORTE

FACILITY LOCATION: LA PORTE TX 775721115

RECEIVING WATER NAME: GALVESTON BAY

REACH NUMBER: 12040204001   FACILITY ON REACH: No   DISCHARGE TYPE: Direct

NPDES PERMIT #: TX0022799   EXPOSED POPULATION: Adult

WWT REMOVAL (%)	RELEASE DAYS	PLANT FLOW (MLD)	PRETREATMENT RELEASE (kg/site/day)	POST-TREATMENT RELEASE (kg/site/day)	BCF (L/kg)
0.	250.	11.83	0.03	3.00E-02	0.00

AQUATIC EXPOSURE ESTIMATES		
MIXING ZONE	DILUTION FACTOR	WATER CONC (µg/l)
ACUTE SCENARIO	1.00	2.54
CHRONIC SCENARIO	1.00	2.54

FACILITY NAME: HONEYWELL INTERNATIONAL INC

FACILITY LOCATION: MONCURE NC 27559

RECEIVING STREAM NAME: HAW R

REACH NUMBER: 03030002001   FACILITY ON REACH: Yes   DISCHARGE TYPE: Direct

NPDES PERMIT NUMBER: NC0001899   GAGING STATION ID: N/A

GAGING STATION   GAGING STATION   NUMBER OF STATIONS ON REACH: N/A

AQUATIC EXPOSURE ESTIMATES

PERIOD OF RECORD:N/A	NUMBER OF OBSERVATIONS: N/A	
MEAN FLOW (MLD): 3876.49	7q10 FLOW (MLD): 144.25	EFFLUENT FLOW (MLD):0.87

FACILITY NAME: HARTSELLE CITY OF UTIL BOARD

FACILITY LOCATION: HARTSELLE AL 35640

RECEIVING STREAM NAME: FLINT CREEK

REACH NUMBER: 06030002032	FACILITY ON REACH: No	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: AL0054674	GAGING STATION ID: N/A	
GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 570.89	7q10 FLOW (MLD): 119.88	EFFLUENT FLOW (MLD):5.16

FACILITY NAME: GENERAL CABLE

FACILITY LOCATION: MARION IN 46953

RECEIVING STREAM NAME: MISSISSINEWA R

REACH NUMBER: 05120103010	FACILITY ON REACH: Yes	DISCHARGE TYPE: Direct
NPDES PERMIT NUMBER: IN0002372	GAGING STATION ID: N/A	
GAGING STATION PERIOD OF RECORD:N/A	GAGING STATION NUMBER OF OBSERVATIONS: N/A	NUMBER OF STATIONS ON REACH: N/A
MEAN FLOW (MLD): 1562.64	7q10 FLOW (MLD): 51.52	EFFLUENT FLOW (MLD):0.28

FACILITY NAME: TYCO ADHESIVES

FACILITY LOCATION: FRANKLIN KY 42134

RECEIVING STREAM NAME: DRAKES CR W FK

REACH NUMBER: 05110002008   FACILITY ON REACH: No   DISCHARGE TYPE: Direct

NPDES PERMIT NUMBER: KY0074659   GAGING STATION ID: 03313700

GAGING STATION   GAGING STATION   NUMBER OF STATIONS ON REACH: 1
PERIOD OF RECORD:10/01/68 - 09/29/89   NUMBER OF OBSERVATIONS: 7670.00

50 %ile FLOW (MLD): 173.66   10 %ile FLOW (MLD): 20.55   EFFLUENT FLOW (MLD):0.82

FACILITY NAME: BOREALIS COMPOUNDS LLC

FACILITY LOCATION: PORT MURRAY NJ 07865

RECEIVING STREAM NAME: MUSCONETCONG R

REACH NUMBER: 02040105008   FACILITY ON REACH: Yes   DISCHARGE TYPE: Direct

NPDES PERMIT NUMBER: NJ0028657   GAGING STATION ID: 01457000

GAGING STATION   GAGING STATION   NUMBER OF STATIONS ON REACH: 3
PERIOD OF RECORD:01/01/66 - 12/11/90   NUMBER OF OBSERVATIONS: 9112.00

50 %ile FLOW (MLD): 503.87   10 %ile FLOW (MLD): 215.25   EFFLUENT FLOW (MLD):0.47

Appendix C   SUMMARY OF USGS AND STORET DATA

Table_Apx C-1. Summary of All USGS-NWIS and STORET Data for Exceedances of the Concentrations of Concern per Media Type

Summary of All USGS-NWIS and STORET Data Sample Type	Parameter Code	Date of Sample	Value Above COC	Station Name	Latitude	Longitude	Data Description (USGS-NWIS data only)[b]
Water – USGS-NWIS	01097: Antimony, water, filtered, µg/L	26-Aug-87	350[c] µg/L	Red River near Quanah, TX	34.4131291	-99.7345385	Historical data
		17-Feb-88	350[c] µg/L	Red River near Quanah, TX	34.4131291	-99.7345385	Historical data
		25-Aug-87	350[c] µg/L	Buck Creek at Loco, TX	34.70283589	-100.1881691	Historical data
		25-Aug-87	350[c] µg/L	Buck Creek near Childress, TX	34.61172738	-100.1109427	Historical data
		26-Aug-87	350[c] µg/L	Salt Fork Red River near Wellington, TX	4.95755294	-100.220949	Historical data
Water – STORET		10-Dec-08	7,600 µg/L	Grays Creek, TN	35.200600	-89.681500	NA
		26-Oct-04	2,000 µg/L	Beech River, TN	36.605000	-88.118800	NA
		3-Nov-04	2,000 µg/L	Muddy Creek-Unnamed Tributary, TN	35.095833	-88.499444	NA
		18-Dec-02	1,000 µg/L	Alexander Creek, TN	36.520600	-82.717800	NA
		26-Oct-04	1,000 µg/L	Big Sandy River, TN	36.232500	-88.100300	NA
		23-Sep-04	1,000 µg/L	Buffalo River, TN	35.457800	-87.775800	NA
		12-Oct-04	1,000 µg/L	Clinch River, TN	36.569400	-83.041700	NA

Summary of All USGS-NWIS and STORET Data Sample Type	Parameter Code	Date of Sample	Value Above COC	Station Name	Latitude	Longitude	Data Description (USGS-NWIS data only)[b]
Sediment – USGS	29816: Antimony, suspended sediment, total digestion, dw	12-Jan-88	90 mg/kg	Kentucky River at Lock 4 at Frankfort, KY	38.2017388	-84.8816171	Historical data
		20-Dec-89	42 mg/kg	Kentucky River at Lock 2 at Lockport, KY	38.43895654	-84.9632844	Historical data
	01098	26-Oct-05	12 mg/kg	Lake Mohawk 1.3 miles upstream of dam near Sparta, NJ	41.0176111	-74.6578056	Presumed satisfactory
	34795	26-Oct-05	12 mg/kg	Lake Mohawk 1.3 miles upstream of dam near Sparta, NJ	41.0176111	-74.6578056	Presumed satisfactory
	29816	14-Apr-95	26 mg/kg	Mohawk River at Cohoes, NY	42.78535559	-73.7076188	Historical data
	29816	14-Apr-95	23 mg/kg	Mohawk River at Cohoes, NY	42.78535559	-73.7076188	Historical data
	29816	17-Sep-96	13 mg/kg	St. Lawrence River at Cornwall Ontario near Massena, NY	45.0061599	-74.7949099	Historical data
	29816	26-Sep-96	20 mg/kg	Rio Grande bl Falcon Dam, TX	26.5572768	-99.1683666	Historical data
	29816	21-Mar-96	14 mg/kg	Rio Grande bl Falcon Dam, TX	26.5572768	-99.1683666	Historical data
	29816	01-May-96	18 mg/kg	Rio Grande at Foster Rh near Langtry, TX	29.78075694	-101.7559523	Historical data
	34795	19-Dec-05	20 mg/kg	ZK-58-27-7 at Avery Deer Spring, TX	30.5075	-97.7491667	Reviewed and accepted
	29816	29-May-96	18 mg/kg	Rio Grande at El Paso, TX	31.80288488	-06.5408218	Historical data
	29816	23-Oct-96	16 mg/kg	Rio Grande at El Paso, TX	31.80288488	-06.5408218	Historical data

Summary of All USGS-NWIS and STORET Data Sample Type	Parameter Code	Date of Sample	Value Above COC	Station Name	Latitude	Longitude	Data Description (USGS-NWIS data only)[b]
	34795	30-Aug-03	18 mg/kg	AF Plant 4 Ofl No. 4 near Bomber Road, Fort Worth, TX	32.77619444	-97.4535833	Reviewed and accepted
Sediment – STORET		21-Sept-0404	88.1 mg/kg	West/East Creeks, NJ	39.176000	-74.900000	NA
		24-Sept-0404	57.2 mg/kg	Great Sound, NJ	39.009000	-74.815000	NA
		28-Sept-0404	35.8 mg/kg	Great Egg Harbor, NJ	39.366000	-74.716000	NA
		8-Aug-005	20.5 mg/kg	Upper NY/NJ Harbor, NY	40.699000	-74.020000	NA
		3-Oct-006	13.7 mg/kg	Raritan Bay, NJ	40.494000	-74.288000	NA
		3-Oct-0606	11.8 mg/kg	Raritan River, NJ	40.503000	-74.292000	NA

[a]USGS-NWIS dataset collected was filtered using the following selection criteria:
1st Data selection factor = Site type name (Site_tp_In). All types listed; those in **blue bold** are the ones selected: Atmosphere; **Coastal - 18 water samples;** Collector or Ranney-type well; Ditch; **Estuary - only sediment samples;** Hyporheic-zone well; **Lake, Reservoir, Impoundment - 279 water samples; Outfall - 27 water samples;** Pavement; Soil hole; **Spring - 174 water samples;** Storm sewer; **Stream - 3,151 water samples;** Test hole not completed as well; Tunnel, shaft, or mine; Unsaturated zone; Wastewater land application; Water-distribution system; Water-use establishment; and Well. *2nd Data selection factor* = Media Name (Medium_nm). All types listed; those in **blue bold** are the ones selected: Animal tissue; Artificial; **Bottom material;** Core material; **Effluent- 1 water sample;** Groundwater; Interstitial water; **Soil; Surface water - 3,648 water samples; Suspended sediment;** and Wet deposition. *Data selection factor for Water samples*: medium_nm = Surface water and Effluent. *Data selection factor for Sediment samples*: Medium Name - Bottom Material, Effluent, Surface water (as reported in mg/kg), Suspended sediment, and wet deposition. *Data selection factor for Soil samples*: medium_nm = Soil.[b]The USGS-NWIS dataset contains a field named 'dqui_nm,' which is defined as 'Data Quality Descriptions.' STORET does not have this field; therefore, NA (Not applicable) is denoted in this column.
[c]½ detection/reported value.